Assessing the effectiveness of UNESCO biosphere reserves - towards a global monitoring tool of ecological functioning

Charlotte Gohr

April 2024

Assessing the effectiveness of UNESCO biosphere reserves - towards a global monitoring tool of ecological functioning

academic dissertation presented to
the faculty of sustainability
of the Leuphana University Lüneburg

for the award of the degree
Doctor of Natural Sciences
- Dr. rer. nat. -

submitted by

Charlotte Gohr

born February 17th, 1992, in Potsdam, Germany

Submitted on April 2nd, 2024
Defended on Jun 21st, 2024
First supervisor and reviewer: Prof. Dr. Pierre L. Ibisch
Second supervisor and reviewer: Prof. Dr. Henrik von Wehrden
Third reviewer: Prof. Dr. Fabian Faßnacht

“The stars don’t look bigger, but they do look brighter.”

Astronaut Sally Ride in space

Structure

SUMMARY 3

ZUSAMMENFASSUNG 5

GRAPHICAL ABSTRACT 7

1. INTRODUCTION 8

1.1. ECOLOGICAL EFFECTIVENESS 9
1.2. APPLIED REMOTE SENSING 10
1.3. FORESTS 11
1.4. BIOSPHERE RESERVES 12
1.5. RESEARCH DESIGN 13

2. SCIENTIFIC RESEARCH ARTICLES 16

2.1. ARTICLE I 16
2.2. ARTICLE II 31
2.3. ARTICLE III 43
2.4. ARTICLE IV 56

3. SYNTHESIS 70

3.1. EFFECTIVENESS & REMOTE SENSING 70
3.2. EFFECTIVENESS & FORESTS 71
3.3. EFFECTIVENESS & CONSERVATION 72
3.4. EFFECTIVENESS & SUSTAINABILITY SCIENCE 73
3.5. FUTURE RESEARCH DIRECTIONS 74

FRAMEWORK REFERENCES 75

APPENDIX 81

A 1 OUTPUT AND FUNDING INFORMATION 82
A 2 SUPPLEMENTARY MATERIAL FOR 2.3 84
A 3 SUPPLEMENTARY MATERIAL FOR 2.4 92
A 4 MANUSCRIPT OF DABARD ET AL. 2024 122
A 5 POSITIONALITY STATEMENT 149
A 6 ACKNOWLEDGEMENTS 150

Summary

Today, ecosystems worldwide are affected by multiple threats, such as loss of biodiversity and extreme weather events. They face habitat fragmentation and are under increasing pressure through human interventions. One approach to mitigate these pressures might be the world network of biosphere reserves, designated by the United Nations Educational, Scientific and Cultural Organization (UNESCO). Designed as learning places for sustainable development, biosphere reserves aim at integrating the protection of biodiversity and the sustainable use of the ecosystems by humans.

Assessing protected area effectiveness with satellite imagery is a common scientific approach. It is time- and cost-effective and facilitates study results at a high temporal and spatial resolution. In the last decades, a vast amount of preprocessed satellite imagery became freely available and processable using cloud-computing platforms. This development allows scientists and practitioners to conduct environmental monitoring worldwide.

Consequently, this thesis investigates pathways to develop a tool to assess the effectiveness of biosphere reserves worldwide using remote sensing in four articles.

(1) Several approaches exist to assess the effectiveness of protected areas using remote sensing. A literature review reveals the focus on single indicator-based local or regional case studies and, with it, the lack of a common framework. Although most studies acknowledge the need for a socio-ecological framework, most studies focus on ecological proxies. The reviewed studies with a focus on forests found that forested protected areas are considered effective if they protect forest cover and prevent forest degradation.

(2) A regional-scale study on ecosystem functions of different land cover types highlights the potential of forests and water bodies to regulate temperatures on a landscape level. Higher shares of forest cover cool the landscape, particularly on hot days and correlate with a higher vitality. Investigating the temperature effects of forests on hot days proves to be a reliable proxy for forest ecosystem function.

(3) Subsequently, connectivity was added as another proxy of forest ecosystem functioning to evaluate temperature regulation and vegetation vitality in a study on forest fragmentation in Germany. It was found that, regardless of ecoregions and forest types, better connected forest patches had a better temperature regulation and higher vitality.

(4) The presented findings build the base for a comprehensive assessment of forest ecosystem functions in 119 biosphere reserves worldwide using multiple proxies

designed with satellite imagery. Four different proxies of primary productivity, temperature regulation, evapotranspiration and connectivity individually indicated signs of effective ecosystem functioning as compared to their surroundings. Yet, when considering multiple ecosystem functions, most forests within the world network of biosphere reserves do not perform better than those outside.

This thesis contributes not only to our knowledge of protected area (socio-) ecological effectiveness but also to the potential of applied remote sensing in monitoring ecosystem functions. To begin with, the effectiveness of ecosystems and protected areas is investigated constantly using remote sensing, but no common socio-ecological framework exists. In addition, hot-day temperature, primary productivity and forest patch connectivity tracked from space are relevant proxies to assess forests worldwide and clearly show the high pressure of human-induced climate change on forest ecosystems in recent years. Finally, UNESCO biosphere reserves claim to create model regions for sustainable development yet need to improve the support for forest ecosystem functions worldwide.

Zusammenfassung

Ökosysteme sind weltweit von zahlreichen Bedrohungen betroffen, wie dem Verlust der biologischen Vielfalt und extremen Wetterereignissen. Lebensräume werden fragmentiert und der Einfluss menschlicher Eingriffe und Veränderungen nimmt zu. Ein Ansatz zur Abschwächung dieser Belastungen könnte das Weltnetz der Biosphärenreservate sein, die von der Organisation der Vereinten Nationen für Erziehung, Wissenschaft und Kultur (UNESCO) ausgewiesen werden. Biosphärenreservate werden als Lernorte für nachhaltige Entwicklung konzipiert und sollen den Schutz der biologischen Vielfalt und die nachhaltige Nutzung der Ökosysteme durch den Menschen miteinander verbinden.
Die Bewertung der Wirksamkeit von Schutzgebieten mit Hilfe von Satellitenbildern ist ein gängiger wissenschaftlicher Ansatz. Sie ist zeit- und kosteneffizient und ermöglicht Ergebnisse in hoher zeitlicher und räumlicher Auflösung. In den letzten Jahrzehnten wurde eine große Menge an vorverarbeiteten Satellitenbildern frei verfügbar, die über Cloud-Computing-Plattformen verarbeitet werden können. Diese Entwicklungen ermöglichen Wissenschaftler*innen und Praktiker*innen, Umweltbeobachtungen durchzuführen.
In dieser Arbeit wird anhand von vier Artikeln untersucht, wie ein Instrument zur Bewertung der Wirksamkeit von Biosphärenreservaten weltweit mithilfe der Fernerkundung entwickelt werden kann.

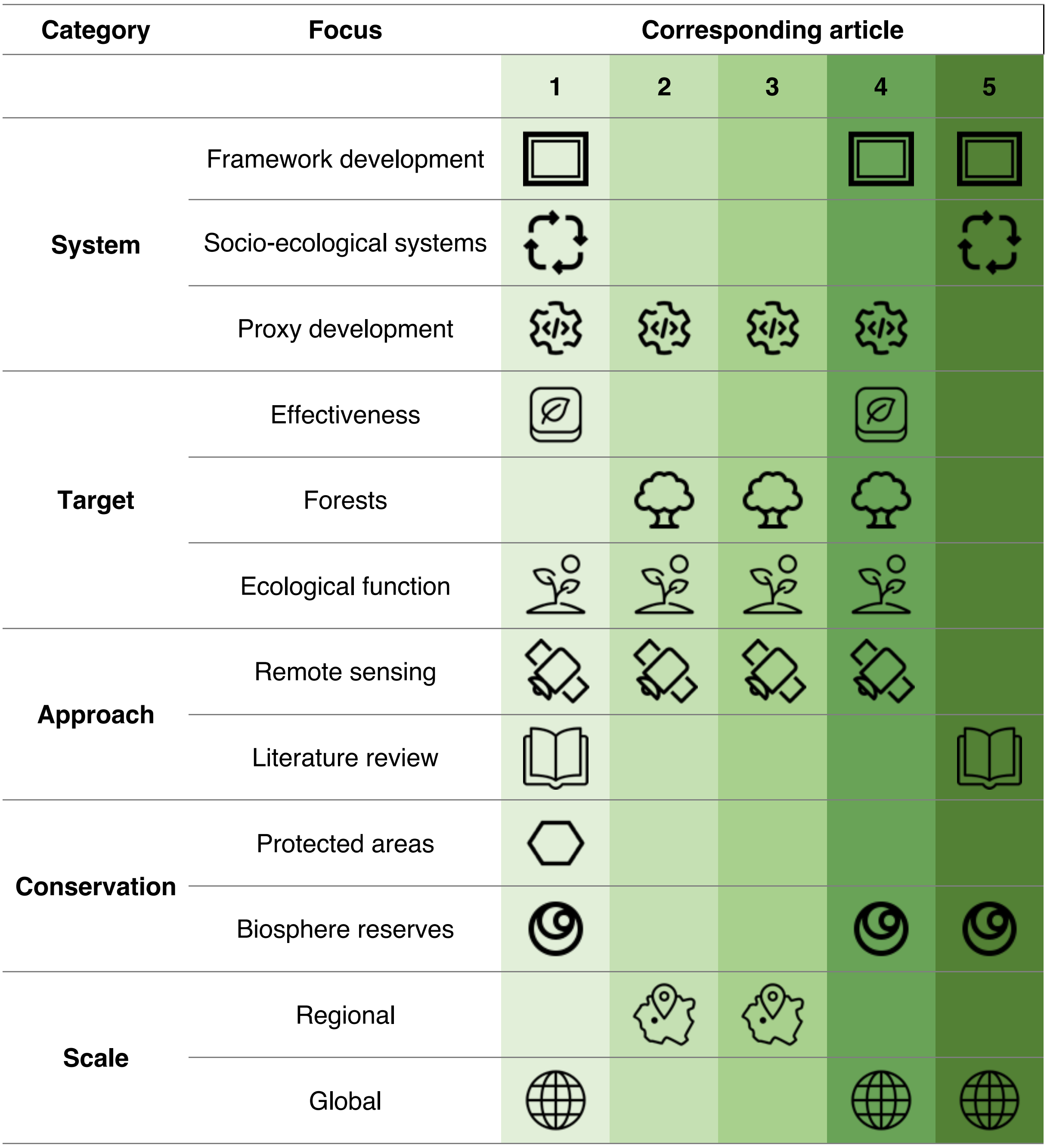
(1) Es gibt verschiedene Ansätze zur Bewertung der Wirksamkeit von Schutzgebieten mit Hilfe der Fernerkundung. Die Auswertung der Literatur macht deutlich, dass der Schwerpunkt auf einzelnen indikatorbasierten lokalen oder regionalen Fallstudien liegt und zeigt damit das Fehlen eines gemeinsamen Rahmenkonzeptes auf. Obwohl die meisten Studien die Notwendigkeit eines sozio-ökologischen Rahmenkonzepts anerkennen, konzentrieren sie sich zumeist auf ökologische Indikatoren. Die untersuchten Artikel mit Waldfokus kamen zu dem Ergebnis, dass bewaldete Schutzgebiete als wirksam angesehen werden, wenn sie die Waldfläche schützen und die Walddegradierung verhindern.
(2) Eine regional angelegte Studie über die Ökosystemfunktion verschiedener Landnutzungstypen unterstreicht das Potenzial von Wäldern und Wasserflächen zur Temperaturregulierung auf Landschaftsebene. Ein höherer Waldanteil kühlt die Landschaft, insbesondere an heißen Tagen, und korreliert mit einer höheren Vitalität. Die Untersuchung der Temperatureffekte von Wäldern an heißen Tagen erweist sich als ein zuverlässiger Proxy für die Ökosystemfunktion von Wäldern.
(3) Darauf aufbauend wurde in einer Studie zur Waldfragmentierung in Deutschland die Konnektivität als weiterer Proxy der Waldökosystemfunktion untersucht, um die

Temperaturregulierung und die Vitalität der Vegetation zu bewerten. Stärker verbundene Waldflächen weisen eine bessere Temperaturregulierung und eine höhere Vitalität auf, unabhängig von Ökoregionen und Waldtypen.

(4) Die vorgenannten Ergebnisse bilden die Grundlage für eine umfassende Bewertung der Funktionen von Waldökosystemen in 119 Biosphärenreservaten weltweit unter Verwendung mehrerer, mit Hilfe von Satellitenbildern entwickelter, Proxies. Vier verschiedene Proxies für die Primärproduktivität, die Temperaturregulierung, die Evapotranspiration und die Konnektivität zeigten im Vergleich zu ihrer Umgebung Anzeichen für effektive Ökosystemfunktionen. Betrachtet man jedoch mehrere Ökosystemfunktionen, so schneiden die meisten Wälder innerhalb des Weltnetzes der Biosphärenreservate nicht besser ab als die Wälder in ihrer Umgebung.

Diese Arbeit trägt nicht nur zum tieferen Verständnis über die (sozio-)ökologische Effektivität von Schutzgebieten bei, sondern zeigt auch das Potenzial der angewandten Fernerkundung bei der Untersuchung von Ökosystemfunktionen auf. Zum einen wird die Wirksamkeit von Ökosystemen und Schutzgebieten stets mit Hilfe der Fernerkundung untersucht, aber ein gemeinsames sozio-ökologisches Rahmenkonzept fehlt. Weiterhin sind die Temperaturen an heißen Tagen, die Primärproduktivität und die Vernetzung von Waldflächen, die aus Satellitendaten gewonnen werden, wichtige Proxies für die Bewertung von Wäldern weltweit. Sie zeigen deutlich den hohen Druck, den der vom Menschen verursachte Klimawandel in den letzten Jahren auf Waldökosysteme ausgeübt hat. Schließlich erheben die UNESCO-Biosphärenreservate den Anspruch, Modellregionen für eine nachhaltige Entwicklung zu schaffen, wobei die Unterstützung der Waldökosystemfunktionen weltweit verbessert werden muss.

Graphical Abstract



1 Remotely sensed effectiveness assessments of protected areas lack a common framework: A review
2 Quantifying the mitigation of temperature extremes by forests and wetlands in a temperate landscape
3 Does fragmentation contribute to the forest crisis in Germany?
4 Low effectiveness of the world network of biosphere reserves in maintaining forest ecosystem functions
5 Biosphere Reserves as model regions for transdisciplinarity? A literature review

Icons by icons8.com

1. Introduction

Protected areas are considered to be one of the most effective means to mitigate the multiple consequences of human-induced climate change that we are facing today (Duncanson et al. 2023; Xu et al. 2022). Still, one-third of the protected areas worldwide are under intense pressure through human interventions (Jones et al. 2018).
While humans profit from functioning ecosystems, they are also driving the loss of biodiversity and are substantially altering ecosystem functioning across the globe (Isbell et al. 2017). The demand for resources and the need to protect nature create a dilemma.
Biosphere reserves have been designated by the United Nations Educational, Scientific and Cultural Organization UNESCO as model regions for sustainability since 1976 (UNESCO 2015). They are intended to resolve this dilemma by addressing both development and conservation within the differentiated zoning of model territories for sustainability. Research in these areas increased in the last decades and foci range from sustainable development and management to conservation and ecology (Kratzer 2018).
To investigate aspects of the world network of biosphere reserves, a comprehensive data basis covering all areas is a prerequisite. Satellite imagery is the only available data source with global coverage. Assessing the ecological effectiveness of biosphere reserves requires the selection of suitable indicators. Remotely sensed proxies of ecosystem functioning can be applied for this purpose (Pettorelli et al. 2018): To determine whether biosphere reserves can mitigate the consequences of human-induced climate change, focusing on extreme heat events and their consequences, relevant proxies are temperature regulation capacities, changes in ecosystem vitality and habitat connectivity (Turner et al. 2020; Zscheischler et al. 2018; Grimm et al. 2013).

This dissertation investigates aspects of the ecological effectiveness of ecosystems and biosphere reserves worldwide on multiple spatial and temporal scales. Its focus lies on forest ecosystems, as most living beings depend on their ecosystem functions and are strongly affected by climate change at the same time. For global coverage, this work aims to find pathways to assess effectiveness using satellite imagery.

The four main research questions resulting from this idea to create a monitoring tool for ecological protected area effectiveness are:

1. Which approaches, methods and gaps of remotely sensed effectiveness assessments of protected areas can be found in the literature?
2. How can the temperature regulation for forests and water ecosystems as a proxy for temperature regulation capacity be quantified?
3. Can we find a higher temperature regulation capacity and vitality in better connected forest fragments and how fragmented are temperate forests?
4. Do biosphere reserves worldwide facilitate healthier ecosystem functions than their surroundings using a multiproxy analysis?

1.1. Ecological effectiveness

Ecological effectiveness frameworks have been defined in many ways, fitted to the respective hypotheses and methods of different research fields with a focus on the effectiveness of conserving ecosystems (Durán, Barbosa, and Gaston 2022; Ghoddousi, Loos, and Kuemmerle 2021; Lee and Abdullah 2019; Chape et al. 2005). The ecological effectiveness of an ecosystem could be defined by its ability to sustain itself: It is often also referred to as ecosystem functions or the health of ecosystems (Rapport, Costanza, and McMichael 1998). The ecological effectiveness of protected areas can be assessed using a multidimensional conceptual framework, rooted in social–ecological theory (Ghoddousi, Loos, and Kuemmerle 2021). Here, the effectiveness of protected areas is derived from ecological and social outcomes combined with social-ecological interactions. Similarly, the socio-ecological systems framework for biosphere reserves management effectiveness suggests assessing the context, progress, input and outcomes of the reserves, including setting the ecological attributes into context (Ferreira et al. 2018). Also, an ecological effectiveness framework for protected areas has been established recently. It focuses on the interconnection between design, location, threat, and management (Durán, Barbosa, and Gaston 2022). Generalized, these frameworks suggest approaches to measure terrestrial and marine protected areas with their potential and impact in effectively maintaining these ecosystem services and functions. Therefore, ecological effectiveness should be assessed by investigating ecosystem functions. Convincing definitions are that ecosystem functions are the direct and indirect benefits of ecosystem processes for a range of species, including humans (Pettorelli et al. 2018), combined with the ability of those ecosystems to adapt to internal and external changes (Freudenberger et al. 2012). Building on these frameworks this study defines the ecological effectiveness of biosphere reserves as the ability to maintain and or increase ecosystem functions within their boundaries and potentially have positive effects on their surroundings.

1.2. Applied remote sensing

Earth observation with satellites started in the 1950s with Russian and US-American missions (Tatem, Goetz, and Hay 2008). Today, more than 1000 satellites are orbiting the Earth, monitoring the water and land surface as well as the atmosphere (USC 2023). Generally, there are two types of sensors aboard satellites. Passive sensors receive reflected or emitted energy across the electromagnetic spectrum using the energy of the sun and send this data to Earth. Active sensors use microwave bands to send and receive information. New developments in sensors allow combined active and passive data collection (Rustamov et al. 2018). Regardless of the way the data was generated, all raw data must be corrected for reflectance, absorption, transmission, cloud cover and other potential artifacts. Subsequently, the preprocessed datasets are then mostly used for applied remote sensing purposes. The focus shifts from technical problem solving to the application of the cleaned data for environmental monitoring to answer societal questions and inform political decision-making (Pettorelli et al. 2014).

The spatial and temporal scales are major aspects of remote sensing. Most freely available satellite imagery, such as data from the Moderate Resolution Imaging Spectroradiometer (MODIS) and Landsat missions, collect data on a landscape level at a one kilometer or 30 meter pixel resolution respectively (NASA 2024a; 2024b). The Landsat missions have provided global coverage of earth imagery for 50 years. An increasing number of studies combine satellite imagery information with unmanned vehicle data and field observations to increase the spatial scale (Nagendra et al. 2013).

In the last decades, cloud computing has emerged to process these ever-growing geospatial datasets. Cloud platforms offer scalable, efficient and cost-effective solutions for storing, sharing, and processing huge datasets. Most prominently used in recent years is the Google Earth Engine (Gorelick et al. 2017). A similar interface, the Multi-Mission Algorithm and Analysis Platform (MAAP), a joint project by the US and European Space Agencies NASA and ESA, was launched in 2021 but is not yet a fully functional platform with a limited data catalog (ESA and NASA 2024). Simple open-access apps, which are world map interfaces with a backend supported by the Google Earth Engine editor, help to share generated knowledge easily. This was harder before, with higher costs for hardware and software and staff training (Pettorelli et al. 2014). Now, less data availability latency, capacity building, and better documentation have led to faster processing of geospatial datasets (Pettorelli et al. 2014). Besides free cloud computing platforms, the availability of free, ever-evolving software to process and visualize geospatial data, such as R (R Core Team 2024) and QGIS (QGIS 2024), has further accelerated this process.

These new possibilities have been matched by an ever-increasing interest in high temporal and spatial resolution images of the Earth. Especially the advances in radar data processing are notable, not only but increasingly for forest ecosystem assessments. Since radar data can penetrate through clouds, it mitigates the impact of weather effects on geospatial datasets and offers more information on vegetation structure and biomass (e.g. Jetz et al. 2016; Saatchi et al. 2011).

For this thesis, preprocessed, freely available passive satellite datasets were used for assessments of ecosystem functioning, with a special focus on forest ecosystems.

1.3. Forests

Forests are critical for Earth's climate entangled in local to global material cycles (Bonan 2008). Forests are important ecosystems as they inhere and provide key ecosystem functions for all living beings.

These include water and carbon cycle regulation, water and air purification, habitat provision, timber supply, and recreational and spiritual values for humans (Brockerhoff et al. 2017; Freudenberger et al. 2012).

Climate change and human activities alter the functionality of forests. A famous example is the Brazilian Amazonas, the largest tropical forest on earth, which has long acted as a major terrestrial carbon sink. Now it is becoming a carbon source additively (Gatti et al. 2021). The reasons are climate change and deforestation. Extreme heat events and droughts induce reduced photosynthetic activity, dieback, a higher vulnerability to insect attacks or the probability of forest fires (Schlesinger et al. 2016; Mildrexler et al. 2016). Droughts also imply water scarcity and the related vapor pressure deficit drives tree mortality (Grossiord et al. 2020).

Tree cover and forests have been assessed over decades with a magnitude of methods ranging from local to global scales. Field studies assessing carbon and water fluxes, or microclimate can be upscaled and linked to satellite imagery (Laura Duncanson et al. 2022; Harris et al. 2021; Kašpar et al. 2021; Haesen et al. 2021). Remotely sensed products such as gross primary productivity or the Normalized Difference Vegetation Index NDVI are used to assess changes in vegetation phenomena (Pettorelli et al. 2018).

Forest management recommendations for a sustainable future under climate change suggest the ban of monocultures and large-scale clearcuts and reforestation (Morecroft et al. 2019). Yet, forest management practices contributed to global warming (Blumröder et al. 2021; Naudts et al. 2016). Extreme heat events lead to increased tree mortality (Allen et al. 2010), making the protection of forests pivotal for maintaining their ecosystem functions. For forests in protected areas in the highest protection category of the catalog of the International Union for Conservation of Nature, IUCN (Dudley 2008), less forest cover loss was recorded as compared to protected area categories with lower protection standards (Leberger et al. 2020). Nevertheless, stricter protection is not necessarily the most important driver in reducing overall forest loss (Ferraro et al. 2013). Reduced forest cover loss inside protected areas can lead to intensified deforestation outside a protected area as well (Fuller et al. 2019). The diversity of influences shows that new ideas are needed to protect people and nature sustainably.

1.4. Biosphere Reserves

Biosphere reserves are designated by the United Nations Educational, Scientific and Cultural Organization (UNESCO) Man and the Biosphere Programme (UNESCO 2015). Launched in 1976 as the world network of biosphere reserves (WNBR), the first biosphere reserves were designated that year with the main objective being to conserve natural areas and their genetic material (Batisse 1982). With the Seville Strategy in 1996, the scope and purpose of these areas were broadened and now included the aim to explore and demonstrate sustainable development on a regional scale (UNESCO 1996). Their aims differ from the general concept of protected areas, which are established to sustainably protect and conserve nature as well as the associated ecosystem services and cultural resources, as defined by the International Union for Conservation of Nature, IUCN (Dudley 2008). Still, these two conservation approaches do spatially overlap at times when a strictly protected area (e.g. IUCN categories Ia - strict nature reserve or Ib – wilderness area) is the core zone of a biosphere reserve (e.g., La Mancha Húmeda biosphere reserve in Spain). For the implementation of biosphere reserves, the Man and the Biosphere Programme relies on the subdivision of each biosphere reserve into three zones that are intended to fulfill different functions. Ideally, each designated area consists of one or more core zones for long-term nature protection and conservation, with adjacent buffer zones and an outer transition area where the sustainable use of resources is promoted. The Man and the Biosphere Programme aims at integrating biological and cultural diversity by recognizing and empowering the role of local knowledge (UNESCO 2015). Over the last decades, the world network constantly grew and as of March 2024, there are 748 biosphere reserves in 134 countries, of which 23 are transboundary reserves (UNESCO 2024).

Research in and on biosphere reserves increased considerably with a propensity towards research in natural sciences (Kratzer 2018). The effectiveness of managing biosphere reserves was systematically reviewed (Ferreira et al. 2020). The Man and the Biosphere Programme monitors the effectiveness of the world network of biosphere reserves through periodic reviews that are ideally handed in every decade by each biosphere reserve handling authority (UNESCO 1996). These periodic reviews evaluate on a local scale the extent to which the biosphere reserves meet the designation criteria and can support the withdrawal from the network in case of non-compliance. Still, they cannot substitute a global standardized assessment method of biosphere reserve effectiveness in maintaining biodiversity and socio-ecological functioning.

Indeed, such frameworks are needed to understand the complex and diverse structures of the world network of biosphere reserves to guide management decisions. However, as a precondition for deeper socio-ecological analyses, updated geospatial information on borders and zonation of biosphere reserves are needed and, to date, is not accessible for the whole network (Palliwoda et al. 2021). This thesis develops a tool for assessing the ecological effectiveness of biosphere reserves using existing data, filling a crucial gap in the evaluation framework.

1.5. Research design

This thesis investigates pathways to assess the ecological effectiveness of UNESCO biosphere reserves using remote sensing.

The first publication reviews the current body of literature on assessing the socio-ecological effectiveness of protected areas using remote sensing. The larger scope includes not only biosphere reserves but all types of protected areas. The review suggests that a multi-proxy framework is needed to assess the socio-ecological effectiveness of protected areas.

The second publication explores temperature regulation as one proxy of ecosystem functioning and ecological effectiveness by developing a new method to quantify the temperature regulation of forest and water ecosystems. We find that in a temperate forest, both the temperature regulation capacity as well as vegetation vitality are connected to the share of forest cover.

The third publication on forest fragmentation in Germany adds connectivity as another proxy of forest ecosystem functioning to assessments on temperature regulation and vegetation vitality. Regardless of ecoregions and forest types, more connected forest patches exhibited better temperature regulation and higher vitality.

The fourth submitted manuscript combines the preceding work and uses a multi-proxy approach to assess the ecological effectiveness in terms of ecosystem functioning of forest ecosystems worldwide in biosphere reserves. This approach confirms the positive impact of biosphere reserves when considering single forest ecosystem functions. Yet most forests within the world network of biosphere reserves do not perform better than their surroundings when considering multiple ecosystem functions.

The thesis is accompanied in the appendix by a systematic literature review on the transdisciplinarity of biosphere reserves research. The review presents patterns within biosphere reserves research and highlights its potential in engaging with sustainability sciences and acknowledges the transformative potential of biosphere reserves.

2. Scientific research articles

2.1. Article I

Remotely sensed effectiveness assessments of protected areas lack a common framework: A review

Received: 14 December 2021 | Accepted: 7 January 2022

DOI: 10.1002/ecs2.4053

SYNTHESIS & INTEGRATION

Socio-Ecological Systems

ECOSPHERE
AN ESA OPEN ACCESS JOURNAL

Remotely sensed effectiveness assessments of protected areas lack a common framework: A review

Charlotte Gohr[1,2,3] | **Henrik von Wehrden**[3] | **Felix May**[4] | **Pierre L. Ibisch**[1,2]

[1]Centre for Econics and Ecosystem Management, Eberswalde University for Sustainable Development, Eberswalde, Germany

[2]Biosphere Reserves Institute, Eberswalde University for Sustainable Development, Eberswalde, Germany

[3]Faculty of Sustainability, Leuphana University Lüneburg, Lüneburg, Germany

[4]Freie Universität Berlin, Institute of Biology, Berlin, Germany

Correspondence
Charlotte Gohr
Email: charlotte.gohr@hnee.de

Funding information
Eberswalde University for Sustainable Development; Ministry of Science, Research and Culture of the federal state of Brandenburg

Handling Editor: Laura López-Hoffman

Abstract

Effective protected areas reflect socio-ecological values, such as biodiversity and habitat maintenance, as well as human well-being. These values, which safeguard ecosystem services in protected areas, are treated as models for the sustainable preservation and use of resources. While there is much research on the effectiveness of protected areas in a variety of disciplines, the question is whether there is a common framework that uses remote sensing methods. We conducted a qualitative and a quantitative analysis of 44 peer-reviewed scientific papers utilizing remote sensing data in order to examine the effectiveness of protected areas. Very few studies to date have a wide or even a global geographical focus; instead, most quantify the effectiveness of protected areas by focusing on local-scale case studies and single indicators such as forest cover change. Methods that help integrate spatial selection approaches, to compare a protected area's characteristics with its surroundings, are increasingly being used. Based on this review, we argue for a multi-indicator-based framework on protected area effectiveness, including the development of a consistent set of socio-ecological indicators for a global analysis. In turn, this will allow for globally applicable use, including a concrete evaluation that considers the diversity of regional parameters, biome-specific variables, and political frameworks. Ideally, such a framework will enhance the monitoring and evaluation of global strategies and conventions.

KEYWORDS
effectiveness, protected areas, remote sensing, socio-ecological indicators

INTRODUCTION

Protected areas are global measures applied to protect and sustain biodiversity, as well as to find ways in which human–nature interactions coincide for mutually beneficial outcomes (UNEP-WCMC, IUCN, and NSG, 2020). There is a growing consensus that—in terms of globally increasing human pressure—protected areas enable to a certain extent the short- to midterm conservation of species and natural ecosystems (Chape et al., 2005; Gaston et al., 2008; Geldmann et al., 2019; Ibisch et al., 2016; Joppa & Pfaff, 2011; Mora & Sale, 2011). The development of protected areas is a key approach in the implementation of global agendas for conservation and sustainable development, such as the Aichi targets set by the Convention of Biological Diversity (CBD), the United Nations' Sustainable Development Goals, and national and supranational (e.g., EU-wide) biodiversity strategies. However, most strategies focus on coverage of protected areas as single indicators for the effective protection of biodiversity (Beresford et al., 2018; Chape et al., 2005; Mora & Sale, 2011). It has already been stressed that maintaining as well as establishing new protected areas on a global scale will not be sufficient to sustain biodiversity in the long term (Mora & Sale, 2011). Even after decades of addressing the challenge of establishing and managing protected areas that are fit for purpose, there is sparse information on their effectiveness regarding quantitative outcomes related to biodiversity conservation and ensuring the ongoing functioning of ecosystems (Ghoddousi et al., 2021). The recently published multidimensional framework to assess the socio-ecological effectiveness of protected areas suggests to measure ecological and social outcomes as well as socio-ecological interactions and indicates remote sensing as one promising method to advance these assessments (Ghoddousi et al., 2021). Quantifying effectiveness is crucial when seeking to provide evidence as to whether the designation of protected areas enables positive changes regarding ecosystem services, biodiversity conservation, an increase in resilience, and other factors (e.g., Chape et al., 2005; dos Santos Ribas et al., 2020; Nagendra et al., 2013). Otherwise, there is the risk that the creation of increasingly more protected areas will encourage displacement behavior and obstruct our view of the key question, namely, safeguarding the global ecosystem (Adams et al., 2019; Watson et al., 2014).

The effectiveness approach, based on remote sensing, attempts to measure to what extent protected areas really safeguard biodiversity and measure their ecosystem functions. Studies on the ecological functioning of protected areas, as well as on measures of management effectiveness, are widely available, and yet, studies on global management effectiveness especially highlight a lack of information that integrates conservation and management efficacy objectives (Leverington et al., 2010). In conservation studies, socio-ecological features are extensively used to identify the effective protection of certain areas (Ghoddousi et al., 2021). Building on a socio-ecological perspective, our understanding of protected area effectiveness is based on three factors: (1) mitigating biodiversity and habitat loss, (2) providing ecosystem services for human well-being, and (3) having a sustainable impact on the surroundings of the protected area by being a flagship region. Within our approach, sustaining biodiversity with the help of protected areas is our effectiveness measure, and it is delimited by efficiently managing protected areas.

Protected areas have been investigated from different viewpoints. The global standard employed to categorize protected areas according to their use restrictions has been the basis for different studies and was initially developed by the International Union for Conservation of Nature (IUCN) (World Conservation Union) (Dudley et al., 2013). These IUCN protection categories have been examined on a global scale in terms of effectiveness in halting forest cover loss (Leberger et al., 2020), and they have also been tested for biodiversity variables (Gray et al., 2016). This study, for example, found higher levels of biodiversity inside terrestrial protected areas than outside (Gray et al., 2016). These categories have been included in a proposal to assess conservational effectiveness, in order to "ensure a global approach to assessment, measure biodiversity conservation effectiveness, and incorporate new data layers and more effective application of IUCN protected area management categories" (Chape et al., 2005). A global study on habitat cover and species population revealed limited evidence of how protected areas benefit flora and fauna (Geldmann et al., 2013), while human pressure on protected areas is increasing across the world (Geldmann et al., 2019). However, encouraging results on multipurpose protected areas have been presented, illustrating that they can benefit biodiversity by integrating the protection of species' habitats with sustainable land use while simultaneously reducing CO_2 emissions (Nelson & Chomitz, 2011). In addition to the establishment of protected areas, the conservation of areas with low levels of anthropogenic disturbance, such as roadless areas, is another measure utilized to preserve biodiversity at all levels and to protect relevant ecosystem services (Ibisch et al., 2016). The delivery of ecosystem services has become a major objective for those seeking to study and enhance the effectiveness of protected areas (Friedlander et al., 2007), showcasing the harmonization of values people perceive from nature. However, (spatial) data on protected areas, especially regarding habitat coverage and geopolitical features, are still in short supply (Chape et al., 2005; Gaston et al., 2008), thereby resulting in clear conservation gaps and a lack of priority setting (Chape et al., 2005).

Since the turn of the century, the paradigm of protected area effectiveness has shifted from studies solely focusing on the conservation of biodiversity toward an increasing integration of human well-being (Nagendra et al., 2010; Naughton-Treves et al., 2005; Nelson & Chomitz, 2011). Protected areas should be perceived more as functional, intact ecosystems on a regional scale rather than areas with strict use constraints and hard borders (Knorn et al., 2012; Moreno et al., 2019). This would encourage critics who warn against isolating protected areas too much from one another and who instead advocate for integrated protected landscapes (DeFries et al., 2010; Southworth et al., 2004)—exemplified in the land-sharing/land-sparing debate (Fischer et al., 2014; von Wehrden et al., 2014).

Remote sensing is a tool widely used to assess the condition of protected areas. It is considered an effective, comparatively inexpensive instrument and is less time-consuming than evaluation by ground-truthing, and it can also be used globally (e.g., Tuholske et al., 2017). In protected area research, remote sensing—among other approaches—can be used to identify potential protected areas (Goodell et al., 2018) or to study the connectivity between protected areas (Crochelet et al., 2016). Herein, we investigated remote sensing studies to review how effectiveness is measured on a large scale, and which indicators are suitable for the remote sensing analysis of protected area effectiveness. We focused particularly on how remote sensing is used to assess the socio-ecological effectiveness of protected areas.

The aim of this review paper is to analyze the existing body of scientific literature utilizing remote sensing to investigate the effectiveness of protected areas. The depth of understanding and the use of effectiveness concepts are parts of this investigation. In addition, we examine which indicators are used to assess the socio-ecological effectiveness of protected areas, and we present a first classification of spatial selection methods for remotely sensed effectiveness indicators.

METHODS

In order to identify existing literature on methods used to assess the effectiveness of protected areas, we brought three research areas together: research utilizing remote sensing, the framework of effectiveness, and protected areas as target areas. To cover a wide range of approaches, we foresaw from including specific terms on socio- and/or ecological effectiveness. We rephrased the approach for all databases on the Web of Science search engine by using the following search string: ("remote sensing "OR "satellite imagery "OR geodata OR geoinformation OR geospatial) AND (effectiveness OR impact* OR efficiency) AND ("biosphere reserve* "OR "protec* area*").

The dataset created via Web of Science on 12 February 2020 revealed a list of 520 published academic papers. Of the 520 identified studies, we first selected 285 based on the abstract. We screened the abstracts for relevance and determined if the study was an empirical investigation using any kind of geoinformation to analyze the effectiveness of and/or impacts on protected areas. After examining the full text of the 285 remaining articles, we excluded a further 241 papers, as they did not include any perception of effectiveness, did not include a comparison approach with adjacent areas, or focused on management effectiveness solely. A final set of 44 relevant papers, written in English and Spanish, remained and was downloaded for our evaluation (Appendix S1). They were then read by one reviewer in detail, and relevant variables were extracted (Table 1).

We counted the number of indicators used to assess the effectiveness of protected areas without co-explaining indicators; that is, we did not include co-indicators (e.g., climate change and human impacts) explaining the main indicator (e.g., forest degradation). The types of indicators were categorized into six groups: land use/land cover change (LUCC), forest cover change, vegetation cover change (e.g., vegetation productivity with nonvegetation cover excluded), fire, habitat ecology (animal observations and migration effects), and marine indicators (e.g., species assemblages). The spatial scale was categorized into five categories, and a local scale was assigned if a particularly small area was examined (<5000 km^2). The regional scale included larger areas with either bigger protected areas or various protected areas in one region. Studies were categorized as "continental" when an entire continent was studied. The zonal scale was applied for studies based on global

TABLE 1 Set of variables to categorize paper content and corresponding description

No.	Variable	Description
1	Year	Year of publication
2	Number of indicators	Number of indicators used to assess the effectiveness of protected areas
3	Type of indicators	Type of indicator, research subject
4	Spatial scale	Study scope with 1, local; 2, regional; 3, continental; 4, zonal; 5, global
5	Effectiveness	Study results indicating e (effective protection), i (ineffectiveness timewise or worse inside than outside), and d (divergent, no effect or positive and negative effects)
6	Spatial selection	Area selection approach: Methods used to select areas for comparison inside and outside of the protected area
7	Time scale	Time series analysis or cross-sectional approach
8	Study area	Continent on which the study was conducted
9	Authors origin	Current residence of the main author
10	Ground truth	Yes or no; additional ground truthing as in field research or household surveys

biome borders, that is, temperate forests or the tropics. Additionally, we focused on the design of the studies in terms of the spatial selection of indicators, to assess the effectiveness of protected areas. There is a simple "inside–outside" approach, where specific parameters are compared inside and outside of protected areas (Gray et al., 2016). For example, it can be verified whether a certain type of forest is more affected by the loss of cover outside a protected area than inside—a corresponding outcome would support a certain level of plausibility in terms of protection effectiveness. To ensure the comparability of variables inside and outside of protected areas, a matching methodology (Beresford et al., 2018), whereby data extraction points are sampled randomly inside and outside protected areas with matching similar socio-ecological characteristics (e.g., distance to human settlements or elevation), is becoming more and more prominent in research. A detailed breakdown of applied spatial selection methods is presented in the "Results" section.

One focus of this study was to establish how existing approaches assess effectiveness by comparing protected areas with their surroundings, because the sole assessment of changes over time inside a protected area can provide insights into variabilities inside the area, but it cannot relate to possible changes in surroundings. Against this backdrop, the selected studies, which all use inside–outside comparisons, were categorized in terms of time series analysis or a cross-sectional approach.

RESULTS

The approaches, spatial scales, selected indicators, and declared levels of effectiveness varied widely across the 44 selected studies. Although the Web of Science database covers publications since 1990, the selected studies were all published between 2004 and 2019, with an increase in publications after 2010 and further after 2015, indicating a rather emerging research field. Most of the studies used only a single indicator to assess the effectiveness of protected areas, and the choice of the indicator was independent of the study's spatial scale (Figure 1a). Regarding the spatial scope of the studies, three used area data on global biomes, that is, tropical forests (Lui & Coomes, 2016; Wright et al., 2007) and the temperate biome (Sommerfeld et al., 2018), one of them dealt with the entire African continent (Beresford et al., 2013), and one conducted a global analysis (Tang et al., 2011) (Figure 1a). In the context of scoping, mostly protected area boundaries were used for delineation, some selected national boundaries for regional analyses, and only a few used biome boundaries for global analysis. As for the perception of protected areas' effectiveness, 18 studies stated an improvement in the situation, while a few studies ($n = 7$) detected a certain ineffectiveness. Almost half of the studies ($n = 19$) presented divergent results (Figure 1a).

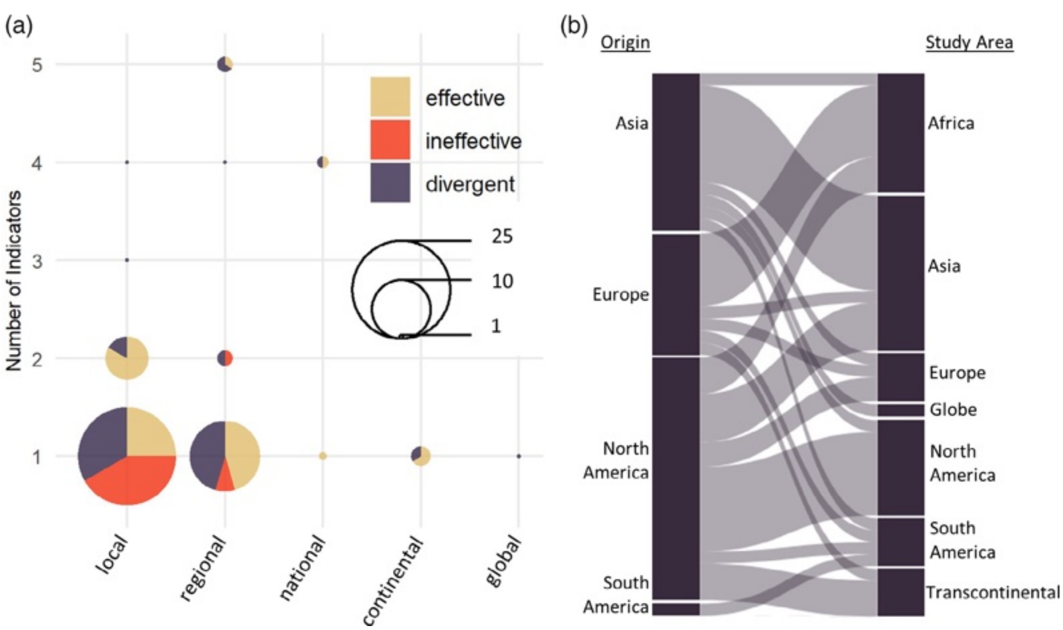


FIGURE 1 (a) An overview of the main characteristics represented across the studies used in this review. The pie chart size represents the number of papers (total $n = 44$). Every pie chart is located according to the study scope (x-axis) and the number of indicators used to assess the effectiveness of protected areas (y-axis). Every pie chart is divided by the number of studies that indicate effective protection (timewise or improvement inside), divergent results (no effect or positive and negative effects), or ineffective protection (timewise or worse inside than outside). (b) The origin of each main author in relation to the examined study area ($n = 44$). The study region "transcontinental" includes studies conducted in the temperate biome, the tropics, and one study conducted in Asia and South America

The majority of studies were conducted on local or regional scales in Asia ($n = 13$), Africa ($n = 10$), North America ($n = 7$), South America ($n = 5$), and Europe ($n = 4$). Studies conducted in Asia and North America were done by indigenous scientists, whereas studies on the African continent were implemented by scientists based in Europe, North America, and Asia (Figure 1b).

Perception and definition of the effectiveness of protected areas

The effectiveness of protected areas was the main focus of studies on rare occasions, and very few studies set out a clear description or definition of how they conceptualized effectiveness (Beresford et al., 2013, 2018; Tang et al., 2011); instead, most of them defined the effectiveness of protected areas as the ability to sustain biodiversity, mostly in terms of selected species (Bonilla-Mejia & Higuera-Mendieta, 2019; Friedlander et al., 2007; Joseph et al., 2009; Kintz et al., 2006; Knorn et al., 2012; Lui & Coomes, 2016; Moreno et al., 2019; Rioja-Nieto et al., 2015; Schulte to Buehne et al., 2017; Xie et al., 2012; Zhang et al., 2017). One of the included papers, Tang et al. (2011), described the effectiveness of protected areas as the ability to "[maintain] ecological functioning" and divided ecological effectiveness into two subfunctions: one part concentrating on existing biodiversity features in protected areas, the other part asking "how well [do] these areas maintain the biodiversity features[?]" (Tang et al., 2011). Beresford et al. (2013), however, defined the effectiveness of protected areas as the ability to "[reduce] deleterious land cover change" and "[prevent] conversion of all habitats in sites of high importance" (Beresford et al., 2013). The follow-up study by Beresford et al. (2018) declared effectiveness in terms of protecting natural habitats, specifically referring to "the degree to which that threat is reduced by designation" (Beresford et al., 2018).

Studies with a main focus on habitat protection assessed the effectiveness of these protected areas among other aims, such as preventing habitat destruction and protecting endangered wildlife (Clerici et al., 2007; Foo & Numata, 2019; Huang et al., 2019; Merkohasanaj et al., 2019; Onojeghuo & Onojeghuo, 2015; Simons-Legaard et al., 2018; Sunderland-Groves et al., 2011; Xie et al., 2012).

Many studies defined protection effectiveness through protecting forest cover (Bonilla-Mejia & Higuera-Mendieta, 2019; Foo & Numata, 2019; Gaveau et al., 2012; Nagendra et al., 2004; Phua et al., 2008; Ren et al., 2015; Wright et al., 2007), or more specifically in terms of protection efficacy by preventing forest fires (Manaswini & Reddy, 2015) or mitigating tree cover loss (Bragina et al., 2015). Others specified the effectiveness of forested protected areas regarding deforestation or even—more specifically—forest degradation (Htun et al., 2009; Southworth et al., 2004).

Overall, 18 studies declared the good effectiveness of their investigated protected areas, always with their individual definition of effectiveness, while seven highlighted the need to improve in this regard. A total of 19 studies presented a mixed picture, highlighting both positive and negative effects. In studies focusing on forest protection, the effective protection of forest cover inside protected areas was identified, but strong pressure on adjacent areas limited any positive effects (Blackman et al., 2015; Clerici et al., 2007; Lui & Coomes, 2016; Nagendra et al., 2009; Xie et al., 2012).

In terms of different approaches on protected area effectiveness, we found that nine of the 44 studies complied with our proposed definition of effectiveness (Figure 2, Group A). In 18 studies, we found two of the three factors (Figure 2, Groups B and C) and 17 studies covered mainly the mitigation of biodiversity loss as definition of protected area effectiveness (Figure 2, Group D). Explicit social aspects (defined as ecosystem services for human well-being) were found in 16 studies (Figure 2, Groups A and C).

Spatial selection of remotely sensed indicators

Different methodological approaches utilized to extract and compare indicators were identified. Twenty-eight studies used either an "inside–outside" or an "inside–outside–buffer" approach (Figures 3a,b). The other main spatial selection approaches used either the protected area as land cover for comparison ($n = 9$; Figure 3c) or a matching method to compare features inside and outside of protected areas ($n = 7$; Figure 3d).

The inside–outside approach (Figure 3a) was mostly just applied in a way that areas inside and outside of protected areas were compared (Bonilla-Mejia & Higuera-Mendieta, 2019; Friedlander et al., 2007; Gaveau et al., 2012; Huang et al., 2019; Leisher et al., 2011; Rioja-Nieto et al., 2015; Sommerfeld et al., 2018), while others compared "outside" areas in the direct periphery of the protected areas (Clerici et al., 2007; Foo & Numata, 2019; Mtui et al., 2017; Nagendra et al., 2009; Sunderland-Groves et al., 2011).

The second large group of studies ($n = 14$; Figure 3b) used an inside–outside–buffer approach, mostly involving separating areas inside and outside of protected areas with a buffer zone in between, to take into account potential edge effects (Fuda et al., 2016; Htun et al., 2009; Lui & Coomes, 2016; Onojeghuo & Onojeghuo, 2015; Phua et al., 2008; Schulte to Buehne et al., 2017; Simons-Legaard et al., 2018; Southworth et al., 2004; Tang et al., 2011; Zhang et al., 2017). Others added continuous buffer zones around

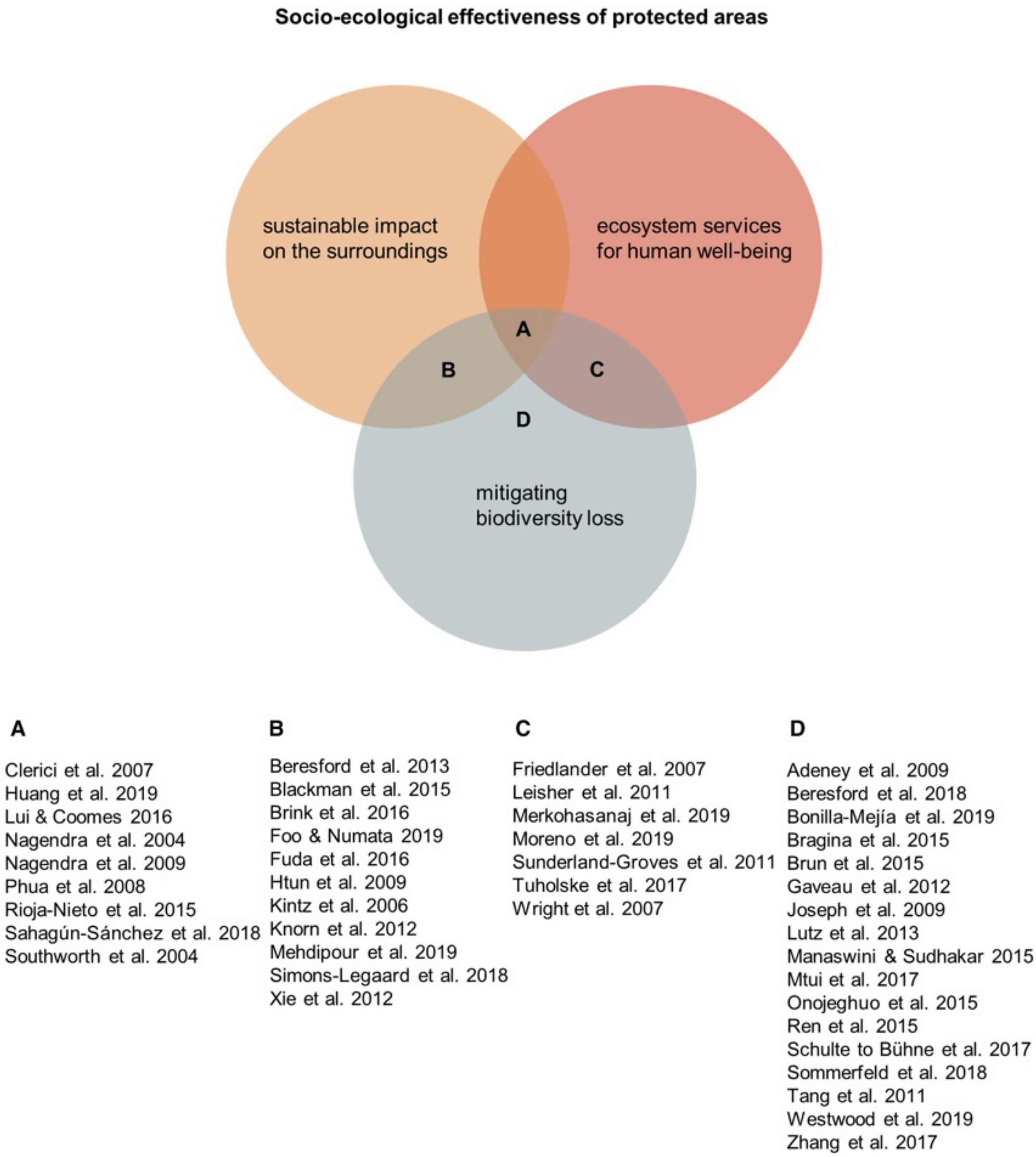


FIGURE 2 Integration of the investigated studies into our perception of socio-ecological effectiveness of protected areas. "A" signifies that the corresponding studies indicated all the factors (mitigating biodiversity loss, ecosystem functions for human well-being, and sustainable impact on the surroundings) in their perception of protected area effectiveness. "B" includes studies that defined effectiveness as mitigating biodiversity loss and having a sustainable impact on the surroundings. "C" lists studies that defined effectiveness as mitigating biodiversity loss and providing ecosystem services for human well-being. "D" includes studies that measured protected area effectiveness investigating the mitigation of biodiversity loss

core zones or the outer border of the protected area (Knorn et al., 2012; Nagendra et al., 2004; Wright et al., 2007). Random sampling integrated into an inside–outside–buffer research framework was noted in two articles (Merkohasanaj et al., 2019; Westwood et al., 2019).

Besides the explicit study of protected areas, several studies defined protected areas as one aspect of land cover and considered whole ecosystems or ecological regimes (Figure 3c). Studies on deforestation (Brun et al., 2015), fire patterns causing deforestation (Adeney et al., 2009; Manaswini & Reddy, 2015), land cover change (Kintz et al., 2006; Mehdipour et al., 2019; Sahagún-Sánchez & Reyes-Hernández, 2018; Tuholske et al., 2017; Xie et al., 2012), and changes in timberline elevation (Lutz et al., 2013) examined whether or not areas under protection suffered less from these threats.

The rather recently proposed matching method (Figures 3d and 4) is complex, and it varies in line with

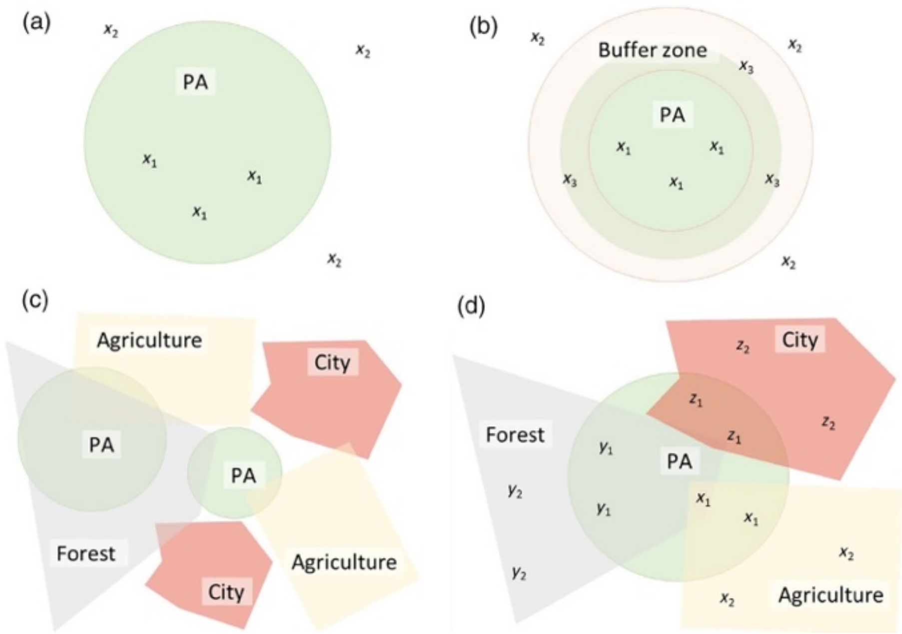


FIGURE 3 Schematic diagram of methodological approaches used to spatially extract and compare indicators, in order to assess the effectiveness of protected areas (PA). (a) Inside–outside; (b) inside–outside and buffer zone; (c) protected area as land cover class; (d) matching indicators inside and outside, based on similar characteristics

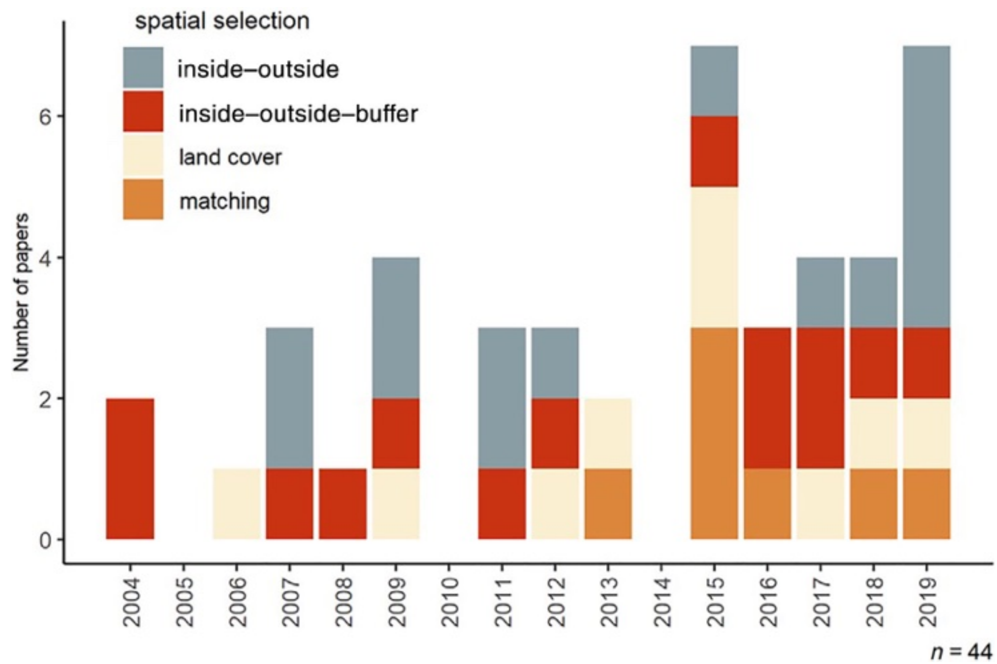


FIGURE 4 The use of different spatial selection methods in selected papers over time. Bar colors indicate the different spatial selection methods for comparing indicators "inside–outside," "inside–outside–buffer" (including a buffer area between the protected area and its surroundings), "land cover" (protected area [PA] as land cover class), and "matching" (random sampling inside and outside *with* similar characteristics)

the focus of the respective study. The selection of similar characteristics to match points inside and outside of protected areas was adapted to the subject of interest: In studies on land cover change, characteristics that might have an impact on land cover change were used to select suitable comparison points (Beresford et al., 2013, 2018).

The same idea, using characteristics possibly influencing the study subject, was used in other studies for habitat types (Brink et al., 2016) and forest canopy cover (Blackman et al., 2015; Bragina et al., 2015; Moreno et al., 2019; Ren et al., 2015).

For the spatial selection of indicators, Landsat derivatives ($n = 18$) were primarily used for remote sensing analysis. MODIS data were analyzed in five studies only, while other studies combined MODIS and Landsat, Landsat and ASTER, or others. Forty-one of the 44 selected studies generated time series for change detection over time. More than half of the studies used any type of inside–outside or inside–outside–buffer approach, followed by a method that defined protected areas through land cover class. The use of the matching approach to select data points randomly occurred first in 2013 in the set of selected articles (Figure 4).

Indicators used to assess the effectiveness of protected areas

Landscape indicators were used in 42 studies, whereas two studies used seascape indicators. The primary focus of the studies was related to LUCC ($n = 23$) analysis, with vegetation or explicit forest cover change ($n = 17$) as the main indicators to assess the effectiveness of protected areas. Additional indicators for the remote sensing on which this review focuses were, for example, avian point count surveys (Westwood et al., 2019), chimpanzee observations (Sunderland-Groves et al., 2011), and household surveys (Leisher et al., 2011).

The overall procedure implemented in studies using LUCC as an indicator of effectiveness involved developing a land use classification via satellite data analysis and the subsequent detection of change over time (Beresford et al., 2013), some with a detailed description of their methods (e.g., Schulte to Buehne et al., 2017). LUCC was used to detect changes to and pressure on wildlife protected areas (Brink et al., 2016; Clerici et al., 2007; Mtui et al., 2017), vegetation cover (Huang et al., 2019), forest cover and forest fragmentation (Nagendra et al., 2009; Rioja-Nieto et al., 2015; Simons-Legaard et al., 2018; Sommerfeld et al., 2018; Southworth et al., 2004), forest leakage (Foo & Numata, 2019; Lui & Coomes, 2016), and landscape fragmentation (Mehdipour et al., 2019; Xie et al., 2012) and to identify loss of natural habitats (Beresford et al., 2018). LUCC was also used to assess the impact of anthropogenic pressure (such as urbanization, agriculture, and tourism) and thereby evaluate the vulnerability and ability of the protected area to reduce pressure on biodiversity (Sahagún-Sánchez & Reyes-Hernández, 2018; Schulte to Buehne et al., 2017; Tuholske et al., 2017; Zhang et al., 2017).

Apart from LUCC, many studies that focused on the effectiveness of forest protected areas used the following indicators: the forest loss/deforestation rate (Bragina et al., 2015; Gaveau et al., 2012; Onojeghuo & Onojeghuo, 2015; Phua et al., 2008; Ren et al., 2015), the disturbance rate (Knorn et al., 2012), both (Htun et al., 2009), fire (Adeney et al., 2009; Manaswini & Reddy, 2015; Wright et al., 2007), or the deforestation rate linked with additional variates (Blackman et al., 2015; Bonilla-Mejia & Higuera-Mendieta, 2019). Indicators such as the normalized difference vegetation index (NDVI) or net primary productivity were used to examine effectiveness (Fuda et al., 2016; Joseph et al., 2009; Leisher et al., 2011; Moreno et al., 2019; Simons-Legaard et al., 2018; Tang et al., 2011). Outstanding approaches used changes over time to the mountainous timberline as an indicator (Lutz et al., 2013) or species distribution models (Sunderland-Groves et al., 2011; Westwood et al., 2019). Indicators for marine protected areas were species habitat distribution models using topographical characteristics and assemblages of animals (Friedlander et al., 2007; Merkohasanaj et al., 2019).

DISCUSSION

Our review identified an increasing research interest in the effectiveness of protected areas from a remote sensing perspective. There is not currently a consistent framework for the effectiveness of protected areas or an explicit indicator system that can be employed to assess socio-ecological effectiveness. Therefore, we recommend two main steps toward establishing such a framework: first, the development of a consistent indicator system that explicitly addresses the socio-ecological effectiveness of protected areas for an evaluation of protection efforts to date and in the future, and second, the further elaboration of the matching method for the spatial selection of indicators for a broad application on local and global scales.

Remotely sensed indicators to assess socio-ecological effectiveness

With regard to commonly used terms, the handling of socio-ecological effectiveness is fitted to specific areas of research without searching for an overarching or a consistent use of terms. There is usually no critical reflection on the fact that a comparison of socio-ecological characteristics inside and outside protected areas is not a consistent measurement being that measured differences are indicators of protection (Tang et al., 2011) or effectiveness. However, there seems to be agreement that safeguarding biodiversity is the main concern of protecting landscapes and seascapes (Knorn et al., 2012; Moreno et al., 2019; Schulte to Buehne et al., 2017; Zhang et al., 2017) (Figure 2). Unfortunately, though, a common definition of the concepts of biodiversity is not available either. Apparently, a couple of studies referred to categories of protection following the IUCN and reached different conclusions regarding the effectiveness of protection strictness. A higher level of protection could prevent more anthropogenic pressure (Schulte to Buehne et al., 2017), and yet, strictly protected areas are not necessarily more protective than less restricted protected areas (Ferraro et al., 2013). Roughly 20% of the investigated studies complied with our proposed definition of socio-ecological effectiveness. The majority (41%) covered two of three factors. We thereby acknowledge that the concepts of socio-ecological effectiveness shift toward more holistic approaches (e.g., Ghoddousi et al., 2021). Although several studies defined protected area effectiveness as being a composite of socio- and ecological factors, only few translated this definition explicitly into a corresponding set of socio-ecological indicators (e.g., Huang et al., 2019; Nagendra et al., 2009). As a result of the investigated diverse range of existing definitions, we propose our introduced understanding of protected area effectiveness as the effectiveness of (1) mitigating biodiversity and habitat loss, (2) providing ecosystem services for human well-being, and (3) having a sustainable impact on the surroundings of the protected area by being a flagship region. Equally, this definition calls for subsequent research to define remotely sensed indicators depicting the socio-ecological effectiveness of protected areas (e.g., Ghoddousi et al., 2021; Pettorelli et al., 2018).

Although there is a significant amount of variety in terms of remotely sensed indicators, there is no systematic framework or harmonization of relevant indicators; most studies focused on one indicator to assess the effectiveness of areas, especially in regard to forested regions. Nevertheless, they still included indicators such as fire and distance to roads, fire, and climate phenomena (Adeney et al., 2009), or logging, transport costs, and protected area status (Brun et al., 2015). With a main focus on biodiversity, most approaches studied features such as species composition, fragmentation rate, or LUCC, while only a few used indirect proxies of ecosystem functioning, such as the NDVI (e.g., Htun et al., 2009; Huang et al., 2019; Tang et al., 2011). Studies using forest cover change as a proxy for protected area effectiveness did not reflect that this proxy cannot be applied for nonforested protected areas and therefore cannot be treated as a general proxy for protected areas effectiveness (Geldmann et al., 2019).

Since only three studies of the selected papers mentioned climate change as an influencing factor and used climate variables as indicators, it should be discussed how—and whether—climate variables should not always and generally be taken into account (Huang et al., 2019; Westwood et al., 2019). In the context of climate change, more holistic approaches that include long-term perspectives and vulnerability to climate change would change the view on the effectiveness of protected areas (Freudenberger et al., 2013; Huang et al., 2019). Another potential indicator could be the effectiveness of protected areas in carbon sequestration (Gaveau et al., 2012) or meso- and microclimatic buffering.

Social aspects of socio-ecological effectiveness

Apart from the general consensus that protected areas are a necessary instrument to counteract the loss of biodiversity and to preserve natural habitats, the issue of environmental justice is neglected in many cases. We consider environmental justice as a key factor in investigating the socio-ecological effectiveness, appropriateness, and long-term success of protected areas. One statement to this effect was made in the literature under consideration: "Often parks are created in areas where poor people depend upon the natural resources for their livelihoods; thus, exclusionary management raises issues of social justice and equity" (Bates & Rudel, 2000; Brockington, 2002, cited in Nagendra et al., 2004).

There are (global) studies on protected areas and their relation to human well-being and social justice (Oldekop et al., 2016; Zafra-Calvo & Geldmann, 2020) that should also be considered. Also, 36% of the investigated articles included aspects of ecosystem services for human well-being into their assessment of protected area effectiveness (Figure 2, Groups A and C), which shows a broadened scope in the field. In this context, UNESCO Biosphere reserves, established for exploring new pathways toward sustainability and directly addressing ecosystem-based strategies to safeguard human well-being, deserve special attention. Biosphere reserves could also be expected to pioneer the implementation and promotion of socially just nature conservation.

While our review design focused on remote sensing approaches, most of the 44 selected studies advocated the inclusion of social science approaches that incorporate the results of studies with the local population, and they cited local participation as one of the most important solutions to increase effectiveness, in addition to expanding the network of protected areas. Occasionally, researchers referred to earlier approaches of effectiveness research and mostly designed their own method to assess this vector, which made it difficult to compare different individual case studies; correspondingly, the findings also support the idea of developing a framework for evaluating effectiveness (Rasheed, 2020; Stephenson et al., 2015). To explore the socio-ecological effectiveness of protected areas and their impact on habitat and species loss, ecosystem services and protection capacity (Beresford et al., 2018)—a more holistic and multi-indicator-based approach—should be considered (e.g., Ghoddousi et al., 2021). The effect of selecting the variables proves relevant, as it influences the results substantially (Beresford et al., 2018; Vanclay, 2001). To select relevant variables, existing studies on essential biodiversity variables (Pereira et al., 2013), biophysical variables (Richter et al., 2012), climate variables (Bojinski et al., 2014), and geodiversity variables (Schrodt et al., 2019) offer a basis on which to build.

Selection methods

The analysis of selection methods within the investigated studies showed that our focus on remote sensing as an explicit method to examine protected area effectiveness helped identify patterns in their different approaches. To estimate the socio-ecological effectiveness of protected areas, indicator values of unprotected and protected areas are generally compared. However, the selection of comparable areas for comparison is rather difficult, albeit it is still highly important. A common selection bias is that protected areas are often established in biodiversity hotspots based on special landscape features, and they are compared with their less diverse surroundings without balancing out this disparity (Beresford et al., 2018; Moreno et al., 2019; Tang et al., 2011). This review represents parts of the debate on the best selection approach, that is, between the inside–outside and matching approaches. Earlier and mostly rather local-scale case studies, using the inside–outside analysis approach with either hard edge borders or an *inside:buffer zone:remote area* ratio, justified the selection of areas directly adjacent to protected areas as having possibly similar biophysical conditions while being aware of a possible bias produced by heterogeneous landscapes (Mtui et al., 2017). However, recent studies using a matching method justified the approach with the bias, because protected areas are not randomly distributed, and most of the time they are located in areas that are either less accessible or of a lower land use value than their surroundings (e.g., Bonilla-Mejia & Higuera-Mendieta, 2019) (Figure 4). Furthermore, to date, there is no common ground on whether designation borders should be considered hard edges or transition areas. Some studies use adjacent areas for comparison (Adeney et al., 2009), while others explicitly exclude them in order to diminish the possible effects of the protected area on them (e.g., "deforestation banned in the reserve spills over to just outside the reserve [...] or isolation spillovers") (Ren et al., 2015).

This review focused on remote sensing approaches, which proved to be successful methods of investigation in all of the selected studies. Nevertheless, the efficacy of this approach is strongly related to the availability and quality of ground-truthing data, not only to test land cover classification, but also for socio-ecological variables and local characteristics. One half of the studies included fieldwork, with some making a clear argument for integrating fieldwork, clearly advocating for ground truthing to clarify anthropogenic activities (Lui & Coomes, 2016; Nagendra et al., 2009). The need to combine local and global monitoring by integrating fieldwork and remote sensing datasets for significant effectiveness statements is clearly visible (Stephenson et al., 2015). However, it has to be proven if local effects of protected areas are visible on a larger scale (Mora & Sale, 2011).

Aside from the spatial scale, temporal dimensions are equally important. The majority of the examined studies used some forms of time series analysis to detect changes effectively over time. Approaches using time series analysis of randomly sampled matching points inside and outside of protected areas are becoming more commonplace (Beresford et al., 2018; Gray et al., 2016). It is a challenging task, though, as multiple factors such as protected area designation, political changes, conflicts, historical context, or population growth have to be considered and are subject to change. Additionally, protected areas are a quite recent phenomenon, and the analysis of effectiveness will be better understood in the future (Moreno et al., 2019).

Prospects

Remote sensing offers tremendous opportunities to examine the land and sea surfaces on various spatial and temporal scales. By taking advantage of new methods and computation technologies such as machine learning and big data analysis, remote sensing can be considered a promising approach to assess the socio-ecological effectiveness of protected areas. A semiautomatic tool for the global assessment of protected area effectiveness, based on a set of appropriate socio-ecological indicators, would help in terms of comparison and support the monitoring and evaluation of global strategies and conventions such as the CBD and the Man and the Biosphere Program, as well as national and supranational biodiversity strategies. However, several questions remain: What indicators should be used to assess the effectiveness of protected areas? Which indicators fit best in terms of region and protection status? And should specific and hard-to-measure indicators such as the genetic diversity of vegetation, the diversity of ecological processes, and the function or the degree of environmental justice be included in assessments?

This study and its results are limited to a certain extent. First, data acquisition was limited to databases listed on the Web of Science. An overview of existing studies, such as the present research, especially if they use large datasets, not only collects information on a topic, but also duplicates uncertainties in the results. The field of protected area effectiveness, and the corresponding spectrum of definitions and theories, is quite a recent research field (this paper's dataset ranges from publishing dates 2004–2019), so research design, definitions, findings, and framework development are in their early stage.

CONCLUSIONS

This paper demonstrates that a globally standardized framework for using remote sensing to assess the socio-ecological effectiveness of protected areas does not currently exist. Several researchers advocate for more holistic approaches on global effectiveness research, which is pending. A global analysis of the effectiveness of protected areas will be a challenging task, mostly due to the demands involved in finding overall matching indicators and reliable data on a global basis. To date, there is not a complete toolset, which incorporates all relevant indicators to assess the effectiveness of protected areas. It is also a question of time and scale to be able to produce global maps of the effectiveness of protected areas while at the same time taking into account local characteristics such as historical context and local and regional environmental management. A global socio-environmental monitoring tool in this regard would not only provide information on the current state of conservation effectiveness around the world, but it would also be valuable to policymakers, stakeholders, and communities seeking to promote sustainable development as a key objective.

ACKNOWLEDGMENTS

This study was funded by Biosphere Reserves Institute and the Innovation and Career Center "ProBio-LaB" by the Ministry of Science, Research and Culture of the federal state of Brandenburg. Pierre L. Ibisch holds a Research Professorship for Ecosystem-based Sustainable Development by Eberswalde University for Sustainable Development. We thank the anonymous reviewers for their valuable contributions.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

AUTHOR CONTRIBUTIONS

Charlotte Gohr and Pierre L. Ibisch conceived the study, and Henrik von Wehrden contributed to the methodological development. Charlotte Gohr wrote the original draft and conducted the analyses. All authors were involved in reviewing and revising the draft versions.

DATA AVAILABILITY STATEMENT

Data (Gohr, 2022) are available from Figshare: https://doi.org/10.6084/m9.figshare.17198771.

ORCID

Charlotte Gohr https://orcid.org/0000-0002-0459-0939
Henrik von Wehrden https://orcid.org/0000-0003-2087-5552
Felix May https://orcid.org/0000-0002-1106-8188
Pierre L. Ibisch https://orcid.org/0000-0001-9820-9272

REFERENCES

Adams, V. M., D. I. Gwenllian, and H. P. Possingham. 2019. "Weighing the Benefits of Expanding Protected Areas Versus Managing Existing Ones." *Nature Sustainability* 2(5): 404–11. https://doi.org/10.1038/s41893-019-0275-5.

Adeney, J. M., N. L. Christensen, and S. L. Pimm. 2009. "Reserves Protect against Deforestation Fires in the Amazon." *PLoS One* 4(4): e5014. https://doi.org/10.1371/journal.pone.0005014.

Beresford, A. E., G. M. Buchanan, B. Phalan, G. W. Eshiamwata, A. Balmford, A. B. Brink, L. D. C. Fishpool, and P. F. Donald. 2018. "Correlates of Long-Term Land-Cover Change and Protected Area Performance at Priority Conservation Sites in Africa." *Environmental Conservation* 45(1): 49–57. https://doi.org/10.1017/S0376892917000157.

Beresford, A. E., G. W. Eshiamwata, P. F. Donald, A. Balmford, B. Bertzky, A. B. Brink, L. D. C. Fishpool, et al. 2013. "Protection Reduces Loss of Natural Land-Cover at Sites of Conservation Importance across Africa." *PLoS One* 8(5): e65370. https://doi.org/10.1371/journal.pone.0065370.

Blackman, A., A. Pfaff, and J. Robalino. 2015. "Paper Park Performance: Mexico's Natural Protected Areas in the 1990s." *Global Environmental Change-Human and Policy Dimensions* 31(3): 50–61. https://doi.org/10.1016/j.gloenvcha.2014.12.004.

Bojinski, S., M. Verstraete, T. C. Peterson, C. Richter, A. Simmons, and M. Zemp. 2014. "The Concept of Essential Climate Variables in Support of Climate Research, Applications, and Policy." *Bulletin of the American Meteorological Society* 95(9): 1431–43. https://doi.org/10.1175/BAMS-D-13-00047.1.

Bonilla-Mejia, L., and I. Higuera-Mendieta. 2019. "Protected Areas under Weak Institutions: Evidence from Colombia." *World Development* 122(10): 585–96. https://doi.org/10.1016/j.worlddev.2019.06.019.

Bragina, E. V., V. C. Radeloff, M. Baumann, K. Wendland, T. Kuemmerle, and A. M. Pidgeon. 2015. "Effectiveness of Protected Areas in the Western Caucasus before and after the Transition to Post-Socialism." *Biological Conservation* 184(4): 456–64. https://doi.org/10.1016/j.biocon.2015.02.013.

Brink, A. B., J. Martinez-Lopez, Z. Szantoi, P. Moreno-Atencia, A. Lupi, L. Bastin, and G. Dubois. 2016. "Indicators for Assessing

Habitat Values and Pressures for Protected Areas—An Integrated Habitat and Land Cover Change Approach for the Udzungwa Mountains National Park in Tanzania." *Remote Sensing* 8(10): 862. https://doi.org/10.3390/rs8100862.

Brun, C., A. R. Cook, J. S. H. Lee, S. A. Wich, L. P. Koh, and L. R. Carrasco. 2015. "Analysis of Deforestation and Protected Area Effectiveness in Indonesia: A Comparison of Bayesian Spatial Models." *Global Environmental Change-Human and Policy Dimensions* 31(3): 285–95. https://doi.org/10.1016/j.gloenvcha.2015.02.004.

Chape, S., J. Harrison, M. Spalding, and I. Lysenko. 2005. "Measuring the Extent and Effectiveness of Protected Areas as an Indicator for Meeting Global Biodiversity Targets." *Philosophical Transactions of the Royal Society B: Biological Sciences* 360(1454): 443–55.

Clerici, N., A. Bodini, H. Eva, J.-M. Grégoire, D. Dulieu, and C. Paolini. 2007. "Increased Isolation of Two Biosphere Reserves and Surrounding Protected Areas (WAP Ecological Complex, West Africa)." *Journal for Nature Conservation* 15(1): 26–40. https://doi.org/10.1016/j.jnc.2006.08.003.

Crochelet, E., J. Roberts, E. Lagabrielle, D. Obura, M. Petit, and P. Chabanet. 2016. "A Model-Based Assessment of Reef Larvae Dispersal in the Western Indian Ocean Reveals Regional Connectivity Patterns—Potential Implications for Conservation Policies." *Regional Studies in Marine Science* 7(9): 159–67. https://doi.org/10.1016/j.rsma.2016.06.007.

DeFries, R., K. K. Karanth, and S. Pareeth. 2010. "Interactions between Protected Areas and Their Surroundings in Human-Dominated Tropical Landscapes." *Biological Conservation* 143(12): 2870–80. https://doi.org/10.1016/j.biocon.2010.02.010.

dos Santos Ribas, L. G., R. L. Pressey, R. Loyola, and L. M. Bini. 2020. "A Global Comparative Analysis of Impact Evaluation Methods in Estimating the Effectiveness of Protected Areas." *Biological Conservation* 246: 108595.

Dudley, N., P. Shadie, and S. Stolton. 2013. *Guidelines for Applying Protected Area Management Categories Including IUCN WCPA Best Practice Guidance on Recognising Protected Areas and Assigning Management Categories and Governance Types. Vol. 21*. Best Practice Protected Area Guidelines Series. Gland: IUCN.

Ferraro, P. J., M. M. Hanauer, D. A. Miteva, G. J. Canavire-Bacarreza, S. K. Pattanayak, and K. R. E. Sims. 2013. "More Strictly Protected Areas Are Not Necessarily More Protective: Evidence from Bolivia, Costa Rica, Indonesia, and Thailand." *Environmental Research Letters* 8(2): 025011. https://doi.org/10.1088/1748-9326/8/2/025011.

Fischer, J., D. J. Abson, V. Butsic, M. J. Chappell, J. Ekroos, J. Hanspach, T. Kuemmerle, H. G. Smith, and H. von Wehrden. 2014. "Land Sparing Versus Land Sharing: Moving Forward." *Conservation Letters* 7(3): 149–57. https://doi.org/10.1111/conl.12084.

Foo, Y. S., and S. Numata. 2019. "Deforestation and Forest Fragmentation in and around Endau-Rompin National Park, Peninsular Malaysia." *Tropics* 28(2): 23–37. https://doi.org/10.3759/tropics.MS18-16.

Freudenberger, L., P. Hobson, M. Schluck, S. Kreft, K. Vohland, H. Sommer, S. Reichle, C. Nowicki, W. Barthlott, and P. L. Ibisch. 2013. "Nature Conservation: Priority-Setting Needs a Global Change." *Biodiversity and Conservation* 22(5): 1255–81. https://doi.org/10.1007/s10531-012-0428-6.

Friedlander, A. M., E. K. Brown, and M. E. Monaco. 2007. "Coupling Ecology and GIS to Evaluate Efficacy of Marine Protected Areas in Hawaii." *Ecological Applications* 17(3): 715–30. https://doi.org/10.1890/06-0536.

Fuda, R. K., S. J. Ryan, J. B. Cohen, J. Hartter, and J. L. Frair. 2016. "Assessing Impacts to Primary Productivity at the Park Edge in Murchison Falls Conservation Area, Uganda." *Ecosphere* 7(10): e01486. https://doi.org/10.1002/ecs2.1486.

Gaston, K. J., S. F. Jackson, L. Cantu-Salazar, and G. Cruz-Piñón. 2008. "The Ecological Performance of Protected Areas." *Annual Review of Ecology, Evolution, and Systematics* 39: 93–113. https://doi.org/10.1146/annurev.ecolsys.39.110707.173529.

Gaveau, D. L. A., L. M. Curran, G. D. Paoli, K. M. Carlson, P. Wells, A. Besse-Rimba, D. Ratnasari, and N. Leader-Williams. 2012. "Examining Protected Area Effectiveness in Sumatra: Importance of Regulations Governing Unprotected Lands." *Conservation Letters* 5(2): 142–8. https://doi.org/10.1111/j.1755-263X.2011.00220.x.

Geldmann, J., M. Barnes, L. Coad, I. D. Craigie, M. Hockings, and N. D. Burgess. 2013. "Effectiveness of Terrestrial Protected Areas in Reducing Habitat Loss and Population Declines." *Biological Conservation* 161: 230–8. https://doi.org/10.1016/j.biocon.2013.02.018.

Geldmann, J., A. Manica, N. D. Burgess, L. Coad, and A. Balmford. 2019. "A Global-Level Assessment of the Effectiveness of Protected Areas at Resisting Anthropogenic Pressures." *Proceedings of the National Academy of Sciences of the United States of America* 116(46): 23209–15. https://doi.org/10.17863/CAM.44811.

Ghoddousi, A., J. Loos, and T. Kuemmerle. 2021. "An Outcome-Oriented, Social–Ecological Framework for Assessing Protected Area Effectiveness." *Bioscience* 114(11): 201–12. https://doi.org/10.1093/biosci/biab114.

Gohr, C. 2022. "Dataset_ProtectedAreaEffectiveness_ArticlesReview." Figshare. Dataset. https://doi.org/10.6084/m9.figshare.17198771.v1.

Goodell, W., K. A. Stamoulis, and A. M. Friedlander. 2018. "Coupling Remote Sensing with in Situ Surveys to Determine Reef Fish Habitat Associations for the Design of Marine Protected Areas." *Marine Ecology Progress Series* 588(2): 121–34. https://doi.org/10.3354/meps12388.

Gray, C. L., S. L. L. Hill, T. Newbold, L. N. Hudson, L. Börger, S. Contu, A. J. Hoskins, S. Ferrier, A. Purvis, and J. P. W. Scharlemann. 2016. "Local Biodiversity Is Higher inside than outside Terrestrial Protected Areas Worldwide." *Nature Communications* 7: 12306.

Htun, N. Z., N. Mizoue, T. Kajisa, and S. Yoshida. 2009. "Deforestation and Forest Degradation as Measures of Popa Mountain Park (Myanmar) Effectiveness." *Environmental Conservation* 36(3): 218–24. https://doi.org/10.1017/S0376892909990415.

Huang, L., Q. Shao, and J. Liu. 2019. "Assessing the Conservation Effects of Nature Reserve Networks under Climate Variability over the Northeastern Tibetan Plateau." *Ecological Indicators* 96(1): 163–73. https://doi.org/10.1016/j.ecolind.2018.08.034.

Ibisch, P. L., M. T. Hoffmann, S. Kreft, G. Pe'er, V. Kati, L. Biber-Freudenberger, D. A. DellaSala, M. M. Vale, P. R. Hobson, and N. Selva. 2016. "A Global Map of Roadless Areas and Their Conservation Status." *Science* 354(6318): 1423–7. https://doi.org/10.1126/science.aaf7166.

Joppa, L. N., and A. Pfaff. 2011. "Global Protected Area Impacts." *Proceedings of the Royal Society B: Biological Sciences* 278(1712): 1633–8. https://doi.org/10.1098/rspb.2010.1713.

Joseph, S., G. A. Blackburn, B. Gharai, S. Sudhakar, A. P. Thomas, and M. S. R. Murthy. 2009. "Monitoring Conservation Effectiveness in a Global Biodiversity Hotspot: The Contribution of Land Cover Change Assessment." *Environmental Monitoring and Assessment* 158(1–4): 169–79. https://doi.org/10.1007/s10661-008-0571-4.

Kintz, D. B., K. R. Young, and K. A. Crews-Meyer. 2006. "Implications of Land Use/Land Cover Change in the Buffer Zone of a National Park in the Tropical Andes." *Environmental Management* 38(2): 238–52. https://doi.org/10.1007/s00267-005-0147-9.

Knorn, J., T. Kuemmerle, V. C. Radeloff, A. Szabo, M. Mindrescu, W. S. Keeton, I. Abrudan, P. Griffiths, V. Gancz, and P. Hostert. 2012. "Forest Restitution and Protected Area Effectiveness in Post-Socialist Romania." *Biological Conservation* 146(1): 204–12. https://doi.org/10.1016/j.biocon.2011.12.020.

Leberger, R., I. M. D. Rosa, C. A. Guerra, F. Wolf, and H. M. Pereira. 2020. "Global Patterns of Forest Loss across IUCN Categories of Protected Areas." *Biological Conservation* 241: 108299. https://doi.org/10.1016/j.biocon.2019.108299.

Leisher, C., R. Brouwer, T. M. Boucher, R. Vogelij, W. R. Bainbridge, and M. Sanjayan. 2011. "Striking a Balance: Socioeconomic Development and Conservation in Grassland through Community-Based Zoning." *PLoS One* 6(12): e28807. https://doi.org/10.1371/journal.pone.0028807.

Leverington, F., K. Lemos Costa, H. Pavese, A. Lisle, and M. Hockings. 2010. "A Global Analysis of Protected Area Management Effectiveness." *Environmental Management* 46(5): 685–98. https://doi.org/10.1007/s00267-010-9564-5.

Lui, G. V., and D. A. Coomes. 2016. "Tropical Nature Reserves Are Losing Their Buffer Zones, but Leakage Is Not to Blame." *Environmental Research* 147(May): 580–9. https://doi.org/10.1016/j.envres.2015.11.008.

Lutz, D. A., R. L. Powell, and M. R. Silman. 2013. "Four Decades of Andean Timberline Migration and Implications for Biodiversity Loss with Climate Change." *PLoS One* 8(9): e74496. https://doi.org/10.1371/journal.pone.0074496.

Manaswini, G., and C. S. Reddy. 2015. "Geospatial Monitoring and Prioritization of Forest Fire Incidences in Andhra Pradesh, India." *Environmental Monitoring and Assessment* 187(10): 616. https://doi.org/10.1007/s10661-015-4821-y.

Mehdipour, N., S. Fakheran, A. Soffianian, and S. Pourmanafi. 2019. "Road-Induced Fragmentation and the Environmental Value of Roadless Areas in a Partly Protected Landscape in Central Iran." *Environmental Monitoring and Assessment* 191(7): 461. https://doi.org/10.1007/s10661-019-7571-4.

Merkohasanaj, M., D. Rodríguez-Rodríguez, M. C. García-Martínez, M. Vargas-Yáñez, J. Guillén, and D. A. Malak. 2019. "Assessing the Environmental Effectiveness of the Spanish Marine Reserve Network Using Remote Sensing." *Ecological Indicators* 107: 105583. https://doi.org/10.1016/j.ecolind.2019.105583.

Mora, C., and P. F. Sale. 2011. "Ongoing Global Biodiversity Loss and the Need to Move beyond Protected Areas: A Review of the Technical and Practical Shortcomings of Protected Areas on Land and Sea." *Marine Ecology Progress Series* 434: 251–66. https://doi.org/10.3354/meps09214.

Moreno, A., M. Neumann, P. M. Mohebalian, C. Thurnher, and H. Hasenauer. 2019. "The Continental Impact of European Forest Conservation Policy and Management on Productivity Stability." *Remote Sensing* 11(1): 87. https://doi.org/10.3390/rs11010087.

Mtui, D. T., C. A. Lepczyk, Q. Chen, T. Miura, and L. J. Cox. 2017. "Assessing Multi-Decadal Land-Cover—Land-Use Change in Two Wildlife Protected Areas in Tanzania Using Landsat Imagery." *PLoS One* 12(9): e0185468. https://doi.org/10.1371/journal.pone.0185468.

Nagendra, H., R. Lucas, J. P. Honrado, R. H. G. Jongman, C. Tarantino, M. Adamo, and P. Mairota. 2013. "Remote Sensing for Conservation Monitoring: Assessing Protected Areas, Habitat Extent, Habitat Condition, Species Diversity, and Threats." *Ecological Indicators* 33: 45–59. https://doi.org/10.1016/j.ecolind.2012.09.014.

Nagendra, H., S. Paul, S. Pareeth, and S. Dutt. 2009. "Landscapes of Protection: Forest Change and Fragmentation in Northern West Bengal, India." *Environmental Management* 44(5): 853–64. https://doi.org/10.1007/s00267-009-9374-9.

Nagendra, H., D. Rocchini, and R. Ghate. 2010. "Beyond Parks as Monoliths: Spatially Differentiating Park-People Relationships in the Tadoba Andhari Tiger Reserve in India." *Biological Conservation* 143(12): 2900–8. https://doi.org/10.1016/j.biocon.2010.04.050.

Nagendra, H., C. Tucker, L. Carlson, J. Southworth, M. Karmacharya, and B. Karna. 2004. "Monitoring Parks through Remote Sensing: Studies in Nepal and Honduras." *Environmental Management* 34(5): 748–60. https://doi.org/10.1007/s00267-004-0028-7.

Naughton-Treves, L., M. B. Holland, and K. Brandon. 2005. "The Role of Protected Areas in Conserving Biodiversity and Sustaining Local Livelihoods." *Annual Review of Environment and Resources* 30(1): 219–52. https://doi.org/10.1146/annurev.energy.30.050504.164507.

Nelson, A., and K. M. Chomitz. 2011. "Effectiveness of Strict vs. Multiple Use Protected Areas in Reducing Tropical Forest Fires: A Global Analysis Using Matching Methods." *PLoS One* 6(8): e22722. https://doi.org/10.1371/journal.pone.0022722.

Oldekop, J. A., G. Holmes, W. E. Harris, and K. L. Evans. 2016. "A Global Assessment of the Social and Conservation Outcomes of Protected Areas." *Conservation Biology* 30(1): 133–41. https://doi.org/10.1111/cobi.12568.

Onojeghuo, A. O., and A. R. Onojeghuo. 2015. "Mapping Forest Transition Trends in Okomu Reserve Using Landsat and UK-DMC-2 Satellite Data." *South African Journal of Geomatics* 4(4): 486–501. https://doi.org/10.4314/sajg.v4i4.9.

Pereira, H. M., S. Ferrier, M. Walters, G. N. Geller, R. H. G. Jongman, R. J. Scholes, M. W. Bruford, N. Brummitt, S. H. M. Butchart, and A. C. Cardoso. 2013. "Essential Biodiversity Variables." *Science* 339(6117): 277–8. https://doi.org/10.1126/science.1229931.

Pettorelli, N., H. Schulte to Bühne, A. Tulloch, G. Dubois, C. Macinnis-Ng, A. M. Queirós, D. A. Keith, et al. 2018. "Satellite Remote Sensing of Ecosystem Functions: Opportunities, Challenges and Way Forward." *Remote Sensing in Ecology and Conservation* 4(2): 71–93. https://doi.org/10.1002/rse2.59.

Phua, M.-H., S. Tsuyuki, N. Furuya, and J. S. Lee. 2008. "Detecting Deforestation with a Spectral Change Detection Approach Using Multitemporal Landsat Data: A Case Study of Kinabalu Park, Sabah, Malaysia." *Journal of Environmental Management* 88(4): 784–95. https://doi.org/10.1016/j.jenvman.2007.04.011.

Rasheed, A. R. 2020. "Marine Protected Areas and Human Well-Being—A Systematic Review and Recommendations." *Ecosystem Services* 41: 101048. https://doi.org/10.1016/j.ecoser.2019.101048.

Ren, G., S. S. Young, L. Wang, W. Wang, Y. Long, R. Wu, J. Li, J. Zhu, and D. W. Yu. 2015. "Effectiveness of China's National Forest Protection Program and Nature Reserves." *Conservation Biology* 29(5): 1368–77. https://doi.org/10.1111/cobi.12561.

Richter, K., T. B. Hank, W. Mauser, and C. Atzberger. 2012. "Derivation of Biophysical Variables from Earth Observation Data: Validation and Statistical Measures." *Journal of Applied Remote Sensing* 6(1): 063557.

Rioja-Nieto, R., J. A. Moreno-Ruiz, and J. Gomez-Valdes. 2015. "Effect of the Management of a Natural Protected Area in the Landscape of the Mangrove Forest in the Yucatan Peninsula." *Hidrobiologica* 25(2): 203–11.

Sahagún-Sánchez, F. J., and H. Reyes-Hernández. 2018. "Impact of Land Use Cover Change on Protected Natural Areas in Central Region of Sierra Madre Oriental, Mexico." *CienciaUAT* 12(2): 6–21.

Schrodt, F., J. J. Bailey, W. D. Kissling, K. F. Rijsdijk, A. C. Seijmonsbergen, D. van Ree, J. Hjort, et al. 2019. "Opinion: To Advance Sustainable Stewardship, We Must Document Not Only Biodiversity but Geodiversity." *Proceedings of the National Academy of Sciences* 116(33): 16155–8. https://doi.org/10.1073/pnas.1911799116.

Schulte to Buehne, H., M. Wegmann, S. M. Durant, C. Ransom, P. de Ornellas, S. Grange, H. Beatty, and N. Pettorelli. 2017. "Protection Status and National Socio-Economic Context Shape Land Conversion in and around a Key Transboundary Protected Area Complex in West Africa." *Remote Sensing in Ecology and Conservation* 3(4): 190–201. https://doi.org/10.1002/rse2.47.

Simons-Legaard, E. M., D. J. Harrison, and K. R. Legaard. 2018. "Ineffectiveness of Local Zoning to Reduce Regional Loss and Fragmentation of Wintering Habitat for White-Tailed Deer." *Forest Ecology and Management* 427(11): 78–85. https://doi.org/10.1016/j.foreco.2018.05.027.

Sommerfeld, A., C. Senf, B. Buma, A. W. D'Amato, T. Despres, I. Diaz-Hormazabal, S. Fraver, et al. 2018. "Patterns and Drivers of Recent Disturbances across the Temperate Forest Biome." *Nature Communications* 9(10): 4355. https://doi.org/10.1038/s41467-018-06788-9.

Southworth, J., H. Nagendra, L. A. Carlson, and C. Tucker. 2004. "Assessing the Impact of Celaque National Park on Forest Fragmentation in Western Honduras." *Applied Geography* 24(4): 303–22. https://doi.org/10.1016/j.apgeog.2004.07.003.

Stephenson, P. J., N. D. Burgess, L. Jungmann, J. Loh, S. O'Connor, T. Oldfield, W. Reidhead, and A. Shapiro. 2015. "Overcoming the Challenges to Conservation Monitoring: Integrating Data from in-situ Reporting and Global Data Sets to Measure Impact and Performance." *Biodiversity* 16(2–3): 68–85. https://doi.org/10.1080/14888386.2015.1070373.

Sunderland-Groves, J. L., D. A. Slayback, M. P. Bessike Balinga, and T. C. H. Sunderland. 2011. "Impacts of Co-Management on Western Chimpanzee (*Pan troglodytes verus*) Habitat and Conservation in Nialama Classified Forest, Republic of Guinea: A Satellite Perspective." *Biodiversity and Conservation* 20(12): 2745–57. https://doi.org/10.1007/s10531-011-0102-4.

Tang, Z., J. Fang, J. Sun, and K. J. Gaston. 2011. "Effectiveness of Protected Areas in Maintaining Plant Production." *PLoS One* 6(4): e19116. https://doi.org/10.1371/journal.pone.0019116.

Tuholske, C., Z. Tane, D. Lopez-Carr, D. Roberts, and S. Cassels. 2017. "Thirty Years of Land Use/Cover Change in the Caribbean: Assessing the Relationship between Urbanization and Mangrove Loss in Roatan, Honduras." *Applied Geography* 88(11): 84–93. https://doi.org/10.1016/j.apgeog.2017.08.018.

UNEP-WCMC, IUCN, and NSG. 2020. *Protected Planet Live Report 2020*. Cambridge; Gland; and Washington, DC: UNEP-WCMC, IUCN, and NGS.

Vanclay, J. K. 2001. "The Effectiveness of Parks." *Science* 293(5532): 1007–7. https://doi.org/10.1126/science.293.5532.1007a.

Watson, J. E. M., N. Dudley, D. B. Segan, and M. Hockings. 2014. "The Performance and Potential of Protected Areas." *Nature* 515(7525): 67–73. https://doi.org/10.1038/nature13947.

von Wehrden, H., D. J. Abson, M. Beckmann, A. F. Cord, S. Klotz, and R. Seppelt. 2014. "Realigning the Land-Sharing/Land-Sparing Debate to Match Conservation Needs: Considering Diversity Scales and Land-Use History." *Landscape Ecology* 29(6): 941–8. https://doi.org/10.1007/s10980-014-0038-7.

Westwood, A. R., C. Staicer, P. Solymos, S. Hache, T. Fontaine, E. M. Bayne, and D. Mazerolle. 2019. "Estimating the Conservation Value of Protected Areas in Maritime Canada for Two Species at Risk: The Olive-Sided Flycatcher (*Contopus cooperi*) and Canada Warbler (*Cardellina canadensis*)." *Avian Conservation and Ecology* 14(1): 16. https://doi.org/10.5751/ACE-01359-140116.

Wright, S. J., G. A. Sanchez-Azofeifa, C. Portillo-Quintero, and D. Davies. 2007. "Poverty and Corruption Compromise Tropical Forest Reserves." *Ecological Applications* 17(5): 1259–66. https://doi.org/10.1890/06-1330.1.

Xie, Z., J. Liu, Z. Ma, X. Duan, and Y. Cui. 2012. "Effect of Surrounding Land-Use Change on the Wetland Landscape Pattern of a Natural Protected Area in Tianjin, China." *International Journal of Sustainable Development and World Ecology* 19(1): 16–24. https://doi.org/10.1080/13504509.2011.583697.

Zafra-Calvo, N., and J. Geldmann. 2020. "Protected Areas to Deliver Biodiversity Need Management Effectiveness and Equity." *Global Ecology and Conservation* 22(6): e01026. https://doi.org/10.1016/j.gecco.2020.e01026.

Zhang, B., Q. Zhang, Q. Feng, B. Cui, and S. Zhang. 2017. "Simulation of the Spatial Stresses Due to Territorial Land Development on Yellow River Delta Nature Reserve Using a GIS-Based Assessment Model." *Environmental Monitoring and Assessment* 189(7): 331. https://doi.org/10.1007/s10661-017-6038-8.

SUPPORTING INFORMATION

Additional supporting information may be found in the online version of the article at the publisher's website.

How to cite this article: Gohr, Charlotte, Henrik von Wehrden, Felix May, and Pierre L. Ibisch. 2022. "Remotely Sensed Effectiveness Assessments of Protected Areas Lack a Common Framework: A Review." *Ecosphere* 13(4): e4053. https://doi.org/10.1002/ecs2.4053

2.2. Article II

Quantifying the mitigation of temperature extremes by forests and wetlands in a temperate landscape

Contents lists available at ScienceDirect

Ecological Informatics

journal homepage: www.elsevier.com/locate/ecolinf

Quantifying the mitigation of temperature extremes by forests and wetlands in a temperate landscape

Charlotte Gohr [a,b,*], Jeanette S. Blumröder [a,b], Douglas Sheil [c,d], Pierre L. Ibisch [a,b]

[a] Centre for Econics and Ecosystem Management, Eberswalde University for Sustainable Development, Eberswalde, Germany
[b] Biosphere Reserves Institute, Eberswalde University for Sustainable Development, Eberswalde, Germany
[c] Forest Ecology and Forest Management Group, Wageningen University & Research, Wageningen, Netherlands
[d] Center for International Forestry Research, P.O. Box 0113 BOCBD, Bogor, Indonesia

ARTICLE INFO

Keywords:
Surface cooling
Climate change mitigation
Ecological indicators
Forests
Land surface temperature
NDVI

ABSTRACT

As a result of ongoing climate change and more frequent heat events, the regulating services of land cover in terms of moderating and mitigating local temperatures are increasingly important. While the reduced temperatures found in forests and wetlands are recognized, their wider contribution to regional landscape cooling remains largely uncharacterized and unquantified. Herein, we propose and test a new method that estimates the temperature response and inertia of landscapes in high temperatures, based on land cover share. In order to achieve this goal, we combined the MODIS daytime land surface temperature (henceforth LST) time series and CORINE land cover data. We classified the time series in two ways, i.e. by stepwise temperature range (−10/−5 °C to +35/+40 °C) and by the occurrence of hot days (days with a mean LST ≥ 30 °C). As an explanatory variable, we developed and used a *greenest pixel composite* of the MODIS normalized difference vegetation index (NDVI) time series. In our study area, covering parts of northeastern Germany and western Poland, the fragmented landscape has heterogeneous temperature patterns, including urban heat islands, warm agricultural areas, cool forests and cold wetlands. We found that at high temperature ranges only forests and wetlands remained comparably cool, with LSTs up to 20.8 °C lower than the maximum LST in the study area. The analysis of land cover shares and LSTs revealed the substantial cooling effect of forests and wetlands in line with increasing land cover share in higher temperature ranges, as well as on hot days. The relation between LST and the NDVI indicated vegetation cover as the cause. We propose the corresponding metrics to quantify landscape-level temperature regulation. Equally, we advocate for management to identify these ecosystem services and their current and potential contributions, along with implications for sustaining and increasing, both tree cover and wetlands and thereby adapting landscapes to climate change.

1. Introduction

Increases in temperature, which are among the most dangerous impacts of climate change, threaten socioeconomic activities (Chen et al., 2020), ecosystem functioning (Fisher et al., 2017) and human health (Luber and McGeehin, 2008; Mora et al., 2017). Human mortality estimates based on data from climate-related heat exposure and deaths in 732 locations over 43 countries suggest a mean of 37.0% (range 20.5–76.3%) between 1991 and 2018, with increased mortality seen on all continents (Vicedo-Cabrera et al., 2021). Heat also contributes to other climate-related challenges such as increased water-stress and drought (Fisher et al., 2017; Teuling et al., 2013). One way to avoid these negative effects is to prevent or moderate temperature extremes (Hatfield and Prueger, 2015).

The relationship between remotely sensed *land surface temperature* (LST) and land cover has been investigated in various contexts (Alkama and Cescatti, 2016; Bonan, 2008; Bright et al., 2017; Jin and Dickinson, 2010). Different land covers are associated with different thermal properties, especially the heat island effects that occur in urban and other built-up areas (Bartesaghi-Koc et al., 2020; Feizizadeh and Blaschke, 2013; Liu et al., 2018; Su et al., 2010; Tran et al., 2017). Land cover proportion has been used in a study of an urban area to investigate to what extent landscape metrics can explain LST (Liu et al., 2018), as well as for a vegetation fraction cover analysis (Duveiller et al., 2018;

* Corresponding author at: Schicklerstr. 5, 16225 Eberswalde, Germany.
E-mail address: charlotte.gohr@hnee.de (C. Gohr).

https://doi.org/10.1016/j.ecoinf.2021.101442
Received 5 July 2021; Received in revised form 22 September 2021; Accepted 27 September 2021

Schwaab et al., 2020). However, the response of hot-day LSTs in relation to land cover shares at different temperature ranges with a focus on multiuse landscapes with urban, forest and agricultural land rather than urban landscapes has not been studied to date.

Greenness, or vegetation productivity measured via the *Normalized Difference Vegetation Index* (NDVI), is commonly applied to quantify and substantiate the impacts of different land cover types on LST. In this regard, a correlation between LST and the NDVI has been detected in various studies (Weng et al., 2004; Xiao and Weng, 2007; Yuan and Bauer, 2007). Vegetated areas with high NDVI show lower LSTs (Deng et al., 2018). The NDVI provides insights into the condition of the vegetation and serves as an explanatory variable for measured LST (Su et al., 2010).

The use of LST to scrutinize the cooling function of forests is well established, as seen in a global study that compared LST and station air temperatures (Mildrexler et al., 2011). Equally, the moderate cooling of temperate forests in summer was observed in a global study using MODIS LST (Li et al., 2015). LST observations have also confirmed cooling resulting from afforestation in China (Peng et al., 2014), while a study of restored oak woodland in Canada found LST declined as the forest matured (Hamberg et al., 2020). Higher LSTs were found following a decline in forest cover due to bark beetle attack following drought conditions in the Czech Republic, thereby demonstrating the utility of remotely-sensed LSTs: rising temperatures in drought-damaged forests led to advection effects and extracted water vapor from the landscape, further promoting landscape desiccation (Hesslerová et al., 2018). In various case studies in Germany, it has been shown that surface temperatures are a good indicator for assessing the functioning of forests and the influence of land use as well as edge effects (e.g. Blumröder et al., 2019a, 2019b; Blumröder et al., 2020; Blumröder et al., 2021; Ibisch et al., 2019).

The threat of high temperature extremes on plants is evident in the drying effect as a result of high air temperatures that absorb more water, while at the same time plants have an increased vapor pressure deficit and atmospheric water vapor demand (Hatfield and Prueger, 2015; Hesslerová et al., 2018). In forest ecosystems, increasing vapor pressure deficit has been linked to increased tree mortality (Breshears et al., 2013; Grossiord et al., 2020; Williams et al., 2013). At the same time, water evaporation from vegetation is an important source of atmospheric moisture (Sheil, 2018). Total water vapor emissions from forests (combined transpiration and evaporation from other sources) are typically higher than for other vegetation, and they can be even higher than for open water (Sheil, 2018). Maintaining cool and healthy forests is also important for preserving microclimatic refuges for many organisms threatened by climate change (Suggitt et al., 2011), while the cooling effects of vegetation contribute to ecosystem-based adaptation and nature-based solutions to heat stress (Bright et al., 2017).

Wetlands, in this study referring equally to water bodies and marshes, influence regional temperatures and play a major role as an ecosystem service in regulating regional climates (Hesslerová et al., 2019; Ramsar Convention Secretariat, 2018; Pokorný et al., 2016). The general cooling function of wetlands in high temperatures is caused by its albedo and evaporation characteristics (Hesslerová et al., 2019). These biophysical processes are not only relevant regarding climate change but functioning wetlands also have a positive effect on the carbon circle and greenhouse gas emissions (Pokorný et al., 2016).

The loss of wetland cover globally is increasing, primarily as a result of human activities (Ramsar Convention Secretariat, 2018). Using remote sensing data, regional studies in China have determined that marshland loss implies warmer local LST (Shen et al., 2020) whereas maintained water bodies and their surroundings have lower LSTs than adjacent urban areas and thereby decrease the UHI effect (Wu and Zhang, 2019). On a global scale, an LST-based analysis distinguished between the regional cooling effect of wetlands in tropical regions throughout the year and seasonal effects in boreal regions with warming effects in winter and cooling effects in summer (Wu et al., 2021). More generally, it is important to understand and foster the daytime cooling capacity of different land covers, particularly when the key factors can be modified through management.

In this paper, we examine the landscape-scale influence of forests and wetlands on moderating high temperature events by applying a new method that relates land cover share to temperature range and to hot days. We use spatial time series based on the large-scale remote sensing data of land surface temperature, classified into temperature ranges and a hot-day composite, to investigate temperature changes in relation to increasing forest and wetland cover. We investigate land cover specific greenness (NDVI) as a possible cause of spatial temperature patterns, and we also discuss the integration of this ecosystem service regulating temperature into landscape management planning.

2. Material and methods

2.1. Study area

We chose a landscape in northeastern Germany sitting on the border with Poland (10,726.8 km^2) and about 100 km south of the Baltic Sea. There is a pronounced land use gradient from the metropolitan region of Berlin in the south to rural ecosystems in the north, comprising both intensively managed agricultural areas and forested regions (Fig. 1). For a European lowland area, the region is quite unique, as within a range of 100 km there is both a large urban area and a forest-dominated landscape, including smaller patches of old-growth forests. In the cultural landscape of the northeastern part of the state of Brandenburg and southeastern Mecklenburg Western Pomerania, large agricultural areas and forests dominate. Roughly 70% of the forest area is dominated by Scots pine (*Pinus sylvestris* L.) plantations, and only smaller parts of the native deciduous broad-leaved forest comprise old-growth forest, most of which is dominated by beech (*Fagus sylvatica* L.) (Ibisch et al., 2018). There are a few scattered lakes of postglacial origin. The soil pattern in the study area is relatively homogenous (Panagos, 2006; Van Liedekerke et al., 2006), and the topography is characterized by small differences in altitude in the range of −10 and 168 m above sea level and a mean slope value of less than 1.7% (Jarvis et al., 2008) (Appendix 1).

2.2. Satellite imagery

We used three preprocessed datasets to examine LST, the NDVI and land cover. Surface temperature information was taken from MODIS (Moderate Resolution Imaging Spectroradiometer) satellite data (Table 1). MODIS is a radio spectrometer on board NASA's Aqua EOS-PM1 satellite, which produces images of the Earth's surface over a wide spectral range and thus allows, among other things, a large-scale investigation of the radiation budget. From the measured radiation intensity in the infrared range (bands 31 & 32 with 10.8–12.3 μm), daily surface temperature is calculated at a resolution of about 1 km, taking into account emissivity and the water vapor content of the air column with the help of the "Generalized Split Window Algorithm" (Wan et al., 2015).

Surface temperature results from various factors such as albedo or emissivity, and it can be locally heterogeneous. Land surfaces can heat up much more during the day than the air above (station air temperature), but there is usually a continuous exchange of heat (Jin and Dickinson, 2010; Mildrexler et al., 2011).

NDVI data was also acquired from a MODIS time series (Table 1). The NDVI is calculated by using the near-infrared and visible spectra provided by satellite imagery, with a 1 km resolution ranging from −1 to 1 (Didan, 2015).

Land cover information was derived from the CORINE dataset of the European Environment Agency's Copernicus program (CORINE: Coordination of Information on the Environment; Table 1). Areas coded in the CORINE dataset as "non-irrigated land" and "pastures" are henceforth referred to as "agricultural land", and areas coded as

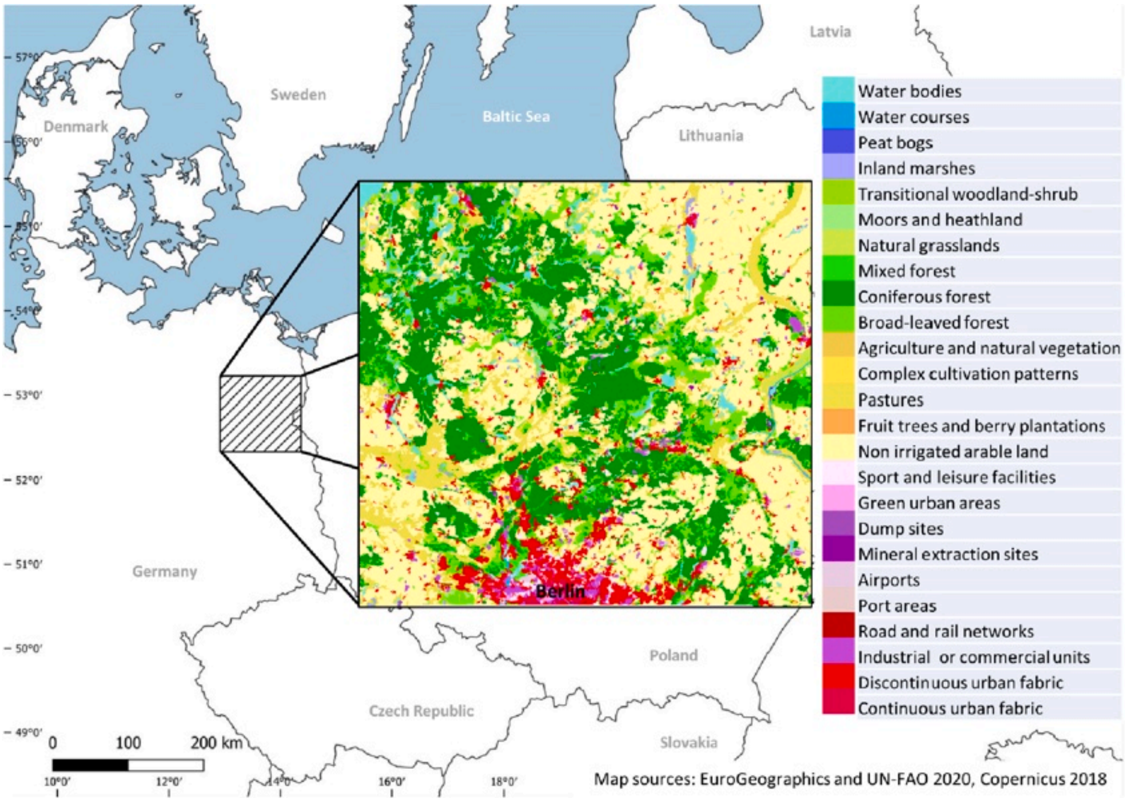


Fig. 1. Map showing the location of the study area (10,726.8 km^2) and its corresponding land cover types.

Table 1
Properties of the used datasets.

Sensor, provider and product ID	Spatial resolution	Temporal coverage	Selected time series	Sequence and time of recording	Number of images
MODIS Aqua, NASA, MYD11A1.006, LST[a]	1 km	04.07.2002 to 13.08.2021[a]	04.07.2002 to 31.12.2020	Time series, 1 day, ~1.30 pm	6618
MODIS Aqua, NASA, MYD13A2.006, NDVI[b]	1 km	04.07.2002 to 20.07.2021[b]	04.07.2002 to 31.12.2020	Time series, 16-day composite, ~1.30 pm	426
Copernicus, EEA, Corine Land Cover[c]	100 m	1990, 2000, 2006, 2012, 2018	2018	Snapshot 2018	1

[a] (Wan et al., 2015). Available time resolution state 17.08.2021. doi:https://doi.org/10.5067/MODIS/MYD11A1.006

[b] To generate the 16-day NDVI composite, the product is run through an algorithm that selects the best pixel with low clouds, a low view angle and the highest NDVI value (Didan, 2015). Available time resolution state 17.08.2021. doi:https://doi.org/10.5067/MODIS/MYD13A2.006

[c] (Copernicus, 2018). https://developers.google.com/earth-engine/datasets/catalog/COPERNICUS_CORINE_V20_100m

"discontinuous urban fabric" and "industrial or commercial units" are henceforth referred to as "urban areas". If not stated otherwise, "forest" is a summary of areas coded as "broad-leaved", "coniferous" and "mixed" forest. The different water body types ("water bodies", "water courses", "inland marshes") are summarized under the term "wetlands".

2.3. Data-processing and statistical analysis

For data acquisition, data-processing and statistical analysis, we used the Google Earth Engine (Google, 2021; Gorelick et al., 2017) as well as the integrated development environment R (R Core Team, 2021).

We generated the temperature range dataset by filtering the mean LST of each image in the whole MODIS LST time series, sorting them stepwise by 5 °C temperature ranges (−10/−5 °C to +35/+40 °C) and calculating a per-pixel mean image for each temperature range. This resulted in ten images with mean values associated with the ten temperature ranges.

The analysis of extreme weather conditions focuses on "hot days" occurring in the period July 2002 – December 2020. The definition of a "hot day" follows Germany's national meteorological service (Deutscher Wetterdienst, DWD), i.e. a maximum air temperature of at least 30 °C (DWD, 2021). In our study, a "hot day" is an image with at least one LST pixel value ≥30 °C. Although the number of days with LST data ≥30 °C is most likely higher than the number of days with corresponding air temperatures, LST data with the chosen threshold represent extreme temperatures at the surface and are therefore suitable for our approach. We reduced the resulting MODIS LST time series with only hot days to a single image with the per-pixel mean of all hot days, which we henceforth refer to as the *hot day composite*.

To quantify the cooling capacities of different land cover types on hot days, we examined the hot-day LST range per land cover type across the area. Land cover changes between 2000 and 2018 were negligible (Appendix 2), so we combined the *hot day composite* (2002−2020) with the land cover data from 2018. The study area includes 13,300 LST pixels at a 1 km resolution, with each pixel comprising approximately 155 land cover pixels at a 100 m resolution. To identify possible differences between land cover types, we grouped and plotted hot-day LSTs per land cover type, attributing LSTs to all land cover shares ranging from 0 to 100%. We also modelled scenarios in which 1, 5 and 10% of agricultural land is replaced by forest to determine whether the temperature in the study area is affected (Appendix 3). For this purpose, the influence on the mean hot-day LST in the study area was calculated when pixels with 100% agricultural land share were replaced by 100% forest. To quantify the cooling capacities of forests and wetlands, we selected temperature pixels consisting of forests and wetlands with a cover share of ≥50%, thereby illustrating the varying ranges of hot-day LSTs for increasing land cover share from 50 to 100%, without the substantial impacts of mixed pixels with shares between 0 and 50%. For simplicity, we focus on single land covers and omit LST responses due to mixed land cover shares.

To search for LST patterns and land cover share across all temperature ranges, we generated 10 mean images per temperature range (−10/−5 °C to +35/+40 °C) and combined them with the land cover image. For each temperature range, we chose all LST values with a forest or wetland cover share of ≥50% and generated linear models for each land cover type and its associated temperature range. This allowed us to compare the linear model coefficients (the slope representing the relation between LST and land cover share increase) of each land cover type per temperature range.

We used the NDVI to examine possible explanations for dissimilarities in hot-day LSTs for different land cover types. However, the NDVI time series is not processed using the hot-days approach, since a distorted picture of vegetation may arise during heat stress. Therefore, we averaged the "greenest" pixels (i.e. the highest NDVI value) of the 16-day summer month composites (June, July, August) in the MODIS NDVI time series 2002–2020 for our region (henceforth the *greenest composite*).

We combined the *hot day composite* (averaged LST image of hot days), the *greenest composite* and the land cover image to find temperature regulation patterns in the study area. We selected only LST and NDVI values with land cover shares ≥50%. For comparison, we centered (subtracted the mean per value) and scaled (divided the standard deviation per value) LST and the NDVI. To determine if LST varies significantly between different land cover types, we performed an analysis of variance (ANOVA). To clarify possible covariance between land cover and the NDVI in explaining LST, we additionally performed an analysis of co-variance (ANCOVA). For a spatial analysis of LST and the NDVI in forested areas, we produced a bivariate map for areas with a ≥ 50% forest share, using the averaged LST image for hot days and the greenest NDVI image.

3. Results

3.1. Landscape temperatures

Deviation in the mean LST indicated land cover-related patterns across the study area for each temperature range considered herein (Fig. 2). Berlin - in the south - was consistently warmer across all temperature ranges. In the higher temperature ranges, from 10 to 15 °C upwards, forests and wetlands tended to show below-average variations, indicating that the heating of these systems lags behind other parts of the landscape. This delayed response allows temperature differences between the faster heating (warmest) and slower heating (coolest) pixels to increase in the higher temperature bands. In the hottest temperature range (35–40 °C), the LST difference between the hottest pixel (44.1 °C) and the coolest pixel (23.3 °C) reached 20.8 °C.

3.2. Land cover shares and temperature effects

The regional distribution of hot-day LST values in relation to the percentage of forest and wetland coverage per pixel indicated a significant negative relationship (Fig. 3). The strongest signal can be observed for wetlands (slope −7.98), followed by coniferous and broad-leaved forests. Data for mixed forests is relatively sparse and yielded no clear pattern (slope 0.36).

Similar patterns were observed when relating the land cover share gradient (= temperature change with increasing share of one land cover from 50 to 100%) to temperatures summarized in 5 °C stepwise temperature ranges, from −10/−5 °C to +35/+40 °C (Fig. 4). The linear model coefficient (LST ~ land cover share) was depicted for differentiated and summarized land cover types per temperature range. With increasing temperatures, the lm-coefficient became increasingly negative for wetlands, as well as, albeit to a lesser extent, for broad-leaved and coniferous forests (Fig. 4 A). The summarized land cover types indicated negative slopes for forests and wetlands when temperatures rise, whereas agricultural and urban areas showed a positive slope with rising temperature ranges (Fig. 4 B).

The *hot day composite* (mean image of the selection of hot-day LSTs

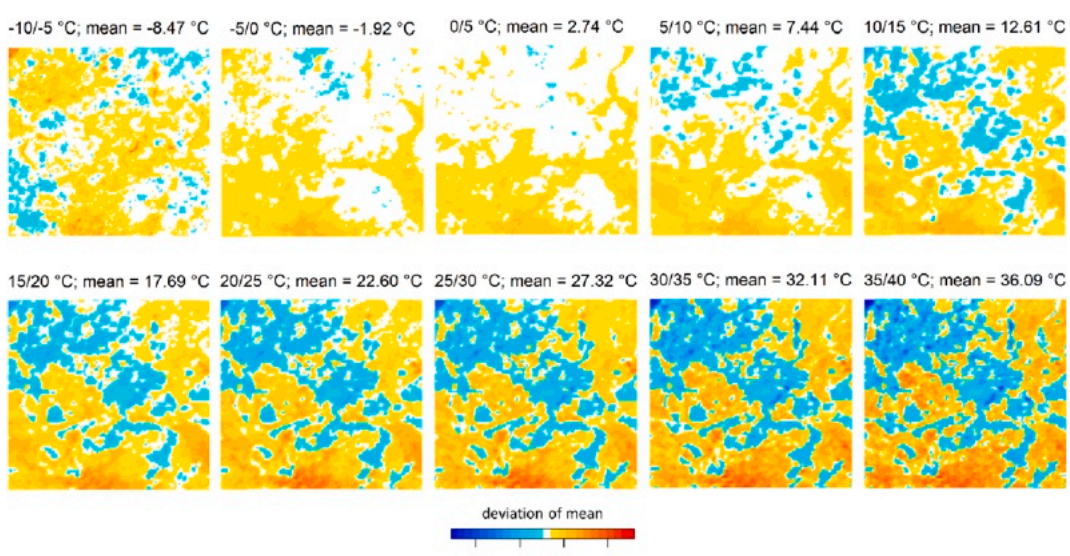


Fig. 2. Ten temperature ranges between −10 and − 5 °C and 35 and 40 °C in 2002–2020 for the study area (as per Fig. 1), showing the per-pixel deviation of mean LSTs. The respective mean value of each range is depicted in white, negative deviations (LSTs cooler than the mean) are blue and positive deviations (LSTs warmer than the mean) are red. For scale and locations, see Fig. 1. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

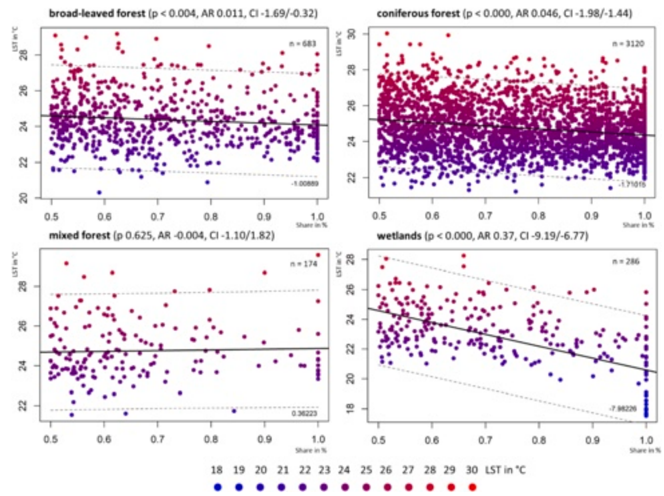


Fig. 3. Hot-day LST values in °C by shares of forest and wetland cover from a 50% to a 100% share. Each LST pixel comprises ~155 land cover pixels. Each LST value is related to the share of each land cover type per LST pixel. 'p' is the probability, 'AR' is the adjusted R-squared and 'CI' is the 95% confidence interval. Numbers of observations ('n') and the slope of the linear model are depicted in the plot.

with at least one regional value ≥30 °C) revealed an anticipated pattern with higher LSTs in urban areas and lower LSTs in forests and wetlands, using all land cover shares ranging from 0 to 100% (Fig. 5 A). Urban areas, i.e. industrial areas, settlements and non-contiguous urban areas, as well as agricultural areas showed values well above 26 °C, while forests (deciduous, coniferous and mixed) and wetlands were between 22 and 25 °C.

The study area comprises 6.6% continuous forest and 13.9% continuous agricultural land. When modelling the replacement of 1%, 5% and 10% forest with agricultural land, a change in the mean temperature on hot days is evident in the study area (Fig. 5 B, Appendix 3). A 10% increase in forest cover would lead to an estimated decrease of 0.9 °C in the mean temperature on hot days in the study area.

3.3. LST and the NDVI

Scaled values for the *hot day composite* and the greenest NDVI image were inversely related (Fig. 6). Only LST and NDVI values with ≥50% of one land cover type were included in the analysis. Urban areas indicated high LST and low NDVI values, whereas forests and wetlands indicated a reverse dispersion with a high NDVI and low LST. Weaker variances in LST and NDVI values were captured for agricultural land, while differences between LSTs for land cover types were significant (ANOVA, n = 13,300, $p < 0.000$). ANCOVA testing (Type III for unbalanced designs) revealed that with and without land cover as a co-variable, the NDVI had a significant relationship with LST (n = 13,300, $p < 0.000$).

Bivariate mapping of the distribution of LST and the NDVI for pixels with a ≥ 50% share of all forest types revealed a distinctive pattern (Fig. 7). Forest areas in the northwest of the area were cold and high in

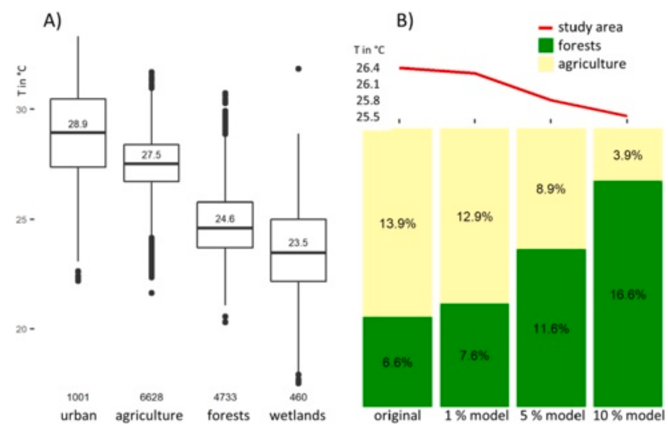


Fig. 5. A) Temperatures comprising different land cover types on days with a maximum temperature ≥ 30 °C (*hot day composite*) in the study area. Pixel count per class is indicated on the x-axis. Median value per class is depicted in each boxplot. B) Coverage and temperature changes in the study area for three models. Bars show scenarios when forests replace agricultural land by 1, 5 and 10%. Line plot shows the declining *hot day composite* temperature in the study area up to 0.9 °C with respective forest cover gain.

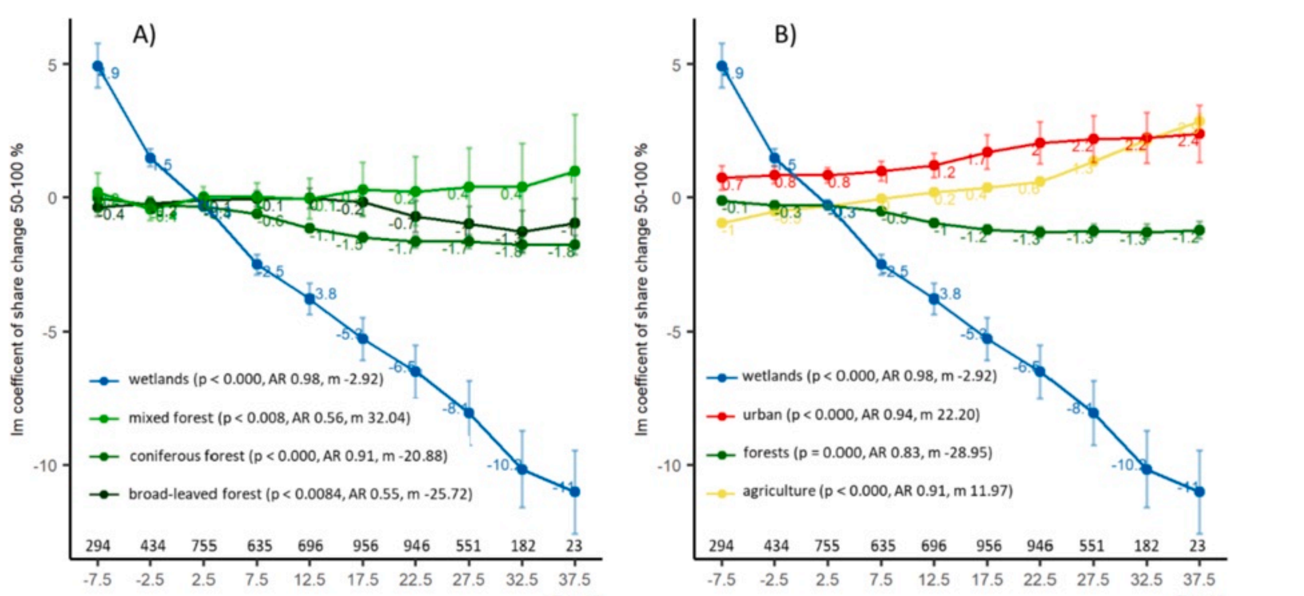


Fig. 4. Relation between the linear model coefficient (LST/range ~ 50–100% share) in stepwise ranges of 5 °C from −10/−5 °C to 35/40 °C for different land covers. Linear model 95% confidence intervals are depicted for each land cover type coefficient in its associated temperature range. Image count per temperature range is indicated on the x-axis. 'p' indicates the probability, 'AR' the adjusted R-squared and 'm' the slope of the linear model. A) Diagram of differentiated forest types and wetlands. B) Diagram of summarized land cover groups, namely wetlands, urban areas, forests and agricultural areas.

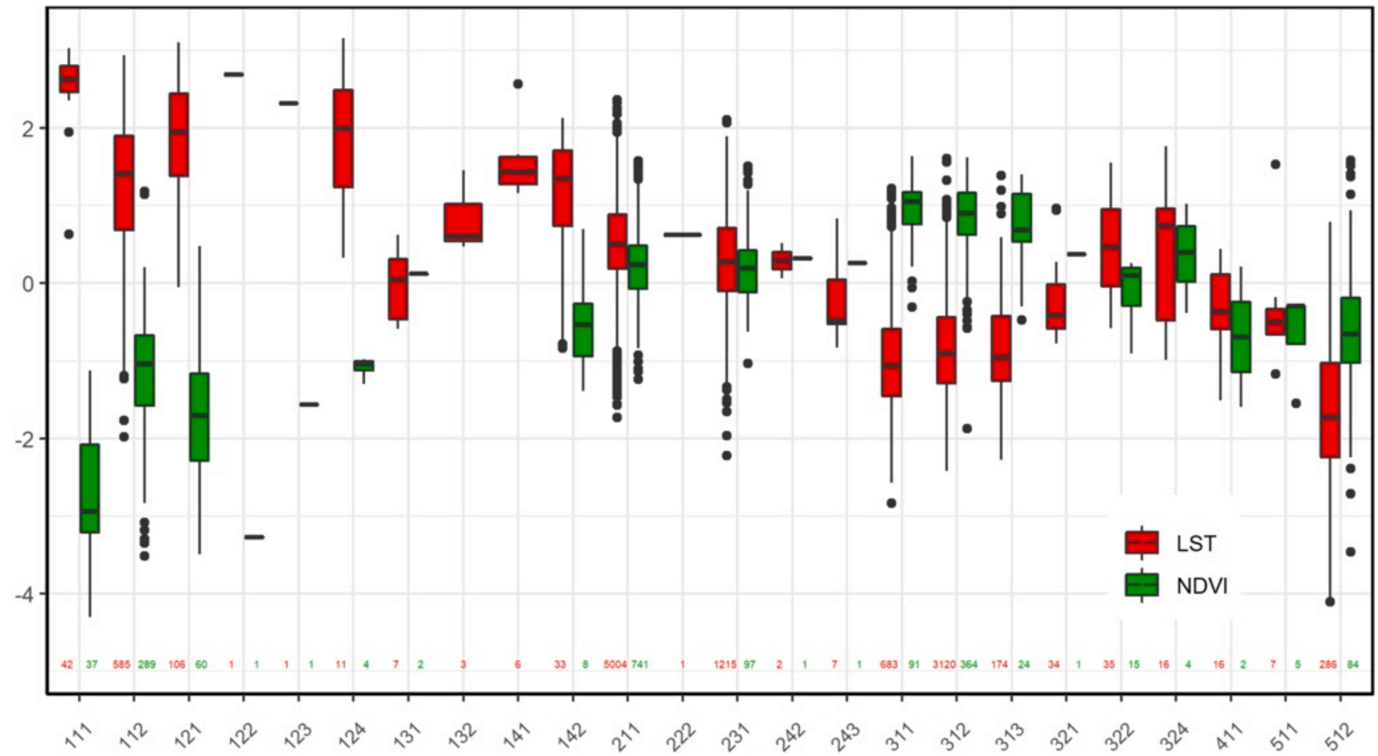


Fig. 6. Scaled distribution of the mean LST of hot days and the greenest NDVI for land cover types with a share of ≥50%. Colored numbers show the numbers of data points per boxplot. Hot-day LSTs are the means of each pixel of the time series 2002–2020 with a daily mean LST ≥ 30 °C. *Greenest composite* NDVI are the maximum values of each pixel of a time series 2002–2020 for the summer months (June, July, August). Both datasets are centered (subtracting the mean per value) and scaled (dividing the standard deviation per value). Designation and order follow the CORINE land cover classification.

greenness (marked in blue), while large conifer plantations at the center of the region were cold but less productive in terms of vegetation (marked in yellow). Smaller forest patches showed a lower NDVI and higher LSTs.

4. Discussion

We have quantified the cooling of landscapes associated with forests and wetlands - or, more precisely, reduced warming - by applying a new method that estimates temperature changes in line with increasing land cover fractions for different temperature ranges. The general temperature mitigation effects of forests and wetlands are consistent with previous studies (e.g. Alkama and Cescatti, 2016; Bonan, 2008; Bright et al., 2017; Frenne et al., 2019; Zellweger et al., 2019). Our quantitative approach is novel, and it demonstrates that we can estimate and manage these contributions and thus moderate temperature extremes across complex landscapes. We now examine the implications of these methods and results.

4.1. Landscape thermal effects of forests and wetlands

We measured daytime thermal effects as the slope of the temperature difference versus the relative share of a specific land cover type per LST pixel for temperatures ranging from −10/−5 °C to +35/+40 °C; all resulting relationships were statistically significant ($p < 0.005$, adjusted R-squared >0.5). Increasing forest cover (all forest types combined, or differentiated by broad-leaved, coniferous and mixed forest) reduces temperatures during heat events compared to other land cover types (Fig. 3 A). Urban areas and agricultural land heat up more (Fig. 3 B), while wetlands reveal the greatest temperature differences at high temperatures and warming at the coldest temperature. Previous studies have established that such thermal reactions in wetlands can be explained not only by albedo and evapotranspiration, but also by soil heat flux, i.e. energy absorbed in the summer and then released in the winter (Shen et al., 2021, 2020; Wu et al., 2021).

We observed that the cooling – or warming – of a given pixel depends on the relative share of each land cover type. The analysis of land cover share influence on hot days (*hot day composite*) indicated that while wetlands have the highest cooling effect when a pixel is 100% covered by wetland (lm coefficient −7.98), coniferous forest (lm coefficient −1.71) and broad-leaved forest (lm coefficient −1.01) also exhibit significantly lower LSTs if the share is closer to 100%. Only the influence of mixed forest (lm coefficient 0.36) was not clearly related to its share (Fig. 3), most likely a result of the lower number of observations leading to inadequate statistical power (Fig. 8). These findings are in congruence with the results of a study that showed how land cover proportion characterizes LST in a Chinese urban landscape (Liu et al., 2018). The stronger influence of coniferous forest in comparison to broad-leaved forest could be a result of more data points and planting densities. However, the absolute values show cooler LSTs in broad-leaved forests than in coniferous forests, which is consistent with a study of forest cover proportions across Europe (Schwaab et al., 2020).

The comparison of *hot day composite* temperatures (landscape mean of a day ≥30 °C) for different land cover types (with 0–100% share) at a scale of 100 m shows that forested areas and wetlands are up to 5 °C cooler than urban areas and agricultural lands (Fig. 5 A). Similar results in the southeastern United States have found that forests are 4–6 °C cooler than grasslands, and air temperatures are 2–3 °C lower (Novick and Katul, 2020). We have not dwelt here on the mechanisms by which this land surface temperature buffering is achieved, but these include both the thermal inertia that results as heat passes from warmer to colder bodies, particularly water, and the energy that is absorbed when water evaporates, both of which rely on the presence of water. The availability of water is of course also influenced by the presence of trees and wetlands and these influences are also potentially complex and involve a range of local to regional scales (Ellison et al., 2017; Sheil,

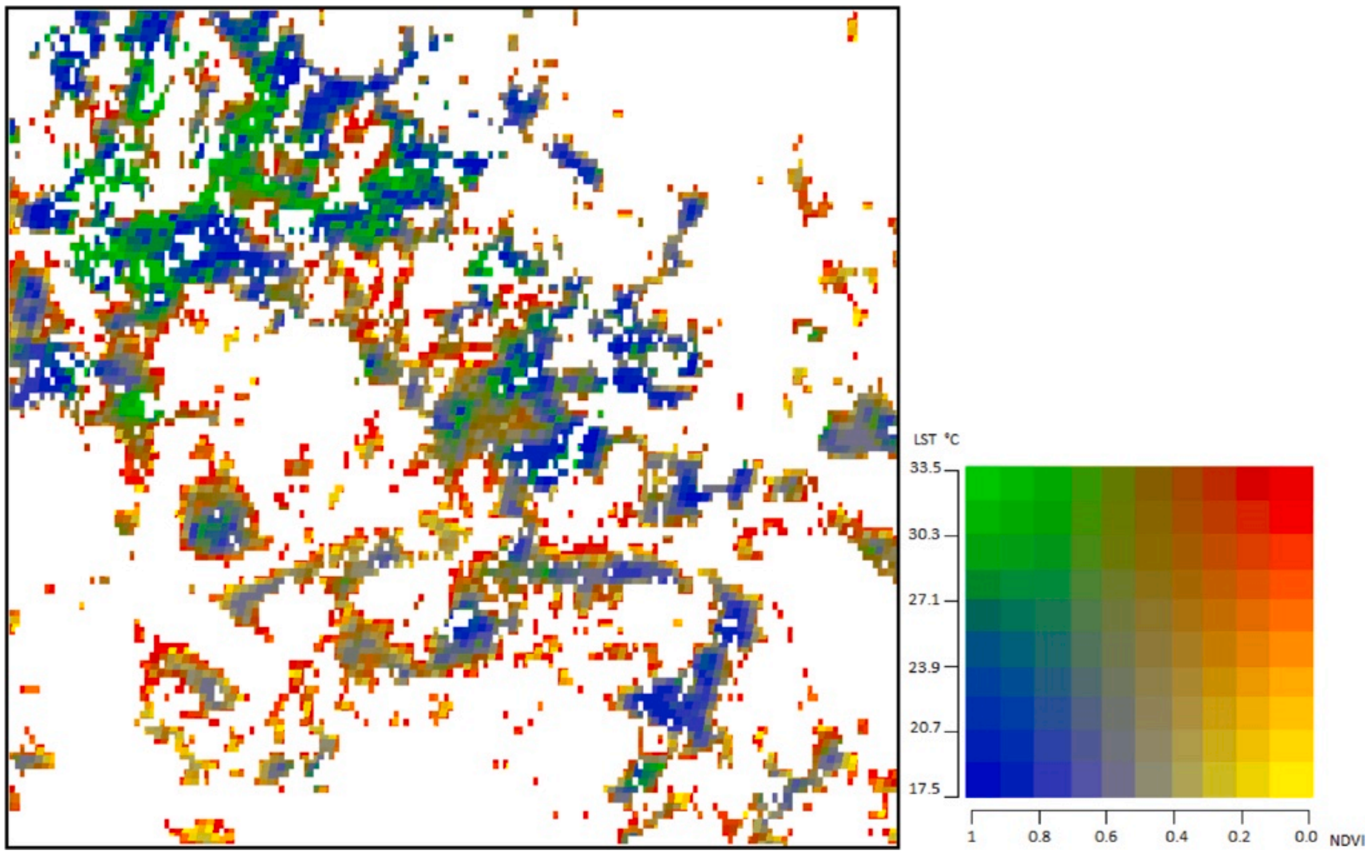


Fig. 7. Bivariate map of LST (means of each pixel of a time series 2002–2020 for hot days) and NDVI (max of each pixel of a time series 2002–2020) of forest areas with ≥50% share. Blue: NDVI high/LST low; Green: NDVI/LST high; Red: NDVI low/LST high; Yellow: NDVI low/ LST low. For scale and locations, see Fig. 1. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

2018). The cooling functions of forests and wetlands at high temperatures are closely linked to ecohydrological functions that support the extraction, recycling and storage of water in the ecosystem (Ellison et al., 2017). Evaporation, transpiration and shade (in forests) offered by forests and wetlands ensure local cooling during the daytime (Ellison et al., 2017; Maes et al., 2011; Shen et al., 2020). However, decreased soil moisture due to heat extremes can interfere with temperature-moderating functions (Teuling et al., 2010). In sum, land cover temperature mitigation provided by forests and wetlands comprises different local and regional factors, such as evaporation, albedo and energy dissipation, as well as supra-regional factors such as landscape land cover composition and clouds (Wu et al., 2021; Shen et al., 2020; Bright et al., 2017; Zeng et al., 2017; Bonan, 2008; Benayas et al., 2008; Zaitchik et al., 2006; Schneider and Kay, 1994).

4.2. *Forest cooling, greenness and landscape modelling*

The pattern of the *hot day composite* of LSTs and the greenest composite of the NDVI shows contradictory relations for urban areas (high LST, low NDVI) and forests (high NDVI, low LST) (Fig. 6). The ANCOVA test for an effect of the NDVI on LST accounting for land cover types, provides a significant result. As the NDVI and LST are negatively correlated, we conclude that more productive vegetation is associated with a lower LST, and the greenest pixel in a time series provides an indicator for LSTs. The local cooling function of forests is known to be maximized in dense biomass-rich stands (De Frenne et al., 2019; Norris et al., 2012; Zellweger et al., 2019; Schwaab et al., 2020), which is consistent with the results of another study on the island of Madeira, where thermal infrared radiation as a function of LST was used to identify different land cover types. The research determined that older ecosystems with more complex structures have lower average temperatures (Avelar et al., 2020).

Our models suggest that replacing 10% of agricultural land with forest would reduce the mean temperature of the *hot day composite* in the study area by 0.9 °C (Fig. 5 B). It is noteworthy that the models remain conservative estimates, as we neglected edge effects where cooling spills from one pixel to its neighbors - an influence that should be evaluated in future work. In general, an increase in forest area can be achieved by the natural succession of abandoned agricultural land, by planting trees or a combination of both. Benayas et al. (2008) proposed “woodland islets”, i.e. many small afforested areas providing ecosystem services and enhancing biodiversity (Benayas et al., 2008). Investigations into the ability of these “woodland islets” to cool the landscape are necessary in a landscape management seeking cooling functions.

4.3. *Implications for ecosystem-based adaptation for climate change*

The advancing climate crisis, with the increasing risk of extreme heat events, highlights the importance of maintaining landscapes where extreme temperatures are avoided as much as possible. To study the regulating ecosystem services of forests and wetlands further, we recommend the analysis of hot-day LSTs (days with a mean ≥30 °C in our region) in combination with the greenest pixel NDVI composite. The use of preprocessed spatial data (MODIS LST and NDVI, CORINE land cover) facilitates analysis. However, inaccuracies in spatiotemporal data, such as database errors, topographic characteristics and land cover changes, influence the results and need to be addressed. The spatial resolution of the MODIS Aqua satellite (1 km) is a limitation; however, it is sufficient to detect temperature buffers in our study region. A higher spatial resolution, available from Landsat, ASTER and Sentinel, would permit assessments on finer scales (e.g. Avelar et al., 2020; Hesslerová et al., 2018; Liu et al., 2018). It is important to note that the measured LSTs are not necessarily the daily maximum values, as the MODIS Aqua satellites take these measurements at around 1.30 pm each day, and the temperature often increases later in the day. This is particularly relevant, because the temperature mitigation of forests and wetlands is even greater in higher temperature bands (Fig. 2). While extreme heat exerts huge stress on ecosystems and people, we see that the presence of forests and wetlands can reduce such impacts (Fig. 3).

The thermal effects of forests and wetlands offer ecosystem-based adaptation, in order to reduce heat stress related to climate change (e.g. Kupika et al., 2019; Nanfuka et al., 2020; Schumacher et al., 2018). Furthermore, forest and wetland loss are therefore not only potential drivers of increasing heat stress, but they also most likely influence relevant biogeochemical functions such as carbon storage and the regulation of greenhouse gas emissions (Laurance et al., 1998; Liu et al., 2019; Pugh et al., 2020; Shen et al., 2020). We need to recognize these functions in landscape management and consider a number of goals and incentives (Lusiana et al., 2017). In addition, further work is required to assess the costs and benefits of different arrangements within a landscape (e.g. Parks and Hardie, 2018). More forests and more wetlands will provide greater adaptation benefits by mitigating extreme heat than any other type of land cover.

5. Conclusion

We quantified the thermal influence of forests and wetlands over a large area north of Berlin, Germany, by developing a novel pixel-based approach. The ability of forests and wetlands to moderate temperatures increases in line with spatial coverage and increased heat. Our approach, using readily available data, accounts for yearly and seasonal variance and shows patterns of expected temperature ranges for all land cover types. Our results highlight the value of forests and wetlands in reducing peak temperatures and thus mitigating some of the more severe impacts of climate change that are increasingly dangerous for human health. Our quantification of landscape cooling is valuable for the analysis of temperate landscapes, and its application could be implemented in other regions, without considerable effort. We recommend that the regulation of ecosystem services, including the avoidance of extreme heat, should play a greater role in landscape planning and management.

Author contributions

PLI, CG and JB designed the research, CG performed the analysis and visualization, CG and PLI wrote the original draft, DS provided comprehensive support in the assessment and interpretation of the results, and all authors contributed to the interpretation of the results and the subsequent revisions of the paper.

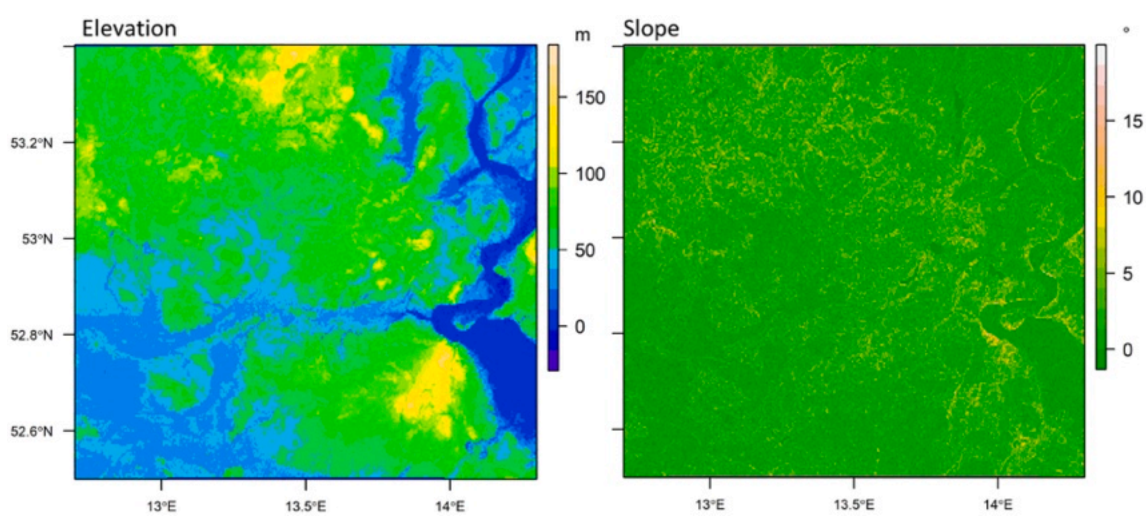
Declaration of Competing Interest

None.

Acknowledgements

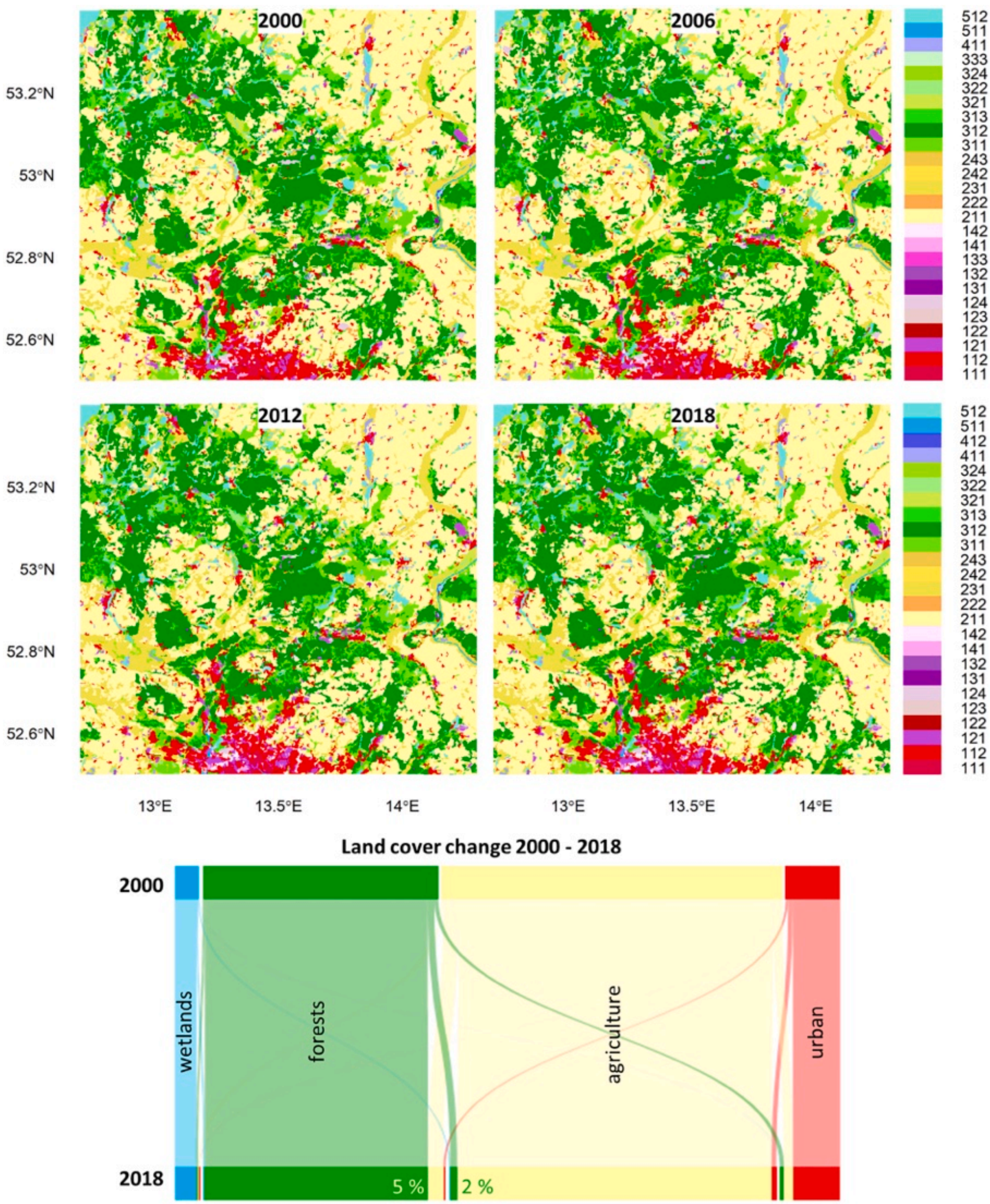
Funding for this study was provided by the 'BERNAU.Pro.Klima' project by the German Federal Ministry of the Environment, Nature Conservation and Nuclear Safety, grant number 03DAS125. The study was partly funded through the "Gläserner Forstbetrieb" project by the German Federal Ministry for Education and Research (BMBF) via the VDI/VDE, grant number 16LC1603C. PLI conceived and supervised the study within the framework of his long-term research program facilitated by the research professorships "Biodiversity and natural resource management under global change" (2009–2015) and as "Ecosystem-based sustainable development" (since 2015) granted by Eberswalde University for Sustainable Development. CG was funded through the Biosphere Reserves Institute and the Innovation and Career Center "ProBio-LaB" by the Ministry of Science, Research and Culture of the federal state of Brandenburg. We thank Steffen Kriewald for performing the initial calculations and providing the first ideas for this analysis. We also wish to thank two anonymous reviewers for their valuable contributions.

Appendix 1



Elevation (range −10 to 168 m) and slope (range 0 to 1.8°) of the study area. More than 70% of areas with a forest cover of >50% lie within an elevation of 50–90 m and with slopes between 0 and 2°.

Appendix 2



Land cover maps for 2000, 2006, 2012 and 2018. The quantification of land cover change between 2000 and 2018 exhibits no significant changes per land cover type. 2% of forest cover in 2000 was replaced by agricultural land in 2018, whereas 5% of agricultural land in 2000 was replaced by forest cover in 2018. For scale, locations and land cover types, see Fig. 1.

Appendix 3

Coverage and temperature changes for the *hot day composite* in the study area for three models. Respective LST for shares with agricultural land replaced by forest cover by 1, 5 and 10%.

	Study area	Forests 100% share	Agricultural land 100% share	Forests and agricultural land	All land cover types without forest and agricultural land
Share of study area in %	100	6.6	13.9	20.5	79.5
Mean of hot-day LST in °C	**26.4**	24.3	27.8	**26.7**	26.1
1% model: forest for agricultural land	26.3 °C	7.6%	12.9%	26.5 °C	26.1 °C
5% model: forest for agricultural land	26.0 °C	11.6%	8.9%	25.8 °C	26.1 °C
10% model: forest for agricultural land	**25.5 °C**	16.6%	3.9%	**25.0 °C**	26.1 °C

References

Alkama, R., Cescatti, A., 2016. Biophysical climate impacts of recent changes in global forest cover. Science 351 (6273), 600–604. https://doi.org/10.1126/science.aac8083.

Avelar, David, Garrett, Pedro, Ulm, Florian, Hobson, Peter, Penha-Lopes, Gil, 2020. Ecological complexity effects on thermal signature of different Madeira Island ecosystems. Ecol. Complex. 43, 100837.

Bartesaghi-Koc, Carlos, Osmond, Paul, Peters, Alan, 2020. Quantifying the seasonal cooling capacity of 'green infrastructure types' (GITs): an approach to assess and mitigate surface urban Heat Island in Sydney, Australia. Landsc. Urban Plan. 203 (November), 103893. https://doi.org/10.1016/j.landurbplan.2020.103893.

Benayas, José M. Rey, Bullock, James M., Newton, Adrian C., 2008. Creating woodland islets to reconcile ecological restoration, conservation, and agricultural land use. Front. Ecol. Environ. 6 (6), 329–336. https://doi.org/10.1890/070057.

Blumröder, J.S., Burova, Natalya, Winter, Susanne, Goroncy, Agnieszka, Hobson, Peter R., Shegolev, Andrey, Dobrynin, Denis, et al., 2019a. Ecological effects of clearcutting practices in a boreal Forest (Arkhangelsk region, Russian Federation) both with and without FSC certification. Ecol. Indic. 106, 105461. https://doi.org/10.1016/j.ecolind.2019.105461.

Blumröder, J.S., Ibisch, P.L., Kriewald, S., 2019b. Hambacher Forst in Der Krise (II) Temperaturmessungen Zur Beurteilung Der Mikroklimatischen Situation Des Waldes Und Des Randbereichs. Greenpeace e.V, Hamburg.

Blumröder, J.S., Hoffmann, Monika T., Ilina, Olga, Winter, Susanne, Hobson, Peter R., Ibisch, Pierre L., 2020. Clearcuts and related secondary dieback undermine the ecological effectiveness of FSC certification in a boreal Forest. Ecol. Process. 9 (1), 10. https://doi.org/10.1186/s13717-020-0214-4.

Blumröder, Jeanette S., May, Felix, Härdtle, Werner, Ibisch, Pierre L., 2021. Forestry contributed to warming of forest ecosystems in northern Germany during the extreme summers of 2018 and 2019. Ecol. Solutions Evidence 2 (3), e12087. https://doi.org/10.1002/2688-8319.12087.

Bonan, G.B., 2008. Forests and climate change: Forcings, feedbacks, and the climate benefits of forests. Science 320 (5882), 1444–1449. https://doi.org/10.1126/science.1155121.

Breshears, David D., Adams, Henry D., Eamus, Derek, McDowell, Nate, Law, Darin J., Will, Rodney E., Williams, A. Park, Zou, Chris B., 2013. The critical amplifying role of increasing atmospheric moisture demand on tree mortality and associated regional die-off. Front. Plant Sci. 4 https://doi.org/10.3389/fpls.2013.00266.

Bright, Ryan M., Davin, Edouard, O'Halloran, Thomas, Pongratz, Julia, Zhao, Kaiguang, Cescatti, Alessandro, 2017. Local temperature response to land cover and management change driven by non-radiative processes. Nat. Clim. Chang. 7 (4), 296–302. https://doi.org/10.1038/nclimate3250.

Chen, Jie, Liu, Yujie, Pan, Tao, Ciais, Philippe, Ma, Ting, Liu, Yanhua, Yamazaki, Dai, Ge, Quansheng, Peñuelas, Josep, 2020. Global socioeconomic exposure of heat extremes under climate change. J. Clean. Prod. 277, 123275.

Copernicus, 2018. CLC 2018 - Copernicus Land Monitoring Service [Data Set]. https://land.copernicus.eu/pan-european/corine-land-cover/clc-2000.

Deng, Yuanhong, Wang, Shijie, Bai, Xiaoyong, Tian, Yichao, Wu, Luhua, Xiao, Jianyong, Chen, Fei, Qian, Qinghuan, 2018. Relationship among land surface temperature and LUCC, NDVI in typical karst area. Sci. Rep. 8 (1), 1–12.

Didan, K., 2015. MOD13A2 MODIS/Terra vegetation indices 16-day L3 global 1km SIN grid V006 [data set]. NASA EOSDIS Land Processes DAAC. https://doi.org/10.5067/MODIS/MOD13A2.006.

Duveiller, Gregory, Hooker, Josh, Cescatti, Alessandro, 2018. The mark of vegetation change on Earth's surface energy balance. Nat. Commun. 9 (1), 679. https://doi.org/10.1038/s41467-017-02810-8.

DWD, 2021. Wetter Und Klima - Deutscher Wetterdienst - German Climate Atlas. https://www.dwd.de/EN/ourservices/germanclimateatlas/climateatlas_node.html.

Ellison, David, Morris, Cindy E., Locatelli, Bruno, Sheil, Douglas, Cohen, Jane, Murdiyarso, Daniel, Gutierrez, Victoria, et al., 2017. Trees, forests and water: cool insights for a hot world. Glob. Environ. Chang. 43, 51–61. https://doi.org/10.1016/j.gloenvcha.2017.01.002.

Feizizadeh, Bakhtiar, Blaschke, Thomas, 2013. Examining urban Heat Island relations to land use and air pollution: multiple endmember spectral mixture analysis for thermal remote sensing. IEEE J. Selected Topics Appl. Earth Observations Remote Sensing 6 (3), 1749–1756. https://doi.org/10.1109/JSTARS.2013.2263425.

Fisher, Joshua B., Melton, Forrest, Middleton, Elizabeth, Hain, Christopher, Anderson, Martha, Allen, Richard, Matthew F. McCabe, et al., 2017. The future of evapotranspiration: global requirements for ecosystem functioning, carbon and climate feedbacks, agricultural management, and water resources. Water Resour. Res. 53 (4), 2618–2626. https://doi.org/10.1002/2016WR020175.

Frenne, De, Pieter, Florian Zellweger, Rodríguez-Sánchez, Francisco, Scheffers, Brett R., Hylander, Kristoffer, Luoto, Miska, Vellend, Mark, Verheyen, Kris, Lenoir, Jonathan, 2019. Global buffering of temperatures under forest canopies. Nature Ecol. Evolut. 3 (5), 744–749. https://doi.org/10.1038/s41559-019-0842-1.

Google, 2021. Introduction | Google Earth Engine API. Google Developers (blog). https://developers.google.com/earth-engine/.

Gorelick, Noel, Hancher, Matt, Dixon, Mike, Ilyushchenko, Simon, Thau, David, Moore, Rebecca, 2017. Google earth engine: planetary-scale geospatial analysis for everyone. Remote Sens. Environ. 202, 18–27. https://doi.org/10.1016/j.rse.2017.06.031.

Grossiord, Charlotte, Buckley, Thomas N., Cernusak, Lucas A., Novick, Kimberly A., Poulter, Benjamin, Siegwolf, Rolf T.W., Sperry, John S., McDowell, Nate G., 2020. Plant responses to rising vapor pressure deficit. New Phytol. 226 (6), 1550–1566. https://doi.org/10.1111/nph.16485.

Hamberg, L. Jonas, Fraser, Roydon A., Robinson, Derek T., Trant, Andrew J., Murphy, Stephen D., 2020. Surface temperature as an Indicator of plant species diversity and restoration in oak woodland. Ecol. Indic. 113, 106249.

Hatfield, Jerry L., Prueger, John H., 2015. Temperature extremes: effect on plant growth and development. Weather Climate Extremes 10, 4–10. https://doi.org/10.1016/j.wace.2015.08.001 (USDA Research and Programs on Extreme Events).

Hesslerová, Petra, Huryna, Hanna, Pokornỳ, Jan, Procházka, Jan, 2018. The effect of Forest disturbance on landscape temperature. Ecol. Eng. 120, 345–354.

Hesslerová, Petra, Pokorný, Jan, Huryna, Hanna, Harper, David, 2019. Wetlands and forests regulate climate via evapotranspiration. In: An, Shuqing, Verhoeven, Jos T.A. (Eds.), Wetlands: Ecosystem Services, Restoration and Wise Use, Ecological Studies. Springer International Publishing, Cham, pp. 63–93. https://doi.org/10.1007/978-3-030-14861-4_4.

Ibisch, P.L., Kriewald, S., Blumröder, J.S., 2019. Hambacher Forst in Der Krise Studie Zur Beurteilung Der Mikro- Und Mesoklimatischen Situation Sowie Randeffekten. Greenpeace e.V, Hamburg. https://www.greenpeace.de/sites/www.greenpeace.de/files/hambacher_forst_ii.pdf.

Ibisch, Pierre L., Kloiber, Judith, Hoffmann, Monika T. (Eds.), 2018. Barnim-Atlas. Lebensraum Im Wandel Eine Ökosystembasierte Betrachtung Des Barnims Zum Wohle Der Menschen. Ehm Welk, Schwedt.

Jarvis, A., Guevara, E., Reuter, H.I., Nelson, A.D., 2008. Hole-Filled SRTM for the Globe: Version 4. [Data Set]. http://srtm.csi.cgiar.org.

Jin, Menglin, Dickinson, Robert E., 2010. Land surface skin temperature climatology: benefitting from the strengths of satellite observations. Environ. Res. Lett. 5 (4), 044004 https://doi.org/10.1088/1748-9326/5/4/044004.

Kupika, Olga Laiza, Gandiwa, Edson, Nhamo, Godwell, Kativu, Shakkie, 2019. Local ecological knowledge on climate change and ecosystem-based adaptation strategies promote resilience in the middle Zambezi biosphere reserve, Zimbabwe. Scientifica 2019 (March), e3069254. https://doi.org/10.1155/2019/3069254.

Laurance, William F., Laurance, Susan G., Delamonica, Patricia, 1998. Tropical forest fragmentation and greenhouse gas emissions. For. Ecol. Manag. 110 (1), 173–180. https://doi.org/10.1016/S0378-1127(98)00291-6.

Li, Yan, Zhao, Maosheng, Motesharrei, Safa, Qiaozhen, Mu, Kalnay, Eugenia, Li, Shuangcheng, 2015. Local cooling and warming effects of forests based on satellite observations. Nat. Commun. 6 (1), 6603. https://doi.org/10.1038/ncomms7603.

Liu, Xiaohui, Zhang, Yuan, Dong, Guihua, Jiang, Ming, 2019. Difference in carbon budget from marshlands to transformed Paddy fields in the Sanjiang plain, Northeast China. Ecol. Eng. 137 (October), 60–64. https://doi.org/10.1016/j.ecoleng.2018.03.013.

Liu, Yanxu, Peng, Jian, Wang, Yanglin, 2018. Efficiency of landscape metrics characterizing urban land surface temperature. Landsc. Urban Plan. 180 (December), 36–53. https://doi.org/10.1016/j.landurbplan.2018.08.006.

Luber, George, McGeehin, Michael, 2008. Climate change and extreme heat events. Am. J. Prev. Med. 35 (5), 429–435. https://doi.org/10.1016/j.amepre.2008.08.021.

Lusiana, Betha, Kuyah, Shem, Öborn, Ingrid, van Noordwijk, Meine, 2017. Typology and metrics of ecosystem services and functions as the basis for payments, rewards and

co-investment. In: Coinvestment in Ecosystem Services: Global Lessons from Payment and Incentive Schemes. World Agroforestry Centre (ICRAF), Nairobi.
Maes, W.H., Pashuysen, Tom, Trabucco, Antonio, Veroustraete, Frank, Muys, Bart, 2011. Does energy dissipation increase with ecosystem succession? Testing the ecosystem exergy theory combining theoretical simulations and thermal remote sensing observations. Ecol. Model. 222 (23–24), 3917–3941.
Mildrexler, David J., Zhao, Maosheng, Running, Steven W., 2011. A global comparison between station air temperatures and MODIS land surface temperatures reveals the cooling role of forests. J. Geophys. Res. Biogeosci. 116 (G3) https://doi.org/10.1029/2010JG001486.
Mora, Camilo, Dousset, Bénédicte, Caldwell, Iain R., Powell, Farrah E., Geronimo, Rollan C., Bielecki, Coral R., Chelsie W. W. Counsell, et al., 2017. Global risk of deadly heat. Nat. Clim. Chang. 7 (7), 501–506. https://doi.org/10.1038/nclimate3322.
Nanfuka, Susan, Mfitumukiza, David, Egeru, Anthony, 2020. Characterisation of ecosystem-based adaptations to drought in the central cattle corridor of Uganda. African J. Range Forage Sci. 37 (4), 257–267. https://doi.org/10.2989/10220119.2020.1748713.
Norris, Catherine, Hobson, Peter, Ibisch, Pierre L., 2012. Microclimate and vegetation function as indicators of forest thermodynamic efficiency. J. Appl. Ecol. 49 (3), 562–570. https://doi.org/10.1111/j.1365-2664.2011.02084.x.
Novick, Kimberly A., Katul, Gabriel G., 2020. The duality of reforestation impacts on surface and air temperature. J. Geophys. Res. Biogeosci. 125 (4) https://doi.org/10.1029/2019JG005543.
Panagos, Panos, 2006. The European soil database. GEO: Connexion 5 (7), 32–33.
Parks, Peter J., Hardie, Ian W., 2018. Least-cost Forest carbon reserves: Cost-effective subsidies to convert marginal agricultural land to forests. In: Economics and Forestry, 1st ed. Routledge, p. 16. https://doi.org/10.4324/9781315182681-24.
Peng, Shu-Shi, Piao, Shilong, Zeng, Zhenzhong, Ciais, Philippe, Zhou, Liming, Li, Laurent Z.X., Myneni, Ranga B., Yin, Yi, Zeng, Hui, 2014. Afforestation in China cools local land surface temperature. Proc. Natl. Acad. Sci. 111 (8), 2915–2919. https://doi.org/10.1073/pnas.1315126111.
Pokorný, Jan, Hesslerová, Petra, Huryna, Hanna, Harper, David, 2016. Indirect and direct thermodynamic effects of wetland ecosystems on climate. In: Vymazal, Jan (Ed.), Natural and Constructed Wetlands: Nutrients, Heavy Metals and Energy Cycling, and Flow. Springer International Publishing, Cham, pp. 91–108. https://doi.org/10.1007/978-3-319-38927-1_7.
Pugh, Thomas A.M., Rademacher, Tim, Shafer, Sarah L., Steinkamp, Jörg, Barichivich, Jonathan, Beckage, Brian, Haverd, Vanessa, et al., 2020. Understanding the uncertainty in global Forest carbon turnover. Biogeosciences 17 (15), 3961–3989. https://doi.org/10.5194/bg-17-3961-2020.
R Core Team, 2021. R: The R Project for Statistical Computing. https://www.r-project.org/.
Ramsar Convention Secretariat, 2018. Global wetland outlook: State of the World's wetlands and their services to people. In: SSRN Scholarly Paper ID 3261606. Social Science Research Network, Rochester, NY. https://papers.ssrn.com/abstract=3261606.
Schneider, Eric D., Kay, James J., 1994. Complexity and thermodynamics: towards a new ecology. Futures 26 (6), 626–647. https://doi.org/10.1016/0016-3287(94)90034-5 (Special Issue Complexity: Fad or Future?).
Schumacher, Paul, Garstecki, Tobias, Mislimshoeva, Bunafsha, Morrison, John, Ibele, Benedikt, Lesk, Corey, Dzhumabaeva, Salamat, Bulbulshoev, Umed, Martin, Shaun, 2018. Using the open standards-based framework for planning and implementing ecosystem-based adaptation projects in the high mountainous regions of Central Asia. In: Alves, Fátima, Filho, Walter Leal, Azeiteiro, Ulisses (Eds.), Theory and Practice of Climate Adaptation, Climate Change Management. Springer International Publishing, Cham, pp. 23–48. https://doi.org/10.1007/978-3-319-72874-2_2.
Schwaab, Jonas, Davin, Edouard L., Bebi, Peter, Duguay-Tetzlaff, Anke, Waser, Lars T., Haeni, Matthias, Meier, Ronny, 2020. Increasing the broad-leaved tree fraction in European forests mitigates hot temperature extremes. Sci. Rep. 10 (1), 14153. https://doi.org/10.1038/s41598-020-71055-1.
Sheil, Douglas, 2018. Forests, atmospheric water and an uncertain future: the new biology of the global water cycle. Forest Ecosystems 5 (1), 1–22.
Shen, Xiangjin, Liu, Binhui, Jiang, Ming, Lu, Xianguo, 2020. Marshland loss warms local land surface temperature in China. Geophys. Res. Lett. 47 (6), e2020GL087648 https://doi.org/10.1029/2020GL087648.
Shen, Xiangjin, Jiang, Ming, Lu, Xianguo, Liu, Xingtu, Liu, Bo, Zhang, Jiaqi, Wang, Xianwei, et al., 2021. Aboveground biomass and its spatial distribution pattern of herbaceous marsh vegetation in China. Sci. China Earth Sci. 64 (7), 1115–1125. https://doi.org/10.1007/s11430-020-9778-7.
Su, Weizhong, Gu, Chaolin, Yang, Guishan, 2010. Assessing the impact of land use/land cover on urban heat island pattern in Nanjing City, China. J. Urban Plan. Dev. 136 (December), 365–372. https://doi.org/10.1061/(ASCE)UP.1943-5444.0000033.
Suggitt, Andrew J., Gillingham, Phillipa K., Hill, Jane K., Huntley, Brian, Kunin, William E., Roy, David B., Thomas, Chris D., 2011. Habitat microclimates drive fine-scale variation in extreme temperatures. Oikos 120 (1), 1–8. https://doi.org/10.1111/j.1600-0706.2010.18270.x.
Teuling, Adriaan J., Seneviratne, Sonia I., Stöckli, Reto, Reichstein, Markus, Moors, Eddy, Ciais, Philippe, Luyssaert, Sebastiaan, et al., 2010. Contrasting response of European forest and grassland energy exchange to heatwaves. Nat. Geosci. 3 (10), 722–727. https://doi.org/10.1038/ngeo950.
Teuling, Adriaan J., Van Loon, Anne F., Seneviratne, Sonia I., Lehner, Irene, Aubinet, Marc, Heinesch, Bernard, Bernhofer, Christian, Grünwald, Thomas, Prasse, Heiko, Spank, Uwe, 2013. Evapotranspiration amplifies European summer drought. Geophys. Res. Lett. 40 (10), 2071–2075. https://doi.org/10.1002/grl.50495.
Tran, Duy X., Pla, Filiberto, Latorre-Carmona, Pedro, Myint, Soe W., Caetano, Mario, Kieu, Hoan V., 2017. Characterizing the relationship between land use land cover change and land surface temperature. ISPRS J. Photogramm. Remote Sens. 124 (February), 119–132. https://doi.org/10.1016/j.isprsjprs.2017.01.001.
Van Liedekerke, Marc, Jones, Arwyn, Panagos, Panos, 2006. ESDBv2 Raster Library - a Set of Rasters Derived from the European Soil Database Distribution v2.0. European Commission; European Soil Bureau Network.
Vicedo-Cabrera, A.M., Scovronick, N., Sera, F., Royé, D., Schneider, R., Tobias, A., Astrom, C., et al., 2021. The burden of heat-related mortality attributable to recent human-induced climate change. Nat. Clim. Chang. 11 (6), 492–500. https://doi.org/10.1038/s41558-021-01058-x.
Wan, Z., Hook, S., Hulley, G., 2015. Collection-6MODIS land surface temperature ProductsUsers' guide. NASA EOSDIS Land Processes DAAC. https://doi.org/10.5067/MODIS/MYD11A1.006.
Weng, Qihao, Lu, Dengsheng, Schubring, Jacquelyn, 2004. Estimation of land surface temperature–vegetation abundance relationship for urban Heat Island studies. Remote Sens. Environ. 89 (4), 467–483. https://doi.org/10.1016/j.rse.2003.11.005.
Williams, A. Park, Allen, Craig D., Macalady, Alison K., Griffin, Daniel, Woodhouse, Connie A., Meko, David M., Thomas W. Swetnam, et al., 2013. Temperature as a potent driver of regional forest drought stress and tree mortality. Nat. Clim. Chang. 3 (3), 292–297. https://doi.org/10.1038/nclimate1693.
Wu, Yuxuan, Xi, Yi, Feng, Maoyuan, Peng, Shushi, 2021. Wetlands cool land surface temperature in tropical regions but warm in boreal regions. Remote Sens. 13 (8), 1439. https://doi.org/10.3390/rs13081439.
Wu, Zhijie, Zhang, Yixin, 2019. Water Bodies' cooling effects on urban land daytime surface temperature: ecosystem service reducing Heat Island effect. Sustainability 11 (3), 787. https://doi.org/10.3390/su11030787.
Xiao, Honglin, Weng, Qihao, 2007. The impact of land use and land cover changes on land surface temperature in a karst area of China. J. Environ. Manag. 85 (1), 245–257. https://doi.org/10.1016/j.jenvman.2006.07.016.
Yuan, Fei, Bauer, Marvin E., 2007. Comparison of impervious surface area and normalized difference vegetation index as indicators of surface urban Heat Island effects in Landsat imagery. Remote Sens. Environ. 106 (3), 375–386. https://doi.org/10.1016/j.rse.2006.09.003.
Zaitchik, Benjamin F., Macalady, Alison K., Bonneau, Laurent R., Smith, Ronald B., 2006. Europe's 2003 heat wave: a satellite view of impacts and land–atmosphere feedbacks. Int. J. Climatol. 26 (6), 743–769. https://doi.org/10.1002/joc.1280.
Zellweger, Florian, Coomes, David, Lenoir, Jonathan, Depauw, Leen, Maes, Sybryn L., Wulf, Monika, Keith J. Kirby, et al., 2019. Seasonal drivers of Understorey temperature buffering in temperate deciduous forests across Europe. Glob. Ecol. Biogeogr. 28 (12), 1774–1786. https://doi.org/10.1111/geb.12991.
Zeng, Zhenzhong, Piao, Shilong, Laurent Z. X. Li, Liming Zhou, Philippe Ciais, Tao Wang, Yue Li, et al., 2017. Climate mitigation from vegetation biophysical feedbacks during the past three decades. Nat. Clim. Chang. 7 (6), 432–436. https://doi.org/10.1038/nclimate3299.

2.3. Article III

Does fragmentation contribute to the forest crisis in Germany?

frontiers | Frontiers in Forests and Global Change

TYPE Original Research
PUBLISHED 20 January 2023
DOI 10.3389/ffgc.2023.1099460

OPEN ACCESS

EDITED BY
William R. Moomaw,
Tufts University, United States

REVIEWED BY
Heather Keith,
Griffith University, Australia
David Joseph Mildrexler,
Eastern Oregon Legacy Lands, United States

*CORRESPONDENCE
Pierre L. Ibisch
✉ Pierre.Ibisch@hnee.de

†These authors have contributed equally to this work and share first authorship

SPECIALTY SECTION
This article was submitted to Forest Management, a section of the journal Frontiers in Forests and Global Change

RECEIVED 15 November 2022
ACCEPTED 02 January 2023
PUBLISHED 20 January 2023

CITATION
Mann D, Gohr C, Blumröder JS and Ibisch PL (2023) Does fragmentation contribute to the forest crisis in Germany? *Front. For. Glob. Change* 6:1099460. doi: 10.3389/ffgc.2023.1099460

Does fragmentation contribute to the forest crisis in Germany?

Deepika Mann[1†], Charlotte Gohr[1,2†], Jeanette S. Blumröder[1] and Pierre L. Ibisch[1,2*]

[1]Centre for Econics and Ecosystem Management, Eberswalde University for Sustainable Development, Eberswalde, Germany, [2]Biosphere Reserves Institute, Eberswalde University for Sustainable Development, Eberswalde, Germany

Intact forests contribute to the ecosystem functionality of landscapes by storing and sequestering carbon, buffering and cooling the microclimate, and providing a range of related ecosystem functions. Forest fragmentation not only poses a threat to many organisms but also reduces the resistance and resilience of the ecosystem, which is especially relevant to the ongoing climate crisis. The effects of recent extreme heat years on forests in Germany have not been studied in detail for the influence of fragmentation. We investigate the relation of forest fragmentation with temperature and vitality in Germany per ecoregion at the canopy level using satellite imagery at 1-km and 30-m resolution. We compiled and correlated forest maps for connectivity based on Thiessen polygons, canopy temperatures on the hottest days based on land surface temperature, and forest vitality based on the maximum normalized difference vegetation index per growing season. We differentiated between ecoregions and main forest types. In 2022, larger intact tree-covered areas that are less fragmented have relatively low temperatures on hot days and higher overall vitality. Nearly 98% of the almost 1.95 million forest fragments at 30-m resolution in Germany are smaller than 1 km^2, which cover nearly 30% of the total forest area. To counteract the forest crisis, forest and landscape management should aim to reduce fragmentation and maintain tree biomass and forest cover in the landscape. Increasing the size of continuous forest fragments contributes to ecosystem-based adaptation to climate change.

KEYWORDS

land surface temperature (LST), normalized difference vegetation index (NDVI), Thiessen connectivity, forest fragmentation, forest cover

1. Introduction

Land use and infrastructure increasingly fragment naturally continuous ecosystems into several smaller fragments more or less isolated from each other (Riitters, 2007; Ibisch et al., 2016). While large unfragmented forest landscapes are considered some of the most vital ecosystems in the world that provide crucial benefits to numerous species (Minnemeyer and Potapov, 2017), fragmentation is a key driver for the loss of ecosystem integrity (Rogers et al., 2022). Large intact forests are the greatest sinks of atmospheric carbon and store disproportionately higher amounts of carbon than fragmented forests, making them an important natural solution in any climate change mitigation and adaptation solution (Potapov et al., 2017; Moomaw et al., 2019). In addition, risks of diminished biodiversity and local extinctions are higher with increasing fragmentation of intact forest landscapes (Betts et al., 2017). Forest fragmentation results in the expansion of forest edges, exposing the forest fragments to higher anthropogenic disturbances (Vieilledent et al., 2018). Higher fragmented and more isolated forest fragments tend to advance changes in local climatic conditions, leading to drier, hotter, and increasingly volatile microclimate (Laurance et al., 2002; Briant et al., 2010; Tuff et al., 2016). Nevertheless, small forest fragments and green canopy cover adjacent to the highly modified anthropogenic landscape also provide substantial benefits for regulating the microclimate (Aalto et al., 2022). However, such effects are more pronounced in large intact forests (Gohr et al., 2021).

The long-term history of forest fragmentation in Central Europe has led to a mosaicked landscape that consists of agricultural lands with scattered fragments of temperate forests. In Central Europe, nearly 40% of the current forested area is located closer than 100-m from the forest edge including the largest continuous mountain forests (Estreguil et al., 2013). Penetration of drought stress and wind into a forest can be measured until several hundred meters from the forest edge, leading to alterations in the forest microclimate and an increase in tree mortality (Laurance et al., 2011). Forest microclimatic changes have been reported at tens to hundreds of meters from the forest edge (Harper et al., 2005; Tuff et al., 2016). This could possibly be one reason for the increased damage due to forest fires every year (Armenteras et al., 2013; Driscoll et al., 2021). The carbon loss that is associated with the edge effects caused by forest fragmentation is another recently recognized factor associated with fragmentation (Silva Junior et al., 2020).

This study refers to all the forest patches as forest fragments irrespective of their size. Smaller forest fragments are largely influenced by the effects of their surrounding edges, and only larger forest fragments with a substantial proportion of interior area can buffer from environmental and biotic changes associated with the edge. The forest edge effect is the outcome of many interacting environmental effects. There are both physio-chemical and biotic impacts from the surroundings on the forest ecosystem. Microclimatic and mesoclimatic impacts are of special relevance for forest vitality as heat and drought stresses have increased over the past decades and are expected to rise substantially in the near future (Jacob et al., 2018). Extreme heat events are more likely to occur with ongoing climate change and contribute to water stress and drought, especially for forest ecosystems (Fisher et al., 2017). In Germany, we speak of a forest crisis based on the severe forest damage in recent years due to droughts, heat waves, pests (especially the bark beetle outbreak in 2018), and mismanagement (Lindner et al., 2014; Schuldt et al., 2020; Blumröder et al., 2021; Ibisch, 2022; Thonfeld et al., 2022). Healthy forests can contribute to landscape cooling, especially on hot days and during heat waves (Gohr et al., 2021).

Although there is a wealth of knowledge on how fragmentation affects temperature and forest vitality in tropical forests (Taubert et al., 2018; Silva Junior et al., 2020), very little research has linked the patterns of fragmentation with temperature and forest vitality altogether in temperate forests, such as forests in Germany. The characteristic features of temperate and tropical forests are inherently different; hence, it is important to understand these linkages specific to temperate forests to develop region or biome-specific forest management strategies. Therefore, it is imperative to understand these effects in temperate forests also. This study aims to expand the current knowledge about the relationship between forest fragments, their sizes, and the associated variations in temperature and forest vitality in Germany. In particular, we addressed the following questions:

(1) Is forest fragmentation associated with spatial variations in landscape temperature and forest vitality?
(2) Are the thermal gradients and variations in forest vitality inside forest fragments influenced by the size and degree of isolation of the forest fragments?

The information obtained in this study provides input to both forest management and landscape planning striving for ecosystem resilience and an ecosystem-based adaptation to climate change. Existing monitoring apps such as the European Forest Condition Monitor (Buras et al., 2021) or the Waldmonitor (Welle et al., 2022) focus on vegetation vitality. Extending this monitoring with analyses of forest fragmentation can contribute to the understanding of forest vulnerability. This study provides evidence-based arguments for reducing forest fragmentation in intensively managed landscapes.

2. Materials and methods

We quantified the relation of forest fragmentation with land surface temperature and forest vitality in Germany per ecoregion at the canopy level using a four-step methodological approach: (i) compilation of annual national forest cover for the year 2022 as well as temperature and vegetation vitality data; (ii) preliminary treatment of all datasets, including standardization of spatial resolution and calculation of annual averages of the hottest days and the maximum vitality within the growing season; (iii) conversion of the forest dataset into a presence–absence forest cover map and calculation of the degree of fragmentation; and (iv) analysis of the relation between forest fragmentation, temperature, and forest vitality using zonal statistics and bivariate choropleth mapping per ecoregion and per forest type. Dataset preprocessing was performed in the code editor of the Google Earth Engine. Post-processing and visualizations were prepared with ArcGIS 10.7 and RStudio version 4.0.3.

2.1. Ecoregions of Germany

To account for the influence of regional characteristics, such as altitude and plant communities, we based our analysis on nine defined ecoregions in Germany (Olson et al., 2001; Bundesamt für Naturschutz [BfN], 2017; **Figure 1**). The justification of ecoregions as a conglomerate of similar geographical and biological characteristics in terms of the assemblage of species is considered proven (Smith J. R. et al., 2018). The influence of altitudinal effects is represented in the segmentation of the ecoregions in Germany. Hence, we consider the investigation of fragmentation patterns within the ecoregion borders appropriate. Since the two northernmost ecoregions in Germany, the Baltic Sea ecoregion and the North Sea ecoregion, exhibit little forest cover, they were excluded from the analysis.

2.2. Forest cover and forest types

We created a forest cover map of Germany at a 30-m resolution for 2022 from the Global Tree Cover 2000 dataset, using a canopy cover threshold of 50%, by subtracting the tree cover loss from 2001 to 2021 (Hansen et al., 2013). We did not account for forest cover gain since no reliable data are available to date. The dataset is derived from the Landsat 7ETM + data, and forest cover is characterized as any vegetation taller than 5 m in height (Hansen et al., 2013).

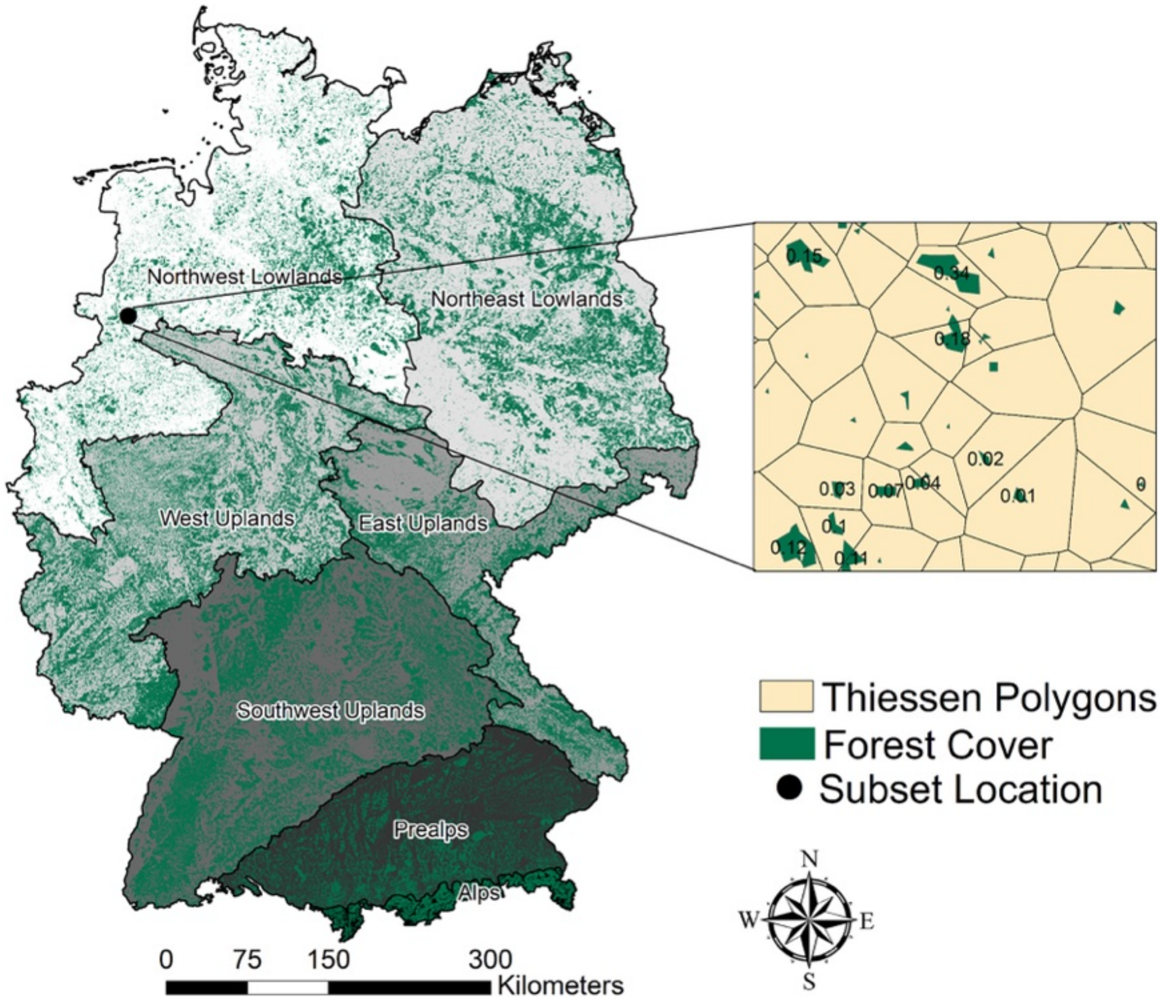


FIGURE 1
Map of Germany categorized by seven ecoregions (left, Bundesamt für Naturschutz [BfN], 2017). The Baltic Sea and North Sea with low forest cover shares are excluded from the original dataset. Forest cover (green) is based on the Global Forest Change dataset (Hansen et al., 2013) with forest areas of 2021 (defined per pixel as more than 50% forest coverage based on Hansen et al., 2013). An example region of a forest-covered area with corresponding Thiessen polygons and corresponding connectivity values in the range of 0–1 is shown on the right.

This threshold is based on the ability to distinguish tall woody vegetation in multispectral imagery, particularly those present in global-scale earth observation systems such as Landsat and MODIS (Hansen et al., 2010). Despite being criticized for not differentiating native and planted forests, these high-resolution maps are suitable for capturing biophysical features that depend on forest cover across the globe (Tropek et al., 2014). The resulting forest coverage of 27.9% for Germany in 2022 (**Supplementary Table 1**) differs from the official statistics. The official statistics, with a forest coverage for 2021 of 31.9% (Bundesministerium für Ernährung und Landwirtschaft [BMBL], 2022), include "legal forest" areas without actual tree cover and do not include some woodlands or tree coverage in urban areas. In comparison, in this study, only forest with measurable tree cover is considered and the relatively small forest patches on small islands and at the coastline of the Baltic Sea and the North Sea ecoregions are omitted, which leads to the smaller area of total forest coverage.

We created a forest type map of Germany at 30-m resolution from the latest available CORINE Land Cover product of 2018 at 100-m by first reclassifying the dataset into three forest type classes, namely, broad-leaved, coniferous, and other forests, and secondly reprojecting, resampling (to 30-m), and masking the dataset to the compiled forest cover map at 30-m resolution.

2.3. Forest fragmentation

Thiessen polygon connectivity (refer to, e.g., Ibisch et al., 2016; Mehdipour et al., 2019; Wu et al., 2019) of the forest fragments was used as a proxy to estimate forest fragmentation. The measure combines both fragment size and isolatedness from other forest fragments and is a unitless value ranging between 0 (high fragmentation) and 1 (low fragmentation). It is defined as the ratio between the size of a forest fragment and its surrounding Thiessen polygon. A Thiessen (or Voronoi) polygon describes the area around a sample point/area where any position taken from inside the polygon is closer to the sample point/area than to any of the other sample points/areas. The greater the value of Thiessen connectivity, the closer the neighboring forest fragments are and hence the lower the fragmentation.

10.3389/ffgc.2023.1099460

The forest cover raster dataset of Germany for 2022 was converted to forest polygons using the centroid method, and one point per forest polygon was generated. Thereafter, Thiessen polygons were created, and Thiessen polygon connectivity values were computed per forest polygon (refer to **Figure 1**). Using these connectivity values based on the forest cover of Germany for 2022, we generated a forest fragmentation map at 30-m resolution.

2.4. Forest temperature

Extreme heat events are more likely to occur with ongoing climate change and contribute to water stress and drought, especially for forest ecosystems (Fisher et al., 2017). In the temperate biome, healthy forests can contribute to landscape cooling (Gohr et al., 2021). At the same time, the forests are heavily impacted by natural and anthropogenic disturbances. The radiative skin temperature of the land surface is the driving force in the exchange of long-wave radiation and turbulent heat fluxes at the surface–atmosphere interface (Li et al., 2013) and has presented valuable results in several sensible heat flux models (Zhan et al., 1996). Land surface temperature closely resembles air temperature trends when analyzing the effects of forest cover on local temperature even at different latitudinal zones (Li et al., 2016) as even with the heat effect of land surface during the day, there is a heat exchange between air and land surfaces (Jin and Dickinson, 2010; Mildrexler et al., 2011). Therefore, the land surface temperature dataset of Germany qualifies for the assessments of forest canopy temperature and is based on the MODIS Aqua MYD11A1 dataset at 1-km spatial resolution, captured every day at ∼1.30 pm Central European Time (CET). The temperature is measured from the radiation intensity in the infrared range (bands 31 and 32 with 10.8–12.3 μm) (Wan et al., 2015). To create a dataset with the per pixel mean temperature of the hottest days in 2022 from January until October, each day was signatured with the mean temperature of Germany and subsequently the resulting 124 days that exceed 30°C were selected (**Supplementary Table 1**, refer to Gohr et al., 2021). This way, we generated a map of the mean temperature on the hottest days in German forests for 2022, the warmest year since records began (Deutscher Wetterdienst [DWD], 2022; together with 2019, the summer of 2022 was the third warmest summer since 1881; Imbery et al., 2022). The same procedure was applied to generate means of the temperature on the hottest days for the years 2013–2022 in respect to ecoregions (**Supplementary Figure 1**).

2.5. Forest vitality

The normalized difference vegetation index (NDVI) is a measure of photosynthetic activity and is commonly used as a proxy for vegetation stress and water balance and therefore indicates vegetation vitality (Lambert et al., 2013; Chakraborty et al., 2022). Furthermore, the NDVI can serve as an explanatory variable for the effects of temperature changes on forest cover (Weng et al., 2004; Deng et al., 2018). The NDVI dataset of Germany is based on the MODIS Aqua MYD13A2 dataset of 16-day composites at 1-km spatial resolution. The composite is preprocessed from MODIS imagery using the near-infrared and visible spectra, is captured every day at ∼1.30 pm CET, and is completed by selecting the best pixel with low clouds, a low view angle, and the highest NDVI (Didan, 2015). For the growing season from May to September, this results in 10 images covering Germany in 2022. We created a map of forest vitality for Germany in 2022 using the maximum value of the growing season. The maximum value per pixel was selected to acknowledge different peaks of "greenness" for different vegetation for the tree-covered areas in 2022. The minimum value was not considered since the influence of fragmentation on healthy forests was the main objective. Minimum values have a diverse range of potential reasons such as outbreaks of pests and diseases, water stress, or other environmental factors. If only the maximum values are taken into account, the loss of vitality may be somewhat underestimated, but this error does not affect the regional assessment of spatial patterns (or temporal changes) substantially. The same procedure was applied to generate means of the greenest NDVI for the years 2013–2022 with respect to forest types and ecoregions (**Supplementary Figures 2**, **3**).

2.6. Analysis

We reclassified the forest fragments into three different categories based on fragment size, that is, small (below 1 km^2), medium (between 1 and 5 km^2), and large (larger than 5 km^2). We resampled the preprocessed temperature and NDVI datasets to 30-m spatial resolution. We extracted the raster datasets using the forest cover mask to prepare temperature, NDVI, Thiessen connectivity, and forest type data for forest cover in Germany. In order to study the spatial relationship between fragmentation, temperature, and forest vitality across different ecoregions in Germany, we prepared bivariate choropleth maps that spatially represent the variation in one variable with respect to another. In addition, we computed statistical information per forest fragment size per ecoregion for Thiessen connectivity, temperature, and vitality, respectively, using the zonal statistics tool in ArcMap 10.7. For a time series of changes in maximum vitality per growing season and changes in the mean temperature on the hottest days, we used a similar approach. For each year from 2013 to 2022 and each ecoregion, we extracted the mean and standard deviation of the temperature and vitality dataset in the corresponding forested areas (by subtracting forest loss of previous years).

3. Results

Conditions regarding fragmentation, temperature, and vitality in the tree-covered areas vary widely across Germany. Larger intact tree-covered areas that are less fragmented (e.g., parts of the Black Forest in the Southwest, **Figure 2A**) exhibit relatively low temperatures on hot days (**Figure 2B**) and higher vitality (**Figure 2C**). Populated areas with low forest coverage and higher fragmentation (e.g., parts of Northwest Germany) feature higher temperatures on hot days and lower vitality in the often very small forest fragments.

3.1. Fragmentation, temperature, and vitality

The current forest distribution in Germany is the result of a long history of anthropogenic land use (Kaplan et al., 2009). The current forest cover in Germany accounts for approximately 9.9 million

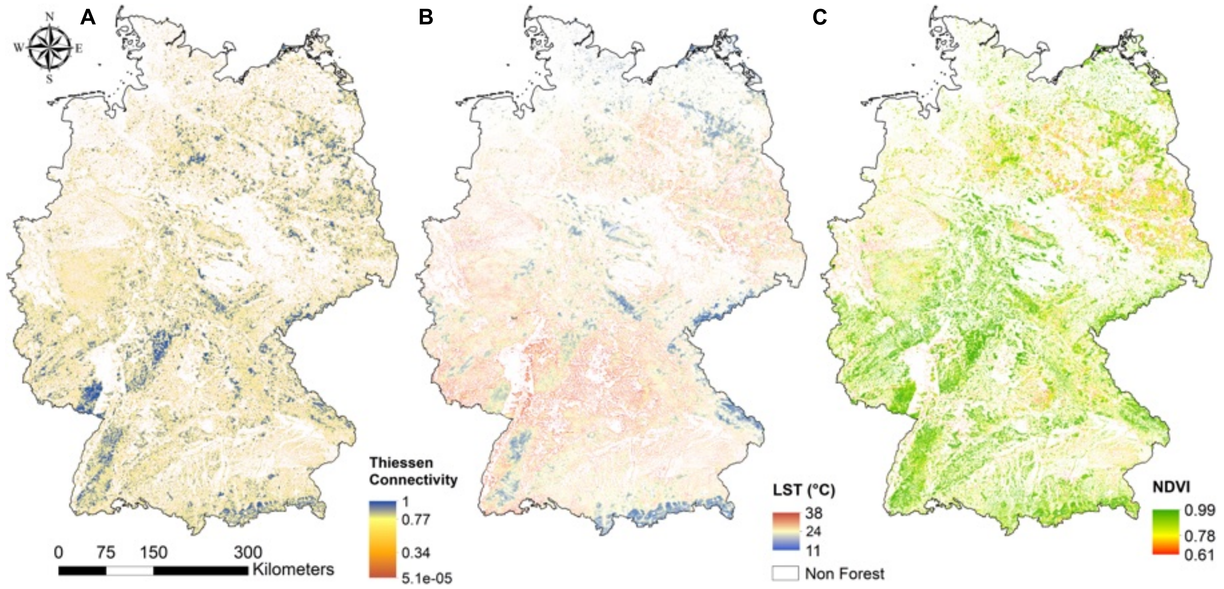


FIGURE 2
Characteristics of tree-covered areas in Germany 2022 across the terrestrial ecoregions in terms of **(A)** forest fragmentation (Thiessen connectivity), **(B)** land surface temperature [land surface temperature (LST); MODIS Aqua MYD11A1] indicated as the per pixel mean of hot days (≥30°C) in 2022, and **(C)** forest vitality (NDVI; MODIS Aqua MYD13A2) indicated as the maximum value of the growing season May–September.

hectares with our study approach [official statistics state 11.4 million hectares (Bundesministerium für Ernährung und Landwirtschaft [BMBL], 2022)] and, at a 30-m resolution, is fragmented into nearly 1.95 million small forest fragments out of which 1.92 million are smaller than 1 km^2 and only around 2,000 forest fragments are larger than 5 km^2 with a maximum size of 3,800 km^2.

With respect to distribution within different ecoregions, 68% (Alps) to 16% (Northwest Lowlands) of Germany is covered by forest (**Supplementary Table 1**). In terms of total area covered by forests, the West Uplands have the highest forest share of 21,960 km^2. A higher value of Thiessen connectivity (closer to 1) indicates higher connectedness of forest fragments and hence less fragmentation. We analyzed the median of Thiessen connectivity values over all the forest fragments in different size classes, and the results indicate that large forest fragments have higher values of Thiessen connectivity in all the ecoregions ranging from 0.75 to 0.87 (**Supplementary Table 2**), while this range is only 0.58–0.70 in medium-sized forest fragments and 0.31–0.51 in the small-sized forest fragments. When comparing Thiessen connectivity across different ecoregions, forest fragments in all the size classes have the highest values in the Alps. Thiessen connectivity across different forest sizes in all the ecoregions is higher in the large-sized forest fragments than small-sized forests by a magnitude of 0.35–0.49, while this value for large- to medium- and medium- to small-sized fragment comparison lies in the range of 0.13–0.27. The median of Thiessen connectivity values across the ecoregions is high when grouping the values per fragment size.

The 2022 hottest day median temperatures of forests with large fragments were lower compared to small fragments in all the ecoregions by 1.29°C in the Alps to as large as 3.28°C in the Southwest Uplands (**Supplementary Table 2**). When comparing medium-sized forest fragments with large fragments within these two ecoregions, this difference is between 0.8 and 2.46°C. When comparing small- and medium-sized forest fragments, the difference in temperature ranges between 0.46 and 0.82°C. The highest variation in temperatures between the different-sized forest fragments was found in the Southwest Uplands. The effect of decreasing temperatures with higher elevation is depicted by the ecoregions since the elevation is one classification variable of ecoregions. This pattern was validated on a pixel basis throughout Germany (refer to **Supplementary Figure 4**). The total of hot days in 2022 varied per ecoregion. In the Alps, 24 days above 30°C were registered, in comparison with the Southwest Highlands 107 days above 30°C in 2022 (full list refer to **Supplementary Table 1**).

The median vitality values, compiled as medians of the maximum NDVI of the growing season in 2022 in Germany, are generally lower in smaller forest fragments and higher in larger forest fragments (**Figure 3** and **Supplementary Table 2**). Since the areas are not discriminated by forest type (**Supplementary Figures 3**, **5**, **6**), a more generalized pattern can be observed. The highest vitality is found in the largest fragments of the Alps, the Western Uplands, and the Southwest Uplands. Vitality values below 0.8 are only observed in the smallest fragments of the Southwest Uplands, and the Northeast and Northwest Lowlands.

Small forest fragments correspond to low connectivity, low vitality, and higher temperatures throughout all ecoregions and their respective altitudinal ranges when looking at the scaled medians across Germany in 2022 (**Figure 3** and **Supplementary Table 2**). The uncertainty (under- or overestimation due to mixed pixel effect) of values in small fragments below 1 km is high since the temperature and forest vitality are originally captured at 1-km spatial resolution. Nevertheless, there are regional specifics for the temperature and forest vitality. The highest temperatures were observed in the small fragments of the Southwest Uplands, and the lowest temperatures could be found in the largest fragments of the Alps. In the small fragments of the Northeast and Northwest Lowlands, the lowest vitality was observed. The highest vitality was registered in the largest fragments of the Alps and the Western Uplands.

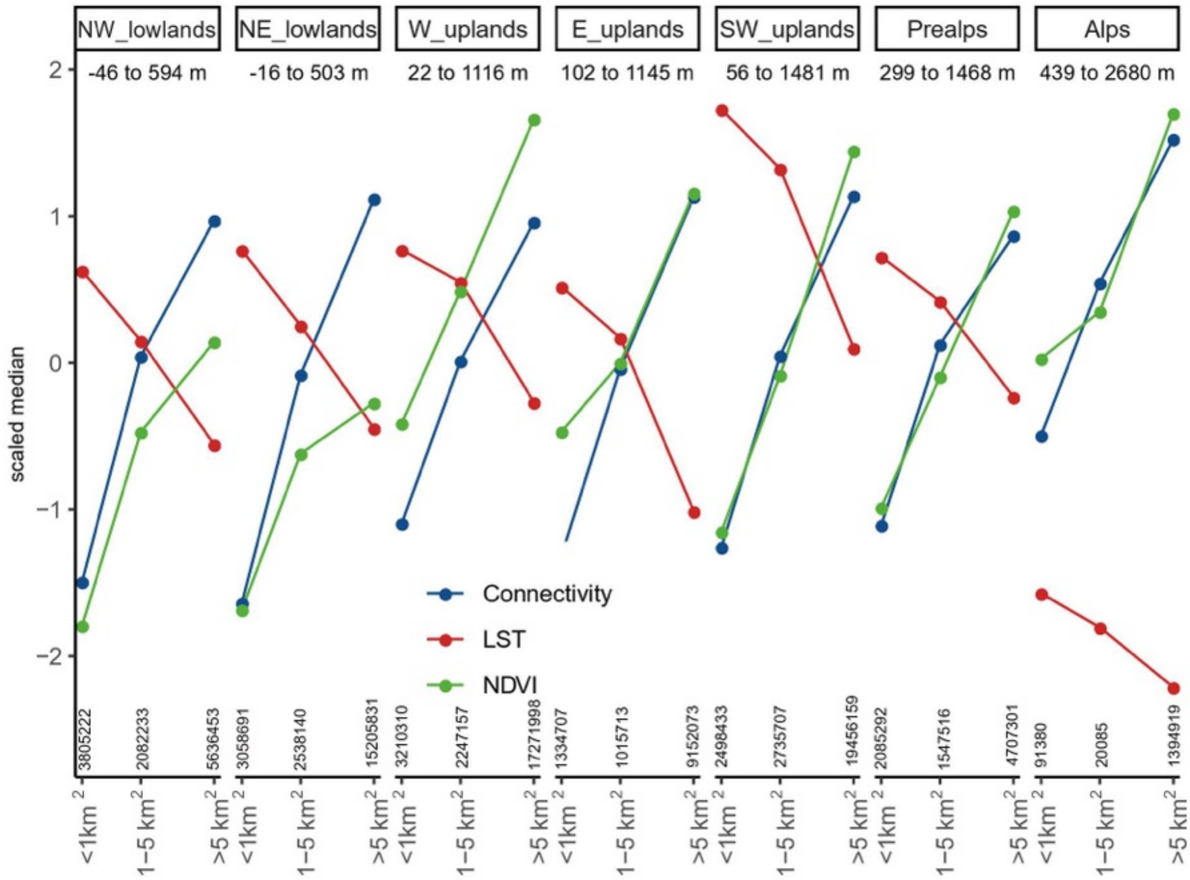


FIGURE 3
Scaled median (median per indicator for tree-covered areas in Germany 2022 divided by the standard deviation) per ecoregion with respect to different forest fragment sizes (<1 km^2, 1–5 km^2, and >5 km^2). The indicators are Thiessen connectivity for forest-covered areas in Germany in 2022 (blue), land surface temperature [land surface temperature (LST); MODIS Aqua MYD11A1] based on the mean of the hottest days in 2022 (red), and the NDVI (MODIS Aqua MYD13A2) based on the maximum value of the growing season 2022 (green). Altitudinal ranges per ecoregion are depicted below the headings. The coverage of each fragment size group is depicted above the *X*-axis in km^2.

3.2. Spatial variations in fragmentation, temperature, and forest vitality

The bivariate choropleth maps between Thiessen connectivity and land surface temperature (**Figure 4A**) and Thiessen connectivity and forest vitality (NDVI) (**Figure 4B**) depict how the studied proxy indicators vary in geographical space with respect to each other. Cyan and magenta color tones on the map indicate a positive association between the two variables, while blue and red color tones indicate a negative association. For instance, the spatial variation between the high connectivity of tree-covered areas in Germany in 2022 and respective low temperatures can be observed in larger, better-connected forest areas (**Figure 4A**, blue areas). This relationship is not only true for large forest landscapes (e.g., part of the Black Forest or the Alps) but can also be found in areas with smaller forest fragments (e.g., Schorfheide in the Northeast Lowlands). Similar patterns can be found between the high connectivity of tree-covered areas and a high vitality (**Figure 4B**, magenta areas). However, the spatial pattern is more scattered. Especially the Northeast Lowlands show higher connectivity with a lower vitality signal. In the higher altitudes (e.g., the Alps), temperatures are low when the connectivity is high, but only some areas show a high vitality with high connectivity, while other areas show reduced connectivity and vitality (**Figure 4B**, Alps ecoregion magenta and green areas). When comparing the bivariate choropleth maps for broad-leaved and coniferous forest types (**Supplementary Figures 5**, **6**), there is a clear indication that broad-leaved forests in Northern and Eastern Germany, representing the natural vegetation, have been severely degraded and fragmented in the past. Coniferous plantations seem to benefit from larger forest blocks and higher connectivity, having a higher NDVI such as the northern Black Forest in the Southwest of Germany.

The range of variations taking into account all the forest fragments per ecoregion can be observed in the scaled data ranges of the connectivity in tree-covered areas in their relation to hot day temperatures and maximum vitality (**Figure 5**). Forest in the Alps is relatively less fragmented than in other ecoregions, is cooler, and has the highest forest vitality. The Lowlands show the lowest median connectivity and lowest vitality, while having similar median temperatures as the Uplands. The highest median temperature is recorded in the Southwest Uplands. In the Western and Eastern Uplands and the Prealps, slightly higher connectivity is accompanied by lower temperatures and higher vitality. When investigating the distribution of Thiessen connectivity per ecoregion regardless of fragment groups, the connectivity is much lower (**Figure 5**), the reason being that all the ecoregions are dominated by the presence

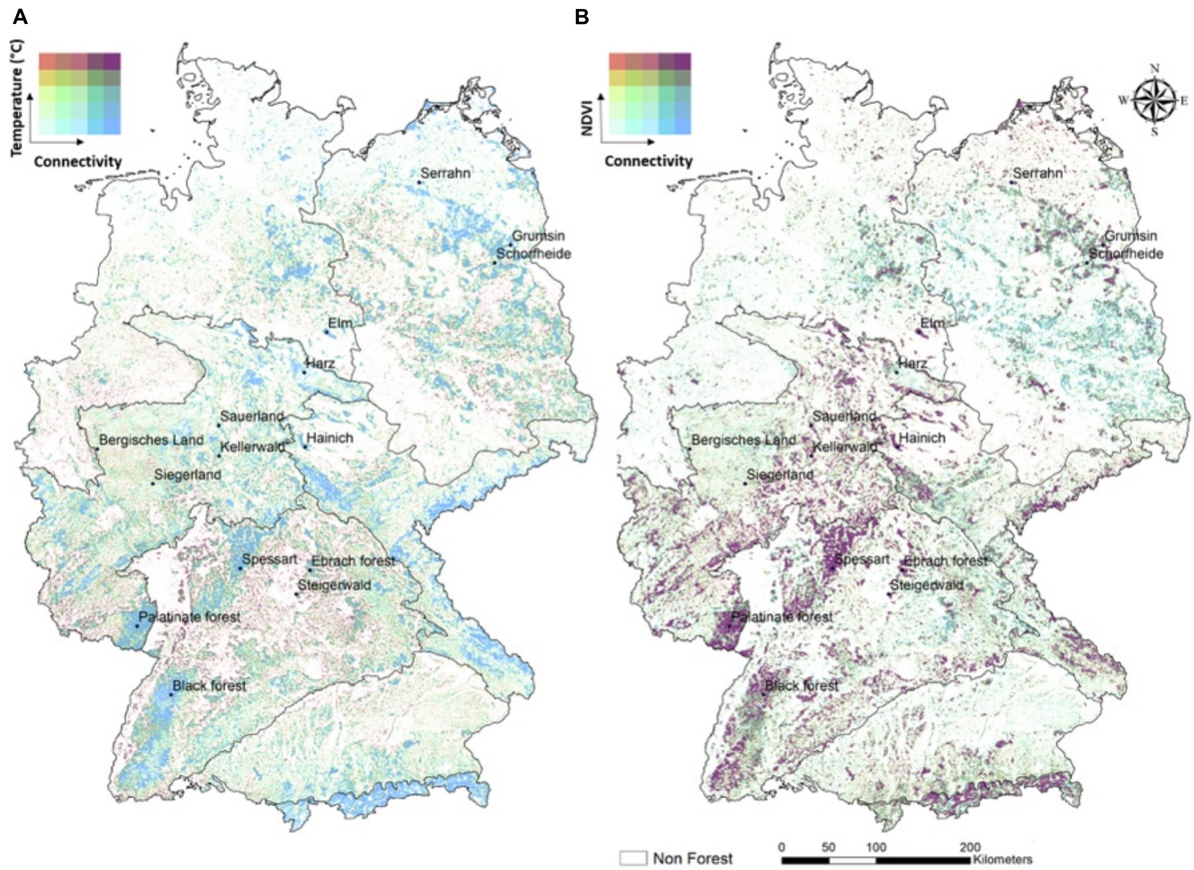


FIGURE 4
Bivariate maps of tree-covered areas in Germany in 2022 and the relation between the degree of fragmentation with temperatures and vegetation vitality. **(A)** Thiessen connectivity and land surface temperature of hot days in 2022. **(B)** Thiessen connectivity and the maximum NDVI per growing season in 2022.

of forest fragments smaller than 1 km^2. Hence, lower overall connectivity values are evident.

4. Discussion

In the studied year 2022, forest fragmentation clearly impacted local temperatures on hot days and forest vitality. Here, we show that these effects vary by ecoregion and we discuss implications for landscape and forest management.

4.1. Regional implications of fragmentation on forest temperature and vitality

Independent of the ecoregion, larger fragments of tree-covered areas show the highest connectivity, the lowest temperatures, and the highest vitality. Lower values of Thiessen connectivity in all ecoregions result from the dominance of small-sized forest fragments indicating a great degree of fragmentation. The area with the least fragmented forest in Germany are the Alps (highest Thiessen connectivity with a median of 0.38 as compared to the maximum value of 1). In comparison with the other ecoregions, this area is less urbanized due to its topography, and therefore, the effects of fragmentation through infrastructure (Ibisch et al., 2016) or cities are less severe. The overall high forest vitality in the Alps ecoregion is congruent with other studies that measured a growing biomass stock and an expanding forest-covered area, despite intensified natural and human-induced disturbances (Bebi et al., 2017). The Alps are the smallest of all ecoregions with 7,796 data points of the tree-covered areas observed. The high elevation and terrain with steep slopes are most likely a reason for the diverse range of temperature values from low to high. Even though fewer hot days in the year 2022 for the Alps ecoregion were detected, the Alps are also threatened by climate change and more frequent temperature extremes (Seidl et al., 2011; Gobiet et al., 2014). A larger forest cover in mountainous regions can help retain permanent snow cover longer than non-tree-covered areas (Hesslerová et al., 2018).

The Prealps are characterized by the highest altitudinal range of all ecoregions from 299- to 1,468-m elevation (refer to **Figure 3**). As a consequence of more intensive land use, the tree-covered areas in the Prealps are less connected and less vital as compared to the Alps, and the land surface temperature median is ca. 5°C higher than in the Alps. The foothills of the Alps are also impacted by climate change (e.g., Thrippleton et al., 2020).

The Southwest Uplands are the ecoregion in Germany with the highest measured temperatures of forest canopies in 2022. Therefore, the forest is rarely connected (Thiessen connectivity median 0.18) with a medium vitality (median NDVI 0.79) and high temperatures

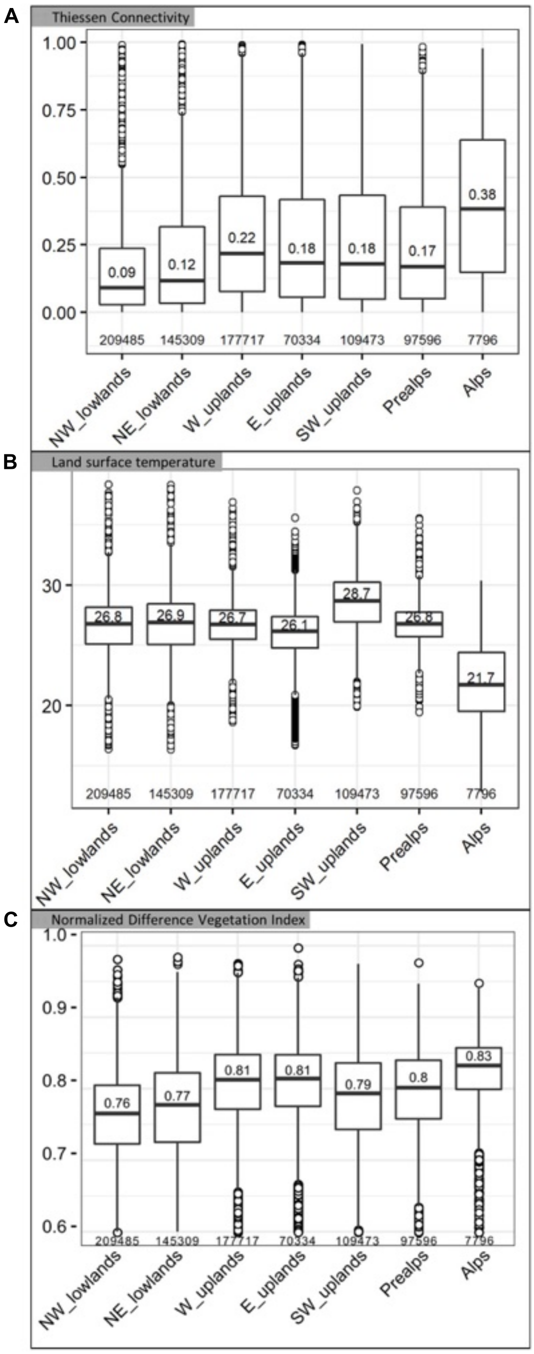


FIGURE 5
Boxplots of connectivity, temperature, and vitality for tree-covered areas in Germany 2022 per ecoregion including all values regardless of the size of a fragment. The median of each dataset per boxplot is depicted as a bold horizontal line, the line above represents the upper quartile of which 75% of the values fall below and the line below the median represents the lower quartile with 25% of the values below. **(A)** Thiessen connectivity for forest-covered areas in Germany as of 2022. **(B)** Land surface temperature (LST) based on the mean of the hottest days in 2022. **(C)** Vitality (NDVI) based on the maximum value of the growing season in 2022 (for better readability, outliers below 0.6 are not depicted). The count of data points per ecoregion is depicted above the *X*-axis.

(median of 28.7°C is 7°C hotter than in the Alps). It is in fact the hottest region in Germany in terms of regional climate and is highly populated. In this ecoregion, there are a few highly connected forests such as the Black Forest in the south-westernmost part but mostly areas with low connectivity and smaller forest fragments. A small-scale study in the area, investigating Scots pine forest edges using dendroecological investigations and close-range remote sensing, found that these forest edges are more vulnerable to drought, showing increased mortality rates and lower tree vitality (Buras et al., 2018). A study conducted in Switzerland using thermal imaging found that the resilience to the drought of broad-leaved forests varies by species (Scherrer et al., 2011). Lower vitality values with a median of 0.71 for the year 2022 can be related to the spruce forest dieback, which is the dominant tree type in the ecoregion. Still, coniferous forests in dense stands seem to benefit from higher connectivity and show higher vitality, especially in the Black Forest (**Supplementary Figures 3**, **5**, **6**). Larger broad-leaved dominated forest areas in the northern part of the ecoregion, such as the Spessart, Steigerwald, or Ebrach forests, show relatively high vitality (**Supplementary Figure 5**).

The Eastern Uplands share a similar value of low connectivity (median 0.18) as the Southwest Uplands but a higher vitality (median 0.81) and 2°C lower temperatures. A strong temperature decline, vitality, and connectivity increase from medium to large forest fragments and can be associated with less intensively managed forests. In the Bavarian forest region, the mainly coniferous forest is relatively vital and connected (**Figure 4**). However, the vegetation is under stress due to climate-related bark beetle outbreaks of spruce before 2012 (Lausch et al., 2013a,b). The same is true for the Erzgebirge with deforestation due to a recent bark beetle outbreak (Gdulová et al., 2021). The Thuringian Forest still can be recognized as a relatively well-connected area with relatively low temperatures and higher vitality, despite Norway spruce being the main tree species (Thiel et al., 2006). But for some years, it already suffers from heavy storms and bark beetle outbreaks, and it is most likely, that in this area, the effects of climate change will become more frequent, such as a rise in temperatures, more frequent temperature extremes, and a decreased water supply during the growing season (Frischbier et al., 2014).

The Western Uplands are characterized by slightly higher overall connectivity than the Eastern and Southwest Uplands but show higher temperatures and not notably higher vitality (**Figure 5A**). A high connectivity and high vitality are visible in the Palatinate Forest (Pfälzerwald), on the southern edge of the ecoregion. In general, the vitality is higher than in the other Uplands, especially, because of the relatively high share of broad-leaf forests with a higher vitality range *per se* (**Supplementary Figure 3**). At the same time, the ecoregion is dominated by large coniferous plantations and experienced strong Norway spruce dieback in the last years. The Western Uplands are one of the formerly largest forest regions in Western Germany, including the Sauerland, Bergisches Land, Siegerland, and Harz mountains. After years of massive dieback and salvage logging, the remaining forests are in relatively poor condition, fragmented, and with reduced vitality (Popkin, 2021).

In the Northeast Lowlands and the Northwest Lowlands, the small size forest fragments show the lowest connectivity. Even in larger forest fragments with higher connectivity detected, the vitality values are low. This is due to a large number of coniferous plantations in both ecoregions (**Supplementary Figure 6**). Many Scots pine plantations do not seem to benefit from the generally somewhat lower temperatures in the north. They are often well-connected, but this does not translate into a better vitality signature. In general, there seems to be a gradient toward higher vitality in the north, possibly due to the buffering impact of the sea with its maritime climate, a higher precipitation potential, and a lower drought potential than in the south of Germany (Zink et al., 2016). This effect of continentality is reflected to some extent by the ecoregions. The median connectivity of 0.09 and median vitality of 0.76 are the lowest of all ecoregions in the Northwest Lowlands and therefore can be attested to the less connected forest cover of all ecoregions. The fragmented landscape leads to low connectivity and higher temperatures, and the coniferous plantations (according to ground-truthing) have low vitality. These effects are repercussions, especially of the extreme years from 2018 onward, and consequent bark beetle outbreaks (refer to **Supplementary Figure 2**). Interestingly, the largest beech forest in Northern Germany, the Elm, lies within the Northwest Lowlands and shows relatively high Thiessen connectivity values with high vitality and low temperatures (refer to **Supplementary Figures 5**, **7**).

4.2. Effects of fragmentation

Independent of the ecoregion, larger fragments of tree-covered areas show the highest connectivity, the lowest temperatures, and the highest vitality (**Figure 3**). The cooling functions of forest fragments increase with higher connectivity. This cooling effect was also observed in a study on reforestation (Huang et al., 2022). Our results regarding the correlation between high connectivity and increasing forest vitality, based on the NDVI in the temperate forest realm, correspond to other studies with similar results. Others used the NDVI, and for connectivity "vegetation continuous fields," and found that higher connectivity in protected areas correlates to high NDVI values (Sun et al., 2021). Other factors influencing the NDVI that are not covered in this study would be local and regional characteristics such as climate, soil moisture, dominant tree species, and the degree of disturbance.

This study focuses on regional assessments and could not scrutinize the effects of fragmentation through skidding trails and small-scale edge effects. These patterns potentially increase the fragmentation impacts and most likely have additional effects on the local forest temperature and vitality (Buckley et al., 2003; Sufo Kankeu et al., 2016; Shirvani et al., 2020). The characteristics of native and planted forests in Germany were not covered in this study. To date, no comprehensive and explicit spatial information on the native and planted forests is available, neither for the globe (Grantham et al., 2020) nor for Germany. It is also important to note that in some areas, the NDVI and Thiessen connectivity are not necessarily positively correlated. There are definitely other relevant effects such as the type of forests. For example, in the Scots pine forests of northeastern Germany, relatively low vitality is observed despite high connectivity. However, especially these forests represent mostly planted even-aged monocultures with low structural diversity, relatively open canopies, and many timber extraction roads and trails. Our analysis could not take into account that there is also internal forest fragmentation, which contributes to the vulnerability of the ecosystem in the form of forest roads and skid trails as there are currently no data available. The situation in Germany is currently worsening because the infrastructure for the expansion of renewable energy production is pushing into the forest. Here, wide permanent access roads and openings for wind turbines increase the forest edges and the edge effects in the midst of the forest.

To promote the protection of remaining old-growth broad-leaved forests, UNESCO recognized some old temperate forests in Europe as natural heritage, which are component parts of the serial UNESCO transnational property "Ancient and Primeval Beech Forests of the Carpathians and Other Regions of Europe" (Voloscuk et al., 2013; Ibisch et al., 2017; Jovanović et al., 2019; UNESCO, 2022). In Germany, the Hainich National Park, Kellerwald-Edersee National Park, Serrahn Forest, and the Grumsin Forest are part of this property and show relatively high connectivity, especially in the core zones, despite being of relatively small size. These areas are highly important not only in terms of being a heritage site but also as a cooling factor in the landscape, a remnant habitat, and are of recreational value for humans.

4.3. Recommendations for landscape and forest management

Forest vitality and functionality are not only impacted by climate change and natural disturbances but also by silvicultural management. The management attempts of the last decades to protect forests in Europe did not mitigate climate warming (Naudts et al., 2016) but even led to increased temperatures within forests (Blumröder et al., 2021). Effective ecosystem management must allow ecosystems to mature through the growth of biomass, information, and network and to maintain or enhance working capacity in the best possible way. The production and storage of biomass and biogenic free energy in the ecosystem—including dead wood, humus, or organic molecules in mineral soil—are the physical basis of all possible natural ecological processes in the ecosystem. Linking to the development of biophysical capacity is also an increase in the ecohydrological capacity of forest ecosystems. The conservation of "green water" stored and mobilized by ecosystems (Ellison et al., 2017; Sheil, 2018; Te Wierik et al., 2021) and microclimatic regulation (Blumröder et al., 2021, 2022) deserve the highest priority in management (Ibisch, 2022). To mitigate hot temperature extremes in European forests, the increase in the broad-leaved tree fraction is a necessary measure (Schwaab et al., 2020). German forests are already highly fragmented. In some forest-poor regions, it is recommended to increase tree and forest cover to buffer temperatures and contribute to forest vitality (Gohr et al., 2021). The regulating function of connected forests within the water, energy, and carbon cycles is more needed than ever (Ellison et al., 2017) since highly fragmented forests with more forest edges provoke more carbon loss (Smith et al., 2018).

Remote sensing and especially new datasets like the dominant tree species of German forests (Welle et al., 2022) will support monitoring fragmentation as seen in other countries (Kupfer, 2006). It is an urgent task for the state rapporteurs of the federal states and the federal ministries responsible for forests to ensure that forest fragmentation is included in the forest status reports and the corresponding analyses of forest health.

Fragmentation is caused by infrastructure expansion and land use changes but also by (past) forest management: When plantations collapse by massive tree dieback, they are often salvage-logged. The regeneration capacity decreases under climate change and potentially these areas are then converted into open lands. It is inevitable that forest fragmentation in the near future will further increase due to large-scale tree dieback in monocultures. The ban on salvage logging and clearcutting seems to be a necessary step in forest management. The existing smaller forest fragments have the potential to be transformed into larger core forest areas for enhanced ecosystem development. This can be achieved by the abandonment of timber extraction, implementing reforestation measures such as assisted restoration, planting native trees, natural regrowth, agroforestry solutions, and commercial plantations.

5. Conclusion

The study reflects that the forests in Germany are highly fragmented, which weakens their ecosystem functionality. This study provides observational evidence to show that highly fragmented forests exhibit higher temperatures and less forest vitality. Thereby, the fragmentation in Germany as of 2022 substantially contributes to the current and ongoing forest crises. The existing small forest fragments have immense potential to be transformed into larger core areas for better ecosystem functioning. With increasing forest fires and extreme climatic events in Germany and worldwide in general, there is an urgent need to advance forest management and restoration efforts that can safeguard the benefits of functional forests as much and as long as possible. Reducing the fragmentation of forests is a crucial contribution to ecosystem-based adaptation to climate change.

Data availability statement

The original contributions presented in this study are included in the article/**Supplementary material**, further inquiries can be directed to the corresponding author.

Author contributions

PI, DM, and CG designed the research. DM and CG performed the analysis, visualization, and wrote the original draft. PI and JB provided comprehensive support in the assessment and interpretation of the results. All authors contributed to the interpretation of the results and the subsequent revisions of the manuscript.

Funding

This study was partly funded by the PYROPHOB project (Grant number: 2219WK50A4, Federal Ministry of Food and Agriculture and the Federal Ministry for the Environment, Nature Conservation, Nuclear Safety, and Consumer Protection) and through the "Gläserner Forstbetrieb" project by the German Federal Ministry for Education and Research (BMBF) *via* the VDI/VDE, grant number: 16LC1603C. PI conceived and supervised the study within the framework of his long-term research program facilitated by the research professorships "Biodiversity and natural resource management under global change" (2009–2015) and "Ecosystem based sustainable development" (since 2015) granted by the Eberswalde University for Sustainable Development. CG was funded through the Biosphere Reserves Institute and the Innovation and Career Center "ProBio-LaB" by the Ministry of Science, Research and Culture of the federal state of Brandenburg.

Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2023.1099460/full#supplementary-material

References

Aalto, I. J., Maeda, E. E., Heiskanen, J., Aalto, E. K., and Pellikka, P. K. E. (2022). Strong influence of trees outside forest in regulating microclimate of intensively modified afromontane landscapes. *Biogeosciences* 19, 4227–4247. doi: 10.5194/bg-19-4227-2022

Armenteras, D., Gonzalez, T. M., and Retana, J. (2013). Forest fragmentation and edge influence on fire occurrence and intensity under different management types in Amazon forests. *Biol. Conserv.* 159, 73–79. doi: 10.1016/j.biocon.2012.10.026

Bebi, P., Seidl, R., Motta, R., Fuhr, M., Firm, D., Krumm, F., et al. (2017). Changes of forest cover and disturbance regimes in the mountain forests of the Alps. *For. Ecol. Manag. Ecol. Mount. For. Ecosyst. Eur.* 388, 43–56. doi: 10.1016/j.foreco.2016.10.028

Betts, M. G., Wolf, C., Ripple, W. J., Phalan, B., Millers, K. A., Duarte, A., et al. (2017). Global forest loss disproportionately erodes biodiversity in intact landscapes. *Nature* 547, 441–444. doi: 10.1038/nature23285

Blumröder, J. S., May, F., Härdtle, W., and Ibisch, P. L. (2021). Forestry contributed to warming of forest ecosystems in Northern Germany during the extreme summers of 2018 and 2019. *Ecol. Solut. Evid.* 2:e12087. doi: 10.1002/2688-8319.12087

Blumröder, J. S., Schmidt, F., Gordon, A., Grosse, S., and Ibisch, P. L. (2022). Ecosystemic resilience of a temperate post-fire forest under extreme weather conditions. *Front. For. Glob. Change* 5:1070958. doi: 10.3389/ffgc.2022.1070958

Briant, G., Gond, V., and Laurance, S. G. W. (2010). Habitat fragmentation and the desiccation of forest canopies: A case study from Eastern Amazonia. *Biol. Conserv.* 143, 2763–2769. doi: 10.1016/j.biocon.2010.07.024

Buckley, D. S., Crow, T. R., Nauertz, E. A., and Schulz, K. E. (2003). Influence of skid trails and haul roads on understory plant richness and composition in managed forest landscapes in Upper Michigan, USA. *For. Ecol. Manag.* 175, 509–520. doi: 10.1016/S0378-1127(02)00185-8

Bundesamt für Naturschutz [BfN] (2017). *Naturräume und Großlandschaften Deutschland*. Bonn: Federal Agency for Nature Conservation.

Bundesministerium für Ernährung und Landwirtschaft [BMBL] (2022). *Ergebnisse der Waldzustandserhebung 2021*. Rochusstr: BMBL, 76.

Buras, A., Rammig, A., and Zang, C. S. (2021). The European forest condition monitor: Using remotely sensed forest greenness to identify hot spots of forest decline. *Front. Plant Sci.* 12:689220. doi: 10.3389/fpls.2021.689220

Buras, A., Schunk, C., Zeiträg, C., Herrmann, C., Kaiser, L., Lemme, H., et al. (2018). Are scots pine forest edges particularly prone to drought-induced mortality? *Environ. Res. Lett.* 13:025001. doi: 10.1088/1748-9326/aaa0b4

Chakraborty, T., Sarkar, S. K., and Morshed, M. D. M. (2022). Big data and remote sensing for multi-decadal drought impact assessment on shorea robusta. *Theor. Appl. Climatol.* 148, 1587–1602. doi: 10.1007/s00704-022-04019-2

Deng, Y., Wang, S., Bai, X., Tian, Y., Wu, L., Xiao, J., et al. (2018). Relationship among land surface temperature and LUCC. NDVI in typical karst area. *Sci. Rep.* 8:641. doi: 10.1038/s41598-017-19088-x

Deutscher Wetterdienst [DWD] (2022). *Deutschlandwetter im herbst 2022.* Available online at: https://www.dwd.de/DE/presse/pressemitteilungen/DE/2022/20221130_deutschlandwetter_herbst2022_news.html (accessed October 10, 2022).

Didan, K. (2015). *MOD13A2 MODIS/Terra vegetation indices 16-Day L3 global 1km SIN Grid V006 [Data Set]. NASA EOSDIS land processes DAAC.* Available online at: https://doi.org/10.5067/MODIS/MOD13A2.006 (accessed June 7, 2022).

Driscoll, D. A., Armenteras, D., Bennett, A. F., Brotons, L., Clarke, M. F., Doherty, T. S., et al. (2021). How fire interacts with habitat loss and fragmentation. *Biol. Rev.* 96, 976–998.

Ellison, D., Morris, C. E., Locatelli, B., Sheil, D., Cohen, J., Murdiyarso, D., et al. (2017). Trees, forests and water: Cool insights for a hot world. *Glob. Environ. Change* 43, 51–61. doi: 10.1016/j.gloenvcha.2017.01.002

Estreguil, C., Caudullo, G., de Rigo, D., and San-Miguel-Ayanz, J. (2013). Forest Landscape in Europe: Pattern, fragmentation and connectivity. *EUR Sci. Tech. Res.* 25717.

Fisher, J. B., Melton, F., Middleton, E., Hain, C., Anderson, M., Allen, R., et al. (2017). The future of evapotranspiration: Global requirements for ecosystem functioning, carbon and climate feedbacks, agricultural management, and water resources. *Water Resour. Res.* 53, 2618–2626. doi: 10.1002/2016WR020175

Frischbier, N., Profft, I., and Hagemann, U. (2014). "Potential impacts of climate change on forest habitats in the biosphere reserve Vessertal-Thuringian forest in Germany," in *Managing protected areas in central and Eastern Europe under climate change*, eds S. Rannow and M. Neubert (Dordrecht: Springer), 243–257.

Gdulová, K., Marešová, J., Barták, V., Szostak, M., Červenka, J., and Moudrý, V. (2021). Use of TanDEM-X and SRTM-C data for detection of deforestation caused by bark beetle in central European mountains. *Remote Sens.* 13:3042. doi: 10.3390/rs13153042

Gobiet, A., Kotlarski, S., Beniston, M., Heinrich, G., Rajczak, J., and Stoffel, M. (2014). 21st century climate change in the European Alps—A review. *Sci. Tot. Environ.* 493, 1138–1151. doi: 10.1016/j.scitotenv.2013.07.050

Gohr, C., Blumröder, J. S., Sheil, D., and Ibisch, P. L. (2021). Quantifying the mitigation of temperature extremes by forests and wetlands in a temperate landscape. *Ecol. Inform.* 66:101442. doi: 10.1016/j.ecoinf.2021.101442

Grantham, H. S., Duncan, A., Evans, T. D., Jones, K. R., Beyer, H. L., Schuster, R., et al. (2020). Anthropogenic modification of forests means only 40% of remaining forests have high ecosystem integrity. *Nat. Commun.* 11, 1–10. doi: 10.1038/s41467-020-19493-3

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A. A., Tyukavina, A., et al. (2013). High-resolution global maps of 21st-century forest cover change. *Science* 342, 850–853. doi: 10.1126/science.1244693

Hansen, M. C., Stehman, S. V., and Potapov, P. V. (2010). Quantification of global gross forest cover loss. *Proc. Natl. Acad. Sci. U.S.A.* 107, 8650–8655. doi: 10.1073/pnas.0912668107

Harper, K. A., Ellen Macdonald, S., Burton, P. J., Chen, J., Brosofske, K. D., Saunders, S. C., et al. (2005). Edge influence on forest structure and composition in fragmented landscapes. *Conserv. Biol.* 19, 768–782.

Hesslerová, P., Huryna, H., Pokorný, J., and Procházka, J. (2018). The effect of forest disturbance on landscape temperature. *Ecol. Eng.* 120, 345–354.

Huang, A., Shen, R., Jia, G., and Xu, X. (2022). Reforestation enhanced landscape connectivity for thermal buffering in China. *Environ. Res. Lett.* 17:014056. doi: 10.1088/1748-9326/ac3fda

Ibisch, P. L. (2022). Ein ökosystembasierter Ansatz für den Umgang mit der Waldkrise in der Klimakrise. *Natur Landschaft* 97, 325–333.

Ibisch, P. L., Hoffmann, M. T., Kreft, S., Pe'er, G., Kati, V., Biber-Freudenberger, L., et al. (2016). A global map of roadless areas and their conservation status. *Science* 354, 1423–1427. doi: 10.1126/science.aaf7166

Ibisch, P. L., Waldherr, M. G., and Knapp, H. D. (2017). Extension nomination of the'Primeval Beech Forests of the Carpathians and Ancient Beech Forests of Germany'as Pan-European UNESCO natural world heritage site. *Natur Landschaft* 92, 109–118.

Imbery, F., Friedrich, K., Fleckenstein, R., Becker, A., Bissolli, P., Haeseler, S., et al. (2022). *Klimatologischer rückblick sommer 2022. Deutscher wetterdienst.* Available online at: https://www.dwd.de/DE/leistungen/besondereereignisse/temperatur/20220921_bericht_sommer2022.pdf?__blob=publicationFile&v=6 (accessed October 10, 2022).

Jacob, D., Kotova, L., Teichmann, C., Sobolowski, S. P., Vautard, R., Donnelly, C., et al. (2018). Climate impacts in Europe Under+ 1.5 C global warming. *Earth's Fut.* 6, 264–285.

Jin, M., and Dickinson, R. E. (2010). Land surface skin temperature climatology: Benefitting from the strengths of satellite observations. *Environ. Res. Lett.* 5:044004. doi: 10.1088/1748-9326/5/4/044004

Jovanović, I., Dragišić, A., Ostojić, D., and Krsteski, B. (2019). Beech forests as world heritage in aspect to the next extension of the ancient and primeval beech forests of the carpathians and other regions of Europe world heritage site. *Zastita Prirode* 69, 15–32. doi: 10.5937/ZasPri1901015J

Kaplan, J. O., Krumhardt, K. M., and Zimmermann, N. (2009). The prehistoric and preindustrial deforestation of Europe. *Quaternary Sci. Rev.* 28, 3016–3034. doi: 10.1016/j.scitotenv.2006.08.013

Kupfer, J. A. (2006). National assessments of forest fragmentation in the US. *Glob. Environ. Change* 16, 73–82. doi: 10.1016/j.gloenvcha.2005.10.003

Lambert, J., Drenou, C., Denux, J.-P., Balent, G., and Cheret, V. (2013). Monitoring forest decline through remote sensing time series analysis. *GISci. Remote Sens.* 50, 437–457. doi: 10.1080/15481603.2013.820070

Laurance, W. F., Camargo, J. L. C., Luizão, R. C. C., Laurance, S. G., Pimm, S. L., Bruna, E. M., et al. (2011). The fate of Amazonian forest fragments: A 32-year investigation. *Biol. Conserv.* 144, 56–67.

Laurance, W. F., Lovejoy, T. E., Vasconcelos, H. L., Bruna, E. M., Didham, R. K., Stouffer, P. C., et al. (2002). Ecosystem decay of Amazonian forest fragments: A 22-year investigation. *Conserv. Biol.* 16, 605–618.

Lausch, A., Heurich, M., Gordalla, D., Dobner, H.-J., Gwillym-Margianto, S., and Salbach, C. (2013a). Forecasting potential bark beetle outbreaks based on spruce forest vitality using hyperspectral remote-sensing techniques at different scales. *For. Ecol. Manag.* 308, 76–89. doi: 10.1016/j.foreco.2013.07.043

Lausch, A., Heurich, M., and Fahse, L. (2013b). Spatio-temporal infestation patterns of Ips typographus (L.) in the Bavarian forest national park, Germany. *Ecol. Indicat. Link. Landscape Struc. Biodivers.* 31, 73–81. doi: 10.1016/j.ecolind.2012.07.026

Li, Y., Zhao, M., Mildrexler, D. J., Motesharrei, S., Mu, Q., Kalnay, E., et al. (2016). Potential and actual impacts of deforestation and afforestation on land surface temperature. *J. Geophys. Res. Atmos.* 121, 14,372–14,386.

Li, Z.-L., Tang, B.-H., Wu, H., Ren, H., Yan, G., Wan, Z., et al. (2013). Satellite-derived land surface temperature: Current status and perspectives. *Remote Sens. Environ.* 131, 14–37.

Lindner, M., Fitzgerald, J. B., Zimmermann, N. E., Reyer, C., Delzon, S., van der Maaten, E., et al. (2014). Climate change and European forests: What do we know, What are the uncertainties, and What are the implications for forest management? *J. Environ. Manag.* 146, 69–83. doi: 10.1016/j.jenvman.2014.07.030

Mehdipour, N., Fakheran, S., Soffianian, A., and Pourmanafi, S. (2019). Road-induced fragmentation and the environmental value of roadless areas in a partly protected landscape in Central Iran. *Environ. Monit. Assess.* 191:461. doi: 10.1007/s10661-019-7571-4

Mildrexler, D. J., Zhao, M., and Running, S. W. (2011). A global comparison between station air temperatures and MODIS land surface temperatures reveals the cooling role of forests. *J. Geophys. Res. Biogeosci.* 116, doi: 10.1029/2010JG001486

Minnemeyer, S., and Potapov, P. (2017). *World's last intact forests are becoming increasingly fragmented.* Washington, DC: World Resource Institute.

Moomaw, W. R., Masino, S. A., and Faison, E. K. (2019). Intact forests in the United States: Proforestation mitigates climate change and serves the greatest good. *Front. For. Glob. Change* 2:27. doi: 10.3389/ffgc.2019.00027

Naudts, K., Chen, Y., McGrath, M. J., Ryder, J., Valade, A., Otto, J., et al. (2016). Europe's forest management did not mitigate climate warming. *Science* 351, 597–600. doi: 10.1126/science.aad7270

Olson, D. M., Dinerstein, E., Wikramanayake, E. D., Burgess, N. D., Powell, G. V. N., Underwood, E. C., et al. (2001). Terrestrial ecoregions of the world: A new map of life on earth: A new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity. *BioScience* 51, 933–938. doi: 10.1641/0006-35682001051[0933:TEOTWA]2.0.CO;2

Popkin, G. (2021). Forest fight. *Science* 374, 1184–1189.

Potapov, P., Hansen, M. C., Laestadius, L., Turubanova, S., Yaroshenko, A., Thies, C., et al. (2017). The last frontiers of wilderness: Tracking loss of intact forest landscapes from 2000 to 2013. *Sci. Adv.* 3:e1600821. doi: 10.1126/sciadv.1600821

Riitters, K. H. (2007). "Forest fragmentation," in *Forest health monitoring: 2005 national technical report. General technical report SRS-104*, Vol. 104, (Asheville, NC: US Department of Agriculture, Forest Service, Southern Research Station), 9–15.

Rogers, B. M., Mackey, B., Shestakova, T. A., Keith, H., Young, V., Kormos, C. F., et al. (2022). Using ecosystem integrity to maximize climate mitigation and minimize risk in international forest policy. *Front. For. Glob. Change* 5:929281. doi: 10.3389/ffgc.2022.929281

Scherrer, D., Bader, M. K. F., and Körner, C. (2011). Drought-sensitivity ranking of deciduous tree species based on thermal imaging of forest canopies. *Agric. For. Meteorol.* 151, 1632–1640. doi: 10.1016/j.agrformet.2011.06.019

Schuldt, B., Buras, A., Arend, M., Vitasse, Y., Beierkuhnlein, C., Damm, A., et al. (2020). A first assessment of the impact of the extreme 2018 summer drought on central European forests. *Basic Appl. Ecol.* 45, 86–103. doi: 10.1016/j.baae.2020.04.003

Schwaab, J., Davin, E. L., Bebi, P., Duguay-Tetzlaff, A., Waser, L. T., Haeni, M., et al. (2020). Increasing the broad-leaved tree fraction in European forests mitigates hot temperature extremes. *Sci. Rep.* 10:14153. doi: 10.1038/s41598-020-71055-1

Seidl, R., Rammer, W., and Lexer, M. J. (2011). Climate change vulnerability of sustainable forest management in the Eastern Alps. *Climatic Change* 106, 225–254. doi: 10.1007/s10584-010-9899-1

Sheil, D. (2018). Forests, atmospheric water and an uncertain future: The new biology of the global water cycle. *For. Ecosyst.* 5, 1–22.

Shirvani, Z., Abdi, O., and Buchroithner, M. F. (2020). A new analysis approach for long-term variations of forest loss, fragmentation, and degradation resulting from road-network expansion using landsat time-series and object-based image analysis. *Land Degradation Dev.* 31, 1462–1481. doi: 10.1002/ldr.3530

Silva Junior, C. H. L., Aragão, L. E. O. C., Anderson, L. O., Fonseca, M. G., Shimabukuro, Y. E., Vancutsem, C., et al. (2020). Persistent collapse of biomass in Amazonian forest edges following deforestation leads to unaccounted carbon losses. *Sci. Adv.* 6:eaaz8360. doi: 10.1126/sciadv.aaz8360

Smith, I. A., Hutyra, L. R., Reinmann, A. B., Marrs, J. K., and Thompson, J. R. (2018). Piecing together the fragments: Elucidating edge effects on forest carbon dynamics. *Front. Ecol. Environ.* 16, 213–221. doi: 10.1002/fee.1793

Smith, J. R., Letten, A. D., Ke, P.-J., Anderson, C. B., Hendershot, J. N., Dhami, M. K., et al. (2018). A global test of ecoregions. *Nat. Ecol. Evol.* 2, 1889–1896. doi: 10.1038/s41559-018-0709-x

Sufo Kankeu, R., Sonwa, D. J., Eba'a Atyi, R., and Nkal, N. M. M. (2016). Quantifying post logging biomass loss using satellite images and ground measurements in Southeast Cameroon. *J. For. Res.* 27, 1415–1426. doi: 10.1007/s11676-016-0277-3

Sun, J., Zheng, Y., and Shi, J. (2021). Conserving the landscape connectivity of natural forest reserves in tourism development. *Prof. Geogr.* 73, 573–578. doi: 10.1080/00330124.2021.1898994

Taubert, F., Fischer, R., Groeneveld, J., Lehmann, S., Müller, M. S., Rödig, E., et al. (2018). Global patterns of tropical forest fragmentation. *Nature* 554, 519–522. doi: 10.1038/nature25508

Te Wierik, S. A., Cammeraat, E. L. H., Gupta, J., and Artzy-Randrup, Y. A. (2021). Reviewing the impact of land use and land-use change on moisture recycling and precipitation patterns. *Water Resour. Res.* 57:e2020WR029234. doi: 10.1029/2020WR029234

Thiel, C., Drezet, P., Weise, C., Quegan, S., and Schmullius, C. (2006). Radar remote sensing for the delineation of forest cover maps and the detection of deforestation. *For. Int. J. For. Res.* 79, 589–597. doi: 10.1093/forestry/cpl036

Thonfeld, F., Gessner, U., Holzwarth, S., Kriese, J., Da Ponte, E., and Kuenzer, C. (2022). *Recent forest cover loss in Germany after the 2018-2020 drought years. Bonn, Germany*. Available online at: https://elib.dlr.de/186889/ (accessed August 23, 2022).

Thrippleton, T., Lüscher, F., and Bugmann, H. (2020). Climate change impacts across a large forest enterprise in the Northern Pre-Alps: Dynamic forest modelling as a tool for decision support. *Eur. J. For. Res.* 139, 483–498. doi: 10.1007/s10342-020-01263-x

Tropek, R., Sedláček, O., Beck, J., Keil, P., Musilová, Z., Šímová, I., et al. (2014). Comment on 'High-resolution global maps of 21st-century forest cover change.' *Science* 344, 981–981.

Tuff, K. T., Tuff, T., and Davies, K. F. (2016). A framework for integrating thermal biology into fragmentation research. *Ecol. Lett.* 19, 361–374. doi: 10.1111/ele.12579

UNESCO (2022). *Ancient and primeval beech forests of the carpathians and other regions of Europe. UNESCO World Heritage Centre*. Available online at: https://whc.unesco.org/en/list/1133/ (accessed October 10, 2022).

Vieilledent, G., Grinand, C., Rakotomalala, F. A., Ranaivosoa, R., Rakotoarijaona, J. R., Allnutt, T. F., et al. (2018). Combining global tree cover loss data with historical national forest cover maps to look at six decades of deforestation and forest fragmentation in Madagascar. *Biol. Conserv.* 222, 189–197.

Voloscuk, I., Pichler, V., and Pichlerová, M. (2013). The primeval beech forests of the carpathians and ancient beech forests of Germany: Joint natural heritage of Europe. *Folia Oecol.* 40:295.

Wan, Z., Hook, S., and Hulley, G. (2015). *Collection-6MODIS land surface temperature productsusers' guide. NASA EOSDIS land processes DAAC*. Available online at: https://doi.org/10.5067/MODIS/MYD11A1.006 (accessed June 7, 2022).

Welle, T., Aschenbrenner, L., Kuonath, K., Kirmaier, S., and Franke, J. (2022). Mapping dominant tree species of German forests. *Remote Sens.* 14:3330. doi: 10.3390/rs14143330

Weng, Q., Lu, D., and Schubring, J. (2004). Estimation of land surface temperature–vegetation abundance relationship for urban heat island studies. *Remote Sens. Environ.* 89, 467–483. doi: 10.1016/j.rse.2003.11.005

Wu, L., Zhou, H., Li, J., Li, K., Sun, X., Lu, S., et al. (2019). Thiessen polygon analysis and spatial pattern evolution of neolithic cultural sites (8.0–4.0 Ka BP) in Huaibei Plain of Anhui, East China. *Quaternary Int.* 521, 75–84. doi: 10.1016/j.quaint.2019.06.005

Zhan, X., Kustas, W. P., and Humes, K. S. (1996). An intercomparison study on models of sensible heat flux over partial canopy surfaces with remotely sensed surface temperature. *Remote Sens. Environ.* 58, 242–256.

Zink, M., Samaniego, L., Kumar, R., Thober, S., Mai, J., Schäfer, D., et al. (2016). The German drought monitor. *Environ. Res. Lett.* 11:074002. doi: 10.1088/1748-9326/11/7/074002

2.4. Article IV

Low effectiveness of the world network of biosphere reserves in maintaining forest ecosystem functions

This article was submitted to *Science* for peer review on March 26, 2024, and follows the journal's corresponding referencing and formatting style.

Low effectiveness of the world network of biosphere reserves in maintaining forest ecosystem functions

Charlotte Gohr[1,2*], Henrik von Wehrden[2], Sassan Saatchi[3,4], Nathalie Pettorelli[5], Pierre L. Ibisch[1]

1 Biosphere Reserves Institute and Centre for Econics and Ecosystem Management, Eberswalde University for Sustainable Development; Eberswalde, 16625, Germany.

2 Center of Methods and Faculty of Sustainability, Leuphana University; Lüneburg, 21335, Germany.

3 Jet Propulsion Laboratory, California Institute of Technology; Pasadena, CA 91109, USA.

4 Institute of Environment, University of California; Los Angeles, CA 90095, USA.

5 Institute of Zoology, Zoological Society of London; London, NW1 4RY, UK.

*Corresponding author. Email: charlotte.gohr@leuphana.de

Abstract. UNESCO biosphere reserves serve as learning areas for sustainable development, where preserving ecosystem functionality is an imperative. However, this critical assumption has yet to be thoroughly examined. To address this knowledge gap, we investigated differences in satellite-derived proxies of ecosystem functions between inside and surrounding areas of forest biosphere reserves, globally. Our findings show that (i) only 18 of 119 forest biosphere reserves exhibited higher values for all ecosystem function proxies inside the reserves compared to outside, (ii) smaller reserves in fragmented forestscapes were more affected by hot day temperatures, and (iii) greater forest cover correlated with increased ecosystem functioning across all biomes. This study underscores the significance of biosphere reserves to biodiversity conservation efforts and the need for the integration of satellite-based, outcome oriented proxies of ecosystem functions in assessments of protected area effectiveness.

One sentence summary. When considering multiple ecosystem functions, most forests within the world network of biosphere reserves do not perform better than those outside.

Main Text. Forests are under pressure globally, largely attributed to human induced climate change with rising temperature, shifting precipitation patterns and an increase in extreme heat events and droughts (*1*–*6*). Human activities threaten forests through fragmentation and degradation, thereby endangering crucial and unique habitats (*7*–*10*). Yet humans depend on forest ecosystems for carbon storage, temperature regulation, soil fertility, resource use and recreational space (*11*, *12*). Protected areas represent a key measure for establishing the protection and conservation of essential ecosystem services and habitats (*13*–*15*). They play a major role in preserving various dimensions of biodiversity across different scales (*16*), mitigating climate change (*13*) and safeguarding ecosystem functions (*17*). Extending the coverage of protected areas is one target of the recently adopted Kunming-Montreal Global Biodiversity Framework 2022 (Target 3, *18*). However, before establishing additional protected areas, it is imperative to evaluate the effectiveness of existing protected areas and their associated outcomes. However, this evaluation cannot be solely based on measuring protection coverage (*19*, *20*). The effectiveness of protected areas has been investigated and defined in various ways, often referring to the achievement of specific conservation targets within these areas (*21*) or equating it to management effectiveness (*22*). Here, we define protected area effectiveness in terms of their contribution to sustaining ecosystem health and functioning.

The International Union of Conservation of Nature (IUCN) established a set of categories as a comprehensive framework to designate protected areas with a primary focus on biodiversity preservation (*23*). UNESCO's biosphere reserves, established under the Man and the Biosphere programme, differ from strict nature reserves in that they serve as integrated areas for both nature and human activities, acting as model areas for sustainable development (*24–26*). This globally applied framework spans across 748 reserves, typically organized into three zones with varying levels of management (*25*, *27*). The core, buffer and transition zones range from stricter protection to sustainable use of resources and the integration of education and research. 174 of these reserves have known forest cover. While these protected, forest-covered areas are widely diverse, they are all united within the world network of biosphere reserves. If biosphere reserves are effective in forest biomes, it should be possible to objectively measure their contribution to biodiversity conservation and to establish that ecosystems inside these model areas for sustainable development are better off than outside. To date, the global analysis of the ecological effectiveness of protected areas has depended on comparable measures based on globally available data (*28*). To our knowledge, this is the first time that the effectiveness of forested UNESCO Biosphere Reserves has been tested using remotely sensed data. This was achieved by analyzing multiple proxies of ecosystem functions in all forest biomes found in biospheres and comparing them to their surroundings. Our research offers insights relevant to optimizing sustainable forest management strategies in Biosphere Reserves and potentially other forested protected areas worldwide.

Remotely sensed forest ecosystem functions

Satellite remote sensing imagery stands as the sole source of information presently available that can be linked to ecosystem functioning, offering global coverage with relatively high temporal and spatial resolutions (*29*). We build upon the definition of ecosystem functions as the direct or indirect dynamics of ecosystem processes, such as primary productivity and evapotranspiration (*29*), and the ability of those ecosystems to adapt to internal and external changes (*30*). Employing multiple proxies for primary productivity helps identify different patterns of ecosystem functioning across a wide range of forested biomes (Fig. 1) (*31*). In this study, we used gross and net primary productivity, the normalized difference vegetation index (NDVI) and the enhanced vegetation index (EVI). Additionally, we incorporated temperature regulation capacity, evapotranspiration and connectivity as pertinent proxies of forest ecosystem function (*29*, *32*, *33*). To assess the effectiveness of forest biosphere reserves worldwide, we compared their ecosystem functioning with their surrounding areas based on the seven proxies at a 1 km resolution. Furthermore, we investigated changes over time in two-time steps from 2010 to 2016 and 2017 to 2022. We highlight the impact of proxy selection on the results, emphasizing the need for multiproxy analyses to obtain a comprehensive understanding of forest ecosystem functioning. Remote sensing offers comparable, freely available global datasets and the spatial and temporal resolution of these datasets will increase even more in the future (*29*). We appreciate the easy access and large computational capacities of cloud-computing platforms such as the Google Earth Engine.

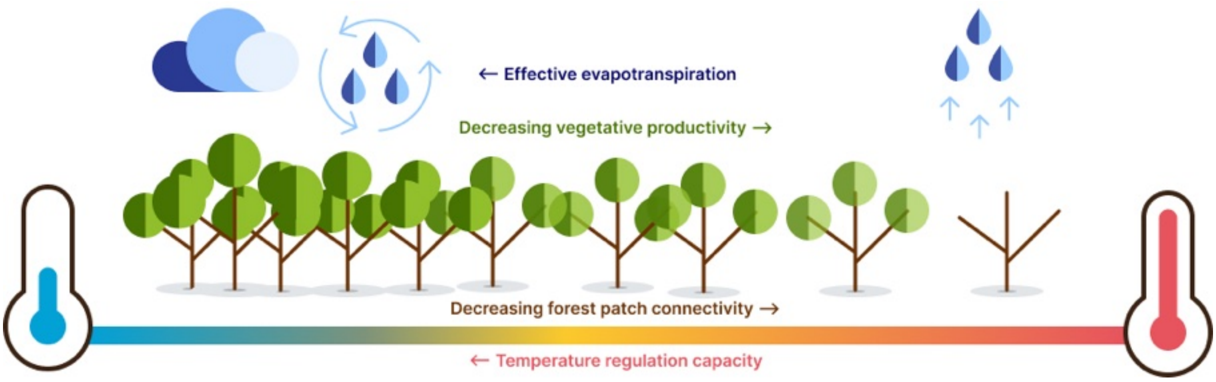


Fig. 1: Graphical interpretation of the forest ecosystem function proxies used in this study. Variations in vegetation vitality assessed via gross and net primary productivity as well as via the enhanced and normalized difference vegetation index; Variations in temperature regulation capacities via hot day land surface temperatures; Variations in precipitation and water regulation via evapotranspiration and variations in ecosystem extents and state via forest patch connectivity.

Effectiveness of forest biosphere reserves

At least 242 (32%) of the global biosphere reserves are in forested biomes, according to the available data. Of those, 119 had correct border information and contain enough tree cover to assess forest ecosystem functions. We defined tree cover as a pixel with >30% at 30m resolution and >50% at 1 km resolution. Biosphere reserves were deemed to contain enough tree cover for analysis when the land cover share of tree covered areas exceeded 20% inside the biosphere reserve (Fig. S1, Tab. S1). We extracted remotely sensed ecosystem functioning proxies within biosphere reserves and in their surroundings, leading to more than

6 million single observations. We defined the surroundings via a buffer using the natural logarithm of the size of each area multiplied by 500. The buffer was then bound to a rectangular format for faster processing (Fig. S1). The study used globally standardized data and methodology to allow for direct comparisons of forest ecosystem functions of biosphere reserves across forested biomes and for two periods of time (Fig. 1). We modeled forest ecosystem functions inside biosphere reserves against their surroundings and the effects of place (biome) and time on seven proxies (Fig. 2, Tab. S2). Our approach on defining the effectiveness of biosphere reserves in maintaining multiple forest ecosystem functions (E) can be read as follows:

$$E_{br} = \ (Pin_1 > Pout_1) \wedge \cdots \wedge \ (Pin_7 > Pout_7)$$

$P_{1\text{-}7}$ are the seven proxies of forest ecosystem functions modeled with their corresponding location per biome for two time steps and concerning their forest cover share. P_{in} and P_{out} define the inside and surroundings of each biosphere reserve respectively. An effective biosphere reserve (E_{br}) is given, if all proxies have higher modeled forest ecosystem functions inside as compared to their surroundings. The explained variance for these seven proxy models was for fixed effects R^2 = 0.11-0.38 and for random effects R^2 = 0.91-0.99 (Tab.S3). The proxy models with the inside-outside comparison as a fixed effect rather than as an explanatory variable showed for fixed effects R^2 = 0.21-0.82 and for random effects R^2 = 0.5-0.91 (Tab.S4, Fig. 3). Of 119 biosphere reserves, only 18 (15%) show higher ecosystem functions inside than outside when considering all functions (Tab. S2, Fig. 2).

Biome-specific characteristics are an underlying pattern for all proxies, especially for gross primary productivity and evapotranspiration (Fig. 2A and 2C). Modeled primary productivity and connectivity were higher for most biosphere reserves inside than outside (Fig. 2A and 2B). Hot day temperatures and evapotranspiration were predicted lower inside than outside (Fig. 2C and 2D). Substantial temporal changes (from dots to triangles) in proxy medians were rare except for forest patch connectivity, which decreased substantially for some biosphere reserves (Fig. 2B). This shows that conditions were maintained over time for single proxies and single biosphere reserves: for example, the Fontainebleau Biosphere Reserve, close to Paris, has a rather low level of connectivity to other forested patches in the landscape, but connectivity was still found to be higher inside the reserve than outside. The same applies to two coastal biosphere reserves, Sian Kaan in Mexico and Georgian Bay in Canada, which have a higher connectivity inside than outside. The Xiriualtique Jiquitizco in El Salvador is a coastal biosphere reserve as well: while for this biosphere reserve gross primary productivity is slightly higher inside, all other proxies (evapotranspiration, hot-day temperature and forest patch connectivity) show higher values outside the biosphere reserve (Fig. S3).

We anticipated and confirmed higher connectivity of forest patches within 85 biosphere reserves than outside them (Tab. S5). Yet substantial changes occurred over time, with a decline observed in and around most biosphere reserves (Fig. S4).

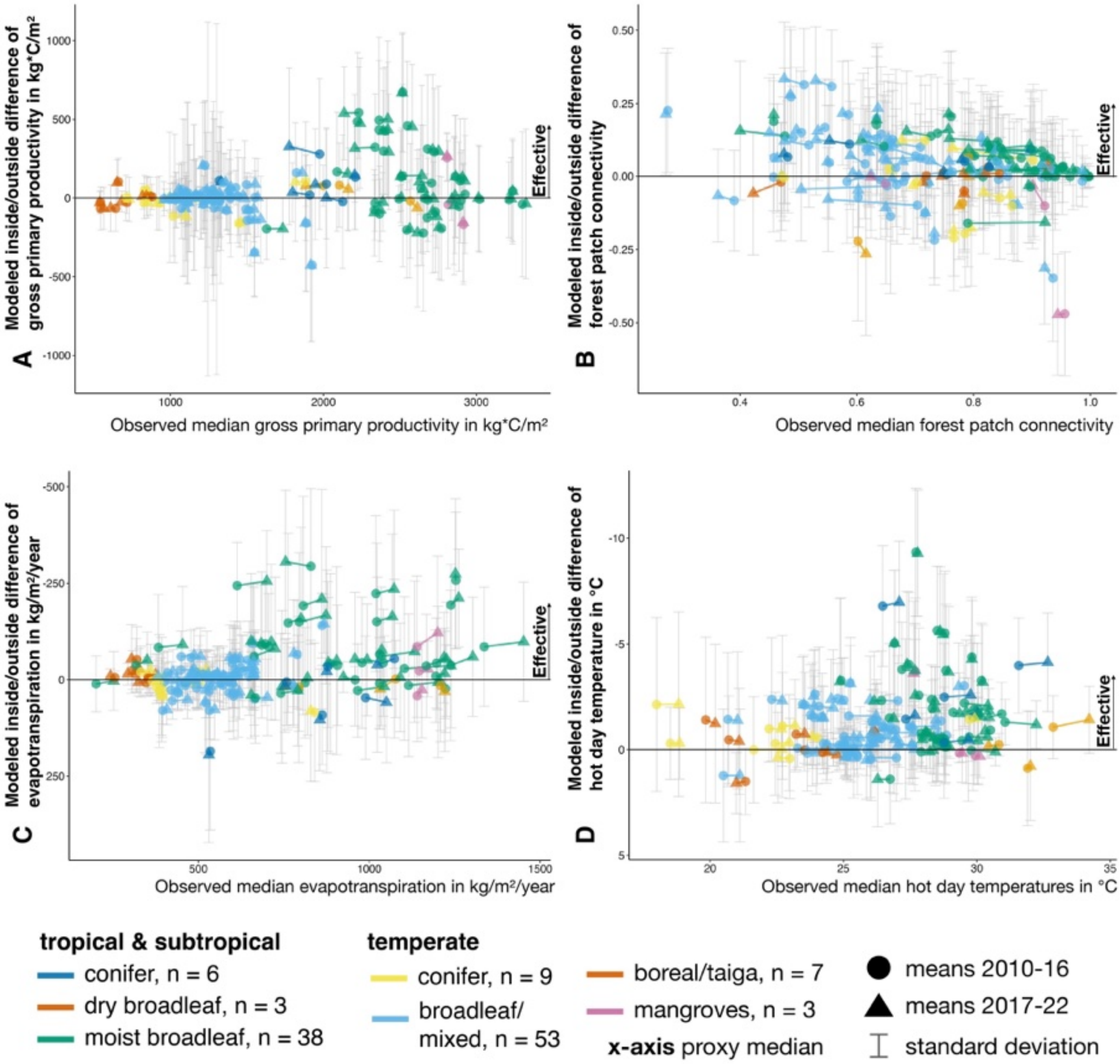


Fig. 2. Model predictions and standard deviation for proxy difference for selected forest ecosystem functions inside and outside the 119 biosphere reserves. Data depicted for two times 2010-16 (dots) and 2017-22 (triangles) against the respective median of the raw data. Predictions above 0 indicate higher proxy values inside than outside. Predictions below 0 indicate the opposite. Colors refer to biomes. Modeled inside/outside difference of (**A**) Gross primary productivity in kg*C/m^2, (**B**) Forest patch connectivity, (**C**) Evapotranspiration in kg/m^2/year, (**D**) Hot day land surface temperature in °C. Difference model with fixed effects of forest cover share biome and two periods of time 2010-16 and 2017-22. Note that y-axes for (C) Evapotranspiration and (D) Hot day land surface temperature are reversed to visualize the effectiveness in water captivity and cooling on top like for (A) Gross primary productivity and (B) Forest patch connectivity.

Modeled primary productivity (gross and net primary productivity, the enhanced and normalized difference vegetation index) was higher inside than outside for most biosphere reserves and increased slightly from 2010-16 to 2017-22 for all but the forests in the tropical dry broadleaved and the boreal biome. This general increase might be attributed to the general trend of global greening (*34*). We detected the highest primary productivity and the highest difference to the surroundings in the Selva el Ocote Biosphere Reserve in Mexico, a tropical moist broadleaved forest area. The modeled effect of lower evapotranspiration within biosphere reserves was more pronounced in tropical and subtropical broadleaved forests than in temperate broadleaved forests (Fig. 2C). Forest ecosystem function proxies in and around biosphere reserves generally decreased over time but showed minor biome-specific differences: predicted primary productivity increased slightly in tropical broadleaved and temperate coniferous forests and decreased slightly in tropical coniferous forests. In all biomes, hot day temperatures increased, and connectivity decreased. Changes were more pronounced outside than inside biosphere reserves, which can be interpreted as an indication of effectiveness (S4). Apart from the assumed higher functionality of temperature and water regulation within biosphere reserves, a general increase in temperatures through global warming was confirmed.

Global distribution of effective forest biosphere reserves varies across biomes

To date, no complete dataset of the world network of biosphere reserves is available. We found 474 datasets, of which 119 inherit enough forest cover to assess ecosystem functions (Fig. 3A, Supplementary Material: area selection). We selected three proxies to display the modeled effect size of each biosphere reserve (Fig. 3B, Tab. S4 and S6). In temperate biosphere reserves, forest patch connectivity is a higher influencing factor than gross primary productivity or temperature: high and low connectivity differences with their surroundings are more pronounced than in other forest biomes (Fig. 3B Europe, blue dots). The fragmentation of temperate forests is caused and intensified by intensive forestry, transport routes and secondary causes from rising temperatures, such as pests and continues to weaken the forest ecosystem functionality (*33*). Tropical and subtropical biomes are defined by temperatures with higher random effects for hot day temperatures than for primary productivity or forest patch connectivity (Fig. 3 Central America and Central Africa, pink and orange dots). This effect can be interpreted as a decreasing resilience due to rising temperatures and decreasing water availability in tropical forests (*6*).

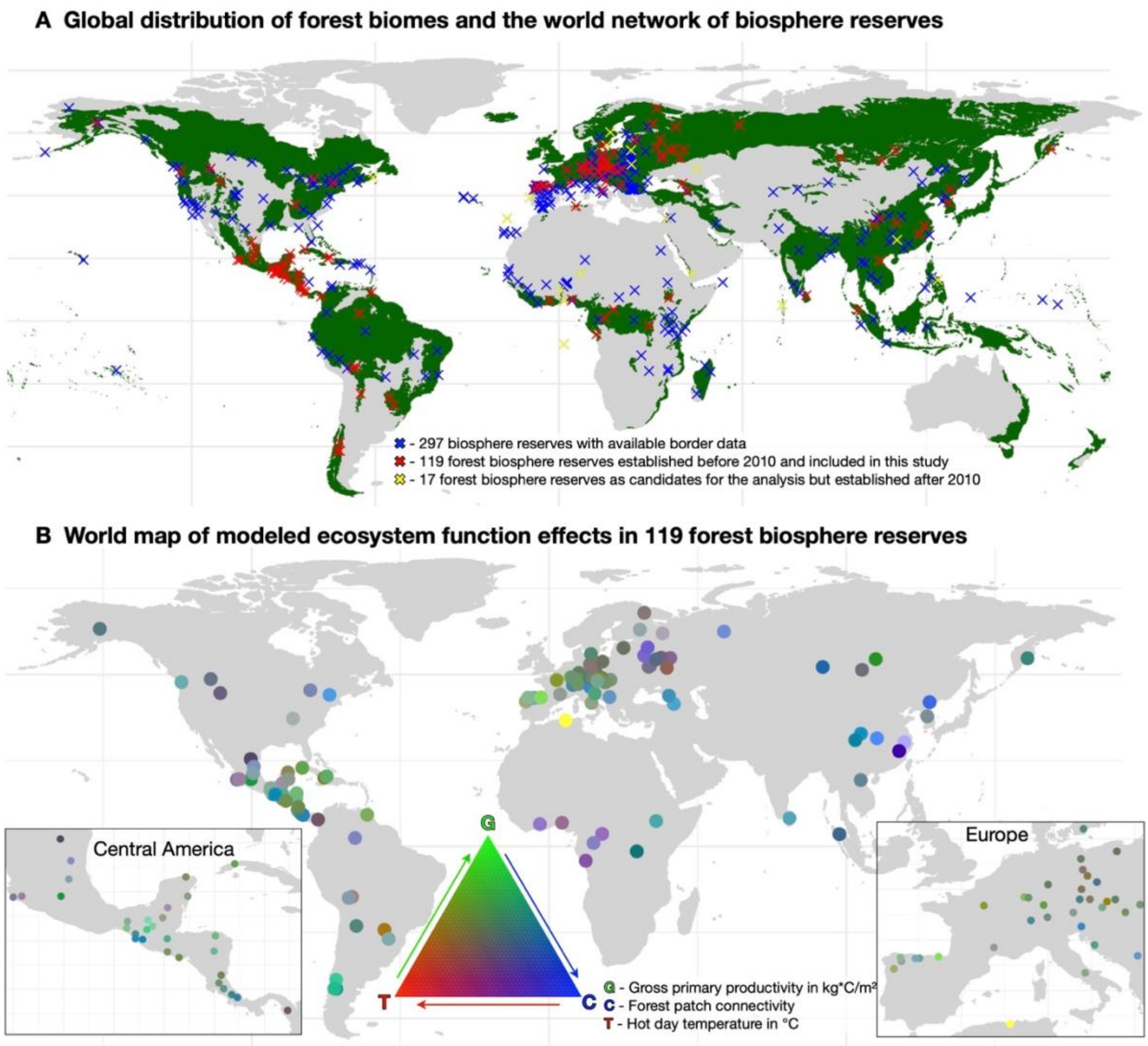


Fig. 3. Forest ecosystem functions in biosphere reserves.[1] (**A**) World map forest biomes and UNESCO biosphere reserves. Forest biomes include boreal forests/taiga, mangroves, temperate broadleaf & mixed forests, temperate conifer forests, tropical & subtropical coniferous forests, tropical & subtropical dry broadleaf forests, tropical & subtropical moist broadleaf forests; biosphere reserves: blue = 297 available border data; of those: yellow = 17 established after 2010; red = 119 forest covered and included in this study. (**B**) World map of ecosystem function effects in 119 biosphere reserves. The biosphere reserves served as random effects for the three modeled proxies: hot-day temperature, connectivity and gross primary productivity with fixed effects of forest cover share, inside or outside the biosphere reserve, biome and two periods of time 2010-16 and 2017-22. Each proxy's random effects were scaled, multiplied by 255, and transformed into RGB values. The colors follow the triangle legend and show the corresponding effect sizes of the three proxies. Smaller windows show the same effects for Central America and Europe.

[1] The color coding in map B differs from the original submission of the thesis framework. It was adapted to this corrected version for this publication.

Across biomes, we predicted for only 18 of 119 forest biosphere reserves a higher primary productivity and connectivity and lower temperatures on hot days and lower evapotranspiration compared to their surroundings in 2017-2022 (Tab. S5). When reducing the number of proxies from seven to only three (Gross primary productivity, temperature and connectivity), the number of effective biosphere reserves increases to 42 (35 %) (Fig. 3B). Smaller reserves displayed more pronounced random effects towards temperature (Fig. 4), suggesting that larger reserves generally exhibit greater temperature regulation capacities, likely attributable to their higher forest cover shares (*32*). Similarly, the random effects of forest patch connectivity are higher for most of the smaller biosphere reserves. Connectivity in larger biosphere reserves might be more balanced.

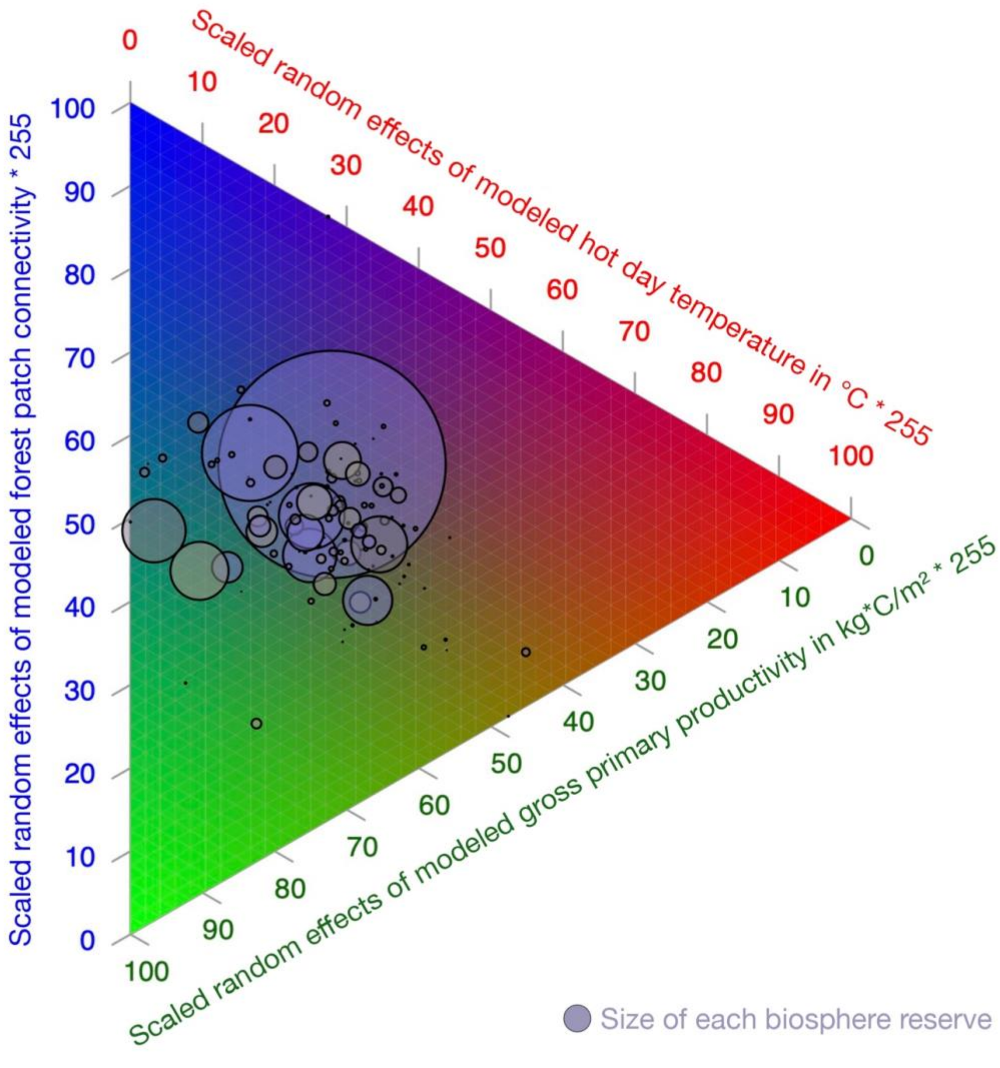


Fig. 4. Modeled random effects of 119 biosphere reserves for three forest ecosystem function proxies. Model with fixed effects of forest cover share, inside or outside the biosphere reserve, biome and two periods of time 2010-16 and 2017-22. Each random effect was scaled, multiplied by 255 and transformed into RGB-values for three ecosystem function proxies namely connectivity, hot-day temperature and gross primary productivity. Each circle represents one biosphere reserve, and the circle size relates to the area size of each reserve. Ternary plot created with the *Ternary*-package in R (*35*).

Higher tree cover share constitutes healthy forests in biosphere reserves worldwide

Increased tree cover shares exert a significant impact on ecosystem functions, as shown by all forest ecosystem function proxies in our study (Fig. 5). We defined tree cover shares as the percentage of tree cover per pixel based on the GLAD tree cover data (Fig. S1, Tab. S1). A linear relationship emerged: as forest cover shares rose, so did the primary productivity, while temperatures declined, evapotranspiration decreased and forest patch connectivity increased. This trend holds true across all forest biomes (Fig. S7). The tropics exhibited a more pronounced increase in primary productivity with higher forest cover shares compared to temperate latitudes. Modeled gross primary productivity and the Enhanced Vegetation Index (EVI) were higher outside at the highest forest cover shares, which is likely related to ecosystem functionality on a landscape level. The EVI's standard deviations were more pronounced than those of other proxies, perhaps partly explained by its sensitivity to greenness in tropical forests. In tropical and subtropical biomes, biosphere reserves exhibited higher primary productivity compared to their surroundings, whereas the reverse holds for temperate forests. In boreal forests, temperatures were nearly the same inside and outside. Biosphere reserves in the boreal biome indicated the lowest primary productivity values and decreased over time. This could be related to the observed boreal forest biome shift with less productivity assessed (*36*). One anomaly was the three biosphere reserves with tropical and subtropical dry broadleaf forests for which higher evapotranspiration outside than inside was predicted. Modeled forest patch connectivity increased with increasing forest cover shares throughout the biomes, except for tropical and subtropical dry broadleaf forests and mangroves (Fig. S7). Generally, higher connectivity did not translate into higher biodiversity, since large connected forest areas do not necessarily harbor greater biodiversity than smaller patches (*37*).

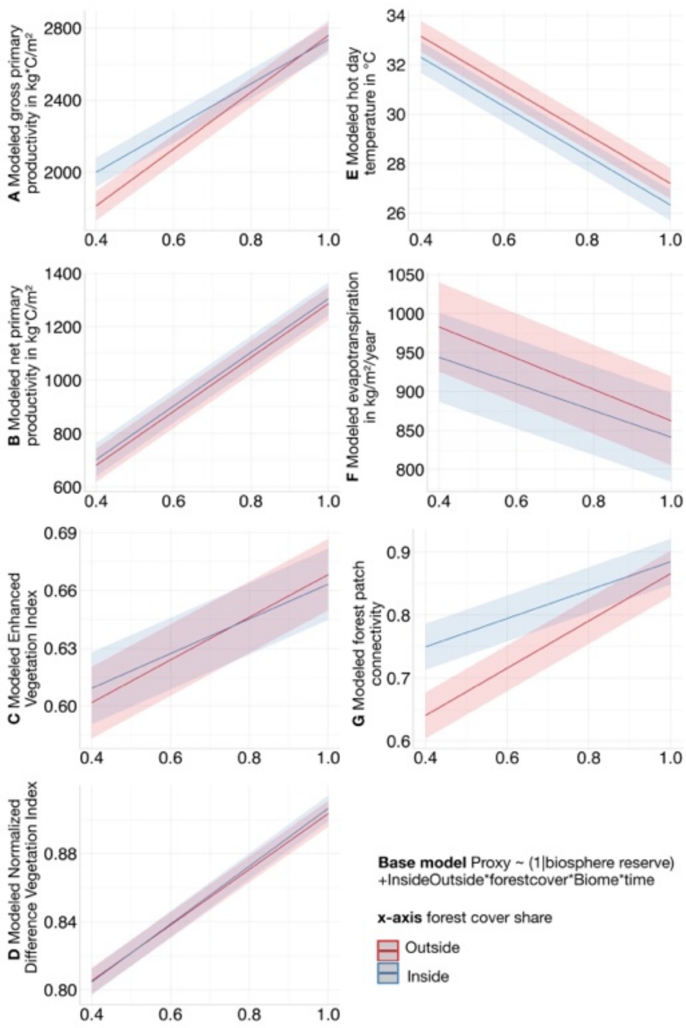


Fig. 5. Modeled ecosystem functioning proxies for forest cover shares in 119 biosphere reserves. (**A**) Gross primary productivity, (**B**) Net primary productivity, (**C**) Enhanced vegetation index, (**D**) Normalized difference vegetation index, (**E**) Hot day land surface temperature, (**F**) Evapotranspiration and (**G**) Forest patch connectivity. Modeled observations are displayed in blue for forests inside biosphere reserves and in red for surrounding forests. Ribbons represent 0.95 confidence intervals. Model with fixed effects of forest cover share, inside or outside the biosphere reserve, biome and two periods of time 2010-16 and 2017-22.

Measuring the effectiveness of biosphere reserves and other area-based conservation instruments

While there is general agreement that the effectiveness of conservation and sustainable development requires an indicator-based assessment, the indicators proposed to date for assessing forest conservation are relatively simple. Rather than measure-based data such as forest area under formal protection or certified management (e.g. CBD's Global Biodiversity Framework) (Target 10, *18*), which only indicates the intention to improve forest management, we urgently need outcome indicators. These must allow us to assess, with high temporal and spatial resolution, whether management promises are being kept. In this context, we need to make better use of available remote sensing data, which can be used to analyze not only tree cover but also the health status and trends of forest ecosystems. The Ad Hoc Technical Expert Group (AHTEG) on Indicators for the Kunming-Montreal Global Biodiversity Framework (*18*) will need to ensure that the indicators to be used do not lag behind rapidly evolving technological capabilities, including in the context of cloud computing and artificial intelligence. We show that a global approach is effective in providing insights into regional characteristics. However, improved monitoring at finer spatial scales is recommended for local management.

In any case, we suggest that a multi-proxy methodology focusing on ecosystem performance, as implemented in this study for UNESCO biosphere reserves, could be applied to other protected area frameworks. The approach could also be adapted to other ecosystems and land cover types. Clearly, a basic requirement is reliable and up-to-date georeferenced data on area-based interventions. In this context, we regret that there is no official source for the boundaries of all UNESCO biosphere reserves, which limits their evaluability.

In some reviews, biosphere reserves are excluded from the assessment of effectiveness (*38–40*), with reference to doubts as to whether they actually represent effective conservation action or merely bureaucratic labels (*41*). Here we would like to disagree and emphasize the role that biosphere reserves can play as modern instruments for safeguarding the functioning of the biosphere - not through fortress conservation, but under a type of governance that includes people and their needs. However, UNESCO biosphere reserves cannot just be model areas for participatory management without delivering benefits to ecosystems. Particularly in the context of the rapidly advancing climate crisis, human well-being will increasingly depend on the health of ecosystems. Any conservation action that does not have a measurable positive impact on ecosystem functioning will lose its credibility and justification.

References and Notes

1. F. Giorgi, F. Raffaele, E. Coppola, The response of precipitation characteristics to global warming from climate projections. Earth System Dynamics 10, 73–89 (2019).
2. X. Zeng, Z. Hu, A. Chen, W. Yuan, G. Hou, D. Han, M. Liang, K. Di, R. Cao, D. Luo, The global decline in the sensitivity of vegetation productivity to precipitation from 2001 to 2018. Global Change Biology 28, 6823–6833 (2022).
3. S. Trumbore, P. Brando, H. Hartmann, Forest health and global change. Science 349, 814–818 (2015).
4. A. Gazol, J. J. Camarero, Compound climate events increase tree drought mortality across European forests. Science of The Total Environment 816, 151604 (2022).
5. W. M. Hammond, A. P. Williams, J. T. Abatzoglou, H. D. Adams, T. Klein, R. López, C. Sáenz-Romero, H. Hartmann, D. D. Breshears, C. D. Allen, Global field observations of tree die-off reveal hotter-drought fingerprint for Earth's forests. Nat Commun 13, 1761 (2022).
6. G. Forzieri, V. Dakos, N. G. McDowell, A. Ramdane, A. Cescatti, Emerging signals of declining forest resilience under climate change. Nature 608, 534–539 (2022).
7. J. Ma, J. Li, W. Wu, J. Liu, Global forest fragmentation change from 2000 to 2020. Nat Commun 14, 3752 (2023).

8. P. L. Ibisch, M. T. Hoffmann, S. Kreft, G. Pe'er, V. Kati, L. Biber-Freudenberger, D. A. DellaSala, M. M. Vale, P. R. Hobson, N. Selva, A global map of roadless areas and their conservation status. Science 354, 1423–1427 (2016).
9. D. M. Lapola, P. Pinho, J. Barlow, L. E. O. C. Aragão, E. Berenguer, R. Carmenta, H. M. Liddy, H. Seixas, C. V. J. Silva, C. H. L. Silva-Junior, A. A. C. Alencar, L. O. Anderson, D. Armenteras, V. Brovkin, K. Calders, J. Chambers, L. Chini, M. H. Costa, B. L. Faria, P. M. Fearnside, J. Ferreira, L. Gatti, V. H. Gutierrez-Velez, Z. Han, K. Hibbard, C. Koven, P. Lawrence, J. Pongratz, B. T. T. Portela, M. Rounsevell, A. C. Ruane, R. Schaldach, S. S. da Silva, C. von Randow, W. S. Walker, The drivers and impacts of Amazon forest degradation. Science 379, eabp8622 (2023).
10. W. Li, W.-Y. Guo, M. Pasgaard, Z. Niu, L. Wang, F. Chen, Y. Qin, J.-C. Svenning, Human fingerprint on structural density of forests globally. Nat Sustain 6, 368–379 (2023).
11. E. Karjalainen, T. Sarjala, H. Raitio, Promoting human health through forests: overview and major challenges. Environ Health Prev Med 15, 1–8 (2010).
12. M. M. Rahman, M. Zimmer, I. Ahmed, D. Donato, M. Kanzaki, M. Xu, Co-benefits of protecting mangroves for biodiversity conservation and carbon storage. Nat Commun 12, 3875 (2021).
13. L. Duncanson, M. Liang, V. Leitold, J. Armston, S. M. Krishna Moorthy, R. Dubayah, S. Costedoat, B. J. Enquist, L. Fatoyinbo, S. J. Goetz, M. Gonzalez-Roglich, C. Merow, P. R. Roehrdanz, K. Tabor, A. Zvoleff, The effectiveness of global protected areas for climate change mitigation. Nat Commun 14, 2908 (2023).
14. C. Wolf, T. Levi, W. J. Ripple, D. A. Zárrate-Charry, M. G. Betts, A forest loss report card for the world's protected areas. Nat Ecol Evol, 1–10 (2021).
15. X. Xu, A. Huang, E. Belle, P. De Frenne, G. Jia, Protected areas provide thermal buffer against climate change. Science Advances 8, eabo0119 (2022).
16. C. L. Gray, S. L. Hill, T. Newbold, L. N. Hudson, L. Börger, S. Contu, A. J. Hoskins, S. Ferrier, A. Purvis, J. P. Scharlemann, Local biodiversity is higher inside than outside terrestrial protected areas worldwide. Nature Communications 7, 12306 (2016).
17. Z. Tang, J. Fang, J. Sun, K. J. Gaston, Effectiveness of Protected Areas in Maintaining Plant Production. Plos One 6, e19116 (2011).
18. CBD, Kunming-Montreal Global Biodiversity Framework. CBD/COP/DEC/15/4. Montreal: Convention on Biological Diversity. 14. (2022).
19. O. Venter, R. A. Fuller, D. B. Segan, J. Carwardine, T. Brooks, S. H. M. Butchart, M. D. Marco, T. Iwamura, L. Joseph, D. O'Grady, H. P. Possingham, C. Rondinini, R. J. Smith, M. Venter, J. E. M. Watson, Targeting Global Protected Area Expansion for Imperiled Biodiversity. PLOS Biology 12, e1001891 (2014).
20. S. Chape, J. Harrison, M. Spalding, I. Lysenko, Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. Phil. Trans. R. Soc. B 360, 443–455 (2005).
21. A. S. L. Rodrigues, S. J. Andelman, M. I. Bakarr, L. Boitani, T. M. Brooks, R. M. Cowling, L. D. C. Fishpool, G. A. B. da Fonseca, K. J. Gaston, M. Hoffmann, J. S. Long, P. A. Marquet, J. D. Pilgrim, R. L. Pressey, J. Schipper, W. Sechrest, S. N. Stuart, L. G. Underhill, R. W. Waller, M. E. J. Watts, X. Yan, Effectiveness of the global protected area network in representing species diversity. Nature 428, 640–643 (2004).
22. J. Geldmann, L. Coad, M. Barnes, I. D. Craigie, M. Hockings, K. Knights, F. Leverington, I. C. Cuadros, C. Zamora, S. Woodley, N. D. Burgess, Changes in protected area management effectiveness over time: A global analysis. Biological Conservation 191, 692–699 (2015).
23. N. Dudley, Guidelines for Applying Protected Area Management Categories (IUCN, 2008; https://portals.iucn.org/library/node/9243).
24. UNESCO, Lima Action Plan for UNESCO's Man and the Biosphere (MAB) Programme and its World Network of Biosphere Reserves (2016-2025). (2016).

25. UNESCO, Statutory framework of the world network of biosphere reserves. (1996).
26. UNESCO, MAB STRATEGY 2015-2025. (2015).
27. UNESCO, World Network of Biosphere Reserves, UNESCO (2024). https://en.unesco.org/biosphere/wnbr.
28. C. Gohr, H. von Wehrden, F. May, P. L. Ibisch, Remotely sensed effectiveness assessments of protected areas lack a common framework: A review. Ecosphere 13, e4053 (2022).
29. N. Pettorelli, H. S. to Bühne, A. Tulloch, G. Dubois, C. Macinnis-Ng, A. M. Queirós, D. A. Keith, M. Wegmann, F. Schrodt, M. Stellmes, R. Sonnenschein, G. N. Geller, S. Roy, B. Somers, N. Murray, L. Bland, I. Geijzendorffer, J. T. Kerr, S. Broszeit, P. J. Leitão, C. Duncan, G. E. Serafy, K. S. He, J. L. Blanchard, R. Lucas, P. Mairota, T. J. Webb, E. Nicholson, Satellite remote sensing of ecosystem functions: opportunities, challenges and way forward. Remote Sensing in Ecology and Conservation 4, 71–93 (2018).
30. L. Freudenberger, P. R. Hobson, M. Schluck, P. L. Ibisch, A global map of the functionality of terrestrial ecosystems. Ecological Complexity 12, 13–22 (2012).
31. L. Huang, Q. Shao, J. Liu, Assessing the conservation effects of nature reserve networks under climate variability over the northeastern Tibetan plateau. Ecological Indicators 96, 163–173 (2019).
32. C. Gohr, J. S. Blumröder, D. Sheil, P. L. Ibisch, Quantifying the mitigation of temperature extremes by forests and wetlands in a temperate landscape. Ecological Informatics 66, 101442 (2021).
33. D. Mann, C. Gohr, J. S. Blumröder, P. L. Ibisch, Does fragmentation contribute to the forest crisis in Germany? Frontiers in Forests and Global Change 6 (2023).
34. S. Piao, X. Wang, T. Park, C. Chen, X. Lian, Y. He, J. W. Bjerke, A. Chen, P. Ciais, H. Tømmervik, R. R. Nemani, R. B. Myneni, Characteristics, drivers and feedbacks of global greening. Nat Rev Earth Environ 1, 14–27 (2020).
35. M. R. Smith, L. Sanselme, Ternary: Create Ternary and Holdridge Plots, version 2.3.1 (2024); https://cran.r-project.org/web/packages/Ternary/index.html.
36. L. T. Berner, S. J. Goetz, Satellite observations document trends consistent with a boreal forest biome shift. Global Change Biology 28, 3275–3292 (2022).
37. F. Riva, C. J. Martin, K. Millard, L. Fahrig, Loss of the world's smallest forests. Global Change Biology 28, 7164–7166 (2022).
38. K. R. Jones, O. Venter, R. A. Fuller, J. R. Allan, S. L. Maxwell, P. J. Negret, J. E. M. Watson, One-third of global protected land is under intense human pressure. Science 360, 788–791 (2018).
39. R. A. Neugarten, R. Chaplin-Kramer, R. P. Sharp, R. Schuster, M. Strimas-Mackey, P. R. Roehrdanz, M. Mulligan, A. van Soesbergen, D. Hole, C. M. Kennedy, Mapping the planet's critical areas for biodiversity and nature's contributions to people. Nature Communications 15, 261 (2024).
40. G. Ceccherini, M. Girardello, P. S. A. Beck, M. Migliavacca, G. Duveiller, G. Dubois, V. Avitabile, L. Battistella, J. I. Barredo, A. Cescatti, Spaceborne LiDAR reveals the effectiveness of European Protected Areas in conserving forest height and vertical structure. Commun Earth Environ 4, 1–13 (2023).
41. K. L. Coetzer, E. T. F. Witkowski, B. F. N. Erasmus, Reviewing Biosphere R eserves globally: effective conservation action or bureaucratic label? Biological Reviews 89, 82–104 (2014).

3. Synthesis

This thesis is structured along the four introduced research questions and corresponding research articles. These are synthesized in the following sections accordingly.

Section	Research question
Effectiveness & remote sensing	Which approaches, methods and gaps of remotely sensed effectiveness assessments of protected areas can be found in the literature?
Effectiveness & forests	How can the temperature regulation for forests and water ecosystems as a proxy for temperature regulation capacity be quantified?
	Can we find a better temperature regulation capacity and higher vitality in more connected forest fragments and how fragmented are temperate forests?
Effectiveness & conservation	Do biosphere reserves worldwide facilitate healthier ecosystem functions than their surroundings using a multiproxy analysis?

The synthesis embeds the results into the wider research landscape with a section on effectiveness and sustainability sciences and closes with a section on future research directions.

3.1. Effectiveness & remote sensing

It is established that earth observation with satellite imagery provides detailed and much-needed information about structures, patterns and changes in the biosphere. We investigated the extent to which these images are used for effectiveness assessments of protected areas through a literature review (Gohr et al. 2022). We found remote sensing to be an effective means of assessing the socio-ecological effectiveness of natural and human-made systems, but to date, there is no commonly used framework. We defined the socio-ecological effectiveness of protected areas as mitigating biodiversity loss while creating ecosystem functions for human well-being as well as having a sustainable impact on the surroundings. Nine of 44 studies analyzed in depth complied with this definition, the others covered one or two of the three aspects.

To cover diverse aspects of effectiveness and for a more holistic approach, we argued for a multi-proxy analysis. Nevertheless, most studies used single indicators to answer their hypotheses, e.g. Normalized Difference Vegetation Index changes (Fuda et al. 2016; Tang et al. 2011) or forest fires (Manaswini and Reddy 2015). We classified the articles corresponding to their selection method of effectiveness indicators that go from a simple inside-outside comparison to a more sophisticated approach using co-variates in a matching algorithm to compare similar structures inside and outside of a protected area. This approach was found in the latest published studies (e.g. Beresford et al. 2013; 2018).
Most effectiveness studies highlight the value and use of satellite imagery, while at the same time calling for more holistic approaches. These are 1) including ground truthing wherever possible, 2) combining multiple indicators and 3) using higher spatial and temporal scales. These needs will be met in the future by growing ground-truthing databases (e.g. GBIF 2024), better cloud computing options (Gorelick et al. 2017; ESA and NASA 2024) and more precise data sets (e.g. GEDI 2024). In addition, learning websites and open communication should promote more democratized access to geospatial information and processing in support of civil society and policymakers (Pettorelli et al. 2014).

3.2. Effectiveness & forests

Healthy forest ecosystems are effective in maintaining their biodiversity and in producing biomass (Marín et al. 2021; Ma et al. 2015). Yet human-induced climate change and global warming create a growing threat to the functioning of forests (Morecroft et al. 2019; Naudts et al. 2016). We quantified these threats by assessing the temperature regulation capacities and vegetation vitality on hot days in a temperate forest region in North West Germany (Gohr et al. 2021). As vegetation cover is especially affected by high temperatures, we related temperature changes with land cover fractions for different temperature ranges. Water bodies and forests stay cool at increasing temperatures on a landscape level compared to agricultural and urban areas, suggesting a regulating effect. In our study area north of Berlin, coniferous forests had a stronger influence on the landscape temperature than broad-leaved forests, which we interpreted as a combination of more data points in these areas and the density of planting. Regardless, broadleaved forests had lower temperatures on hot days than coniferous forests in absolute values. This pattern is observed throughout Europe as well (Schwaab et al. 2020). The greenest pixel composite based on the Normalized Difference Vegetation Index (NDVI) correlated negatively with hot day temperatures and suggests that forests with a high vitality have a high effect on temperature regulation.

Fragmentation caused by deforestation or construction presents another big challenge for forests in terms of maintaining their ecosystem functionality. Therefore, we investigated the state of fragmentation in German forests and assessed the extent to which forests adhere to a better temperature regulation capacity and vitality in better connected forest fragments (Mann et al. 2023). We used Thyssen polygons of Landsat imagery at 30 meter resolution and investigated connectivity for each ecoregion to account for elevation-based characteristics. For all ecoregions, we found that larger intact tree covered areas are cooler on hot days and more vital than more fragmented tree cover patches. Forest patch connectivity, hot day temperatures and the NDVI of temperate forests suggest that higher connectivity relates to effective forest ecosystem functioning. This holds true regardless of the ecoregion.

Forests are pertinent for Earth’s climate, and they need to be safeguarded. Both studies include old-growth beech forests, which are designated areas of UNESCO’s World Heritage “Ancient and Primeval Beech Forests of the Carpathians and Other Regions of Europe”. Old-growth forests are less fragmented and cooler than, for example, needle-leaved monocultures (unpublished analysis, expected Adhikari et al. 2024). It is advised that those areas be protected to maintain their ecological effectiveness.

3.3. Effectiveness & conservation

Protected areas have been investigated and their effectiveness in safeguarding biodiversity or storing carbon more than their unprotected surroundings was quantified and confirmed (Duncanson et al. 2023). We have examined whether this applies to the world network of biosphere reserves and developed a remote sensing method using multiple proxies of forest ecosystem functions (Article IV, expected Gohr et al. 2024). We defined effectiveness in terms of the biosphere reserves’ contribution to sustaining the ecosystem's health and functioning based on seven forest ecosystem function proxies. The rather low number of 18 out of 119 investigated forest biosphere reserves with positive proxy differences inside as compared to their surroundings suggests a rather low effectiveness of forest biosphere reserves worldwide. Nevertheless, we confirmed higher ecosystem functions with higher forest cover share across all forest biomes. In another UNESCO category, the world heritage sites, the forest cover integrity was measured to have a positive spillover effect on its surroundings: Comparing different IUCN categories, the highest forest integrity values were found in areas where conservation and sustainable use of natural resources within the sites were combined (Hyland and Quinn 2023).

Still, the approach of biosphere reserve management to combine ecosystem function maintenance and sustainable development needs to improve. Since UNESCO biosphere reserves are designed as learning sites for sustainable development, their aim exceeds maintaining (forest) ecosystem functions. Based on 3304 research articles, we found an increasing amount of transdisciplinary research in biosphere reserves (expected Dabard et al. 2024, Appendix A3). These developments can be a starting point for a more holistic research agenda and thus for better-informed policy decisions. It is debatable how these investigations can translate into practical management recommendations, while it is contested whether or not healthy ecosystems can maintain themselves on their own (Aplet and Cole 2010).

Our assessment of forest ecosystem functions in biosphere reserves worldwide contributed to research on conservation effectiveness by developing a global monitoring tool for ecological conditions of forest biosphere reserves. Still, assessments of conservation effectiveness are based on various frameworks and are either outcome-oriented socio-ecological effectiveness or progress-oriented, focusing on management. Global holistic assessments on various spatial and temporal scales seem complex and unsolvable. It can be called a wicked problem, taking into account the various aspects of biomes, political management regimes, resources, and pressure, including climate change-induced threats (DeFries and Nagendra 2017).

3.4. Effectiveness & Sustainability Science

In Sustainability science, effectiveness can be a useful concept to assess human-nature relations and the transformative potential of studied systems (Kates 2011). The potential of using socio-ecological effectiveness as a baseline for decision processes in combining environmental protection with economic development for a sustainable future is high, yet to be further developed (Ghoddousi, Loos, and Kuemmerle 2021). Future studies will show how advancing sustainability knowledge and sustainable science research and combining transdisciplinary knowledge can be developed (expected Dabard et al. 2024, Appendix A3). International agreements and frameworks such as the Sustainable Development Goals, the Paris Agreement on Climate Change or the recently adopted Kunming-Montreal Global Biodiversity Framework (CBD 2022) guide and control research directions (Obura 2023). Sustainability considerations also include questions of (environmental) justice (Griggs et al. 2013). Studies focusing solely on supply changes or forest loss or comprehensive analyses including local knowledge are difficult to implement but possible (Nagendra, Rocchini, and Ghate 2010; Ostrom and Nagendra 2006).

This thesis covers components of social and ecological effectiveness of ecosystems and biosphere reserves on multiple spatial and temporal scales. Still, it cannot be considered a holistic framework to assess effectiveness in sustainability science. The world network of biosphere reserves aims to establish learning sites for sustainable development, which could be translated into relevant study sites of effectiveness in sustainability science. Participatory conservation, defined as community-based sustainable use of resources is a concept firmly established in the concept of biosphere reserves (Stoll-Kleemann, de la Vega-Leinert, and Schultz 2010). With it, the transformative potential of biosphere reserves can be enhanced with transdisciplinary research agendas to guide management decisions on multiple scales and encourage sustainable actions in those areas.

3.5. Future research directions

This dissertation can serve as a starting point for conducting socio-ecological assessments of ecosystem function proxies using remote sensing and policy-informed methodologies. In the following, I propose three research directions for further exploration.

Firstly, forest ecosystems are embedded into landscapes of various land cover types. By designing, integrating, and comparing ecosystem functioning proxies of other land cover types, we would better understand landscapes in a changing climate. The world network of biosphere reserves would be the ideal frame for exploring their potential as model regions for sustainable development, and evidence of the effectiveness of area-based approaches to conservation would need to be provided.

Secondly, the ecological proxies of ecosystem function should be augmented with social and economic proxies such as the Human footprint and the gross domestic product per capita within and beyond biosphere reserves. This would support understanding the drivers of inequalities and pressures on nature and people and would potentially present leverage points for sustainable change.

Lastly, integrating findings of ecosystem functionality with social functionality by involving community-driven approaches can lead the way to more applied remote sensing assessments and, ultimately, to transdisciplinary research agendas.

Framework references

Allen, Craig D., Alison K. Macalady, Haroun Chenchouni, Dominique Bachelet, Nate McDowell, Michel Vennetier, Thomas Kitzberger, et al. 2010. "A Global Overview of Drought and Heat-Induced Tree Mortality Reveals Emerging Climate Change Risks for Forests." *Forest Ecology and Management*, Adaptation of Forests and Forest Management to Changing Climate, 259 (4): 660–84. https://doi.org/10.1016/j.foreco.2009.09.001.

Aplet, Gregory H., and David N. Cole. 2010. "The Trouble with Naturalness: Rethinking Park and Wilderness Goals." In *Beyond Naturalness: Rethinking Park and Wilderness Stewardship in an Era of Rapid Change*, 12–29.

Batisse, Michel. 1982. "The Biosphere Reserve: A Tool for Environmental Conservation and Management." *Environmental Conservation* 9 (2): 101–11. https://doi.org/10.1017/S0376892900019937.

Beresford, Alison E., Graeme M. Buchanan, Ben Phalan, George W. Eshiamwata, Andrew Balmford, Andreas B. Brink, Lincoln D.C. Fishpool, and Paul F. Donald. 2018. "Correlates of Long-Term Land-Cover Change and Protected Area Performance at Priority Conservation Sites in Africa." *Environmental Conservation* 45 (1): 49–57. https://doi.org/10.1017/S0376892917000157.

Beresford, Alison E., George W. Eshiamwata, Paul F. Donald, Andrew Balmford, Bastian Bertzky, Andreas B. Brink, Lincoln D. C. Fishpool, et al. 2013. "Protection Reduces Loss of Natural Land-Cover at Sites of Conservation Importance across Africa." Edited by Guy J-P. Schumann. *PLOS ONE* 8 (5): e65370. https://doi.org/10.1371/journal.pone.0065370.

Blumröder, Jeanette S., Felix May, Werner Härdtle, and Pierre L. Ibisch. 2021. "Forestry Contributed to Warming of Forest Ecosystems in Northern Germany during the Extreme Summers of 2018 and 2019." *Ecological Solutions and Evidence* 2 (3): e12087. https://doi.org/10.1002/2688-8319.12087.

Bonan, G. B. 2008. "Forests and Climate Change: Forcings, Feedbacks, and the Climate Benefits of Forests." *Science* 320 (5882): 1444–49. https://doi.org/10.1126/science.1155121.

Brockerhoff, Eckehard G., Luc Barbaro, Bastien Castagneyrol, David I. Forrester, Barry Gardiner, José R. González-Olabarria, et al. 2017. "Forest Biodiversity, Ecosystem Functioning and the Provision of Ecosystem Services." *Biodiversity and Conservation* 26 (13): 3005–35. https://doi.org/10.1007/s10531-017-1453-2.

CBD. 2022. "Kunming-Montreal Global Biodiversity Framework. CBD/COP/DEC/15/4. Montreal: Convention on Biological Diversity. 14."

Chape, S, J Harrison, M Spalding, and I Lysenko. 2005. "Measuring the Extent and Effectiveness of Protected Areas as an Indicator for Meeting Global Biodiversity Targets." *Philosophical Transactions of the Royal Society B: Biological Sciences* 360 (1454): 443–55. https://doi.org/10.1098/rstb.2004.1592.

DeFries, Ruth, and Harini Nagendra. 2017. "Ecosystem Management as a Wicked Problem." *Science* 356 (6335): 265–70. https://doi.org/10.1126/science.aal1950.

Dudley, Nigel. 2008. *Guidelines for Applying Protected Area Management Categories*. IUCN. https://doi.org/10.2305/IUCN.CH.2008.PAPS.2.en.

Duncanson, Laura, M. Liang, V. Leitold, J. Armston, S. M. Krishna Moorthy, R. Dubayah, S. Costedoat, et al. 2023. "The Effectiveness of Global Protected Areas for Climate Change Mitigation." *Nature Communications* 14 (1): 2908. https://doi.org/10.1038/s41467-023-38073-9.

Duncanson, Laura, James R. Kellner, John Armston, Ralph Dubayah, David M. Minor, Steven Hancock, Sean P. Healey, et al. 2022. "Aboveground Biomass Density Models for NASA's Global Ecosystem Dynamics Investigation (GEDI) Lidar Mission." *Remote Sensing of Environment* 270 (March): 112845. https://doi.org/10.1016/j.rse.2021.112845.

Durán, América P., Olga Barbosa, and Kevin J. Gaston. 2022. “Understanding the Interacting Factors That Determine Ecological Effectiveness of Terrestrial Protected Areas.” *Journal for Nature Conservation* 70 (December): 126264. https://doi.org/10.1016/j.jnc.2022.126264.

ESA, and NASA. 2024. “Multi-Mission Algorithm and Analysis Platform (MAAP).” https://scimaap.net/.

Ferraro, Paul J., Merlin M. Hanauer, Daniela A. Miteva, Gustavo Javier Canavire-Bacarreza, Subhrendu K. Pattanayak, and Katharine R. E. Sims. 2013. “More Strictly Protected Areas Are Not Necessarily More Protective: Evidence from Bolivia, Costa Rica, Indonesia, and Thailand.” *Environmental Research Letters* 8 (2): 025011. https://doi.org/10.1088/1748-9326/8/2/025011.

Ferreira, A. F., Heike Zimmermann, Rui Santos, and Henrik von Wehrden. 2020. “Biosphere Reserves’ Management Effectiveness—A Systematic Literature Review and a Research Agenda.” *Sustainability* 12 (14): 5497.

Ferreira, Ana F., Heike Zimmermann, Rui Santos, and Henrik von Wehrden. 2018. “A Social–Ecological Systems Framework as a Tool for Understanding the Effectiveness of Biosphere Reserve Management.” *Sustainability* 10 (10): 3608.

Freudenberger, Lisa, Peter R. Hobson, Martin Schluck, and Pierre L. Ibisch. 2012. “A Global Map of the Functionality of Terrestrial Ecosystems.” *Ecological Complexity* 12 (December): 13–22. https://doi.org/10.1016/j.ecocom.2012.08.002.

Fuda, Rebecca K., Sadie J. Ryan, Jonathan B. Cohen, Joel Hartter, and Jacqueline L. Frair. 2016. “Assessing Impacts to Primary Productivity at the Park Edge in Murchison Falls Conservation Area, Uganda.” *Ecosphere* 7 (10): e01486. https://doi.org/10.1002/ecs2.1486.

Fuller, Carley, Stefania Ondei, Barry W. Brook, and Jessie C. Buettel. 2019. “First, Do No Harm: A Systematic Review of Deforestation Spillovers from Protected Areas.” *Global Ecology and Conservation* 18 (4). https://doi.org/10.1016/j.gecco.2019.e00591.

Gatti, Luciana V., Luana S. Basso, John B. Miller, Manuel Gloor, Lucas Gatti Domingues, Henrique L. G. Cassol, Graciela Tejada, et al. 2021. “Amazonia as a Carbon Source Linked to Deforestation and Climate Change.” *Nature* 595 (7867): 388–93. https://doi.org/10.1038/s41586-021-03629-6.

GBIF. 2024. “Global Biodiversity Information Facility. Free and Open Access to Biodiversity Data.” 2024. https://www.gbif.org/.

GEDI. 2024. “Global Ecosystem Dynamics Investigation (GEDI). High Resolution Laser Ranging of Earth’s Forests and Topography from the International Space Station (ISS).” GEDI. 2024. https://gedi.umd.edu/.

Ghoddousi, Arash, Jacqueline Loos, and Tobias Kuemmerle. 2021. “An Outcome-Oriented, Social–Ecological Framework for Assessing Protected Area Effectiveness.” *BioScience*, no. biab114 (November). https://doi.org/10.1093/biosci/biab114.

Gohr, Charlotte, Jeanette S. Blumröder, Douglas Sheil, and Pierre L. Ibisch. 2021. “Quantifying the Mitigation of Temperature Extremes by Forests and Wetlands in a Temperate Landscape.” *Ecological Informatics* 66 (December): 101442. https://doi.org/10.1016/j.ecoinf.2021.101442.

Gohr, Charlotte, Henrik von Wehrden, Felix May, and P. L. Ibisch. 2022. “Remotely Sensed Effectiveness Assessments of Protected Areas Lack a Common Framework: A Review.” *Ecosphere*.

Gorelick, Noel, Matt Hancher, Mike Dixon, Simon Ilyushchenko, David Thau, and Rebecca Moore. 2017. “Google Earth Engine: Planetary-Scale Geospatial Analysis for Everyone.” *Remote Sensing of Environment* 202: 18–27. https://doi.org/10.1016/j.rse.2017.06.031.

Griggs, David, Mark Stafford-Smith, Owen Gaffney, Johan Rockström, Marcus C. Öhman, Priya Shyamsundar, Will Steffen, Gisbert Glaser, Norichika Kanie, and Ian Noble. 2013. “Sustainable Development Goals for People and Planet.” *Nature* 495 (7441): 305–7. https://doi.org/10.1038/495305a.

Grimm, Nancy B, F Stuart Chapin III, Britta Bierwagen, Patrick Gonzalez, Peter M Groffman, Yiqi Luo, Forrest Melton, et al. 2013. “The Impacts of Climate Change on Ecosystem Structure and Function.” *Frontiers in Ecology and the Environment* 11 (9): 474–82. https://doi.org/10.1890/120282.

Grossiord, Charlotte, Thomas N. Buckley, Lucas A. Cernusak, Kimberly A. Novick, Benjamin Poulter, Rolf T. W. Siegwolf, John S. Sperry, and Nate G. McDowell. 2020. “Plant Responses to Rising Vapor Pressure Deficit.” *New Phytologist* 226 (6): 1550–66. https://doi.org/10.1111/nph.16485.

Haesen, Stef, Jonas J. Lembrechts, Pieter De Frenne, Jonathan Lenoir, Juha Aalto, Michael B. Ashcroft, Martin Kopecký, et al. 2021. “ForestTemp – Sub-Canopy Microclimate Temperatures of European Forests.” *Global Change Biology* 27 (23): 6307–19. https://doi.org/10.1111/gcb.15892.

Harris, Nancy L., David A. Gibbs, Alessandro Baccini, Richard A. Birdsey, Sytze de Bruin, Mary Farina, Lola Fatoyinbo, et al. 2021. “Global Maps of Twenty-First Century Forest Carbon Fluxes.” *Nature Climate Change* 11 (3): 234–40. https://doi.org/10.1038/s41558-020-00976-6.

Hyland, Emilia B., and John E. Quinn. 2023. “Conservation Spillover Effect of UNESCO World Heritage Sites into Surrounding Landscapes.” *PeerJ* 11 (October): e15858. https://doi.org/10.7717/peerj.15858.

Isbell, Forest, Andrew Gonzalez, Michel Loreau, Jane Cowles, Sandra Díaz, Andy Hector, Georgina M. Mace, et al. 2017. “Linking the Influence and Dependence of People on Biodiversity across Scales.” *Nature* 546 (7656): 65–72. https://doi.org/10.1038/nature22899.

Jetz, Walter, Jeannine Cavender-Bares, Ryan Pavlick, David Schimel, Frank W. Davis, Gregory P. Asner, Robert Guralnick, et al. 2016. “Monitoring Plant Functional Diversity from Space.” *Nature Plants* 2 (3): 1–5. https://doi.org/10.1038/nplants.2016.24.

Jones, Kendall R., Oscar Venter, Richard A. Fuller, James R. Allan, Sean L. Maxwell, Pablo Jose Negret, and James E. M. Watson. 2018. “One-Third of Global Protected Land Is under Intense Human Pressure.” *Science* 360 (6390): 788–91. https://doi.org/10.1126/science.aap9565.

Kašpar, Vít, Lucia Hederová, Martin Macek, Jana Müllerová, Jiří Prošek, Peter Surový, Jan Wild, and Martin Kopecký. 2021. “Temperature Buffering in Temperate Forests: Comparing Microclimate Models Based on Ground Measurements with Active and Passive Remote Sensing.” *Remote Sensing of Environment* 263 (September): 112522. https://doi.org/10.1016/j.rse.2021.112522.

Kates, Robert W. 2011. “What Kind of a Science Is Sustainability Science?” *Proceedings of the National Academy of Sciences* 108 (49): 19449–50. https://doi.org/10.1073/pnas.1116097108.

Kratzer, Armin. 2018. “Biosphere Reserves Research: A Bibliometric Analysis.” *Eco.Mont (Journal on Protected Mountain Areas Research)* 10 (2): 36–49. https://doi.org/10.1553/eco.mont-10-2s36.

Leberger, Roxanne, Isabel M. D. Rosa, Carlos A. Guerra, Florian Wolf, and Henrique M. Pereira. 2020. “Global Patterns of Forest Loss across IUCN Categories of Protected Areas.” *Biological Conservation* 241: 108299. https://doi.org/10.1016/j.biocon.2019.108299.

Lee, Woon Hang, and Saiful Arif Abdullah. 2019. “Framework to Develop a Consolidated Index Model to Evaluate the Conservation Effectiveness of Protected Areas.” *Ecological Indicators* 102 (July): 131–44. https://doi.org/10.1016/j.ecolind.2019.02.034.

Ma, Jianyong, Xiaodong Yan, Wenjie Dong, and Jieming Chou. 2015. “Gross Primary Production of Global Forest Ecosystems Has Been Overestimated.” *Scientific Reports* 5 (1): 10820. https://doi.org/10.1038/srep10820.

Manaswini, G., and C. Sudhakar Reddy. 2015. “Geospatial Monitoring and Prioritization of Forest Fire Incidences in Andhra Pradesh, India.” *Environmental Monitoring and Assessment* 187 (10): 616. https://doi.org/10.1007/s10661-015-4821-y.

Mann, Deepika, Charlotte Gohr, Jeanette S. Blumröder, and Pierre L. Ibisch. 2023. “Does Fragmentation Contribute to the Forest Crisis in Germany?” *Frontiers in Forests and Global Change* 6. https://www.frontiersin.org/articles/10.3389/ffgc.2023.1099460.

Marín, Ana Isabel, Dania Abdul Malak, Annemarie Bastrup-Birk, Gherardo Chirici, Anna Barbati, and Stefan Kleeschulte. 2021. “Mapping Forest Condition in Europe: Methodological Developments in Support to Forest Biodiversity Assessments.” *Ecological Indicators* 128 (September): 107839. https://doi.org/10.1016/j.ecolind.2021.107839.

Mildrexler, David, Zhiqiang Yang, Warren B. Cohen, and David M. Bell. 2016. “A Forest Vulnerability Index Based on Drought and High Temperatures.” *Remote Sensing of Environment* 173 (February): 314–25. https://doi.org/10.1016/j.rse.2015.11.024.

Morecroft, Michael D., Simon Duffield, Mike Harley, James W. Pearce-Higgins, Nicola Stevens, Olly Watts, and Jeanette Whitaker. 2019. “Measuring the Success of Climate Change Adaptation and Mitigation in Terrestrial Ecosystems.” *Science* 366 (6471). https://doi.org/10.1126/science.aaw9256.

Nagendra, Harini, Richard Lucas, João Pradinho Honrado, Rob H.G. Jongman, Cristina Tarantino, Maria Adamo, and Paola Mairota. 2013. “Remote Sensing for Conservation Monitoring: Assessing Protected Areas, Habitat Extent, Habitat Condition, Species Diversity, and Threats.” *Ecological Indicators* 33: 45–59. https://doi.org/10.1016/j.ecolind.2012.09.014.

Nagendra, Harini, Duccio Rocchini, and Rucha Ghate. 2010. “Beyond Parks as Monoliths: Spatially Differentiating Park-People Relationships in the Tadoba Andhari Tiger Reserve in India.” *Biological Conservation* 143 (12): 2900–2908. https://doi.org/10.1016/j.biocon.2010.04.050.

NASA, National Aeronautics and Space Administration. 2024a. “Landsat.” 2024. https://landsat.gsfc.nasa.gov/.

NASA, National Aeronautics and Space Administration. 2024b. “MODIS.” 2024. https://modis.gsfc.nasa.gov/data/.

Naudts, K., Y. Chen, M. J. McGrath, J. Ryder, A. Valade, J. Otto, and S. Luyssaert. 2016. “Europes Forest Management Did Not Mitigate Climate Warming.” *Science* 351 (6273): 597–600. https://doi.org/10.1126/science.aad7270.

Obura, David. 2023. “The Kunming-Montreal Global Biodiversity Framework: Business as Usual or a Turning Point?” *One Earth* 6 (2): 77–80. https://doi.org/10.1016/j.oneear.2023.01.013.

Ostrom, Elinor, and Harini Nagendra. 2006. “Insights on Linking Forests, Trees, and People from the Air, on the Ground, and in the Laboratory.” *Proceedings of the National Academy of Sciences of the United States of America* 103 (51): 19224–31. https://doi.org/10.1073/pnas.0607962103.

Palliwoda, Julia, Andrea Büermann, Julia Fischer, Roland Kraemer, and Matthias Schröter. 2021. “Zoning of UNESCO Biosphere Reserves: A Comprehensive Set of Geodata for Europe.” *Frontiers in Ecology and Evolution*, 673.

Pettorelli, Nathalie, Henrike Schulte to Bühne, Ayesha Tulloch, Grégoire Dubois, Cate Macinnis-Ng, Ana M. Queirós, David A. Keith, et al. 2018. “Satellite Remote Sensing of Ecosystem Functions: Opportunities, Challenges and Way Forward.” *Remote Sensing in Ecology and Conservation* 4 (2): 71–93. https://doi.org/10.1002/rse2.59.

Pettorelli, Nathalie, William F. Laurance, Timothy G. O’Brien, Martin Wegmann, Harini Nagendra, and Woody Turner. 2014. “Satellite Remote Sensing for Applied Ecologists: Opportunities and Challenges.” *Journal of Applied Ecology* 51 (4): 839–48. https://doi.org/10.1111/1365-2664.12261.

QGIS. 2024. “A Free and Open Source Geographic Information System.” http://www.qgis.org/.

R Core Team. 2024. “R: The R Project for Statistical Computing.” 2024. https://www.r-project.org/.
Rapport, D J, R Costanza, and A J McMichael. 1998. “Assessing Ecosystem Health” 13 (10).
Rustamov, Rustam B., Sabina Hasanova, and Mahfuza H. Zeynalova. 2018. *Multi-Purposeful Application of Geospatial Data*. BoD – Books on Demand.
Saatchi, Sassan S., Nancy L. Harris, Sandra Brown, Michael Lefsky, Edward T. A. Mitchard, William Salas, Brian R. Zutta, et al. 2011. “Benchmark Map of Forest Carbon Stocks in Tropical Regions across Three Continents.” *Proceedings of the National Academy of Sciences* 108 (24): 9899–9904. https://doi.org/10.1073/pnas.1019576108.
Schlesinger, William H., Michael C. Dietze, Robert B. Jackson, Richard P. Phillips, Charles C. Rhoades, Lindsey E. Rustad, and James M. Vose. 2016. “Forest Biogeochemistry in Response to Drought.” *Global Change Biology* 22 (7): 2318–28. https://doi.org/10.1111/gcb.13105.
Schwaab, Jonas, Edouard L. Davin, Peter Bebi, Anke Duguay-Tetzlaff, Lars T. Waser, Matthias Haeni, and Ronny Meier. 2020. “Increasing the Broad-Leaved Tree Fraction in European Forests Mitigates Hot Temperature Extremes.” *Scientific Reports* 10 (1): 14153. https://doi.org/10.1038/s41598-020-71055-1.
Stoll-Kleemann, Susanne, A. Cristina de la Vega-Leinert, and Lisen Schultz. 2010. “The Role of Community Participation in the Effectiveness of UNESCO Biosphere Reserve Management: Evidence and Reflections from Two Parallel Global Surveys.” *Environmental Conservation* 37 (3): 227–38.
Tang, Zhiyao, Jingyun Fang, Jinyu Sun, and Kevin J. Gaston. 2011. “Effectiveness of Protected Areas in Maintaining Plant Production.” *Plos One* 6 (4): e19116. https://doi.org/10.1371/journal.pone.0019116.
Tatem, Andrew, Scott Goetz, and Simon Hay. 2008. “Fifty Years of Earth-Observation Satellites.” *American Scientist* 96 (5): 390. https://doi.org/10.1511/2008.74.390.
Turner, Monica G., W. John Calder, Graeme S. Cumming, Terry P. Hughes, Anke Jentsch, Shannon L. LaDeau, Timothy M. Lenton, et al. 2020. “Climate Change, Ecosystems and Abrupt Change: Science Priorities.” *Philosophical Transactions of the Royal Society B: Biological Sciences* 375 (1794): 20190105. https://doi.org/10.1098/rstb.2019.0105.
UNESCO, United Nations Educational, Scientific and Cultural Organization. 1996. *Biosphere Reserves: The Seville Strategy and the Statutory Framework of the World Network*. Paris.
UNESCO. 2015. “MAB STRATEGY 2015-2025.”
UNESCO. 2024. “World Network of Biosphere Reserves.” UNESCO. 2024. https://en.unesco.org/biosphere/wnbr.
USC, Union of Concerned Scientists. 2023. “Satellite Database | Union of Concerned Scientists.” UCS Satellite Database. 2023. https://www.ucsusa.org/resources/satellite-database.
Xu, Xiyan, Anqi Huang, Elise Belle, Pieter De Frenne, and Gensuo Jia. 2022. “Protected Areas Provide Thermal Buffer against Climate Change.” *Science Advances* 8 (44): eabo0119. https://doi.org/10.1126/sciadv.abo0119.
Zscheischler, Jakob, Seth Westra, Bart J. J. M. Van Den Hurk, Sonia I. Seneviratne, Philip J. Ward, Andy Pitman, Amir AghaKouchak, et al. 2018. “Future Climate Risk from Compound Events.” *Nature Climate Change* 8 (6): 469–77. https://doi.org/10.1038/s41558-018-0156-3.

Appendix

<u>Appendix overview</u>

Appendix A 1 Outputs and funding information
Appendix A 2 Supplementary material for 2.3
Appendix A 3 Supplementary material for 2.4
Appendix A 4 Manuscript of Dabard et al. 2024
Appendix A 5 Positionality statement
Appendix A 6 Acknowledgements

A 1 Output and funding information

Tab. 1: Overview of articles relevant to this thesis

Article	Title	Journal	Status	Cited as of March 2024	Author contribution	Weighting factor
1	Remotely sensed effectiveness assessments of protected areas lack a common framework: A review	Ecosphere	Published 2021	4	First author	1
2	Quantifying the mitigation of temperature extremes by forests and wetlands in a temperate landscape	Ecological Informatics	Published 2022	20	First author	1
3	Does fragmentation contribute to the forest crisis in Germany?	Frontiers in Forests and Global Change	Published 2023	5	Co-first author	1
4	Low effectiveness of the world network of biosphere reserves in maintaining forest ecosystem functions	Science	submitted		First author	1
Appendix	Biosphere Reserves as model regions for transdisciplinarity? A literature review	Sustainability Science	first revision 2024		Co-first author	0

Tab. 2: Overview of relevant articles and corresponding authors contribution

Article	Title reference	Authors contribution
1	Remotely sensed effectiveness assessments of protected areas lack a common framework: A review	Charlotte Gohr and Pierre L. Ibisch conceived the study, and Henrik von Wehrden contributed to the methodological development. Charlotte Gohr wrote the original draft and conducted the analyses. All authors were involved in reviewing and revising the draft versions.
2	Quantifying the mitigation of temperature extremes by forests and wetlands in a temperate landscape	PLI, CG and JB designed the research, CG performed the analysis and visualization, CG and PLI wrote the original draft, DS provided comprehensive support in the assessment and interpretation of the results, and all authors contributed to the interpretation of the results and the subsequent revisions of the paper.
3	Does fragmentation contribute to the forest crisis in Germany?	PI, DM, and CG designed the research. DM and CG performed the analysis, visualization, and wrote the original draft. PI and JB provided comprehensive support in the assessment and interpretation of the results. All authors contributed to the interpretation of the results and the subsequent revisions of the manuscript.
4	Low effectiveness of the world network of biosphere reserves in maintaining forest ecosystem functions	Conceptualization: CG, PLI, HVW, SS, NP Methodology: CG, PLI, HVW, SS, NP Formal Analysis: CG Investigation: CG Visualization: CG Funding acquisition: PLI, CG Project administration: PLI, CG Supervision: PLI, HVW, SS Writing – original draft: CG, PLI Writing – review & editing: CG, PLI, HVW, SS, NP
Appendix	Biosphere Reserves as model regions for transdisciplinarity? A literature review	Conceptualization: CHD, CG, FW, HVW, AFF, JH, CM, PLI, VL Investigation: CHD, CG, FW, JH, CM, FN, SH, BDFA Data curation: CHD, CG, FW, HVW, AFF, JJ Formal analysis: CG, FW, HVW Methodology: CHD, CG, FW, HVW, AFF Visualization: CG, FW, HVW Funding acquisition: PLI, VL, JH, CHD, CG, AFF Project administration: CHD, CG, AFF Supervision: CHD, CG, HVW, AFF Writing—original draft: CHD, CG, HVW, JJ, FN, SH, BDFA Writing—review & editing: CHD, CG, FW, HVW, AFF, PLI, CM, JJ, JH, VL

Tab. 3: Overview of additional output

Type	Output
Reviews for journals. (8, counting revisions)	Ecological Informatics (4) Biodiversity and Conservation (2) Sustainability Science (2)
Supervised thesis (2)	1. Comparison of air- and surface temperature recordings between forest interiors, forest edges and clear-cuts during the fall and winter season 2021/2022 in biosphere reserve Roztochya, Ukraine. N. Bachstein and L. Grabsch. Bachelor thesis in International Forest Ecosystem Management, B.Sc. Eberswalde University. Second supervision. 2. Monitoring forests under a changing management regime – use cases of Loreley valley and Oberbergischer Kreis. Valentina García. Master thesis in Global change management. Second supervision.
Teaching (3)	1. Tutorial for 'Mathematics and Statistics' lecture at Leuphana University 2. Co-teaching lecture 'Scientific methods – different pathways to knowledge' at Leuphana University 3. Presentations and Workshop at Eberswalde University on Environmental Justice (2019), Research in Forests (2020), Literature Reviews (2020), Remote Sensing (2020, 2021, 2022)
Non peer-reviewed articles (3)	1. Gohr, C. (2023): Analyse der Wirksamkeit von UNESCO-Biosphärenreservaten - Entwicklung eines globalen Monitorings auf Grundlage von sozio-ökologischen Parametern, ed. 2023. In: Treffpunkt Biologische Vielfalt XX. Hrsg: J. Stadler. DE: Bundesamt für Naturschutz. https://doi.org/10.19217/skr664. 2. Ibisch, P. L.; Blumröder, J. S.; Gohr, C.; Schmidt, L. (2021): Konzept zur Förderung der Funktionen und Leistungen von Waldökosystemen in Deutschland. Centre for Econics and Ecosystem Management an der Hochschule für nachhaltige Entwicklung Eberswalde für die Bundestagsfraktion Bündnis 90/Die Grünen. Eberswalde, Berlin. 3. Gohr, C.; Blumröder, J. S.; Ibisch, P. L. (2020): Thermische Wirkungen von Waldökosystemen und Autobahnen unter den Bedingungen des Klimawandels Beurteilung der mikro- und mesoklimatischen Effekte des geplanten Baus des Autobahnabschnittes der A49 auf den Dannenröder Forst, Hessen. Greenpeace e.V., Hamburg.
Conferences (4)	1. Tracking human impact on land surface temperature (2022). Presentation at the Google Geo for Good Summit, Mountain View. USA. 2. Biosphere Reserves from space: Assessing their ecological effectiveness using Earth Engine (2022). Presentation at the International Conference: Science and Research in, for and with UNESCO Biosphere Reserves. Eberswalde, Werbellinsee. Germany. 3. Quantifying Landscape Temperature Mitigation of Forests and Wetlands (2021). Presentation at the Annual Meeting of AK Fernerkundung: Fernerkundung für die Welt von morgen: Herausforderungen und Lösungsansätze für eine nachhaltige Entwicklung. GeoWoche. Online. 4. Participation in the Fall meeting of the American Geophysical Union (AGU) 2022. Advancing Earth and Space Sciences. Chicago. USA.
Research stays (1)	10/2022-03/2023 Visiting student research program. NASA/Caltech/Jet Propulsion Laboratory, Pasadena, USA

Tab. 4: Funding and Networks

Type	Description
Funding and supporting institutions	Ministry of Science, Research and Culture of the federal state of Brandenburg, Germany Biosphere Reserves Institute at the Eberswalde University for Sustainable Development, Germany Qualifizierungsfonds, Leuphana University Lüneburg, Germany Professorinnenprogramm Eberswalde University for Sustainable Development, Germany DAAD-Nachwuchsprogramm für Künstliche Intelligenz und Informatik: Internationale Forschungsaufenthalte für Informatikerinnen & Informatiker Erasmus+ NASA Jet Propulsion Laboratory Pasadena, USA CALTECH California Institute of Technology Pasadena, USA DAAD German Exchange Agency Bonn, Germany
Networks	Women+ in Geospatial mentorship programme, global ProViae Mentoring programme, Leuphana University JPL Visiting Student Research Program, NASA JPL

Supplementary Material

Does fragmentation contribute to the forest crisis in Germany?

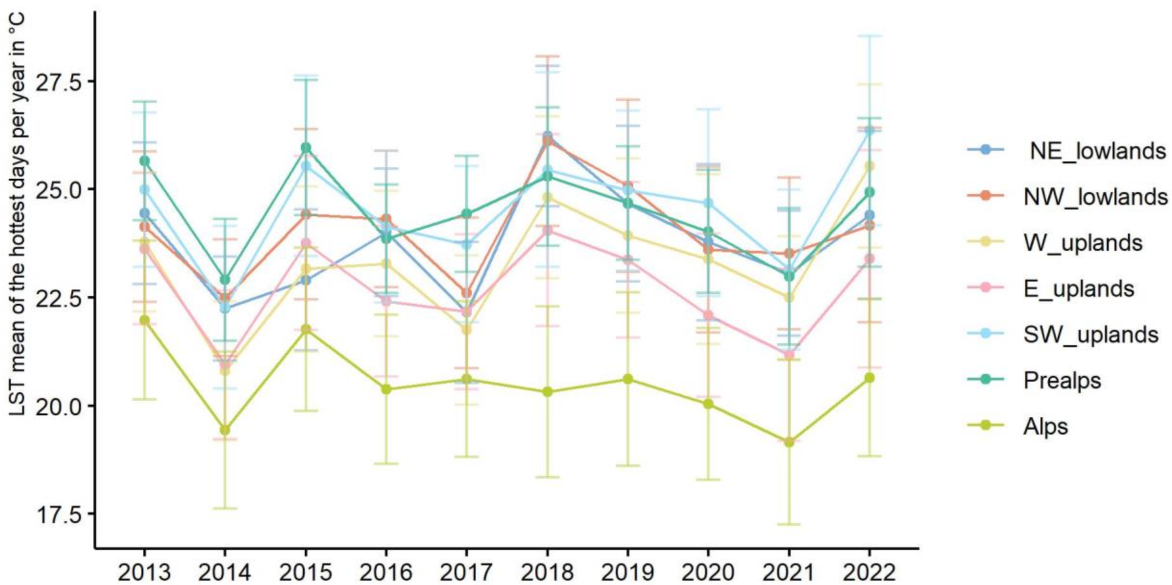
Deepika Mann[1†], **Charlotte Gohr**[1,2†], **Jeanette S. Blumröder**[1], **Pierre L. Ibisch**[1,2*]

1 Centre for Econics and Ecosystem Management, Eberswalde University for Sustainable Development, Eberswalde, Germany

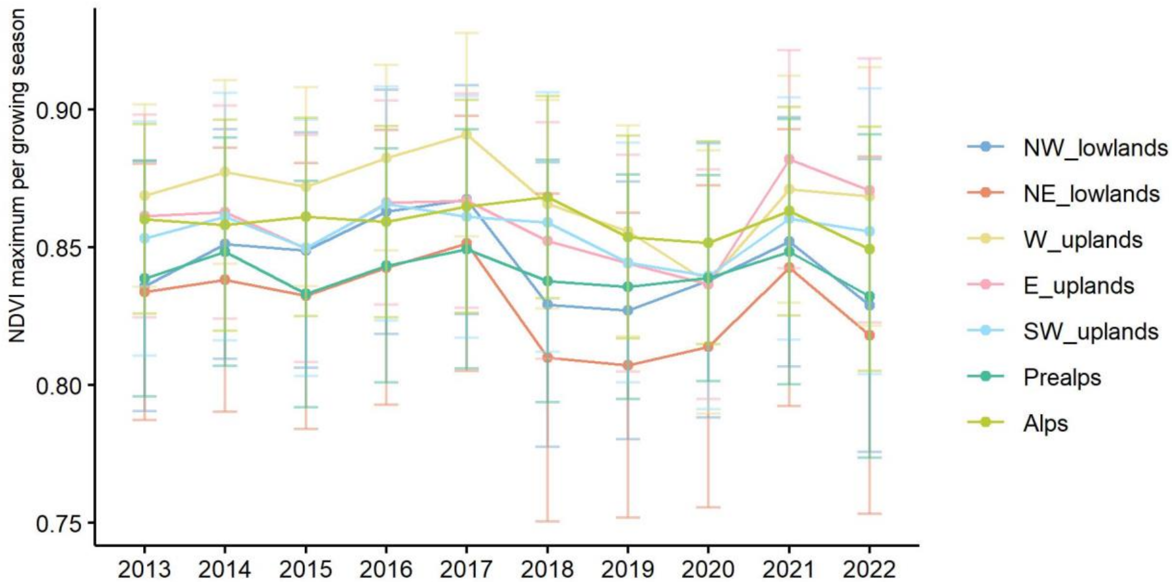
2 Biosphere Reserves Institute, Eberswalde University for Sustainable Development, Eberswalde, Germany

†These authors contributed equally to this work and share first authorship

*** Correspondence:** Pierre L. Ibisch: pierre.ibisch@hnee.de



Supplementary Figure 1. The LST (MODIS Aqua MYD11A1) mean value and standard deviation of forest areas (Hansen et al. 2013) in different ecoregions in Germany of the annual hottest days 2013-2022. The year 2022 shows extreme heat peaks, similar to the year 2018. The Alps show the same trends as other ecoregions at lower mean temperatures except for 2018, where the relatively cool mean temperature on hot days did not rise. In the Prealps, some of the highest mean temperatures are recorded in 2013 to 2015. The temperatures of the Southwest Uplands peak in 2022.



Supplementary Figure 2. The NDVI (MODIS Aqua MYD13A2) mean value and standard deviation of forest areas (Hansen et al. 2013) in different ecoregions in Germany per growing season (May-September) 2013-2022. The Uplands and the Alps record a higher mean greenness per growing season than the Lowlands and the Prealps. However, the decrease in vitality in 2018 and the following years are significant for all ecoregions.

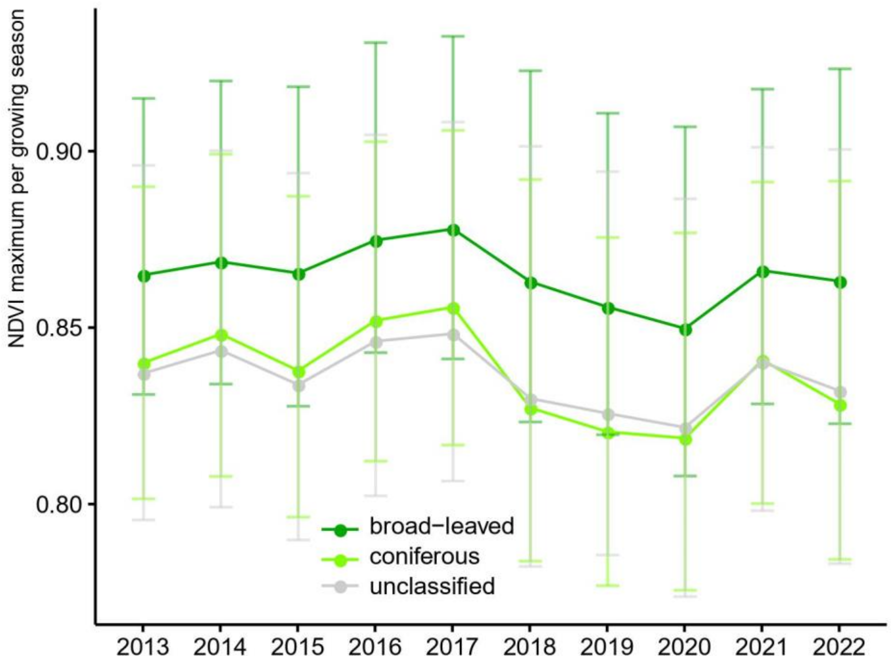
Supplementary Table 1. Overview of area specifics for number of forest fragments, the size of the largest fragment, the area in km^2, forest area in km^2, the percentage forest cover and the days above 30 °C in forests in 2022.

Area	**No. of fragments**	**max size fragment in km^2**	**Area in km^2**	**Forest area in km^2*1**	**% forest cover**	**Days above 30°C in forests 2022*2**
Germany	1931934	3758	357588	98768.35	27.6*1	124
Alps	19106	1300	3816.30	2598.55	68.09	24
Prealps	238162	441	30636.16	8219.63	26.83	80
E_uplands	168553	1279	30528.82	11048.23	36.19	77
SW_uplands	259848	3758	63118.01	23382.1	37.05	107
W_uplands	433428	1372	57020.72	21960.29	38.51	91
NE_lowlands	335306	2369	76616.65	19911	25.99	96
NW_lowlands	477531	1134	71213.83	11408.29	16.02	92

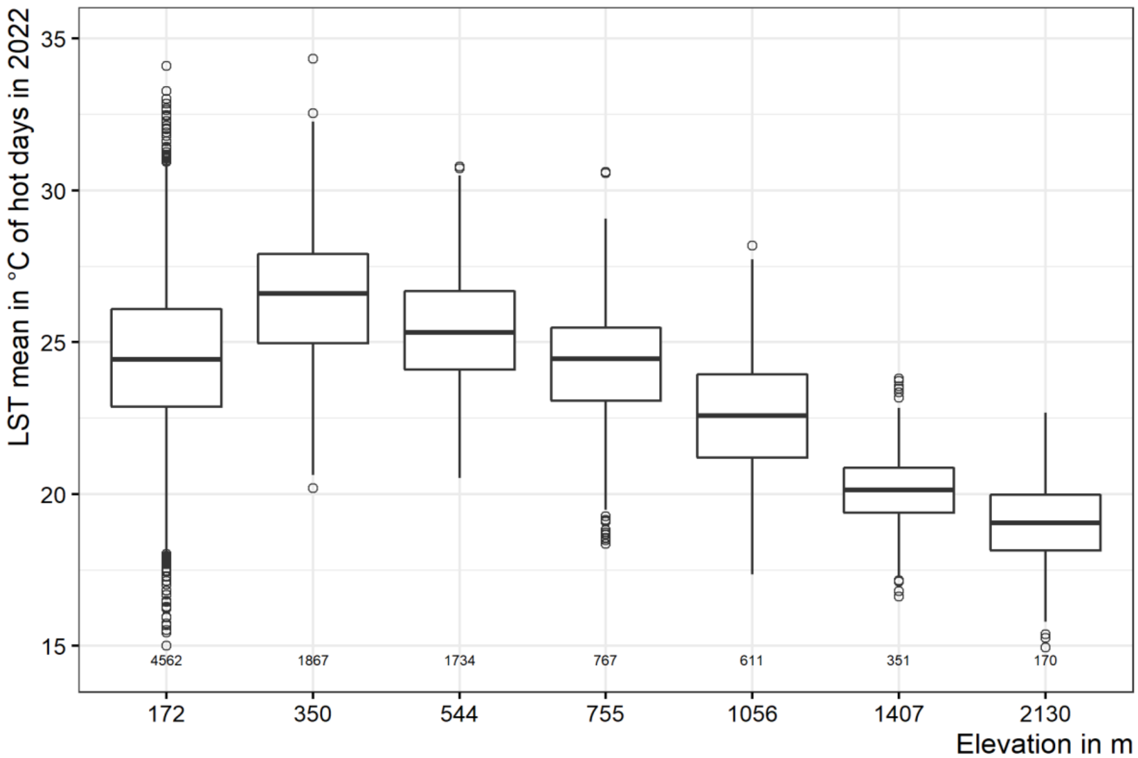
*1 based on Hansen et al. 2013 tree cover data at Landsat 30 m pixel resolution. The values differ from official statistics as they count forest areas without trees (and exclude some woodlands, urban forests that are no legal forest).
*2 based on MODIS Aqua MYD11A1 Land surface temperature at 1 km spatial resolution.

Supplementary Table 2. Number of fragments (count), median values of Thiessen connectivity (TP), Land surface temperature (LST) and Normalized Difference Vegetation Index (NDVI) in different grouped forest fragments per ecoregion in Germany in 2022.

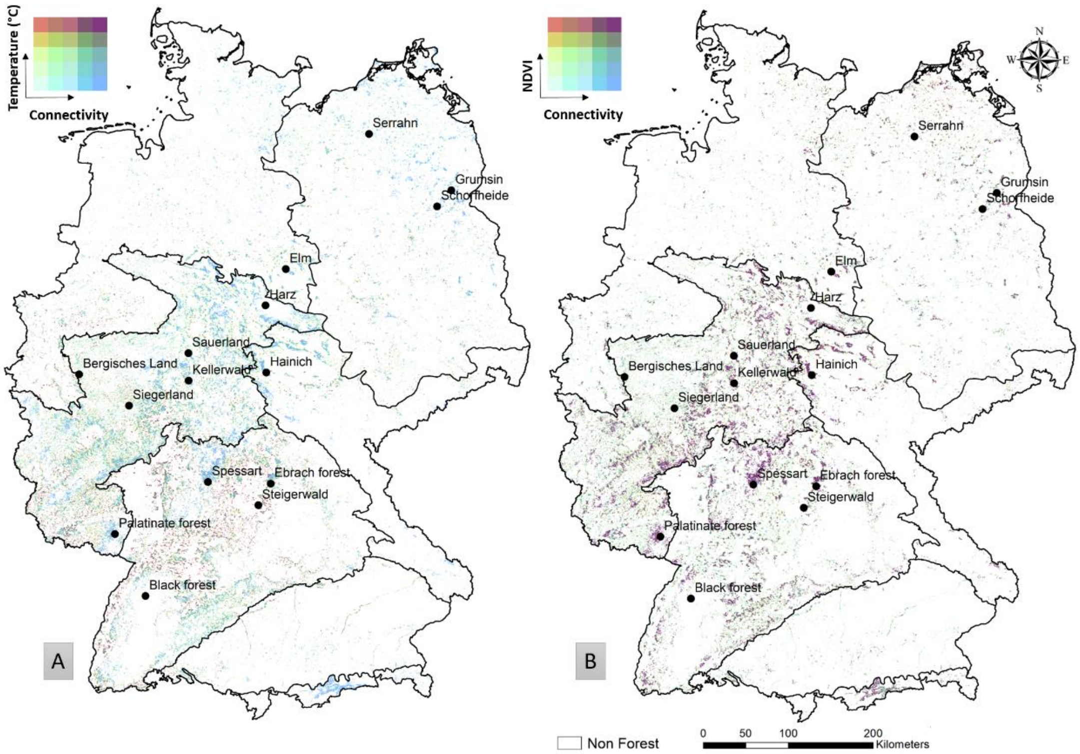
Ecoregion		**Alps**				**Prealps**			
Indicator		count	TP	LST	NDVI	Count	TP	LST	NDVI
Fragment	< 1 km	91380	0.52	22.22	0.83	2085292	0.41	26.84	0.80
	1-5 km	20085	0.70	21.75	0.83	1547516	0.76	26.23	0.82
	> 5 km	2627582	0.88	20.93	0.87	4707301	0.63	24.92	0.85
Ecoregion		**E_uplands**				**SW_uplands**			
Indicator		count	TP	LST	NDVI	Count	TP	LST	NDVI
Fragment	< 1 km	1334707	0.37	26.40	0.81	2498433	0.38	28.86	0.79
	1-5 km	1015713	0.60	25.72	0.86	2735707	0.61	28.04	0.82
	> 5 km	9152073	0.81	23.35	0.82	19456159	0.81	25.58	0.87
Ecoregion		**NE_lowlands**				**NW_lowlands**			
Indicator		count	TP	LST	NDVI	Count	TP	LST	NDVI
Fragment	< 1 km	3058691	0.31	26.93	0.78	3805222	0.34	26.65	0.77
	1-5 km	2538140	0.59	25.89	0.81	2082233	0.61	25.68	0.81
	> 5 km	15205831	0.81	24.48	0.82	5636453	0.78	24.26	0.83
Ecoregion		**W_uplands**							
Indicator		count	TP	LST	NDVI				
Fragment	< 1 km	3210310	0.41	26.93	0.81				
	1-5 km	2247157	0.61	26.49	0.84				
	> 5 km	17271998	0.77	24.84	0.87				



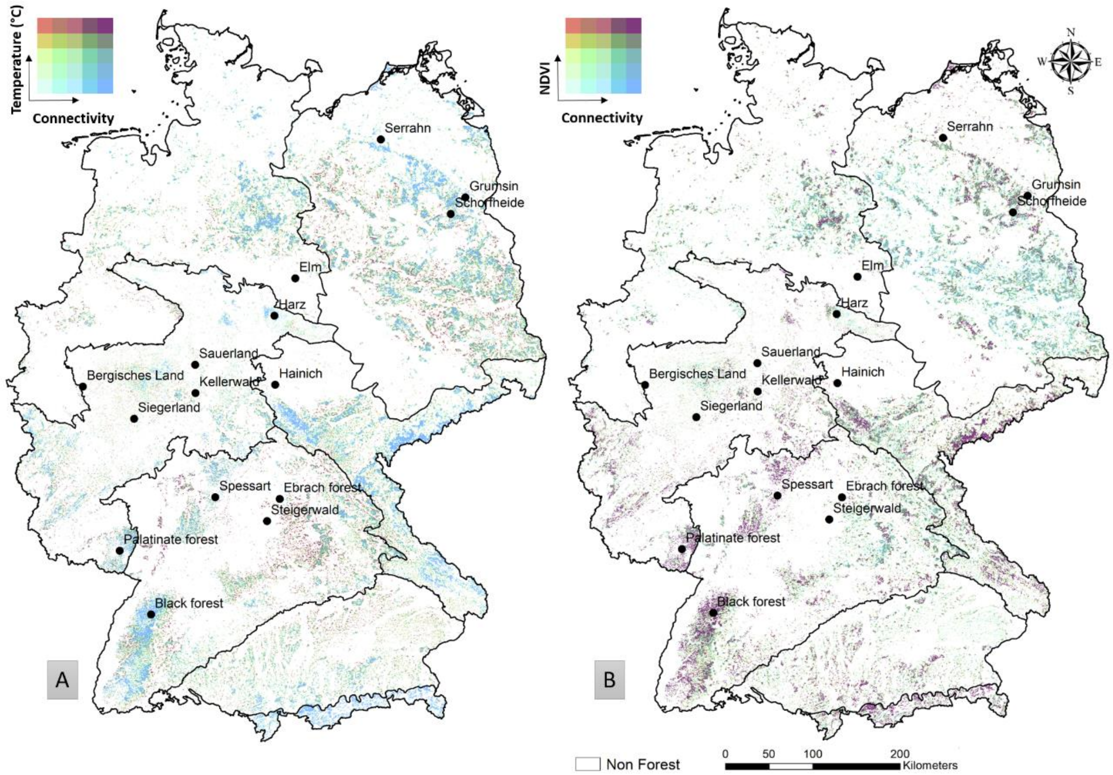
Supplementary Figure 3. Mean values of the maximum vitality (NDVI) per pixel in the growing season May to September of forest areas in Germany from 2013 to 2022 differentiated by broad-leaved forest, coniferous forest and unclassified forest areas. The standard deviations of the mean values are shown for each year and forest class.



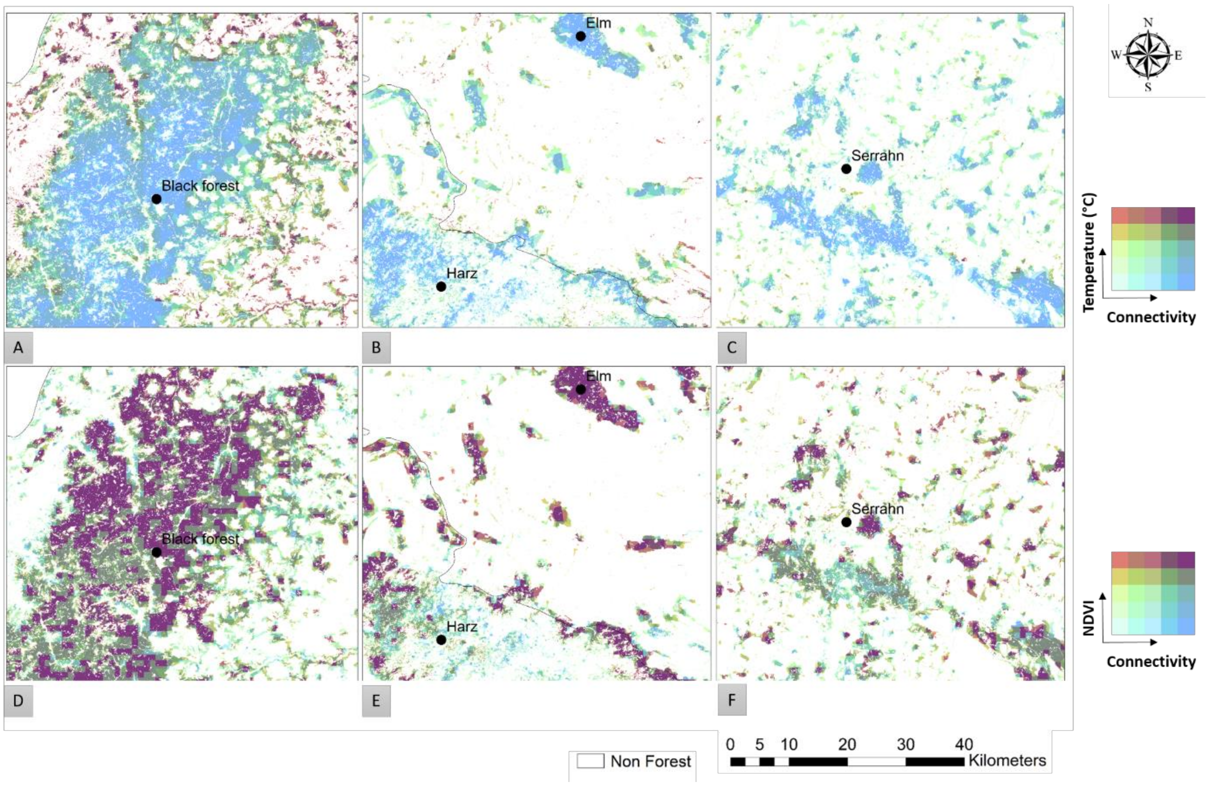
Supplementary Figure 4. Land surface temperature (MODIS Aqua MYD11A1) on hot days 2022 (at 1 km resolution of forested areas 2022 (Hansen et al. 2013, 30 m) and elevation based on the SRTM 30 m (NASA JPL 2020) with Jenks natural breaks for seven classes. The count of data points at 1 km resolution per elevation break is depicted above the x-axis. The higher the forest covered areas, the cooler the temperature on hot days.



Supplementary Figure 5. Bivariate maps of broadleaved tree covered areas in Germany 2022 and their relation between degree of fragmentation with temperatures and vegetation vitality. A) Thiessen Connectivity and Land surface temperature of hot days in 2022 (MODIS Aqua MYD11A1). B) Thiessen Connectivity and the maximum NDVI (MODIS Aqua MYD13A2) per growing season in 2022.



Supplementary Figure 6. Bivariate maps of coniferous tree covered areas in Germany 2022 and their relation between degree of fragmentation with temperatures and vegetation vitality. A) Thiessen Connectivity and Land surface temperature of hot days in 2022. B) Thiessen Connectivity and the maximum NDVI per growing season in 2022.



Supplementary Figure 7. Zoomed in view of some important locations depicted in bivariate maps of tree covered areas in Germany 2022. Subsets A, B and C shows the relation between Thiessen Connectivity and Land surface temperature of hot days in 2022. Subsets D, E and F shows the relation between Thiessen Connectivity and the maximum NDVI per growing season in 2022.

References

Hansen, Matthew C., Peter V. Potapov, Rebecca Moore, Matt Hancher, S. A. A. Turubanova, Alexandra Tyukavina, David Thau, S. V. Stehman, S. J. Goetz, and Thomas R. Loveland. 2013. "High-Resolution Global Maps of 21st-Century Forest Cover Change." *Science* 342 (6160): 850–53. https://doi.org/10.1126/science.1244693.

NASA JPL. 2020. "NASADEM Merged DEM Global 1 Arc Second V001." NASA EOSDIS Land Processes DAAC. https://doi.org/10.5067/MEASURES/NASADEM/NASADEM_HGT.001. Access. Jun. 7, 2022.

A 3 Supplementary material for 2.4

Supplementary Materials for

Low effectiveness of the world network of biosphere reserves in maintaining forest ecosystem functions

Charlotte Gohr, Henrik von Wehrden, Sassan Saatchi, Nathalie Pettorelli, Pierre L. Ibisch
Corresponding author: charlotte.gohr@leuphana.de

This article was submitted to *Science* for peer review on March 26, 2024, and follows the journal's corresponding referencing and formatting style.

Materials and Methods

Summary

The main datasets based on satellite imagery used were the GLAD forest cover 2010 product (*42*) and seven different preprocessed products of NASA's Moderate Resolution Imaging Spectroradiometer (MODIS) with derived forest ecosystem functioning proxies, all accessed and processed through the cloud-computing infrastructure Google Earth Engine and the connected large repository of geospatial datasets (*43*). The primary vector dataset is a collection of border information from the world network of biosphere reserves. All datasets are described in Table S1. For a spatio-temporal analysis at 1 km, we collected a 20 percent sample for each of the seven forest ecosystem functioning proxies inside and outside for two time spans 2010-2016 and 2017-2022.

Forest definition and area selection

The global GLAD tree cover dataset 2010 at 30 m resolution (https://glad.umd.edu/dataset/global-2010-tree-cover-30-m) served as the basis for our definition of forest. We filtered tree cover at 30 m with a threshold of ≥ 30 % and transformed the data to a binary tree cover/no tree cover format. We used the binary tree cover at 30 m to define tree cover at 1 km with a threshold of ≥ 50 % (*44*). This tree cover dataset 2010 at 1 km with values from 50 to 100 % coverage was used in the analyses and to define and select forested biosphere reserves (example: Fig. S1). The set of UNESCO biosphere reserves border data was acquired from the World Database on Protected Areas (*45*), a European Collection of biosphere reserves' border information (*46*), and directly from local biosphere reserves administrations.

We excluded biosphere reserves that 1) were available as point information only, 2) were established after 2010 since we wanted to detect changes over time under designated areas,

3) had not only less than 20 % forest cover inside but also less than 10 % forest cover in their surroundings (example: Fig. S1), 4) were outside of forest biomes, namely tropical & subtropical moist and dry broadleaf and coniferous forests, temperate broadleaf and mixed and conifer forests, boreal forests/taiga, and mangroves. The selection criteria are comparable to standard practice (e.g., *14*).

Of 748 biosphere reserves, we acquired 474 biosphere reserves with geospatial border datasets, of which 308 were established before 2010, of which 174 had correct border information and lie within forest cover biomes, of which 119 had sufficient forest cover. Due to the wide range of area sizes of all biosphere reserves, the outside border was set with a buffer calculated with the natural logarithm of the size of each area multiplied by 500. The buffer was then bound to a rectangular format for faster processing. We did not exclude designated areas, such as other protected areas or biosphere reserves, from the outside area since we wanted to depict the real-world characteristics of both the inside and outside biosphere reserves. Nevertheless, we accounted for this fact as a source of uncertainty and extracted the percentage of protected area cover in the surroundings of the biosphere reserves as a moderating variable.

<u>Proxies of forest ecosystem functions</u>

The selected proxies follow the overview of significant, convincing, and potentially available remotely sensed ecosystem functioning indicators suitable for forest ecosystems (*29*). The proxies are forest connectivity, evapotranspiration, the Normalized Difference and Enhanced Vegetation Index, gross and net primary productivity, and land surface temperature. Forest connectivity for each biosphere reserve and its surroundings for both time steps was produced by 1) creating tree cover 2016 (excluding tree cover loss before 2016) and tree cover 2022 (excluding tree cover loss before 2022) datasets, 2) computing Thiessen polygons around forest fragment centroids, 3) computing the ratio of each Thiessen polygon size and the corresponding forest cover size, which results in values ranging from 0 to 1, 4) sampling 10 % inside and 10 % outside and 5) transform to table data. Except for forest connectivity, all proxies were processed as raster data for each biosphere reserve and its surroundings by 1) if required, being scaled to 1 km, 2) masked to tree cover 2010 at 1 km, 3) temporally filtered to means per pixel for the time steps 2010-2016 and 2017-2022, 4) defined/masked as inside or outside, 5) sampled 10 % inside and 10 % outside and 6) transformed to table data. The per-pixel formula for both time steps 2010-2016 and 2017-2022 for each proxy was defined as follows: Forest connectivity 2016 and 2022, mean of annual sums of evapotranspiration in kg/m²/year, mean of yearly maxima of the Normalized Difference and Enhanced Vegetation

Index, mean of the annual sums of the gross and net primary productivity in kg*C/km², mean of days ≥ 30 °C of land surface temperature in °C (as a proxy for the cooling function of forests (*32*)), mean of the annual sums of precipitation in mm/year. For a detailed product and processing overview, see table S1.

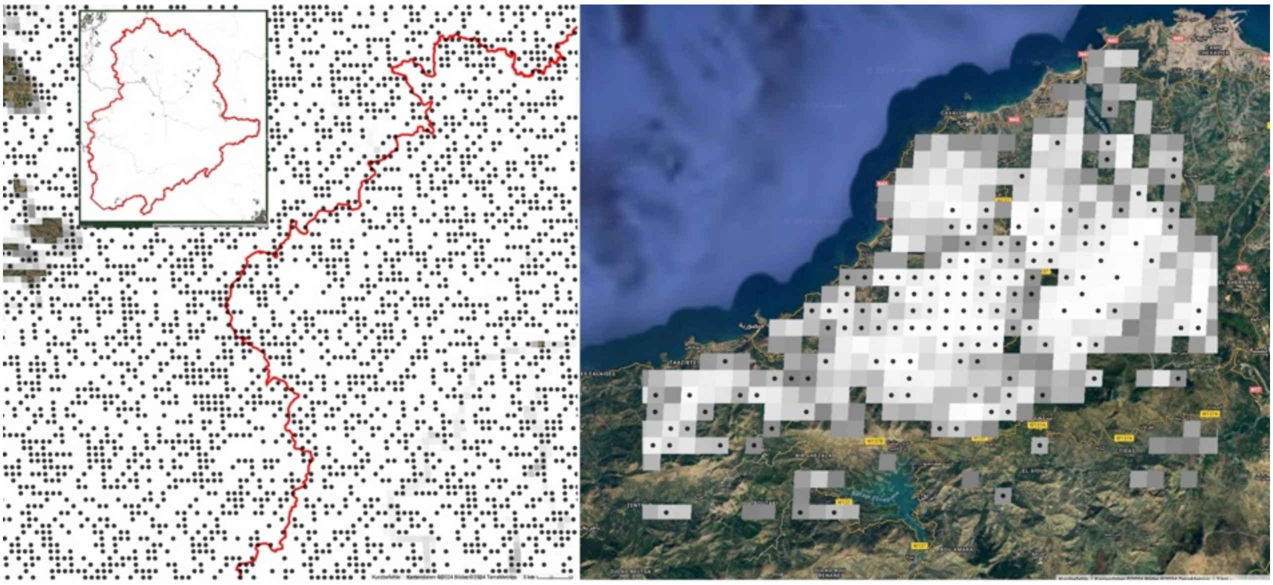


Fig. S1: Sampling illustration screenshot from the Google Earth Engine.
Forest cover from 50 to 100 % in dark gray to light gray. Sample points as 30 % of the forest cover area (10% inside and 20% outside). biosphere reserves' borders in red. Left: Northwest corner of the largest biosphere reserve, Alto Orinoco Casiquiare in Venezuela, with 8.5 Mio ha and > 150.000 sampling points (inside right of the red line) and total area in the top left corner. Right: The smallest biosphere reserve, Taza, in Algeria, with 1.600 ha and 147 sampling points.

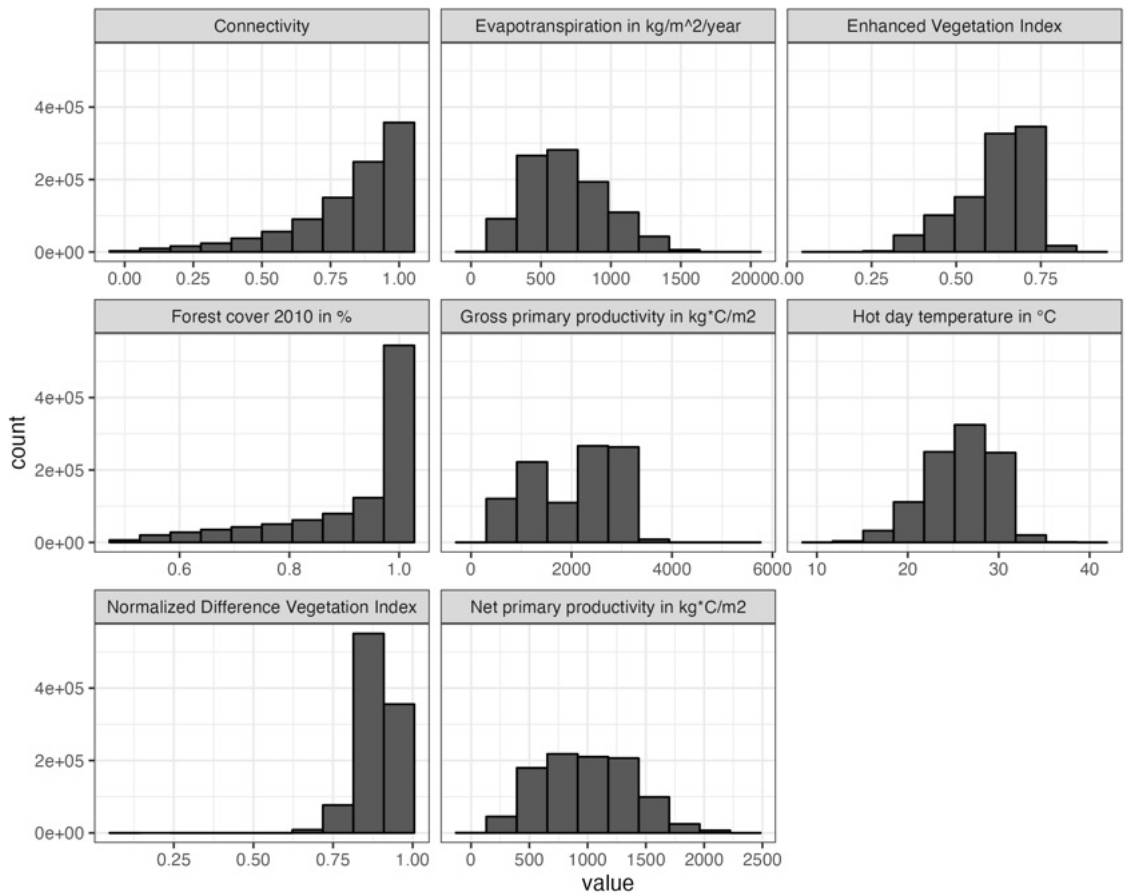


Fig. S2: Number of observations and value ranges of the processed satellite imagery used in this study. Each of the seven ecosystem functioning proxies is displayed with the per pixel means for the two time spans of 2010-16 and 2017-22. The forest coverage is from 2010. The observations cover the inside and surroundings of the investigated 119 biosphere reserves leading up to a total of more than 6 million observations.

Area characteristics

For each biosphere reserve, elevation and slope from the global multi-resolution terrain elevation data GMTED 2010 (*47*) were masked from water areas and scaled to 1 km. We did not use the commonly used SRTM data, since we required data above 60 °N and the overall accuracy of GMTED data is good (*48*). The water bodies distribution was based on the MODIS Terra land water mask at 250 m and the water mask 2010 (*49*) was selected and scaled to 1 km.

The land use land cover change data from the MODIS land cover type yearly data at 500 m (*50*) was processed by selecting land cover 2010, 2016 and 2021, comparing 2010 with 2016 and 2016 with 2021 pixel-wise and upscaling to 1 km (if >50 % land use land cover change in 1 km = 1). The burned area data from the MODIS Burned Area Monthly Global 500 m (*51*) indicating the day of a year a fire happened was processed by selecting the time series for each time step (2010-2016 and 2017-2022), transforming values to 1 if burned and 0 if not burned and scaling to 1 km. The metadata for each biosphere reserve, based on the described datasets above contain the year of establishment, biome and ecoregion information (*52*), tree cover percentage for 2010, 2016 and 2022, water cover percentage for 2010, burned area percentage for 2010-2016 and 2017-2022 and protected area cover percentage 2023 (*45*). For a detailed product and processing overview see the Supplementary Material (Table S2).

Statistics and linear mixed effect models

The dataset generated in the Google Earth Engine encompasses 7,938,752 values. They correspond to 119 biosphere reserves and their forest cover inside and outside for seven ecosystem functioning proxies and two time steps. We used the R programming language for all statistics with RStudio 2023.06.1+524 (*53*). To investigate if ecosystem functions in biosphere reserves differ from forests in their surroundings over time and worldwide, we used a linear mixed effect model framework, specifically the *lmer* - function which is part of the *lme4* –package (*54*), integrating the nested structure of our data into those models. All forest ecosystem functioning proxy data is continuous. To find the most fitting model, we run various models for each proxy. The biosphere reserves served as random effects. We fitted as fixed effects forest cover percentage, biome, time steps and the inside and outside to explore which combination best describes the behavior of each proxy. The Aikake Information Criterion (AIC) was used to decide for the best model for our dataset. The general structure of all models using the maximum likelihood where:

(1) proxy ~ Inside_Outside, random = ~ 1|Biosphere_Reserve
(2) proxy ~ Inside_Outside *Forest_Cover_2010, random = ~ 1|Biosphere_Reserve
(3) proxy ~ Inside_Outside * Forest_Cover_2010*biome, random = ~ 1| Biosphere_Reserve
(4) proxy ~ Inside_Outside * Forest_Cover_2010*biome*time, random = ~ 1| Biosphere_Reserve
(5) proxy ~ Inside_Outside * Forest_Cover_2010*biome+time* Inside_Outside, random = ~ 1| Biosphere_Reserve

To model the difference between the inside of the biosphere reserves and their surroundings, we build on the same model structure and used the median proxy difference of inside and outside:

(6) proxyDifference_Inside_Outside ~ Forest_Cover_2010_median*biome*time, random = ~ 1| Biosphere_Reserve

Uncertainty and influencing factors

In the course of this analysis, we investigated various sources of uncertainty. Since the number of biosphere reserves per biome is not balanced, we modeled the data with and without biomes with few numbers (n=3). Since no significant differences were observed, the biomes with small numbers of biosphere reserves remained in the dataset. We modeled and mapped the standard deviation of the biosphere reserves as random factors (Fig. S4). We compared the standard deviation of each model with the size of the biosphere reserves (Fig. S5). Other data on the number of observations per pixel per proxy per time step, population density, overlapping protected areas, water coverage, elevation and slope are available upon request. Other sources of uncertainty that were not investigated were globally different governance and management regimes and forest types.

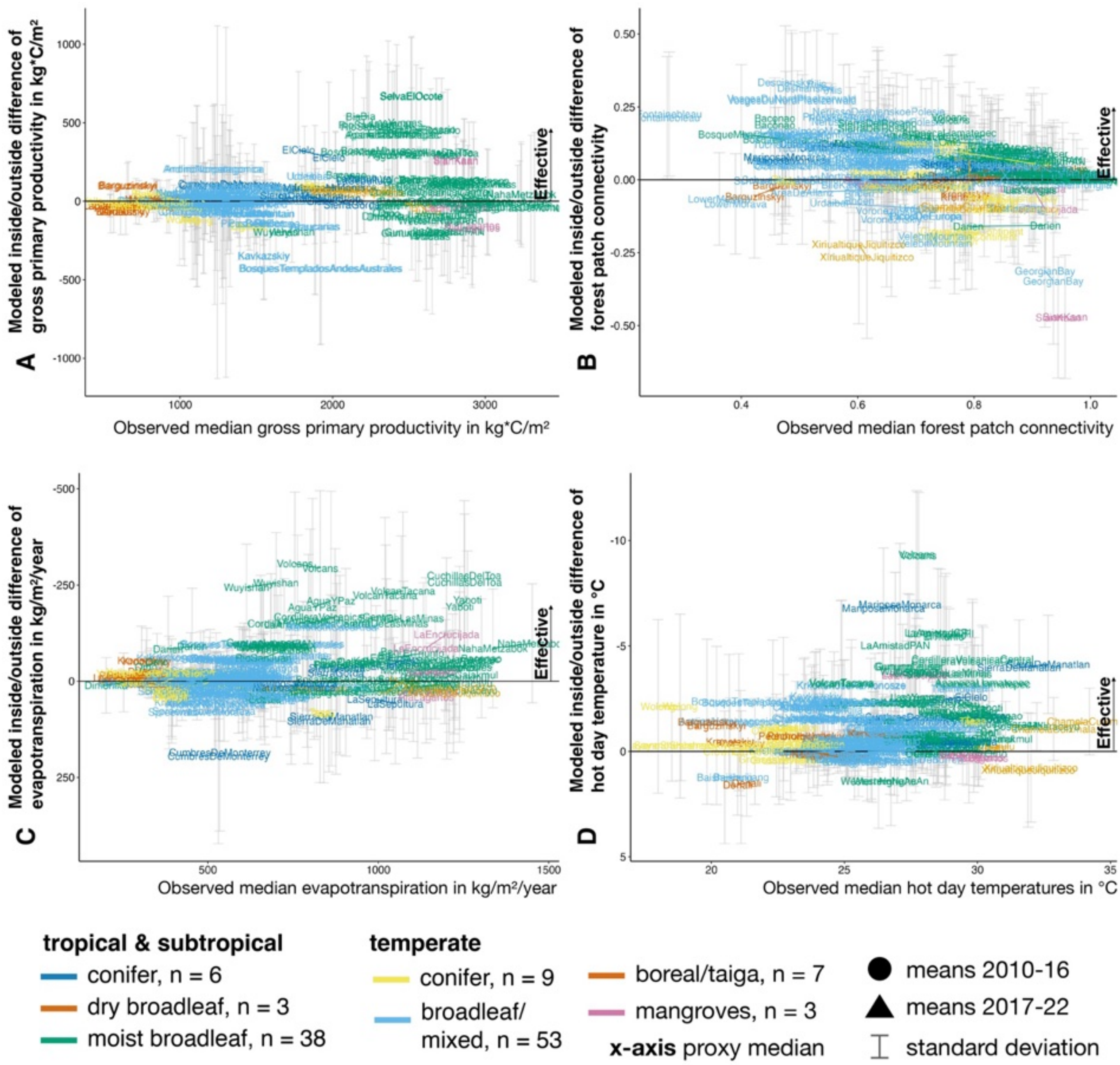


Fig. S3. Model predictions and standard deviation for the proxy difference inside and outside for each of the 119 biosphere reserves.
Data is depicted for two times 2010-16 (dots) and 2017-22 (triangles) against the respective median of the raw data. Predictions above 0 indicate higher proxy values inside than outside. Predictions below 0 indicate the opposite. Colors refer to biomes.
Model: (6) proxyDifference_Inside_Outside ~ Forest_Cover_2010_median*biome*time, random = ~ 1l Biosphere_Reserve.
A) Gross primary productivity, B) Connectivity, C) Evapotranspiration, D) Hot day temperature.

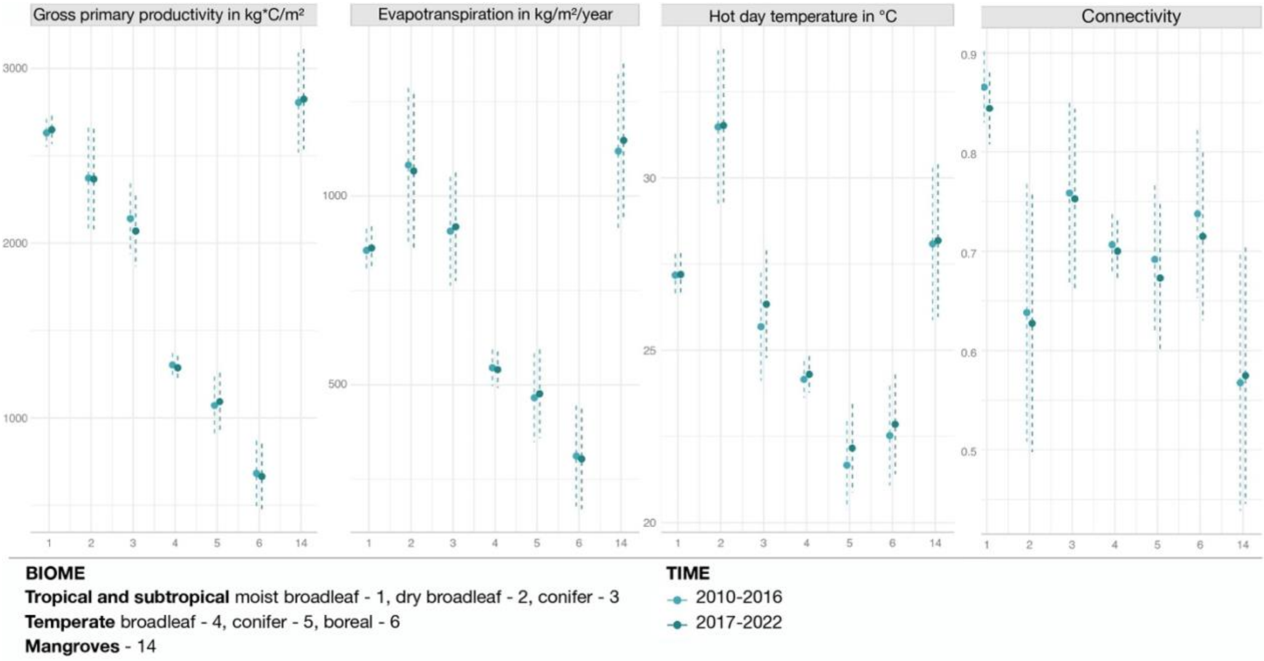


Fig. S4. Model predictions and confidence intervals.
Each proxy for each of the 119 biosphere reserves for two times 2010-16 (light blue) and 2017-22 (dark blue) in forested biomes for the surroundings (0) and the inside of the biosphere reserve (1). A) Gross primary productivity, B) Evapotranspiration, C) Hot-day temperature, D) Connectivity.

Fig. S5. World map of modeled standard deviation (SD) of the random effects of each biosphere reserve (n = 119).[2] Each standard deviation per biosphere reserve is scaled, multiplied by 255 and transformed into RGB-values for three ecosystem function proxies namely forest patch connectivity, hot day temperature and gross primary productivity. Smaller windows show the same effects for Central America and Europe.

[2] The color coding in map B differs from the original submission of the thesis framework. It was adapted to this corrected version for this publication.

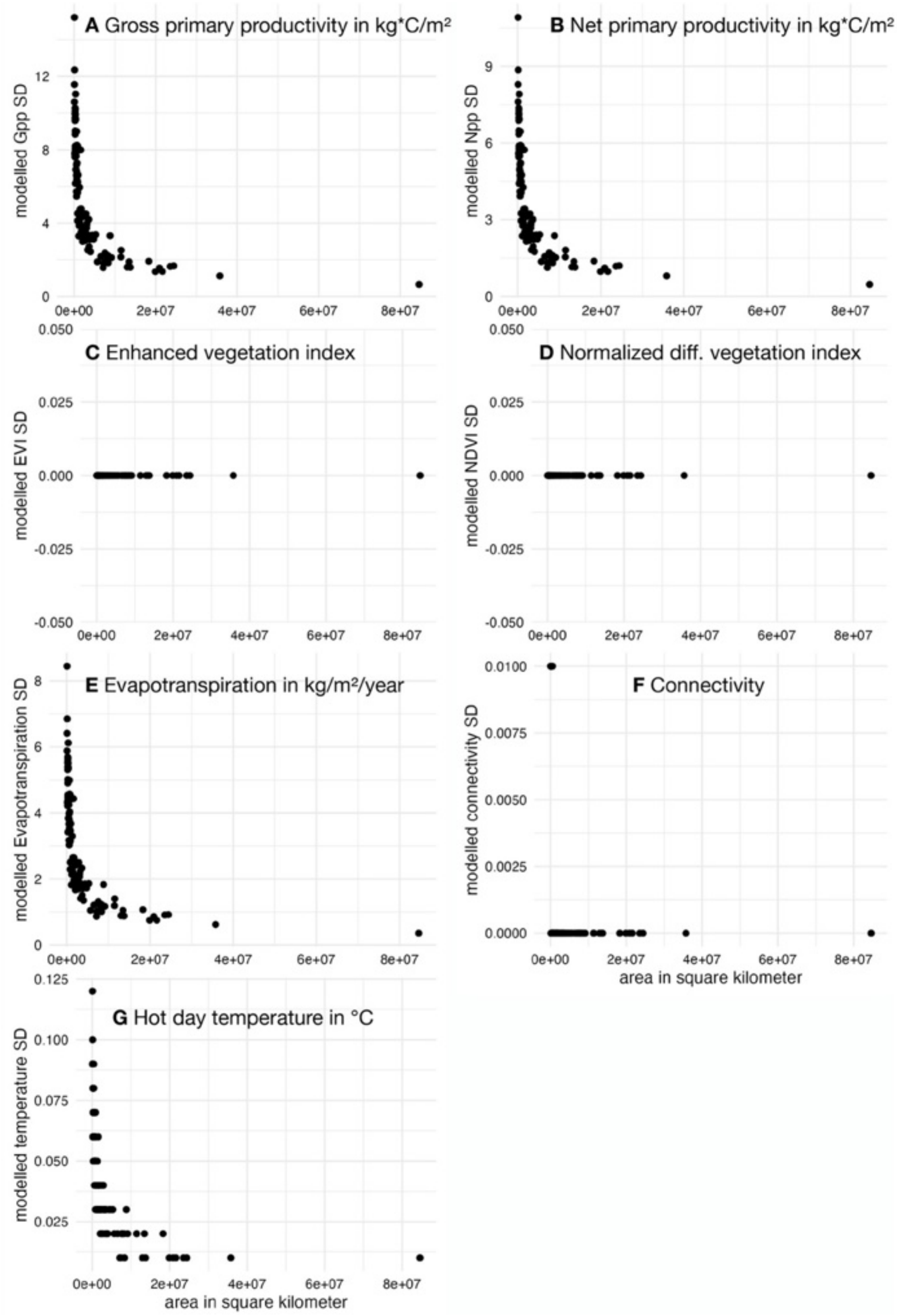


Fig. S6. Relations between area size of each biosphere reserve (n=119) and the corresponding modeling standard deviation for each forest ecosystem proxy.
Model: proxy ~ Inside_Outside * Forest_Cover_2010*biome*time, random = ~ 1| Biosphere_Reserve
A) Gross primary productivity, B) Net primary productivity, C) Enhanced Vegetation Index, D) Normalized Difference Vegetation Index, E) Evapotranspiration, F) Connectivity, G) Hot day temperature.

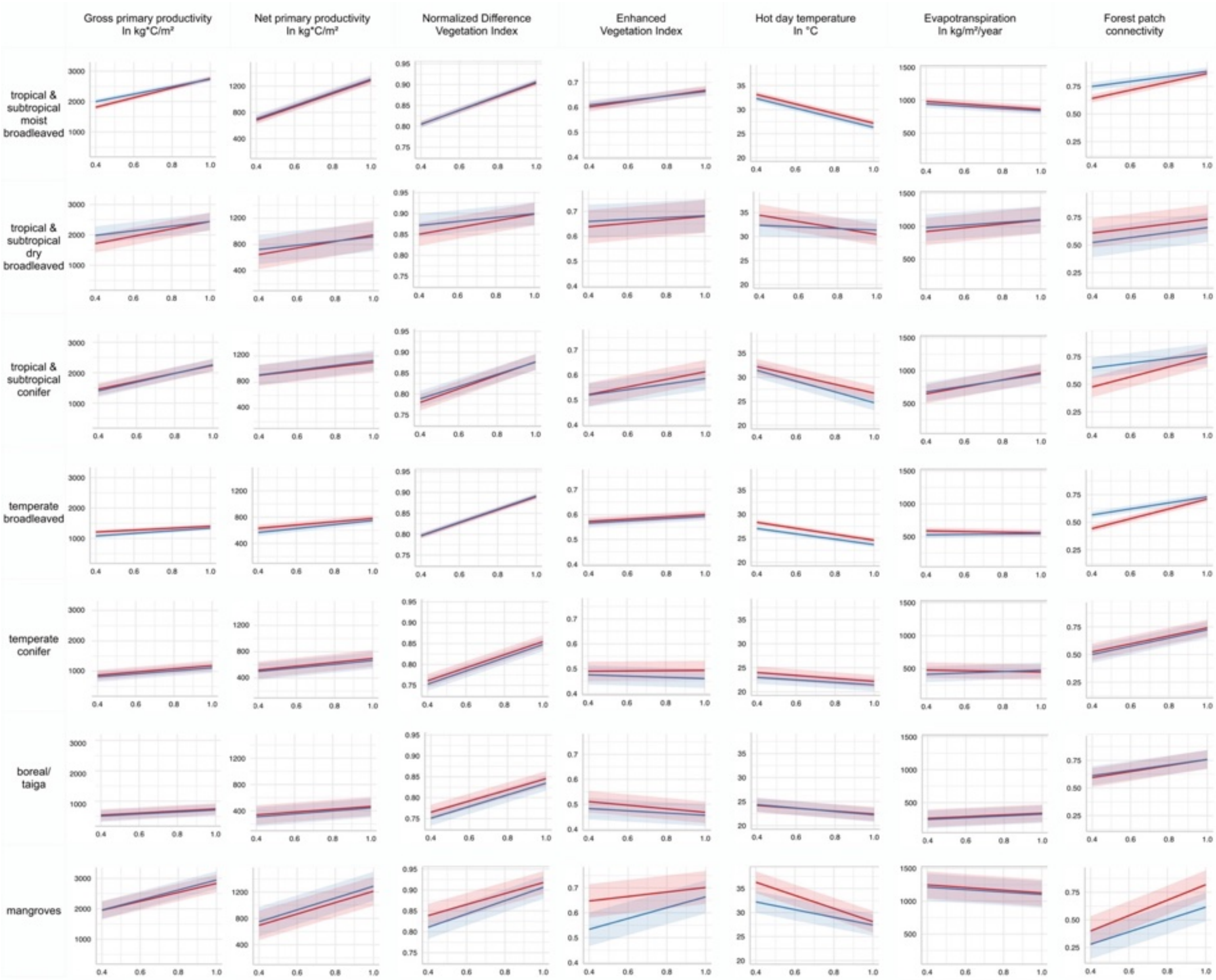


Fig. S7. Predicted forest ecosystem functioning for forest cover shares by biome. Biomes: tropical & subtropical moist broadleaf, dry broadleaf and conifer forests, temperate broadleaf and mixed and conifer forests, boreal forests and mangroves. Modeled observations are displayed for forests inside biosphere reserves in blue and for surrounding forests in red. Ribbons represent 0.95 confidence intervals. X-axis is forest cover share. Model with fixed effects of forest cover share, inside or outside the biosphere reserve, biome and two periods of time 2010-16 and 2017-22.

Table S1. Overview of datasets used in this study. Each dataset's processing is described in alphabetical order, and its use is justified.

Name	Data basis	DOI	Processing	General product	Pixel dimensions	Function	Justification
Biome	RESOLVE ecoregions 2017	https://academic.oup.com/bioscience/article/67/6/534/3102935 https://doi.org/10.1093/biosci/bix014	adding biome data to the biosphere reserves polygons	biome and ecoregions characteristics for each biosphere reserve	polygon	area characteristics	based on the commonly used biome and ecoregions definition of Olson et al. 2001
Biosphere reserves	UNESCO WNBR list, most recent WDPA shapes, Palliwoda et al. 2021 Europe	www.protectedplanet.net ; https://doi.org/10.3389/fevo.2021.736358	select if > 20 % forest cover inside and >10 % outside, surrounding area defined as log(area inside)*500	biosphere reserves borders and surrounding as polygons and binary raster (inside = 1, outside = 0)	0 = outside, 1 = inside	Area identification	WDPA most comprehensive database of protected areas currently available, Palliwoda et al. 2021 curated shapefiles for Europe
Burned Area	MCD64A1.061 MODIS Burned Area Monthly Global 500m	https://doi.org/10.5067/MODIS/MCD64A1.061	select burned area time series, transform values to 1 if burned and 0 if not burned in two time spans, upscale to 1 km	Feature and Proxy of two time spans (2010-2016 and 2017-2022) for forest cover	binary, 0 = no burn, 1 = burn	Feature: counts transformed to coverage per time span, output = table; Proxy: raster binary	burned area more reliable per pixel than detecting fires (hard to estimate the size of each fire); therefore the binary approach allows to see if fire was an influencing factor of ecosystem functions or not
Connectivity	Global Forest Cover Change (GFCC) Tree Cover Multi-Year Global 30m	https://doi.org/10.5067/MEaSUREs/GFCC/GFCC30TC.003; adapted from Justin Braaten: https://code.earthengine.google.com/94f412c3272d5c02139879a8ac4a7860	generate forest cover for 2016 and 2022, select and upscale as in forest cover 2016 and 2022 (see above); compute Thiessen Polygons around centroids of each forest fragment; compute size of Thiessen Polygons and size of forest fragments; calculate ratio between forest cover and corresponding Thiessen Polygon size	Proxy of two time spans (2010-2016 and 2017-2022) for forest cover	Connectivity value for each forest pixel at 1 km, range 0 to 1 (1 = most connected)	Proxy	based on Mann et al. 2023, works for 1 km resolution as an estimate of fragmentation
Elevation and Slope	GMTED2010	Danielson, J.J., Gesch, D.B., 2011, Global multi-resolution terrain elevation data 2010 (GMTED2010): U.S. Geological Survey Open-File Report 2011–1073, 26 p.	select elevation/slope, upscale to 1km, mask water areas	Characteristics for year 2010 for all land cover and for forest	continuous per pixel value of elevation and slope year 2010	area characteristics	values above 60°N; Accuracy assessments: https://ntrs.nasa.gov/api/citations/20110009936/downloads/20110009936.pdf

APPENDIX

Name	Data basis	DOI	Processing	General product	Pixel dimensions	Function	Justification
Evapo-transpi-ration	MOD16A2 .006: Terra Net Evapotranspiration 8-Day Global 500m	https://doi.org/10.5067/MODIS/MOD16A2.006	for each time span, compute the annual sums from 8-day dataset and generate the per pixel mean, upscale to 1 km, get for all land cover (except water) and for only forest	Proxy of two time spans (2010-2016 and 2017-2022) for for forest cover	unit kg/m²/8day, mean of annual sums in two time spans with unit kg/m²/year, range 0 to approx. 30000	Proxy	means of annual sums avoids the problems of seasonality for a global study
Forest cover	GLAD tree cover 2010 at 30 m	https://glad.umd.edu/Potapov/TCC_2010/	download from website, clip to selected BRs, upload to Google Earth Engine, >30 % tree cover to binary, upscale to 1 km	Forest cover >50 % at 1 km	0 to 100 %	Mask for Proxies, base for fragmentation product	For global datasets threshold of >30 % tree cover is used to identify forest; 50 % threshold justified by upscaling, if >50 % of the 1 km pixel is covered with trees, it is considered forest
Forest cover 2016 and 2022	Global Forest Cover Change (GFCC) Tree Cover Multi-Year Global 30m	https://doi.org/10.5067/MEaSUREs/GFCC/GFCC30TC.003	generate forest cover for 2016 and 2022, my masking forest loss (2001-2015 for forest cover 2016 and 2017-2021 for forest cover 2022), making tree cover % binary, upscale to 1km and use forest cover at 1 km for the two time spans with > 50 % and make binary again to compute % inside and outside	Feature for two years 2016 and 2022, coverage in side and outside	1 = feature exists, other is masked; 1 counted to compute coverage percentage for two years for inside and outside	Feature, output = table	best available dataset for forest cover change globally
Gross primary produc-tivity and net primary produc-tivity	Aqua Net Primary Production Gap-Filled Yearly Global 500m	https://doi.org/10.5067/MODIS/MYD17A3HGF.061	for each time span, generate the per pixel mean from the annual sums, upscale to 1 km	Proxy of two time spans (2010-2016 and 2017-2022) for forest cover	unit kg*C/m^2, mean of annual sums in two time spans, GPP range 0 to approx. 6000, NPP range -3000 to 3000	Proxy	means of annual sums avoids the problems of seasonality for a global study
Hot day tempe-rature	MYD11A1 .061 Aqua Land Surface Temperature Daily Global 1km	https://doi.org/10.5067/MODIS/MYD11A1.061	Filter days with maximum daytime temperature values in the area above 30 °C for two time spans; generate the per pixel mean of hot days	Proxy of two time spans (2010-2016 and 2017-2022) for forest cover	hot day means, continuous, pixel at 1 km range approx. -5 to 40°C	Proxy	based on Gohr et al. 2021, means of hot days avoid the problems of seasonality in a global study
Land cover change	MCD12Q1 .061 MODIS Land Cover Type Yearly Global 500m	https://doi.org/10.5067/MODIS/MCD12Q1.061	select land cover 2010, 2016 and 2021, compare 2010 with 2016 and 2016 with 2021, upscale to 1km (if >50% LULCC in 1km = 1)	Feature and Proxy of two time spans (2010-2016 and 2017-2022) for forest cover	binary, 0 = no change, 1 = change	Feature: counts transformed to coverage per time span, output = table; Proxy: raster binary	MODIS land cover type best available global and yearly updated dataset, focus on if there was a change or not rather than which land cover type was there at which time

Name	Data basis	DOI	Processing	General product	Pixel dimensions	Function	Justification
NDVI and EVI	MYD13A2 .061 Aqua Vegetation Indices 16-Day Global 1km	https://doi.org/10.5067/MODIS/MYD13A2.061	select NDVI/EVI time series, mask snow and clouds and water, generate annual max per pixel, get per pixel mean for each time span	Proxy of two time spans (2010-2016 and 2017-2022) for forest cover	max means, continuous, pixel at 1 km range 0 to 1	Proxy	using the means for six years of the annual maximum values allows for a global comparison and seasonality does not have to be considered
Popula-tion density	GPWv411	https://doi.org/10.7927/H49C6VHW.	select census 2010, water already masked	Feature and Proxy of 2010 (contains census collected between 2005-2014) for forest cover	Feature: counts transformed to coverage for 2010, output = table; Proxy: raster continuous per pixel value of estimated number of people per square kilometer	area characteristics	Selected only census of 2010 as year for human density, since other years are estimations. https://sedac.ciesin.columbia.edu/binaries/web/sedac/collections/gpw-v4/gpw-v4-documentation-rev11.pdf
Water bodies	MOD44W.006 Terra Land Water Mask Derived From MODIS and SRTM Yearly Global 250m	https://doi.org/10.5067/MODIS/MOD44W.006	select water mask 2010, upscale to 1 km	Watermask: Mask for water not to be included in the Proxies all land cover types raster; Water bodies: include water bodies for % cover of features	Watermask: water = masked, no water = 1; Water bodies: water = 1, no water= masked	Watermask for Proxies for all land cover types, base for water bodies cover % for features	best available dataset for water bodies globally

Table S2. Model outputs for model selection.
mod1: proxy ~ Inside_Outside, random = ~ 1|BR
mod2: proxy ~ Inside_Outside *Forest_Cover_2010, random = ~ 1|Biosphere_Reserve
mod3: proxy ~ Inside_Outside * Forest_Cover_2010*biome, random = ~ 1|
Biosphere_Reserve
mod4: proxy ~ Inside_Outside * Forest_Cover_2010*biome*time, random = ~ 1|
Biosphere_Reserve
mod5: proxy ~ Inside_Outside * Forest_Cover_2010*biome+time* Inside_Outside, random =
~ 1| Biosphere_Reserve

Proxy	model	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Gpp	mod4	58	13864046	0.000	1,00E+06	1,00E+06	-6931965	1
Gpp	mod5	32	13865453	1.407.156	2,75E-300	2,75E-300	-6932695	1
Gpp	mod3	30	13865501	1.455.401	9.195668e-317	9.195668e-317	-6932721	1
Gpp	mod2	6	13923896	59.850.167	0.000000e+00	0.000000e+00	-6961942	1
Gpp	mod1	4	14031885	167.838.729	0.000000e+00	0.000000e+00	-7015938	1
Npp	mod4	58	13203471	0.000	1	1	-6601677	1
Npp	mod5	32	13205040	1.569.013	0	0	-6602488	1
Npp	mod3	30	13205204	1.732.948	0	0	-6602572	1
Npp	mod2	6	13261095	57.624.839	0	0	-6630542	1
Npp	mod1	4	13360342	156.871.853	0	0	-6680167	1
EVI	mod4	58	-2773562	0.0000	1,00E+06	1,00E+06	1386839	1
EVI	mod5	32	-2772969	5.933.531	1,43E-123	1,43E-123	1386516	1
EVI	mod3	30	-2770697	28.650.273	0.000000e+00	0.000000e+00	1385379	1
EVI	mod2	6	-2755193	183.690.818	0.000000e+00	0.000000e+00	1377602	1
EVI	mod1	4	-2743433	301.290.690	0.000000e+00	0.000000e+00	1371720	1
NDVI	mod4	58	-4299942	0.000	1	1	2150029	1
NDVI	mod5	32	-4298344	1.598.215	0	0	2149204	1
NDVI	mod3	30	-4298300	1.642.324	0	0	2149180	1
NDVI	mod2	6	-4287592	12.350.479	0	0	2143802	1
NDVI	mod1	4	-3982400	317.542.496	0	0	1991204	1
LST	mod4	58	4290312	0.000	1,00E+06	1,00E+06	-2145098	1
LST	mod5	32	4291589	1.276.813	5,54E-272	5,54E-272	-2145762	1
LST	mod3	30	4292489	2.177.208	0.000000e+00	0.000000e+00	-2146215	1
LST	mod2	6	4316507	26.194.863	0.000000e+00	0.000000e+00	-2158247	1
LST	mod1	4	4437764	147.452.492	0.000000e+00	0.000000e+00	-2218878	1
ET	mod4	58	12691774	0.000	1	1	-6345829	1
ET	mod5	32	12694163	2.388.878	0	0	-6347049	1
ET	mod3	30	12694172	2.398.196	0	0	-6347056	1
ET	mod2	6	12713311	21.537.396	0	0	-6356650	1
ET	mod1	4	12718489	26.714.972	0	0	-6359240	1
Conn	mod4	58	-966936.7	0.000	1,00E+06	1,00E+06	483526.3	1
Conn	mod5	32	-965924.7	1.011.987	1,78E-214	1,78E-214	482994.3	1
Conn	mod3	30	-961476.3	5.460.352	0.000000e+00	0.000000e+00	480768.2	1
Conn	mod2	6	-950414.7	16.521.933	0.000000e+00	0.000000e+00	475213.4	1
Conn	mod1	4	-883623.6	83.313.020	0.000000e+00	0.000000e+00	441815.8	1

Table S3. Model outputs of fixed and random effects for forest ecosystem proxies. The table includes marginal and conditional r^2 for all modeled forest ecosystem function proxies differences between inside and outside following the model: proxyDifference_Inside_Outside ~ Forest_Cover_2010_median*biome*time, random = ~ 1l Biosphere_Reserve

	Gppdiff			Nppdiff			EVIdiff			NDVIdiff			ETdiff			LSTdiff			Conndiff		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
(Intercept)	1642.06	1057.91 – 2226.21	<0.001	1047.65	679.76 – 1415.53	<0.001	0.03	-0.11 – 0.17	0.644	0.08	0.02 – 0.15	0.011	-385.53	-642.08 – -128.99	0.003	-5.16	-11.45 – 1.12	0.107	0.52	0.08 – 0.97	0.022
FOR10 median	-1650.35	-2280.01 – -1020.69	<0.001	-1012.58	-1409.13 – -616.04	<0.001	-0.04	-0.19 – 0.11	0.600	-0.08	-0.14 – -0.01	0.029	341.98	65.45 – 618.52	0.016	3.42	-3.36 – 10.19	0.321	-0.51	-0.99 – -0.03	0.039
Biome [2]	-989.65	-3748.43 – 1769.13	0.480	-733.58	-2470.98 – 1003.83	0.406	-0.18	-0.84 – 0.49	0.597	-0.03	-0.33 – 0.27	0.824	525.34	-686.27 – 1736.94	0.394	-1.80	-31.48 – 27.87	0.905	-1.50	-3.61 – 0.61	0.163
Biome [3]	-2122.40	-3899.92 – -344.88	0.020	-1541.66	-2661.09 – -422.23	0.007	0.28	-0.15 – 0.71	0.195	0.20	0.01 – 0.40	0.039	1274.80	494.15 – 2055.46	0.001	-6.10	-25.22 – 13.02	0.530	-0.42	-1.78 – 0.94	0.539
Biome [4]	-1405.76	-2160.42 – -651.10	<0.001	-917.27	-1392.54 – -442.01	<0.001	0.02	-0.16 – 0.20	0.851	-0.05	-0.13 – 0.03	0.207	378.64	47.21 – 710.08	0.025	5.57	-2.55 – 13.69	0.178	-0.09	-0.67 – 0.48	0.751
Biome [5]	-1202.13	-2462.36 – 58.09	0.061	-871.17	-1664.82 – -77.51	0.032	0.16	-0.15 – 0.46	0.307	-0.09	-0.22 – 0.05	0.208	282.07	-271.40 – 835.54	0.316	7.28	-6.27 – 20.84	0.291	-0.14	-1.10 – 0.82	0.776
Biome [6]	-1033.53	-3201.16 – 1134.10	0.348	-679.81	-2044.93 – 685.30	0.327	-0.02	-0.54 – 0.50	0.939	-0.02	-0.25 – 0.22	0.879	418.87	-533.12 – 1370.85	0.387	4.32	-19.00 – 27.63	0.716	-0.67	-2.33 – 0.99	0.428
Biome [14]	-6007.28	-13307.52 – 1292.97	0.106	-4902.88	-9500.38 – -305.38	0.037	0.46	-1.30 – 2.22	0.605	0.49	-0.30 – 1.29	0.221	-1114.83	-4320.97 – 2091.31	0.494	-72.96	-151.49 – 5.57	0.068	4.97	-0.61 – 10.56	0.081
time [1722]	-71.69	-155.33 – 11.95	0.093	-20.06	-70.07 – 29.95	0.430	0.01	-0.02 – 0.05	0.389	-0.01	-0.03 – 0.01	0.195	-65.31	-136.98 – 6.36	0.074	0.66	-0.19 – 1.51	0.127	0.08	-0.13 – 0.30	0.444
FOR10 median × Biome [2]	911.78	-2431.72 – 4255.28	0.591	600.57	-1505.08 – 2706.22	0.575	0.23	-0.58 – 1.03	0.579	0.02	-0.34 – 0.39	0.907	-499.77	-1968.18 – 968.64	0.503	4.92	-31.05 – 40.89	0.788	1.60	-0.96 – 4.16	0.219

	Gppdiff			Nppdiff			EVIdiff			NDVIdiff			ETdiff			LSTdiff			Conndiff		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
FOR10 median × Biome [3]	2337.78	261.11 – 4414.46	0.028	1613.59	305.76 – 2921.43	0.016	-0.35	-0.85 – 0.15	0.175	-0.24	-0.47 – -0.02	0.036	-1345.54	-2257.58 – -433.50	0.004	6.64	-15.70 – 28.98	0.559	0.46	-1.13 – 2.05	0.568
FOR10 median × Biome [4]	1358.75	519.36 – 2198.15	0.002	852.66	324.03 – 1381.29	0.002	-0.02	-0.22 – 0.18	0.847	0.05	-0.04 – 0.14	0.280	-341.18	-709.83 – 27.47	0.070	-4.90	-13.93 – 4.13	0.286	0.06	-0.59 – 0.70	0.862
FOR10 median × Biome [5]	1125.13	-293.67 – 2543.94	0.119	796.85	-96.68 – 1690.37	0.080	-0.20	-0.54 – 0.15	0.259	0.08	-0.07 – 0.23	0.308	-205.92	-829.03 – 417.20	0.515	-6.65	-21.91 – 8.62	0.392	0.02	-1.07 – 1.10	0.975
FOR10 median × Biome [6]	971.85	-1401.96 – 3345.65	0.421	594.39	-900.57 – 2089.35	0.434	0.01	-0.56 – 0.58	0.979	-0.00	-0.26 – 0.26	0.997	-394.47	-1437.00 – 648.06	0.457	-2.76	-28.30 – 22.78	0.831	0.67	-1.15 – 2.48	0.469
FOR10 median × Biome [14]	6137.41	-1340.50 – 13615.31	0.107	5005.68	296.30 – 9715.06	0.037	-0.51	-2.31 – 1.29	0.575	-0.52	-1.34 – 0.29	0.205	1171.77	-2112.39 – 4455.94	0.483	75.50	-4.94 – 155.94	0.066	-5.33	-11.05 – 0.39	0.068
FOR10 median × time [1722]	78.58	-11.58 – 168.74	0.087	17.76	-36.15 – 71.66	0.517	-0.01	-0.05 – 0.02	0.456	0.01	-0.01 – 0.03	0.203	58.00	-19.26 – 135.25	0.140	-0.66	-1.58 – 0.26	0.156	-0.08	-0.31 – 0.15	0.495
Biome [2] × time [1722]	338.83	-56.20 – 733.86	0.092	106.38	-129.79 – 342.55	0.376	-0.02	-0.18 – 0.14	0.824	-0.02	-0.10 – 0.07	0.690	-102.58	-441.05 – 235.90	0.551	-3.67	-7.69 – 0.35	0.073	0.27	-0.74 – 1.29	0.595
Biome [3] × time [1722]	-179.40	-433.93 – 75.12	0.166	-90.19	-242.36 – 61.98	0.244	0.02	-0.09 – 0.12	0.747	-0.00	-0.06 – 0.05	0.902	50.36	-167.73 – 268.44	0.649	-1.21	-3.80 – 1.38	0.357	-0.01	-0.67 – 0.64	0.968
Biome [4] × time [1722]	86.94	-21.12 – 194.99	0.114	14.67	-49.93 – 79.28	0.655	0.00	-0.04 – 0.04	0.991	0.01	-0.01 – 0.03	0.367	52.21	-40.38 – 144.80	0.268	-1.06	-2.16 – 0.04	0.058	-0.26	-0.54 – 0.01	0.063
Biome [5] × time [1722]	30.60	-149.85 – 211.05	0.738	19.76	-88.13 – 127.64	0.718	-0.01	-0.08 – 0.06	0.819	0.04	0.00 – 0.08	0.047	58.14	-96.48 – 212.75	0.459	-0.93	-2.76 – 0.91	0.322	-0.28	-0.75 – 0.18	0.230
Biome [6] × time [1722]	-1.80	-312.18 – 308.58	0.991	-63.65	-249.21 – 121.92	0.500	-0.03	-0.15 – 0.10	0.691	-0.02	-0.08 – 0.05	0.605	22.82	-243.13 – 288.77	0.866	0.57	-2.59 – 3.72	0.724	-0.40	-1.20 – 0.40	0.325
Biome [14] × time [1722]	338.44	-706.87 – 1383.76	0.524	5.21	-619.75 – 630.16	0.987	-0.10	-0.53 – 0.32	0.627	-0.15	-0.36 – 0.07	0.176	-526.85	-1422.52 – 368.82	0.248	1.27	-9.37 – 11.91	0.814	1.28	-1.40 – 3.97	0.347

	Gppdiff			Nppdiff			EVIdiff			NDVIdiff			ETdiff			LSTdiff			Conndiff		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
(FOR10 median × Biome [2]) × time [1722]	-434.66	-913.41 – 44.09	0.075	-123.64	-409.87 – 162.59	0.395	0.02	-0.17 – 0.21	0.843	0.02	-0.08 – 0.12	0.660	163.08	-247.13 – 573.30	0.434	4.17	-0.70 – 9.04	0.093	-0.40	-1.64 – 0.83	0.517
(FOR10 median × Biome [3]) × time [1722]	236.03	-61.32 – 533.39	0.119	110.18	-67.60 – 287.96	0.223	-0.02	-0.14 – 0.10	0.780	0.01	-0.06 – 0.07	0.866	-26.12	-280.91 – 228.66	0.840	1.14	-1.89 – 4.16	0.460	0.01	-0.76 – 0.77	0.989
(FOR10 median × Biome [4]) × time [1722]	-96.35	-216.54 – 23.84	0.116	-11.71	-83.56 – 60.15	0.748	-0.00	-0.05 – 0.04	0.874	-0.01	-0.04 – 0.01	0.366	-46.16	-149.14 – 56.83	0.378	1.17	-0.06 – 2.39	0.062	0.31	-0.00 – 0.62	0.052
(FOR10 median × Biome [5]) × time [1722]	-35.65	-238.81 – 167.51	0.730	-19.38	-140.84 – 102.08	0.753	0.01	-0.07 – 0.09	0.834	-0.04	-0.08 – 0.00	0.051	-54.02	-228.10 – 120.05	0.541	0.94	-1.13 – 3.01	0.372	0.32	-0.20 – 0.85	0.222
(FOR10 median × Biome [6]) × time [1722]	1.91	-337.99 – 341.82	0.991	73.83	-129.39 – 277.04	0.475	0.02	-0.11 – 0.16	0.741	0.02	-0.05 – 0.09	0.653	-14.03	-305.28 – 277.21	0.924	-0.60	-4.06 – 2.86	0.734	0.41	-0.46 – 1.29	0.354
(FOR10 median × Biome [14]) × time [1722]	-337.18	-1407.93 – 733.58	0.535	2.02	-638.14 – 642.19	0.995	0.11	-0.32 – 0.54	0.620	0.15	-0.07 – 0.37	0.174	529.25	-388.22 – 1446.71	0.257	-1.28	-12.18 – 9.62	0.817	-1.29	-4.04 – 1.46	0.357
Random Effects																					
σ^2	209.12			74.75			0.00			0.00			153.53			0.02			0.00		
τ_{00}	20189.42 $_{BR}$			8015.60 $_{BR}$			0.00 $_{BR}$			0.00 $_{BR}$			3780.96 $_{BR}$			2.34 $_{BR}$			0.01 $_{BR}$		
ICC	0.99			0.99			0.97			0.96			0.96			0.99			0.88		
N	119 $_{BR}$			119 $_{BR}$			119 $_{BR}$			119 $_{BR}$			119 $_{BR}$			119 $_{BR}$			119 $_{BR}$		
Observations	238			238			238			238			238			238			238		
Marginal R^2 / Conditional R^2	**0.306 / 0.993**			**0.384 / 0.994**			**0.106 / 0.974**			**0.219 / 0.971**			**0.317 / 0.973**			**0.204 / 0.993**			**0.251 / 0.913**		

Table S4. Model outputs of fixed and random effects for forest ecosystem proxies. Table includes marginal and conditional r^2 for all modeled forest ecosystem function proxies following the model:
proxy ~ Inside_Outside *Forest_Cover_2010*biome*time, random = ~ 1| Biosphere_Reserve

	Gpp			Npp			EVI			NDVI			LST			ET			Conn		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
(Intercept)	1183.53	1102.23 – 1264.84	**<0.001**	277.86	215.62 – 340.10	**<0.001**	0.56	0.54 – 0.58	**<0.001**	0.74	0.73 – 0.75	**<0.001**	37.12	36.49 – 37.75	**<0.001**	1063.32	1005.86 – 1120.78	**<0.001**	0.49	0.45 – 0.53	**<0.001**
InOut	325.37	307.31 – 343.42	**<0.001**	22.92	9.98 – 35.86	**0.001**	0.02	0.01 – 0.02	**<0.001**	-0.00	-0.01 – -0.00	**0.001**	-0.84	-0.98 – -0.69	**<0.001**	-50.85	-60.85 – -40.84	**<0.001**	0.17	0.16 – 0.18	**<0.001**
FOR10	1575.35	1562.98 – 1587.72	**<0.001**	1009.02	1000.15 – 1017.89	**<0.001**	0.11	0.11 – 0.11	**<0.001**	0.16	0.16 – 0.16	**<0.001**	-9.92	-10.02 – -9.83	**<0.001**	-200.88	-207.73 – -194.03	**<0.001**	0.38	0.37 – 0.38	**<0.001**
Biome [2]	46.44	-267.20 – 360.08	0.772	169.18	-69.71 – 408.07	0.165	0.05	-0.02 – 0.13	0.151	0.08	0.05 – 0.11	**<0.001**	-0.00	-2.44 – 2.43	0.998	-259.09	-477.23 – -40.96	**0.020**	0.03	-0.11 – 0.18	0.645
Biome [3]	-243.62	-465.69 – -21.55	**0.032**	496.13	326.31 – 665.95	**<0.001**	-0.10	-0.15 – -0.05	**<0.001**	-0.02	-0.05 – -0.00	**0.022**	-1.25	-2.97 – 0.47	0.154	-631.74	-788.16 – -475.31	**<0.001**	-0.20	-0.30 – -0.10	**<0.001**
Biome [4]	-100.58	-207.44 – 6.27	0.065	251.72	169.95 – 333.49	**<0.001**	-0.00	-0.03 – 0.02	0.789	-0.01	-0.02 – 0.00	0.224	-6.34	-7.17 – -5.52	**<0.001**	-453.59	-529.02 – -378.16	**<0.001**	-0.23	-0.27 – -0.18	**<0.001**
Biome [5]	-518.46	-705.26 – -331.66	**<0.001**	125.93	-16.97 – 268.84	0.084	-0.07	-0.11 – -0.03	**0.001**	-0.04	-0.06 – -0.02	**<0.001**	-11.89	-13.34 – -10.45	**<0.001**	-565.57	-697.31 – -433.83	**<0.001**	-0.11	-0.20 – -0.03	**0.010**
Biome [6]	-774.10	-979.89 – -568.31	**<0.001**	-24.53	-182.10 – 133.04	0.760	-0.02	-0.07 – 0.03	0.447	-0.03	-0.05 – -0.01	**0.005**	-11.74	-13.33 – -10.15	**<0.001**	-856.37	-1001.90 – -710.83	**<0.001**	-0.01	-0.10 – 0.09	0.893
Biome [14]	177.60	-136.52 – 491.71	0.268	70.43	-168.78 – 309.64	0.564	0.05	-0.02 – 0.13	0.142	0.05	0.02 – 0.08	**0.003**	4.63	2.19 – 7.06	**<0.001**	266.20	47.85 – 484.55	**0.017**	-0.37	-0.52 – -0.23	**<0.001**
time [1722]	10.20	-5.51 – 25.91	0.203	-56.60	-67.87 – -45.34	**<0.001**	-0.01	-0.01 – -0.01	**<0.001**	-0.00	-0.00 – 0.00	0.595	-0.23	-0.36 – -0.10	**<0.001**	154.81	146.11 – 163.51	**<0.001**	-0.09	-0.10 – -0.08	**<0.001**
InOut × FOR10	-348.03	-366.95 – -329.11	**<0.001**	-4.80	-18.37 – 8.76	0.488	-0.02	-0.03 – -0.02	**<0.001**	0.01	0.00 – 0.01	**<0.001**	-0.04	-0.19 – 0.11	0.629	29.76	19.28 – 40.24	**<0.001**	-0.15	-0.16 – -0.14	**<0.001**

	Gpp			Npp			EVI			NDVI			LST			ET			Conn		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
InOut × Biome [2]	127.01	-17.83 – 271.84	0.086	126.20	22.37 – 230.03	**0.017**	0.02	-0.01 – 0.05	0.283	0.04	0.02 – 0.05	**<0.001**	-3.28	-4.45 – -2.12	**<0.001**	142.75	62.52 – 222.98	**<0.001**	-0.26	-0.34 – -0.18	**<0.001**
InOut × Biome [3]	-437.82	-502.08 – -373.56	**<0.001**	-30.87	-76.94 – 15.20	0.189	-0.00	-0.02 – 0.01	0.839	0.02	0.01 – 0.02	**<0.001**	0.87	0.36 – 1.39	**0.001**	116.24	80.64 – 151.84	**<0.001**	0.10	0.06 – 0.14	**<0.001**
InOut × Biome [4]	-494.47	-520.73 – -468.20	**<0.001**	-108.33	-127.16 – -89.50	**<0.001**	-0.02	-0.03 – -0.02	**<0.001**	0.00	0.00 – 0.01	**0.019**	-0.68	-0.89 – -0.47	**<0.001**	-44.21	-58.76 – -29.66	**<0.001**	0.02	0.01 – 0.04	**0.001**
InOut × Biome [5]	-367.78	-413.82 – -321.74	**<0.001**	-38.90	-71.90 – -5.90	**0.021**	-0.02	-0.03 – -0.01	**0.001**	-0.01	-0.01 – -0.00	**0.033**	-0.37	-0.74 – -0.00	**0.049**	-74.77	-100.27 – -49.27	**<0.001**	-0.20	-0.22 – -0.17	**<0.001**
InOut × Biome [6]	-353.75	-392.77 – -314.74	**<0.001**	-58.44	-86.41 – -30.48	**<0.001**	-0.05	-0.06 – -0.05	**<0.001**	-0.01	-0.02 – -0.01	**<0.001**	1.26	0.95 – 1.58	**<0.001**	32.98	11.37 – 54.59	**0.003**	-0.14	-0.16 – -0.12	**<0.001**
InOut × Biome [14]	-384.52	-521.45 – -247.60	**<0.001**	17.94	-80.22 – 116.11	0.720	-0.18	-0.21 – -0.15	**<0.001**	-0.04	-0.05 – -0.02	**<0.001**	-5.52	-6.62 – -4.42	**<0.001**	10.56	-65.29 – 86.42	0.785	-0.23	-0.31 – -0.15	**<0.001**
FOR10 × Biome [2]	-366.93	-488.33 – -245.53	**<0.001**	-510.39	-597.41 – -423.36	**<0.001**	-0.04	-0.07 – -0.01	**0.004**	-0.08	-0.10 – -0.07	**<0.001**	3.25	2.28 – 4.23	**<0.001**	490.26	423.01 – 557.51	**<0.001**	-0.17	-0.24 – -0.10	**<0.001**
FOR10 × Biome [3]	-275.05	-325.51 – -224.60	**<0.001**	-680.09	-716.26 – -643.92	**<0.001**	0.04	0.03 – 0.05	**<0.001**	-0.00	-0.01 – 0.00	0.516	0.75	0.35 – 1.16	**<0.001**	736.59	708.64 – 764.54	**<0.001**	0.08	0.05 – 0.11	**<0.001**
FOR10 × Biome [4]	-1255.75	-1275.09 – -1236.41	**<0.001**	-758.72	-772.58 – -744.85	**<0.001**	-0.06	-0.07 – -0.06	**<0.001**	-0.01	-0.01 – -0.01	**<0.001**	3.73	3.57 – 3.88	**<0.001**	150.74	140.03 – 161.46	**<0.001**	0.07	0.06 – 0.08	**<0.001**
FOR10 × Biome [5]	-1053.49	-1090.41 – -1016.57	**<0.001**	-715.67	-742.13 – -689.20	**<0.001**	-0.10	-0.11 – -0.10	**<0.001**	-0.01	-0.01 – -0.00	**<0.001**	6.86	6.57 – 7.16	**<0.001**	150.57	130.12 – 171.02	**<0.001**	-0.01	-0.03 – 0.01	0.208
FOR10 × Biome [6]	-1238.38	-1268.16 – -1208.61	**<0.001**	-796.25	-817.60 – -774.90	**<0.001**	-0.18	-0.19 – -0.18	**<0.001**	-0.03	-0.03 – -0.03	**<0.001**	6.97	6.73 – 7.21	**<0.001**	332.59	316.10 – 349.09	**<0.001**	-0.10	-0.12 – -0.08	**<0.001**
FOR10 × Biome [14]	-101.98	-208.14 – 4.18	0.060	-143.59	-219.69 – -67.49	**<0.001**	-0.02	-0.05 – 0.00	0.090	-0.03	-0.04 – -0.02	**<0.001**	-3.73	-4.58 – -2.88	**<0.001**	-1.42	-60.23 – 57.38	0.962	0.32	0.26 – 0.38	**<0.001**

		Gpp			Npp			EVI			NDVI			LST			ET			Conn	
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
InOut × time [1722]	-29.19	-54.57 – -3.82	**0.024**	-6.33	-24.52 – 11.86	0.495	0.00	-0.01 – 0.01	0.808	-0.00	-0.01 – -0.00	**0.002**	-0.40	-0.61 – -0.20	**<0.001**	-8.99	-23.05 – 5.06	0.210	-0.01	-0.02 – 0.01	0.296
FOR10 × time [1722]	3.96	-12.75 – 20.67	0.642	63.61	51.63 – 75.59	**<0.001**	0.01	0.01 – 0.02	**<0.001**	-0.00	-0.00 – 0.00	0.137	0.31	0.17 – 0.44	**<0.001**	-163.75	-173.01 – -154.50	**<0.001**	0.07	0.06 – 0.08	**<0.001**
Biome [2] × time [1722]	-77.54	-217.07 – 62.00	0.276	17.09	-82.94 – 117.12	0.738	0.01	-0.02 – 0.04	0.670	-0.00	-0.02 – 0.01	0.719	0.87	-0.25 – 1.99	0.127	-137.21	-214.51 – -59.92	**0.001**	0.08	0.01 – 0.16	**0.036**
Biome [3] × time [1722]	3.43	-56.36 – 63.22	0.910	90.08	47.22 – 132.95	**<0.001**	-0.00	-0.01 – 0.01	0.878	0.00	-0.00 – 0.01	0.303	0.81	0.33 – 1.29	**0.001**	-176.50	-209.62 – -143.38	**<0.001**	0.08	0.05 – 0.11	**<0.001**
Biome [4] × time [1722]	-4.34	-28.17 – 19.50	0.721	48.79	31.71 – 65.88	**<0.001**	0.01	0.01 – 0.02	**<0.001**	0.01	0.00 – 0.01	**<0.001**	0.58	0.39 – 0.77	**<0.001**	-111.21	-124.41 – -98.01	**<0.001**	0.06	0.04 – 0.07	**<0.001**
Biome [5] × time [1722]	12.20	-32.81 – 57.21	0.595	66.52	34.25 – 98.78	**<0.001**	0.03	0.02 – 0.04	**<0.001**	0.03	0.02 – 0.03	**<0.001**	0.13	-0.23 – 0.49	0.480	-113.77	-138.71 – -88.84	**<0.001**	0.10	0.07 – 0.12	**<0.001**
Biome [6] × time [1722]	-3.24	-40.61 – 34.13	0.865	54.11	27.32 – 80.90	**<0.001**	0.04	0.03 – 0.05	**<0.001**	0.02	0.02 – 0.03	**<0.001**	-0.50	-0.80 – -0.20	**0.001**	-134.56	-155.26 – -113.86	**<0.001**	0.07	0.05 – 0.09	**<0.001**
Biome [14] × time [1722]	64.07	-75.42 – 203.57	0.368	115.00	15.00 – 215.00	**0.024**	0.01	-0.02 – 0.04	0.569	0.02	0.00 – 0.03	**0.012**	-0.31	-1.43 – 0.81	0.586	47.29	-29.98 – 124.57	0.230	-0.00	-0.08 – 0.08	0.964
(InOut × FOR10) × Biome [2]	-106.01	-283.63 – 71.60	0.242	-174.15	-301.48 – -46.82	**0.007**	-0.01	-0.05 – 0.03	0.617	-0.04	-0.06 – -0.02	**<0.001**	5.05	3.62 – 6.47	**<0.001**	-115.88	-214.27 – -17.49	**0.021**	0.17	0.07 – 0.27	**0.001**
(InOut × FOR10) × Biome [3]	483.35	409.46 – 557.25	**<0.001**	40.87	-12.11 – 93.85	0.131	-0.02	-0.04 – -0.00	**0.015**	-0.02	-0.03 – -0.01	**<0.001**	-1.98	-2.57 – -1.38	**<0.001**	-117.20	-158.13 – -76.26	**<0.001**	-0.09	-0.13 – -0.05	**<0.001**
(InOut × FOR10) × Biome [4]	453.68	424.96 – 482.39	**<0.001**	57.75	37.16 – 78.33	**<0.001**	0.02	0.01 – 0.03	**<0.001**	-0.00	-0.01 – -0.00	**0.004**	0.65	0.42 – 0.88	**<0.001**	53.85	37.94 – 69.76	**<0.001**	-0.02	-0.04 – -0.01	**0.005**
(InOut × FOR10) × Biome [5]	316.88	265.04 – 368.71	**<0.001**	-11.32	-48.48 – 25.84	0.550	-0.01	-0.02 – 0.00	0.107	-0.00	-0.01 – 0.00	0.089	0.52	0.11 – 0.94	**0.014**	123.52	94.81 – 152.23	**<0.001**	0.16	0.13 – 0.19	**<0.001**

	Gpp			Npp			EVI			NDVI			LST			ET			Conn		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
(InOut × FOR10) × Biome [6]	340.54	297.52 – 383.56	**<0.001**	17.69	-13.16 – 48.53	0.261	0.05	0.04 – 0.06	**<0.001**	-0.00	-0.01 – 0.00	0.848	-0.60	-0.94 – - 0.25	**0.001**	-27.78	-51.61 – -3.94	**0.022**	0.12	0.10 – 0.15	**<0.001**
(InOut × FOR10) × Biome [14]	518.35	374.24 – 662.47	**<0.001**	39.83	-63.48 – 143.15	0.450	0.15	0.11 – 0.18	**<0.001**	0.02	0.01 – 0.04	**0.006**	5.69	4.54 – 6.85	**<0.001**	-13.95	-93.79 – 65.88	0.732	0.01	-0.07 – 0.09	0.801
(InOut × FOR10) × time [1722]	36.32	9.73 – 62.91	**0.007**	6.43	-12.64 – 25.49	0.509	-0.00	-0.01 – 0.00	0.726	0.00	0.00 – 0.01	**0.003**	0.41	0.19 – 0.62	**<0.001**	11.12	-3.62 – 25.85	0.139	0.02	0.01 – 0.04	**0.008**
(InOut × Biome [2]) × time [1722]	-116.90	-319.55 – 85.75	0.258	3.45	-141.83 – 148.73	0.963	-0.03	-0.08 – 0.01	0.139	-0.01	-0.03 – 0.01	0.496	1.63	0.00 – 3.26	**0.050**	-29.72	-141.98 – 82.54	0.604	0.11	-0.00 – 0.23	0.058
(InOut × Biome [3]) × time [1722]	-8.60	-99.25 – 82.04	0.852	-5.28	-70.26 – 59.70	0.874	-0.02	-0.04 – 0.01	0.144	-0.01	-0.02 – 0.00	0.241	0.86	0.14 – 1.59	**0.020**	5.42	-44.79 – 55.63	0.832	0.01	-0.04 – 0.06	0.716
(InOut × Biome [4]) × time [1722]	57.61	20.67 – 94.54	**0.002**	23.39	-3.08 – 49.87	0.083	0.00	-0.01 – 0.01	0.496	0.00	-0.00 – 0.01	0.080	0.34	0.04 – 0.64	**0.025**	23.20	2.74 – 43.66	**0.026**	0.01	-0.01 – 0.03	0.199
(InOut × Biome [5]) × time [1722]	39.29	-25.63 – 104.21	0.236	13.93	-32.61 – 60.47	0.558	-0.01	-0.02 – 0.01	0.469	0.00	-0.00 – 0.01	0.435	0.77	0.25 – 1.29	**0.004**	5.12	-30.85 – 41.08	0.780	0.01	-0.03 – 0.05	0.655
(InOut × Biome [6]) × time [1722]	69.52	14.68 – 124.35	**0.013**	33.26	-6.05 – 72.57	0.097	0.00	-0.01 – 0.01	0.977	0.00	-0.00 – 0.01	0.686	-0.27	-0.71 – 0.17	0.237	20.92	-9.46 – 51.29	0.177	-0.02	-0.05 – 0.02	0.310
(InOut × Biome [14]) × time [1722]	1.15	-190.05 – 192.34	0.991	-63.96	-201.03 – 73.10	0.360	0.01	-0.04 – 0.05	0.708	-0.02	-0.04 – 0.00	0.131	1.01	-0.52 – 2.55	0.196	-94.19	-200.10 – 11.72	0.081	0.13	0.02 – 0.23	**0.023**
(FOR10 × Biome [2]) × time [1722]	74.81	-95.71 – 245.32	0.390	-33.31	-155.55 – 88.93	0.593	-0.01	-0.05 – 0.03	0.689	0.00	-0.01 – 0.02	0.730	-0.72	-2.09 – 0.65	0.305	121.25	26.80 – 215.71	**0.012**	-0.06	-0.16 – 0.04	0.211
(FOR10 × Biome [3]) × time [1722]	-113.19	-183.85 – -42.54	**0.002**	-196.02	-246.67 – - 145.37	**<0.001**	-0.01	-0.02 – 0.01	0.392	-0.01	-0.01 – 0.00	0.159	-0.12	-0.69 – 0.45	0.678	198.06	158.92 – 237.19	**<0.001**	-0.06	-0.10 – - 0.02	**0.005**
(FOR10 × Biome [4])× time [1722]	-30.12	-56.76 – -3.48	**0.027**	-72.15	-91.24 – -53.05	**<0.001**	-0.01	-0.01 – - 0.00	**0.014**	-0.01	-0.01 – - 0.00	**<0.001**	-0.59	-0.80 – - 0.38	**<0.001**	110.44	95.69 – 125.20	**<0.001**	-0.04	-0.06 – - 0.03	**<0.001**

APPENDIX

	Gpp			Npp			EVI			NDVI			LST			ET			Conn		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
(FOR10 × Biome [5])× time [1722]	-2.18	-52.80 – 48.44	0.933	-65.78	-102.07 – -29.49	**<0.001**	-0.02	-0.04 – -0.01	**<0.001**	-0.02	-0.03 – -0.02	**<0.001**	0.40	-0.01 – 0.80	0.056	132.10	104.06 – 160.14	**<0.001**	-0.09	-0.12 – -0.06	**<0.001**
(FOR10 × Biome [6])× time [1722]	-32.10	-73.60 – 9.40	0.129	-78.72	-108.46 – -48.97	**<0.001**	-0.03	-0.04 – -0.02	**<0.001**	-0.02	-0.02 – -0.02	**<0.001**	0.73	0.39 – 1.06	**<0.001**	135.87	112.88 – 158.85	**<0.001**	-0.06	-0.08 – -0.03	**<0.001**
(FOR10 × Biome [14]) × time [1722]	-68.00	-214.33 – 78.32	0.362	-133.75	-238.65 – -28.85	**0.012**	-0.01	-0.05 – 0.02	0.468	-0.02	-0.04 – -0.00	**0.011**	0.31	-0.87 – 1.48	0.611	-12.30	-93.36 – 68.75	0.766	0.02	-0.07 – 0.10	0.695
(InOut × FOR10 × Biome [2]) × time [1722]	112.40	-136.00 – 360.79	0.375	-15.13	-193.19 – 162.94	0.868	0.04	-0.01 – 0.10	0.134	0.01	-0.02 – 0.04	0.508	-2.00	-3.99 – -0.00	**0.050**	37.25	-100.35 – 174.85	0.596	-0.14	-0.28 – 0.00	0.050
(InOut × FOR10 × Biome [3]) × time [1722]	22.01	-82.22 – 126.24	0.679	12.23	-62.49 – 86.95	0.748	0.02	-0.00 – 0.05	0.069	0.01	-0.00 – 0.02	0.169	-1.02	-1.86 – -0.18	**0.017**	-5.02	-62.76 – 52.72	0.865	-0.02	-0.08 – 0.04	0.439
(InOut × FOR10 × Biome [4]) × time [1722]	-65.24	-105.62 – -24.86	**0.002**	-24.93	-53.88 – 4.02	0.091	-0.00	-0.01 – 0.00	0.304	-0.00	-0.01 – 0.00	0.254	-0.28	-0.60 – 0.05	0.094	-26.46	-48.82 – -4.09	**0.020**	-0.02	-0.04 – 0.00	0.123
(InOut × FOR10 × Biome [5]) × time [1722]	-50.03	-123.13 – 23.08	0.180	-16.43	-68.84 – 35.98	0.539	0.01	-0.01 – 0.03	0.294	-0.00	-0.01 – 0.01	0.591	-0.86	-1.45 – -0.27	**0.004**	-8.81	-49.31 – 31.68	0.670	-0.02	-0.07 – 0.02	0.263
(InOut × FOR10 × Biome [6]) × time [1722]	-79.10	-139.56 – -18.63	**0.010**	-33.23	-76.58 – 10.12	0.133	-0.00	-0.02 – 0.01	0.865	-0.00	-0.01 – 0.00	0.355	0.44	-0.04 – 0.93	0.074	-26.50	-59.99 – 7.00	0.121	-0.01	-0.04 – 0.03	0.672
(InOut × FOR10 × Biome [14]) × time [1722]	-1.86	-203.07 – 199.35	0.986	62.47	-81.77 – 206.71	0.396	-0.00	-0.05 – 0.04	0.931	0.02	-0.00 – 0.04	0.085	-0.99	-2.61 – 0.63	0.229	88.00	-23.46 – 199.46	0.122	-0.12	-0.24 – -0.01	**0.038**
Random Effects																					
σ^2	68329.38			35115.84			0.00			0.00			4.41			20967.30			0.02		
τ_{00}	64066.78 $_{BR}$			37638.78 $_{BR}$			0.00 $_{BR}$			0.00 $_{BR}$			3.82 $_{BR}$			32252.16 $_{BR}$			0.01 $_{BR}$		
ICC	0.48			0.52			0.49			0.43			0.46			0.61			0.37		
N	119 $_{BR}$			119 $_{BR}$			119 $_{BR}$			119 $_{BR}$			119 $_{BR}$			119 $_{BR}$			119 $_{BR}$		
Observations	992344			992344			992344			992344			992344			992344			992344		
Marginal R^2 / Conditional R^2	**0.821 / 0.908**			**0.578 / 0.796**			**0.383 / 0.684**			**0.405 / 0.658**			**0.339 / 0.646**			**0.436 / 0.778**			**0.213 / 0.504**		

Table S5. Modeled difference inside and outside for each biosphere reserve and proxy. Sorted by biome and effectiveness. Model: proxyDifference_Inside_Outside ~ Forest_Cover_2010_median*biome*time, random = ~ 1| Biosphere_Reserve
Gpp and Npp are Gross and Net primary productivity in kg*C/m^2, EVI and NDVI are Enhanced and Normalized Difference Vegetation Index, ET is Evapotranspiration in kg/km^2/year, LST is Hot day temperature in °C and Conn is forest patch connectivity. ET and LST are effective if negative with low emissions and low temperatures.

biome	**biosphere reserve**	**modeled difference Inside Outside 2017-2022**								
		Gpp	**Npp**	**EVI**	**NDVI**	**ET**	**LST**	**Conn**	**modeled**	**raw**
Tropical and subtropical moist broadleaf	SelvaElOcote	668.42	278.98	0.02	0.03	-5.96	-1.56	0.07	**Effective**	**Effective**
	Bia	539.26	667.73	0.02	0.05	-7.37	-1.83	0.01	**Effective**	**Ineffective**
	SierraDelRosario	452.53	276.71	0.01	0.01	-53.19	-0.47	0.19	**Effective**	**Effective**
	ElTriunfo	434.02	185.20	0.01	0.02	-72.87	-5.45	0.12	**Effective**	**Effective**
	ApanecaLlamatepec	424.51	163.93	0.02	0.01	-57.34	-3.19	0.16	**Effective**	**Effective**
	Kafa	297.50	162.81	0.02	0.00	-53.59	-1.47	0.04	**Effective**	Ineffective
	Baconao	154.51	180.53	0.03	0.01	-59.50	-1.52	0.21	**Effective**	**Effective**
	Beni	136.84	3.33	0.05	0.03	-76.98	-0.88	0.01	**Effective**	Ineffective
	Bosawas	136.66	105.06	0.04	0.02	-4.79	-0.55	0.07	**Effective**	Ineffective
	DeltaDelOrinoco	132.19	89.67	0.03	0.01	-51.44	-0.82	0.07	**Effective**	**Effective**
	RegionDeCalakmul	102.77	83.16	0.00	0.00	-16.62	-0.59	0.03	**Effective**	**Effective**
	NahaMetzabok	42.98	90.89	0.00	0.02	-98.56	-1.70	0.01	**Effective**	**Effective**
	Yaboti	15.14	38.24	0.04	0.05	-211.70	-2.07	0.09	**Effective**	**Effective**
	BasseLobaye	01.02	4.92	0.00	0.00	-7.14	-0.13	0.00	**Effective**	Ineffective
	LasYungas	501.58	394.24	0.02	0.01	27.74	-2.56	-0.03	Ineffective	Ineffective
	RioSanJuan	475.42	236.04	0.00	0.00	-81.15	0.03	0.13	Ineffective	Ineffective
	BosqueMbaracayu	316.08	59.18	0.02	0.02	5.25	-1.18	0.16	Ineffective	Ineffective
	CuchillasDelToa	306.47	239.93	-0.03	0.02	-274.72	-2.26	0.03	Ineffective	Ineffective
	AguaYPaz	292.96	240.74	-0.02	0.00	-209.15	-1.78	0.15	Ineffective	Ineffective
	VolcanTacana	143.38	125.78	-0.02	-0.01	-234.93	-3.22	0.01	Ineffective	Ineffective
	SierraDeLasMinas	108.24	101.37	-0.02	0.03	-163.75	-3.67	0.03	Ineffective	Ineffective
	CordilleraVolcanicaCentral	54.09	275.17	-0.05	0.00	-166.99	-4.27	0.07	Ineffective	Ineffective
	Odzala	4.81	2.88	0.00	0.00	39.58	0.11	0.00	Ineffective	Ineffective
	RioPlatano	-1.19	-2.11	0.01	0.02	-49.40	-0.78	0.02	Ineffective	Ineffective
	MontesAzules	-4.65	-8.88	0.02	0.02	-36.77	-0.22	0.03	Ineffective	Ineffective
	AltoOrinocoCasiquiare	-4.65	34.03	0.00	0.00	17.90	-0.47	0.00	Ineffective	Ineffective
	PilonLajas	-9.12	-10.55	0.00	0.00	-44.60	-0.49	0.04	Ineffective	Ineffective
	Darien	-25.42	39.98	-0.02	0.00	-91.12	-1.94	-0.16	Ineffective	Ineffective

APPENDIX

biome		biosphere reserve	modeled difference Inside Outside 2017-2022								
Tropical and subtropical	moist broadleaf	LagunasDeMontebello	-36.91	27.14	-0.04	0.00	17.82	-2.29	0.01	Ineffective	Ineffective
		Maya	-48.27	-40.27	0.01	0.02	6.16	-0.42	0.09	Ineffective	Ineffective
		Omo	-76.93	-37.71	-0.01	-0.01	-50.74	0.10	0.13	Ineffective	Ineffective
		LaAmistadCRI	-86.35	229.11	-0.07	0.01	-158.58	-5.59	0.10	Ineffective	Ineffective
		Dimonika	-94.51	-20.29	-0.02	0.00	03.09	-0.52	0.02	Ineffective	Ineffective
		WesternNgheAn	-118.63	-161.22	0.01	0.00	-93.76	1.41	0.02	Ineffective	Ineffective
		LaAmistadPAN	-187.98	9.71	-0.07	-0.01	-97.89	-5.02	0.07	Ineffective	Ineffective
		Wuyishan	-195.13	49.82	-0.04	0.02	-255.56	-3.74	0.11	Ineffective	Ineffective
		GunungLeuser	-196.91	3.23	-0.06	0.01	-100.37	-4.02	0.09	Ineffective	Ineffective
		Volcans	-221.30	-34.16	0.04	0.04	-305.57	-9.29	0.21	Ineffective	Ineffective
	dry broadleaf	ChamelaCuixmala	76.27	-2.96	0.00	0.01	17.34	-1.42	-0.10	Ineffective	Ineffective
		XiriualtiqueJiquitizco	54.09	1.69	0.01	0.00	30.18	0.78	-0.26	Ineffective	Ineffective
		Hurulu	-63.24	-71.00	0.02	0.01	23.85	-0.17	0.00	Ineffective	Ineffective
	conifer	ElCielo	327.36	82.84	-0.03	0.01	-39.11	-2.58	0.09	Ineffective	Ineffective
		LaSepultura	145.27	68.64	0.00	0.01	58.14	-1.70	0.01	Ineffective	Ineffective
		CumbresDeMonterrey	102.97	-63.83	0.08	0.07	194.78	-1.62	0.12	Ineffective	Ineffective
		MariposaMonarca	86.97	-31.52	-0.02	0.03	15.88	-6.96	0.08	Ineffective	Ineffective
		SierraDeManatlan	36.06	5.85	-0.06	-0.01	103.42	-4.13	0.06	Ineffective	Ineffective
		SierraGorda	1.29	14.04	-0.01	0.00	-21.50	-0.46	0.04	Ineffective	Ineffective
temperate	broadleaf and mixed	MammothCaveArea	87.84	146.66	0.01	0.03	-145.31	-1.39	0.11	**Effective**	**Effective**
		Desnianskyi	9.95	3.19	0.03	0.02	-15.23	-0.89	0.33	**Effective**	**Effective**
		ChamplainAdirondack	9.36	57.23	0.02	0.02	-58.16	-1.51	0.02	**Effective**	**Effective**
		Pilis	1.28	3.39	0.01	0.02	-13.72	-1.29	0.33	**Effective**	**Effective**
		AndinoNorpatagonica	206.68	114.98	0.03	0.03	57.35	-1.38	0.13	Ineffective	Ineffective
		Urdaibai	160.93	79.72	0.01	0.01	-29.79	-1.02	-0.08	Ineffective	Ineffective
		OsAncaresLucenses	99.82	35.37	0.01	0.00	29.82	-0.23	0.05	Ineffective	Ineffective
		CollemeluccioMontedimezzo	68.26	19.79	0.02	0.01	12.74	-1.08	0.16	Ineffective	Ineffective
		NorthVidzeme	52.60	26.69	0.04	0.01	31.82	-0.16	0.01	Ineffective	Ineffective
		GeorgianBay	50.57	53.96	-0.06	-0.02	12.57	0.33	-0.31	Ineffective	Ineffective
		Aggtelek	47.72	39.34	0.00	0.00	7.35	0.39	0.13	Ineffective	Ineffective
		Entlebuch	47.33	10.34	0.00	-0.01	-10.54	0.15	0.08	Ineffective	Ineffective
		Spreewald	32.99	4.95	0.10	0.03	77.17	-0.48	-0.01	Ineffective	Ineffective
		LowerMorava	32.26	-25.84	0.08	0.03	76.97	-0.23	-0.07	Ineffective	Ineffective

biome		biosphere reserve	modeled difference Inside Outside 2017-2022								
temperate	broadleaf and mixed	Baishuijiang	26.86	12.17	0.00	0.00	26.17	1.23	-0.10	Ineffective	Ineffective
		Rhoen	19.46	6.30	0.03	0.00	13.03	-0.23	-0.05	Ineffective	Ineffective
		TrebonBasin	14.58	-9.27	-0.03	0.00	1.60	-0.27	0.16	Ineffective	Ineffective
		KristianstadVattenrike	14.22	5.13	0.01	0.01	1.88	-0.03	0.00	Ineffective	Ineffective
		AreaDeAllariz	12.96	-12.12	0.04	0.02	45.51	-0.70	-0.04	Ineffective	Ineffective
		Fontainebleau	10.20	19.34	-0.05	-0.01	-19.91	-0.01	0.21	Ineffective	Ineffective
		Cevennes	8.42	-19.29	0.02	0.02	31.23	-1.10	0.14	Ineffective	Ineffective
		SchwaebischeAlb	5.83	-8.47	0.03	0.00	5.33	-0.01	0.00	Ineffective	Ineffective
		NerussoDesnianskoePolesie	3.96	4.89	-0.03	0.00	-10.12	0.04	0.23	Ineffective	Ineffective
		RiEoOscosYTerrasDeBurron	3.54	-14.01	0.02	0.01	38.60	0.38	0.06	Ineffective	Ineffective
		SchorfheideChorin	03.02	-7.19	0.03	0.01	23.88	-0.55	0.07	Ineffective	Ineffective
		OberlausitzerHeideUndTeichl and	2.54	10.04	0.00	0.00	1.94	-0.57	0.05	Ineffective	Ineffective
		PriokskoTerrasnyi	1.12	10.18	-0.03	0.01	-22.57	-0.58	0.22	Ineffective	Ineffective
		BileKarpaty	0.90	-14.17	-0.01	0.00	4.44	-0.19	0.01	Ineffective	Ineffective
		SmolenskLakeland	-1.03	1.19	-0.01	0.00	-1.13	-0.12	0.09	Ineffective	Ineffective
		Krivoklatsko	-1.84	-11.20	0.06	0.03	51.93	-0.48	0.15	Ineffective	Ineffective
		Tianmushan	-2.81	-11.04	0.04	0.04	-46.97	-2.98	0.04	Ineffective	Ineffective
		Bliesgau	-5.60	-0.47	-0.01	0.01	18.10	-0.80	0.06	Ineffective	Ineffective
		Ugra	-6.05	8.84	-0.05	-0.01	-18.11	0.05	0.05	Ineffective	Ineffective
		GwangneungForest	-6.31	2.84	-0.02	-0.02	-9.66	0.49	-0.02	Ineffective	Ineffective
		Voronezhskiy	-6.75	-13.81	-0.01	0.02	-5.36	-0.87	-0.11	Ineffective	Ineffective
		Valdaiskiy	-11.48	0.93	-0.04	-0.01	-10.24	-0.34	0.04	Ineffective	Ineffective
		Wienerwald	-12.55	-28.60	0.00	0.01	-8.14	0.20	0.02	Ineffective	Ineffective
		Okskiy	-14.07	-3.50	0.00	0.00	-10.59	-0.15	0.05	Ineffective	Ineffective
		VosgesDuNordPfaelzerwald	-18.39	-13.66	-0.04	0.01	-15.40	-1.58	0.28	Ineffective	Ineffective
		Shennongjia	-19.81	-2.65	-0.01	0.00	-15.09	-1.97	0.02	Ineffective	Ineffective
		TucholaForest	-20.13	-13.47	-0.04	-0.02	-21.49	0.36	0.12	Ineffective	Ineffective
		KrkokonoseKarkonosze	-30.68	-11.46	-0.06	0.01	-63.32	-3.12	0.10	Ineffective	Ineffective
		Sumava	-41.01	-22.10	-0.05	0.01	-60.42	-2.38	0.14	Ineffective	Ineffective
		JulianAlps	-62.53	-11.31	0.01	0.01	-56.03	-1.60	0.03	Ineffective	Ineffective
		Changbaishan	-67.20	4.25	-0.05	0.00	-27.32	-2.04	0.03	Ineffective	Ineffective
		Camili	-71.63	-51.67	0.01	0.00	-13.21	-1.48	0.13	Ineffective	Ineffective
		GolijaStudenica	-77.07	-41.69	-0.02	0.00	-55.35	-2.34	0.01	Ineffective	Ineffective

biome		biosphere reserve	modeled difference Inside Outside 2017-2022								
temperate	broadleaf and mixed	VelebitMountain	-79.05	-71.22	-0.04	-0.01	-10.35	-2.29	-0.19	Ineffective	Ineffective
		Redes	-137.00	-62.65	-0.01	0.00	-25.95	-1.11	0.05	Ineffective	Ineffective
		PicosDeEuropa	-141.39	-97.55	0.01	0.02	-34.55	-1.37	-0.12	Ineffective	Ineffective
		Araucarias	-159.97	-83.49	-0.01	0.01	-68.52	-2.12	0.11	Ineffective	Ineffective
		Kavkazskiy	-348.50	-174.92	-0.05	0.00	-54.40	-2.52	0.03	Ineffective	Ineffective
		BosquesTempladosAndes Australes	-427.45	-217.73	-0.03	0.01	-97.05	-2.31	0.10	Ineffective	Ineffective
	conifer	Taza	101.21	-2.04	0.02	0.03	80.49	-1.46	0.00	Ineffective	Ineffective
		Baikalskyi	46.02	19.29	-0.01	0.01	14.82	-1.13	-0.19	Ineffective	Ineffective
		GrossesWalsertal	-2.92	2.19	0.03	0.01	3.59	0.38	0.02	Ineffective	Ineffective
		BerchtesgadenerLand	-4.09	-0.74	-0.02	0.00	-25.77	-0.60	0.12	Ineffective	Ineffective
		YellowstoneGrandTeton	-7.63	4.65	-0.02	-0.03	-24.66	-0.33	-0.03	Ineffective	Ineffective
		SayanoShushenskiy	-30.28	0.01	0.00	0.00	0.06	-0.29	-0.06	Ineffective	Ineffective
		CrownOfTheContinent	-35.11	-3.29	-0.01	0.00	-7.18	-0.01	-0.18	Ineffective	Ineffective
		Wolong	-118.11	-61.99	-0.03	0.00	25.78	-2.15	-0.06	Ineffective	Ineffective
		Olympic	-158.24	-84.60	-0.10	-0.02	40.12	-0.97	0.10	Ineffective	Ineffective
boreal	boreal	Barguzinskyi	97.72	55.85	-0.03	0.01	-13.58	-1.23	-0.06	Ineffective	Ineffective
		Kenozersky	9.82	-1.73	-0.03	0.00	6.34	-0.82	0.01	Ineffective	Ineffective
		NorthKarelian	09.05	-5.88	0.00	0.00	-5.90	0.25	0.01	Ineffective	Ineffective
		Pechorollychskiy	-22.87	-7.89	-0.02	0.00	-14.69	-0.74	0.04	Ineffective	Ineffective
		Laplandskiy	-29.82	-22.35	0.00	0.00	-9.20	0.14	-0.02	Ineffective	Ineffective
		Kronotskiy	-65.02	-68.90	0.04	0.00	-54.13	-0.39	-0.06	Ineffective	Ineffective
		Denali	-66.40	-42.68	-0.09	-0.07	-27.31	1.59	0.00	Ineffective	Ineffective
mangroves	mangroves	SianKaan	265.37	217.11	-0.05	-0.02	-27.98	0.16	-0.47	Ineffective	Ineffective
		LaEncrucijada	-20.15	-28.68	-0.02	0.01	-121.63	-3.61	-0.03	Ineffective	Ineffective
		RiaLagartos	-155.41	-44.95	-0.06	-0.02	25.87	0.31	0.00	Ineffective	Ineffective

Table S6. Conditional means and conditional standard deviation of the random effects as each biosphere reserve for all forest ecosystem function proxies.

Biosphere reserve	Gpp	Gpp SD	Npp	Npp SD	EVI	EVI SD	NDVI	NDVI SD	LST	LST SD	ET	ET SD	Conn	Conn SD
Aggtelek	62.73	7.94	24.57	5.69	0.08	0	0.03	0	0.58	0.06	52.28	4.4	-0.05	0
AguaYPaz	16.86	2.12	-149.05	1.52	0.03	0	0.01	0	0.28	0.02	-187.91	1.17	-0.06	0
AltoOrinoco Casiquiare	-70.64	0.66	-105.17	0.47	0	0	0.02	0	0.92	0.01	-83	0.36	0.13	0
Andino Norpatagonica	65.26	1.64	142.26	1.18	-0.06	0	-0.02	0	-4.46	0.01	7.91	0.91	0.01	0
Apaneca Llamatepec	41.4	6.82	31.13	4.89	0.02	0	0	0	1	0.05	304.69	3.78	-0.08	0
Araucarias	430.68	2.15	375.42	1.54	-0.05	0	-0.02	0	-2.4	0.02	177.55	1.19	0.03	0
AreaDeAllariz	211.42	9.59	146.22	6.88	-0.01	0	-0.03	0	2.6	0.08	143.26	5.31	-0.09	0.01
Baconao	-129.53	8.2	-170.25	5.88	0.05	0	0.02	0	1	0.07	343.16	4.54	-0.24	0
Baikalskyi	-235.98	4.59	-108.74	3.29	-0.02	0	0.01	0	0.2	0.04	-71.12	2.55	-0.04	0
Baishuijiang	13.51	3.76	84.12	2.69	0.05	0	0.04	0	-4.19	0.03	-21.27	2.08	0.14	0
Barguzinskyi	30.31	4.2	31.28	3.01	-0.05	0	-0.02	0	-3.11	0.03	38.44	2.33	-0.24	0
BasseLobaye	-266.99	7.61	-335.96	5.46	-0.01	0	0.01	0	2.1	0.06	90.2	4.22	0.13	0
Beni	-111.92	4.32	-197.7	3.1	0.05	0	0	0	2.47	0.03	132.99	2.39	-0.05	0
Berchtesgadener Land	93.49	5.68	47.08	4.07	0.06	0	0.03	0	1.18	0.05	52.68	3.15	0.03	0
Bia	-181.7	6.17	-459.12	4.42	0.01	0	-0.01	0	1.75	0.05	-390.71	3.42	0.15	0
BileKarpaty	-72.84	5.63	-85.97	4.04	0.06	0	0.02	0	-0.65	0.05	24.34	3.12	-0.09	0
Bliesgau	316.62	9.65	-13	6.92	0.03	0	0.01	0	1.48	0.08	-32.84	5.35	-0.13	0.01
Bosawas	303.78	1.35	109.71	0.97	0.05	0	0.01	0	0.06	0.01	-59.15	0.75	0.01	0
Bosque Mbaracayu	-181.04	4.5	-118.24	3.22	0.02	0	0	0	2.81	0.04	314.69	2.49	-0.28	0
BosquesTempladosAndesAustrales	330.18	1.36	300.91	0.97	-0.02	0	0	0	-3.32	0.01	35.46	0.75	0.01	0
Camili	157.12	8.06	114.22	5.78	0.08	0	0.03	0	-1.56	0.06	104.48	4.47	0.16	0
Cevennes	112.65	3.9	111.5	2.8	-0.07	0	-0.04	0	1.75	0.03	67.7	2.16	0.02	0
ChamelaCuixmala	-171.83	8.25	17.08	5.91	0.05	0	0.02	0	0.79	0.07	-213.41	4.57	0.04	0
Champlain Adirondack	104.54	1.12	95.15	0.8	0.09	0	0.03	0	-1.36	0.01	14.05	0.62	0.22	0
Changbaishan	-285.43	3.04	-110.94	2.18	0.03	0	0.02	0	-2.15	0.02	-103.29	1.69	0.21	0
Collemeluccio Montedimezzo	162.79	9.03	9.8	6.48	0.09	0	0.01	0	1.51	0.07	224.14	5.01	-0.02	0.01
Cordillera VolcanicaCentral	74.09	2.18	30.96	1.56	-0.01	0	0	0	-0.87	0.02	-121.28	1.21	0.02	0
CrownOfThe Continent	-195.58	3.34	-102.72	2.4	-0.05	0	-0.02	0	-0.3	0.03	-79	1.85	-0.02	0
CuchillasDelToa	229.05	4.34	131.68	3.11	-0.04	0	0	0	-0.69	0.03	234.11	2.41	-0.09	0
CumbresDe Monterrey	-512.19	4.62	-285	3.31	-0.11	0	-0.06	0	-1.22	0.04	-182.99	2.56	-0.09	0
Darien	-420.48	1.89	-456.36	1.36	0.02	0	0.02	0	0.25	0.02	-475.54	1.05	-0.08	0
DeltaDelOrinoco	222.43	2.21	-178.77	1.59	-0.03	0	-0.01	0	0.79	0.02	153.78	1.23	-0.11	0

Biosphere reserve	Gpp	Gpp SD	Npp	Npp SD	EVI	EVI SD	NDVI	NDVI SD	LST	LST SD	ET	ET SD	Conn	Conn SD
Denali	-144.04	1.67	-67.39	1.2	-0.01	0	-0.03	0	-1.08	0.01	-4.23	0.92	-0.03	0
Desnianskyi	-230.93	6.3	-141.92	4.52	-0.03	0	0	0	1.17	0.05	-26.52	3.49	-0.08	0
Dimonika	-441.4	4.06	-179.96	2.91	-0.1	0	-0.02	0	1.4	0.03	-607.38	2.25	0.08	0
ElCielo	4.12	4.77	-29.35	3.42	0.01	0	0	0	0.99	0.04	72.21	2.65	0.14	0
ElTriunfo	8.12	3.87	90.64	2.77	-0.01	0	0.01	0	-1.84	0.03	183.89	2.14	0.01	0
Entlebuch	-14.65	7.68	50.21	5.51	-0.01	0	0	0	-1.76	0.06	14.38	4.26	0.03	0
Fontainebleau	70.66	7.99	52.45	5.73	0.05	0	0	0	1.42	0.06	10.77	4.43	-0.22	0
GeorgianBay	-58.22	3.19	17.95	2.29	-0.02	0	-0.01	0	0.61	0.03	22.74	1.77	0.09	0
GolijaStudenica	49.49	6.6	57.42	4.73	0.05	0	0.02	0	-0.28	0.05	73.5	3.66	0.14	0
GrossesWalsertal	-6.23	9.96	2.93	7.14	0.06	0	0.03	0	0.59	0.08	37.19	5.52	-0.02	0.01
GunungLeuser	-293.98	1.83	-135.98	1.31	-0.07	0	0	0	-2.06	0.01	-235.92	1.01	0.01	0
Gwangneung Forest	12.1	8.84	-63.03	6.34	0.05	0	0.02	0	0.68	0.07	-10.47	4.9	0.03	0.01
Hurulu	172.33	8.2	66.45	5.88	-0.07	0	-0.02	0	-1.07	0.07	10.03	4.54	0.1	0
JulianAlps	-38.94	3.74	12.92	2.68	0.01	0	0.01	0	-2.39	0.03	13	2.07	0.14	0
Kafa	189.84	1.86	481.02	1.33	0.01	0	-0.02	0	-1.59	0.01	65.85	1.03	0.08	0
Kavkazskiy	33.68	3.06	61.34	2.19	0.07	0	0.03	0	-1.88	0.02	26.68	1.7	0.1	0
Kenozersky	162.77	3.87	75.04	2.77	0.01	0	0.02	0	3.05	0.03	23.48	2.14	0.1	0
Kristianstad Vattenrike	-27.7	6.62	44.86	4.74	-0.01	0	0	0	-1.19	0.05	-80.07	3.67	-0.06	0
Krivoklatsko	-163.69	7.91	-103.13	5.67	-0.04	0	-0.01	0	0.97	0.06	-82.89	4.38	-0.13	0
Krkokonose Karkonosze	-202.66	6.21	-100.21	4.46	-0.07	0	0	0	-1.36	0.05	-111.03	3.44	0	0
Kronotskiy	-63.05	2.52	-71.67	1.81	0.17	0	0.07	0	-2.27	0.02	-16.56	1.4	-0.03	0
LaAmistadCRI	-25.33	1.89	17.93	1.35	-0.02	0	0	0	-2.18	0.02	-160.98	1.04	-0.01	0
LaAmistadPAN	-88.52	3.08	17.85	2.21	-0.03	0	0	0	-3.11	0.02	-208.61	1.71	0.09	0
LaEncrucijada	-59.66	4.49	126.94	3.22	-0.01	0	0	0	-2.9	0.04	-29.2	2.49	0.11	0
LagunasDe Montebello	488.9	11.56	592.6	8.29	-0.06	0	-0.02	0	-2.04	0.09	140.92	6.41	-0.01	0.01
Laplandskiy	-150.75	3.41	-60.56	2.45	-0.05	0	-0.03	0	1.16	0.03	-66.01	1.89	-0.04	0
LaSepultura	387.45	4.79	152.5	3.43	0.1	0	0.04	0	0.99	0.04	188.29	2.65	0	0
LasYungas	-43.01	1.89	362.46	1.36	0	0	0.01	0	-1.08	0.02	-74.76	1.05	-0.03	0
LowerMorava	-55	11.04	-152.85	7.91	0	0	-0.01	0	1.95	0.09	33.51	6.12	-0.33	0.01
MammothCave Area	250	9.73	-20.61	6.98	0.11	0	0.04	0	2.69	0.08	233.98	5.39	0.07	0.01
MariposaMonarca	84.74	7.19	154.21	5.16	-0.06	0	-0.01	0	-4.07	0.06	-111.17	3.98	-0.18	0
Maya	131.72	1.55	-90.07	1.11	0.03	0	0.01	0	2.2	0.01	309.53	0.86	-0.02	0
MontesAzules	465.01	2.54	296.48	1.82	0.04	0	0.01	0	1.07	0.02	297.61	1.41	0.04	0
NahaMetzabok	556.78	7.77	400.36	5.57	0.01	0	-0.01	0	0.61	0.06	485	4.31	0.09	0
NerussoDesnians koePolesie	-237.56	4.65	-147.98	3.34	-0.02	0	0	0	0.52	0.04	-33.11	2.58	0.01	0
NorthKarelian	182.33	1.92	119.99	1.37	-0.02	0	-0.01	0	1.81	0.02	45.67	1.06	0.07	0

APPENDIX

Biosphere reserve	Gpp	Gpp SD	Npp	Npp SD	EVI	EVI SD	NDVI	NDVI SD	LST	LST SD	ET	ET SD	Conn	Conn SD
NorthVidzeme	-199.44	3.12	-71.98	2.24	-0.03	0	-0.01	0	-0.07	0.03	-96.52	1.73	-0.1	0
OberlausitzerHeideUndTeichland	-157.7	9.72	-120.53	6.97	-0.13	0	-0.05	0	1.1	0.08	-146.98	5.39	-0.17	0.01
Odzala	-95.58	1.59	-63.88	1.14	-0.03	0	0	0	1.29	0.01	-184.92	0.88	0.13	0
Okskiy	-292.67	5.46	-181.27	3.92	-0.05	0	-0.01	0	1.16	0.04	-77.6	3.03	0.01	0
Olympic	225.2	2.72	121.88	1.95	0.03	0	0.03	0	-0.19	0.02	-68.82	1.51	0.08	0
Omo	-251.59	4.55	-401.75	3.26	-0.01	0	-0.03	0	2.09	0.04	-571.55	2.52	0.03	0
OsAncares Lucenses	152.04	5.72	132	4.1	-0.05	0	-0.02	0	1.63	0.05	61.75	3.17	0.08	0
Pechorollychskiy	-17.57	1.57	-26.68	1.13	-0.05	0	0	0	0.44	0.01	-20.8	0.87	0.18	0
PicosDeEuropa	175.85	6.38	224.9	4.57	0.02	0	-0.01	0	0.08	0.05	69.72	3.53	0	0
Pilis	10.78	10.23	-93.27	7.33	0.08	0	0.02	0	0.61	0.08	49.99	5.67	-0.09	0.01
PilonLajas	101.73	2.44	75.61	1.75	0.07	0	0.01	0	1.29	0.02	3.51	1.35	0.08	0
PriokskoTerrasnyi	-306.85	8.18	-157.32	5.86	-0.03	0	0	0	0.42	0.07	-77.98	4.53	-0.04	0
Redes	187.51	6.92	225.6	4.96	0.03	0	-0.01	0	0.61	0.06	76.71	3.84	0.13	0
RegionDe Calakmul	-30.08	1.83	-248.5	1.31	0.04	0	0.01	0	2.85	0.01	335.08	1.01	0.09	0
Rhoen	-100.1	4.38	-65.27	3.14	0.02	0	0.01	0	-0.51	0.04	-75.57	2.43	-0.1	0
RiaLagartos	-33.77	9	-79.03	6.45	0	0	0	0	1.45	0.07	3.2	4.99	-0.12	0.01
RiEoOscosY TerrasDeBurron	304.46	4.6	228.05	3.3	-0.05	0	-0.02	0	1.3	0.04	81.45	2.55	0.02	0
RioPlatano	319.2	2.23	148.61	1.6	0.02	0	0.01	0	0.28	0.02	50.37	1.23	-0.05	0
RioSanJuan	-16.71	1.92	-283.75	1.38	0.05	0	0.02	0	0.01	0.02	-239.02	1.07	-0.15	0
Sayano Shushenskiy	-293.54	2.37	-136.23	1.7	-0.06	0	0	0	-3.57	0.02	-88.77	1.31	0.08	0
Schorfheide Chorin	-118.73	5.96	-91.73	4.27	-0.07	0	-0.02	0	0.18	0.05	-106.11	3.3	-0.09	0
SchwaebischeAlb	103.98	6.24	31.7	4.48	0.05	0	0.02	0	-0.23	0.05	11.37	3.46	-0.16	0
SelvaElOcote	226.44	4.49	233.31	3.22	0.01	0	0	0	1.36	0.04	177.51	2.49	0.03	0
Shennongjia	-27.77	4.52	-15.75	3.24	0.09	0	0.03	0	-1.35	0.04	4.84	2.5	0.27	0
SianKaan	93.42	3.37	-47.9	2.41	0.01	0	0	0	1.45	0.03	26	1.86	0.01	0
SierraDeLas Minas	255.48	3.38	340.46	2.42	-0.03	0	-0.02	0	-1.25	0.03	91.32	1.87	0.01	0
SierraDelRosario	159.06	10.11	-136.07	7.25	0.07	0	0.03	0	-0.27	0.08	303.66	5.6	-0.15	0.01
SierraDeManatlan	-112.19	4.48	-127.69	3.21	0.06	0	0.02	0	2.2	0.04	32.37	2.48	0.06	0
SierraGorda	148.07	3.11	135.34	2.23	0	0	0	0	1.1	0.02	1.28	1.72	0.08	0
Smolensk Lakeland	-290.24	3.95	-157.5	2.84	0.03	0	0.02	0	0.69	0.03	-101.27	2.19	0.15	0
Spreewald	-191.06	7.65	-150.91	5.49	-0.11	0	-0.05	0	1.28	0.06	-121.32	4.24	-0.05	0
Sumava	-120.39	3.51	-42.48	2.52	-0.07	0	-0.01	0	-1.46	0.03	-87.81	1.95	0	0
Taza	830.12	15.22	309.53	10.91	0.07	0	-0.01	0	6.75	0.12	405.06	8.44	-0.16	0.01
Tianmushan	191.01	7.87	-69.64	5.64	0.05	0	-0.01	0	3.09	0.06	64.87	4.36	0.28	0
TrebonBasin	-115.99	7.28	-42.29	5.22	-0.11	0	-0.03	0	0.93	0.06	-87.75	4.03	-0.1	0

Biosphere reserve	Gpp	Gpp SD	Npp	Npp SD	EVI	EVI SD	NDVI	NDVI SD	LST	LST SD	ET	ET SD	Conn	Conn SD
TucholaForest	-207.8	4.04	-103.28	2.9	-0.13	0	-0.05	0	0.14	0.03	-138.36	2.24	-0.11	0
Ugra	-274.64	3.29	-149.37	2.36	0	0	0.02	0	-0.85	0.03	-63.13	1.82	-0.02	0
Urdaibai	590.75	10.28	409.02	7.37	-0.04	0	0	0	1.1	0.08	84.31	5.69	-0.11	0.01
Valdaiskiy	-362.15	3.01	-163.9	2.16	-0.05	0	0	0	0.31	0.02	-166.18	1.67	0.17	0
VelebitMountain	150.26	3.68	69.6	2.64	0.08	0	0.02	0	-1.23	0.03	102.9	2.04	-0.04	0
Volcans	-174.65	6.94	286.11	4.98	-0.03	0	-0.03	0	-4.9	0.06	-229.98	3.85	-0.05	0
VolcanTacana	-37.61	10.61	212.3	7.61	-0.07	0	-0.02	0	-4.53	0.09	113.02	5.88	0.1	0.01
Voronezhskiy	-335.85	9.93	-237.9	7.12	-0.05	0	-0.03	0	1.69	0.08	-50.3	5.5	-0.13	0.01
VosgesDuNord Pfaelzerwald	47.74	3.74	-52.87	2.68	0.01	0	0.01	0	-0.05	0.03	-31.13	2.07	-0.05	0
WesternNgheAn	-68.15	1.6	72.68	1.15	0.04	0	0.01	0	-0.45	0.01	-216.97	0.89	0.02	0
Wienerwald	191.21	12.36	-115.31	8.86	0.07	0	0.02	0	0.43	0.1	41.84	6.85	0.02	0.01
Wolong	-76.17	3.14	26.85	2.25	0.06	0	0.04	0	-4.63	0.03	-61.56	1.74	0.07	0
Wuyishan	-1064.01	4.13	-500.71	2.96	-0.08	0	-0.01	0	-2.05	0.03	-337.54	2.29	0.06	0
Xiriualtique Jiquitizco	-0.5	8.03	-83.52	5.76	0.02	0	0	0	0.29	0.06	203.38	4.45	-0.15	0
Yaboti	203.03	3.35	279.38	2.4	0.01	0	0.01	0	1.03	0.03	254.33	1.86	0.08	0
Yellowstone GrandTeton	-341.32	3.31	-160.57	2.37	-0.14	0	-0.1	0	-0.04	0.03	-125.66	1.83	-0.01	0

A 4 Manuscript of Dabard et al. 2024

Biosphere Reserves as model regions for transdisciplinarity? A literature review

Caroline Hélène Dabard [1,2†*], Charlotte Gohr [1,3,4†], Fabio Weiss [1,3,5], Henrik von Wehrden [6], Frederike Neumann [1], Solomiia Hordasevych [1], Bruno Arieta [1], Jenny Hammerich [1,3], Caroline Meier [1,8], Janine Jargow [1,7], Vera Luthardt [1], Pierre L. Ibisch [1,4], Ana Filipa Ferreira [1]

1) Biosphere Reserves Institute, Eberswalde University for Sustainable Development, Schicklerstraße 5, 16225 Eberswalde, Germany

2) Social-ecological Systems Institute, Leuphana University Lüneburg, Universitätsallee 1, 21335 Lüneburg, Germany

3) Leuphana University Lüneburg, Universitätsallee 1, 21335 Lüneburg, Germany

4) Centre for Econics and Ecosystem Management, Eberswalde University for Sustainable Development, Schicklerstraße 5, 16225 Eberswalde, Germany

5) Cawthron Institute, 98 Halifax Street East Nelson 7010, New Zealand

6) Center of Methods & Faculty of Sustainability, Leuphana University, Universitätsallee 1, 21335 Lüneburg, Germany

7) Technische Universität Dresden, Fakultät Mathematik und Naturwissenschaften, Professur für Allgemeine Psychologie, 01062 Dresden, Germany

8) Faculty of Philology and History, Augsburg University, Universitätsstraße 10, 86159 Augsburg, Germany

† These authors have contributed equally to this work and share first authorship

*corresponding author: caroline.dabard@hnee.de

This article was submitted to *Sustainability Science*, is currently under peer review, and follows the journal's corresponding referencing and formatting style.

Abstract. The UNESCO Man and the Biosphere program advocates and designates Biosphere Reserves as learning sites for sustainable development. Yet the extent to which research aligns with their core objectives - biodiversity conservation, economic development and capacity building - remains uncertain. In response, transdisciplinary research in conservation and development aims at implementing more diverse, participatory methods to improve effective management as well as governance. This study provides a systematic screening of scientific research in and on Biosphere Reserves published since 1975. Research fields in Biosphere Reserves are diverse and range from social to political to ecological investigations. We identified an emerging field of transdisciplinary science in research related to or conducted in UNESCO Biosphere Reserves, highlighting progress in author gender equality as compared to studies that did not build on a transdisciplinary mode. Most transdisciplinary studies were conducted in Mexican and Indian Biosphere Reserves. Transdisciplinary research in Biosphere Reserves calls for high-impact knowledge, addressing deep leverage points and the inclusive participation of underrepresented and discriminated groups. Thereby, Biosphere Reserves as specialized areas for sustainable development could play a vital role.

Keywords. Systematic literature review, leverage points, knowledge types, transformative research, participation

Highlights

- Biosphere Reserves related research is mostly conducted in North America, Asia and Europe and the annual number of publications increased by a mean of 10.5% per annum over the last 20 years.
- Mexico and India are champions in transdisciplinary research in Biosphere Reserves.
- Research topics in Biosphere Reserves refer to a diverse range of research fields, including aspects of participation or ethnobotany.
- Gender disparities and restricted participation are identified in research in Biosphere Reserves. Whilst more female authors in transdisciplinary research are identified.
- Transdisciplinary research in Biosphere Reserves calls for the engagement of underrepresented groups and for addressing deep leverage points to foster sustainable change.

Introduction

Acknowledging the need for integrated approaches to mainstream sustainable development, the UNESCO Man and the Biosphere (MAB) Programme promotes Biosphere Reserves as areas dedicated both to nature conservation and sustainable human development (UNESCO 2017). Biosphere Reserves have three core missions: (1) biodiversity conservation, (2) economic development and (3) logistic support and capacity building, in particular through research (UNESCO 1996). To meet those goals, the MAB Programme emphasizes the contribution of local actors - specifically of Women, Youth, and Indigenous People - to Biosphere Reserves' effective, equitable and participatory planning (UNESCO 2017). To support logistics and capacity building, scientific research is expected to contribute to the other missions, i.e. conservation and development. However, recent reviews have shown that research on Biosphere Reserves has been largely confined to the natural sciences (Kratzer 2018, Pool-Stanvliet and Coetzer 2020). Research conducted in Biosphere Reserves merely used these study sites for a broad range of issues, but hardly focussed on the factors influencing a successful implementation of the MAB Programme (Ferreira et al. 2020, Pool-Stanvliet and Coetzer 2020). Biosphere Reserves research has seen an increasing call for more co-productive, inter- and transdisciplinary approaches to investigate pressing issues, such as effective management and participatory governance (van Cuong et al. 2017, Ferreira et al. 2020, Barraclough et al. 2023).

Transdisciplinary sustainability research is increasingly expected to promote solution-finding processes for real-world sustainability issues (Kates et al. 2001, Lang et al. 2012, Norström et al. 2020). This field emerged rather recently and encompasses a diverse array of approaches, e.g. participatory research, transformative research or knowledge co-production (Norström et al. 2020, Chambers et al. 2021). These approaches share the mission to produce knowledge in a participatory manner or mode (Pohl and Hirsch Hadorn 2008, Lang et al. 2012, Brandt et al. 2013), while inducing positive societal effects in the social-ecological systems they investigate (Pereira et al. 2020, Jahn et al. 2021, Lawrence et al. 2022). Three key features of transdisciplinary research can be highlighted: (1) a focus on real-world problems and developing solutions to wicked problems (Chambers et al. 2021, Lawrence et al. 2022), (2) the combination of different types of knowledge, including Indigenous and Local Knowledge (Brandt et al. 2013, Knapp et al. 2019, Bartlett et al. 2012, Reed et al. 2023) and (3) involvement of practitioners and relevant non-academic actors (Talwar et al. 2011, Lang et al. 2012, Jahn et al. 2021).

The current MAB strategy and its Lima Action Plan highlight the role of Biosphere Reserves in the operationalization of transdisciplinarity sustainability science, as it calls for involvement of local communities and relevant actors in Biosphere Reserves research and governance (UNESCO 2017). Scholars also have increasingly called for co-creative, participative and transdisciplinary approaches in Biosphere Reserves research (Ishwaran et al. 2008, Schultz et al. 2011, Coetzer et al. 2014, Barraclough et al. 2023). Yet, there is still a lack of evidence about the extent of transdisciplinarity in the global context of Biosphere Reserves and only few empirical studies have investigated the application and benefits of transdisciplinary research in Biosphere Reserves. Nevertheless, a body of literature has been dedicated to understanding how to improve the governance of Biosphere Reserves through adaptive co-management (Olsson et al. 2007, Schultz and Lundholm 2010, Plummer et al. 2017) and through bridging and facilitating organizations (Schultz et al. 2011, Reed and Abernathy 2018, Walk et al. 2020). Multiple studies were conducted within recent transdisciplinary programs in Canadian Biosphere Reserves, e.g. on the practice of collective learning, participatory research, knowledge co-production (Reed et al. 2014, Reed and Abernathy 2018), the challenges of community engagement (George and Reed 2017), or the ethics of transdisciplinary research conducted with and by Indigenous People (Reed et al. 2023). Another strand of literature anchored in South African Biosphere Reserves has explored e.g. the contributions of the MAB programme for conservation and development (Stanvliet et al. 2004, Coetzer et al. 2014), inter- and transdisciplinary processes to establish Biosphere Reserves (Pool-Stanvliet et al. 2018), and collaborative learning processes (Pool-Stanvliet and Coetzer 2020). Concerning the actual outcomes of applying transdisciplinary approaches in Biosphere Reserves, empirical evidence is limited. One example is a comparative study of four co-productive projects in the area of Kristianstad Vattenrike Biosphere Reserve, in Sweden (Malmborg et al. 2022).

There is also a need to better assess the contributions of Biosphere Reserves research - and specifically of transdisciplinary research - in addressing the need for diversity and participation. The most recent MAB strategy calls to enhance the participation of a diversity of actors, namely scientists, policy-makers, local communities (including Indigenous People) and the private sector, in facilitating sustainability science in Biosphere Reserves (UNESCO 2017). This call for participation and diversity has been echoed in sustainability science, particularly in transdisciplinary research (Staffa et al. 2022, Vogel and O'Brien 2022, Caniglia and Vogel 2023). However, when it comes to the concrete involvement of local actors in research processes, a stark contrast has been identified between ideal transdisciplinarity (i.e. methodologies committed to strong collaboration and empowerment of non-academic actors) and the wide-spread application of transdisciplinary approaches, often limited to consultations with non-academic actors (Brandt et al. 2013, Zscheischler and Rogga 2015, Jahn et al. 2021). Recent studies have argued that the discrepancy between the geographic location of researchers and their study sites - where researchers from Global North study the Global South - shows an academic neocolonial pattern (Brandt et al. 2013, Ghosh 2020, Sultana 2022, Zonta et al. 2023). In Biosphere Reserves research, the limited involvement of non-academic actors has been pointed out as well (Stoll-Kleeman et al. 2010, Reed 2016, Barraclough et al. 2021). While such neocolonial mechanisms, as well as the underrepresentation of Women and minorities have been identified in scientific research (Hofstra et al. 2020, Zonta et al. 2023), there is so far little evidence about such aspects in transdisciplinary research in Biosphere Reserves.

Transdisciplinary approaches ideally address real-world challenges and explore transformative solutions (Pereira et al. 2020, Lawrence et al. 2022). While it is expected that transdisciplinary research may support the successful implementation of the MAB Programme (Reed 2016, UNESCO 2017, Ferreira et al. 2020, Barraclough et al. 2023), there is still a lack of evidence about the outcomes of transdisciplinary approaches in terms of transformative impacts. It is a widespread challenge for transdisciplinary researchers to monitor and report about concrete societal impacts and there is no standard procedure so far (Newig et al. 2019, Chambers et al. 2021, Schäfer et al. 2021). Hence, to address the transformative potential of transdisciplinary research in Biosphere Reserves, we follow recent reviews (Brandt et al. 2013, Riechers et al. 2021b, Zimmermann et al. 2023) and examine (1) the different types of knowledge (systems, target, transformation and process knowledge) produced in scientific publications, from descriptive to more transformative types (Brandt et al. 2013, Lawrence et al. 2022) and (2) the leverage points addressed, i.e. the potential interventions, policies, innovations or practices and their more or less systemic impacts in focal situations (Meadows 1999, Meadows 2012, Abson et al. 2017).

This article aims to provide a comprehensive review of Biosphere Reserves research to date, delineating research contributions to sustainability science and to the MAB Programme. In this article, we refer to Biosphere Reserves research as the research conducted in, with or about Biosphere Reserves. We carried out a systematic literature review, in two steps: (1) we analyzed 3,304 scientific studies conducted in Biosphere Reserves through meta-data and word occurrence analysis, and (2) we analyzed in depth the contributions of 336 articles from the latter, general data set, which applied transdisciplinary approaches. We aim to answer the following questions:

1. How is Biosphere Reserves research -in particular transdisciplinary research- spatially distributed and how has it evolved over time?
2. What topics has Biosphere Reserves research -and specifically transdisciplinary research addressed so far?
3. How diverse is Biosphere Reserves research and specifically transdisciplinary research in terms of gender and actor participation?
4. What transformative potential does transdisciplinary research in Biosphere Reserves display?

Methods

Data extraction

We used Web of Science and Scopus to download all publications containing “biosphere reserve*” OR “biosphere region*” OR “biosphere area*” in the title, keywords, abstract and text (later referred to as “general data set”). To identify as many transdisciplinary publications as possible within the general data set (later referred to as “transdisciplinary data set”), we used a broad range of keywords (Table S1), based on recent reviews or conceptualisations of transdisciplinary science (e.g. Lang et al. 2012, Brandt et al. 2013, Knapp et al. 2019, Chambers et al. 2021, Schäfer et al. 2021).

Following the preliminary articles identification via keywords, we screened the general data set and excluded articles based on the following criteria. We only selected articles written in English and original research articles (e.g opinion papers were excluded). Additional inclusion criteria were that the research took place in at least one UNESCO Biosphere Reserve and that articles full text should be accessible online. Based on these criteria, we retained 3,304 articles for the overview of Biosphere Reserves research. Furthermore, we

screened the preliminary selection of transdisciplinary articles and excluded articles that did not involve any non-academic actors in the research process, resulting in the selection of 336 transdisciplinary publications for in-depth analysis.
The data extraction and organization was carried out by C.G. The transdisciplinary keywords were selected by C.H.D and approved by all authors. For transparency and accountability purposes, we follow the MeRIT guidelines proposed by Nakagawa et al. (2023) and report throughout the methods section who has contributed to which steps of the study.

Coding process
The selected 3,304 articles, including the 336 transdisciplinary articles, were then coded as follows. First, we coded all 3,304 articles according to geographic location of study sites and author affiliations, and authors gender distribution, based on title and abstract. We then coded the 336 transdisciplinary articles based on full-text analysis, according to participation of non-academic actors, knowledge types and leverage points (section 2.3). The coding was conducted by: F.N., S.H., B.d.F.A., C.H.D., C.G., F.W., C.M. and J.H. and the data cleaning by : C.H.D., C.G. and F.W. See Table S.1 for a summary of the review process.
To ensure validity of results across the 10 coders, the following measures were implemented. First, C.H.D., C.H. and F.W. created guidelines for all coding variables. The guidelines contained comprehensive descriptions of the coding variables and facilitated a shared understanding and consistent application of these variables across coders. In a one-day workshop, the objectives, variables, guidelines and coding procedure were presented by C.H.D., J.H., C.G. and F.W. to all coders and A.F.F. and H.v.W.. Additionally, all coders engaged in parallel analysis of several articles over the course of two weeks. They compared and discussed results in daily meetings, which ensured a consistent interpretation of the coding guidelines. Finally, a set of the variables was independently coded by two different coders in 292 randomly selected articles. By comparing these duplicates, we were able to calculate the accuracy rate in coding and quantify potential coder bias (calculations by F.W., see Fig. S4).

Coding variables
To gain an overview of the general characteristics and topics of interest in Biosphere Reserves research and of their evolution over time, we used meta data such as publication date, and word occurrence analysis. We examined all publications in terms of geographic location and gender distribution, to identify potential bias in representation and participation (Table 1). We categorized all articles according to the geographic location of first and last authors' affiliation. We also categorized the continent and country of study area. Articles studying several Biosphere Reserves in different continents were classified as

“transcontinental”. We then recorded the gender of the first and last authors, classified as female, male or unknown, according to genderize.io. We used this algorithm to categorize the gender of first names with a probability of at least 95% of accuracy, as proposed by e.g. Fox et al. (2019), although we have to acknowledge the limitations and potential bias of this binary gender approach.

Table 1: Description and operationalisation of review variables

Variables	Description
Diversity and inclusivity	
Geographic location of study area	Continent and country of the studied Biosphere Reserve(s)
Geographic location of first and last authors’ affiliations	Continent and country of the first affiliation attributed to the first and last authors
Gender of first and last authors	Categorization of first and last authors’ first names as female/male/unknown using genderize.io
Participation in transdisciplinary research	
Number of non-academic co-authors	Number of publication authors without academic affiliation
Acknowledgement of non-academic actors	Whether or not non-academic actors are acknowledged in the publication
Type of actors	Types of actors that are involved in the transdisciplinary research process, e.g. Biosphere Reserves representatives, Indigenous People, land users
Level of actor involvement	The extent of involvement of non-academic actors in the transdisciplinary research process, categorized as consultation, collaboration and/or empowerment
Transformative potential of transdisciplinary research	
Knowledge types	Types of knowledge produced in the publication, categorized as system, target, transformation and/or process knowledge
Leverage points	Types of leverage points targeted in the publication, categorized as parameters, feedbacks, design and/or intent

To assess in more detail the quality of participation and diversity in transdisciplinary research, we categorized publications according to the following variables (Table 1). First, we counted the number of authors, whose affiliations were not academic, to appraise whether transdisciplinary projects fostered the participation of non-academic actors in scientific outputs. Second, we recorded whether or not the publications mentioned non-academic actors in the acknowledgements, as a further indicator of the involvement and recognition of non-academic actors in the research process. With in-depth analysis of the articles, we then categorized the type of actors involved in the research process, adapting the list of Ferreira et al. (2020). We also recorded their level of involvement, categorized as consultation, collaboration or empowerment (Brandt et al. 2013, Fritz and Binder 2020, Jahn et al. 2021). To capture the transformative potential of transdisciplinary research in Biosphere Reserves, we categorized transdisciplinary articles according, first, to the type of knowledge they produced (Brandt et al. 2013, Lawrence et al. 2022) and, second, to the leverage points they addressed (Dorninger et al. 2020, Riechers et al. 2021a, Zimmermann et al. 2023) (Box 1). With regard to knowledge types, we categorized the contributions of the transdisciplinary articles as systems, target, transformation or process knowledge (Table 1). Furthermore, to better appraise the contributions of transdisciplinary research to the understanding and development of solutions, we categorized articles according to the leverage points they addressed (Meadows 1999, Meadows 2012, Abson et al. 2017). The leverage points framework distinguishes interventions, such as policies or innovations, according to the more or less radical impacts they might have on the systems they target.

Box 1. Assessing transformative impacts of transdisciplinary publications through knowledge types and leverage points
1. Knowledge types
Different knowledge types have been identified based on the different objects of study in transdisciplinary research (e.g. Hirsch Hadorn et al. 2006, Brandt et al. 2013, Lawrence et al. 2022). Systems knowledge explores the history, root causes and functioning of specific situations and systems, e.g. exploring the root causes of ecosystem degradation. Target knowledge contributes insights into how a situation should or could be, for example studying local actors' preferences towards different land use and management systems. Transformation knowledge explores how to change a situation to the desired outcomes and how solutions could be implemented, for instance studying how to foster value and behaviour shifts. Finally, process knowledge addresses how to carry out transdisciplinary research, e.g. sharing insights on ethical requirements for transdisciplinary processes or developing new methodologies for actor involvement. In this review, we assumed that transdisciplinary studies have stronger transformative impacts when they produce target or transformation knowledge rather than systems knowledge.
2. Leverage points
Leverage points refer to interventions' shallow or deep impacts on a targeted system, i.e. the capacity of interventions to radically change a system (Meadows 1999, Meadows 2012, Abson et al. 2017). For instance, the level of parameters targets very shallow leverage points, such as adapting the level of resource use quota (Fischer and Riechers 2019). These shallow leverage points are rather easy to implement but have limited systemic impacts. Feedbacks refer to systemic interactions and feedback loops between elements of a system, such as delays and time in how the ozone hole can change after a stop on emissions (Fischer and Riechers 2019). Design leverages are more radical in that they affect information flows, the way systems are structured and organized and the power to change the systems rules, such as changes in policies or self-regulation of communities (Abson et al. 2017). Finally, leverages on the intent level, such as value shifts and institutional change, are more difficult to implement but are expected to have strong systemic, radical outcomes (Abson et al. 2017, Riechers et al. 2022). We categorized transdisciplinary publications according to whether they produced knowledge about interventions targeting parameters, feedbacks, design and/or intent and assumed that transdisciplinary studies have stronger transformative impacts when they address deep (design and intent), rather than shallow (parameters and feedbacks) leverage points.

The selection of the variables resulted from a test round of coding conducted by C.H.D, C.G., F.W., C.M., J.H.. The final selection of variables was decided by C.H.D, C.G., F.W., H.v.W. and A.F.F.

<u>Data analysis</u>

In order to identify clusters within the body of literature examined by us, we used a multivariate statistical approach first developed by Abson et al. (2014). We created a corpus containing all words within each individual paper, and reduced this exhaustive list to words included in at least 5 % of the papers. This list was then manually refined to contain only words that transport a meaning, thereby excluding stopwords. Groups were then derived based on a cluster analysis using Wards method, aiming to identify relatively equally sized groups. Based on an Indicator species analysis (Dufrene and Legendre 1997) we identified significant indicator words for each group, which were subsequently visualized in a

Detrended Correspondence Analysis. This linguistic analysis was conducted both for the whole dataset as well as for the subset only containing papers containing a transdisciplinary approach. We used the R programming language v.4.3.1 (R Core Team 2023) with RStudio v.2023.06.1+524 (Rstudio Team, 2023) for all descriptive statistics and analyses.
C.G., F.W., H.v.W. analyzed and visualized the data sets.

Results

<u>Research in Biosphere Reserves</u>

Spatial distribution
Our analysis of 3,304 publications on Biosphere Reserves showed a research focus in North America with 27 %, Asia with 25 % and Europe with 22% of all publications (Fig. 1). Mexican and Indian Biosphere Reserves were studied the most with 825 and 390 publications, respectively (see Box 2). As of 2023, 35.3% of the 748 Biosphere Reserves are located in Europe, followed by Asia and North America. The share of studies per continent for all studies and transdisciplinary studies were relatively indifferent. Europe had the highest number of general studies and, together with Mexico, the highest number of transdisciplinary studies. Studies with a transdisciplinary approach are predominantly conducted in Europe, in North America and particularly in Mexico and India (see Box 2). Having a high number of designated Biosphere Reserves does not translate, however, in a high number of studies in the same country, with Mexico being a notable exception (Fig. S5).

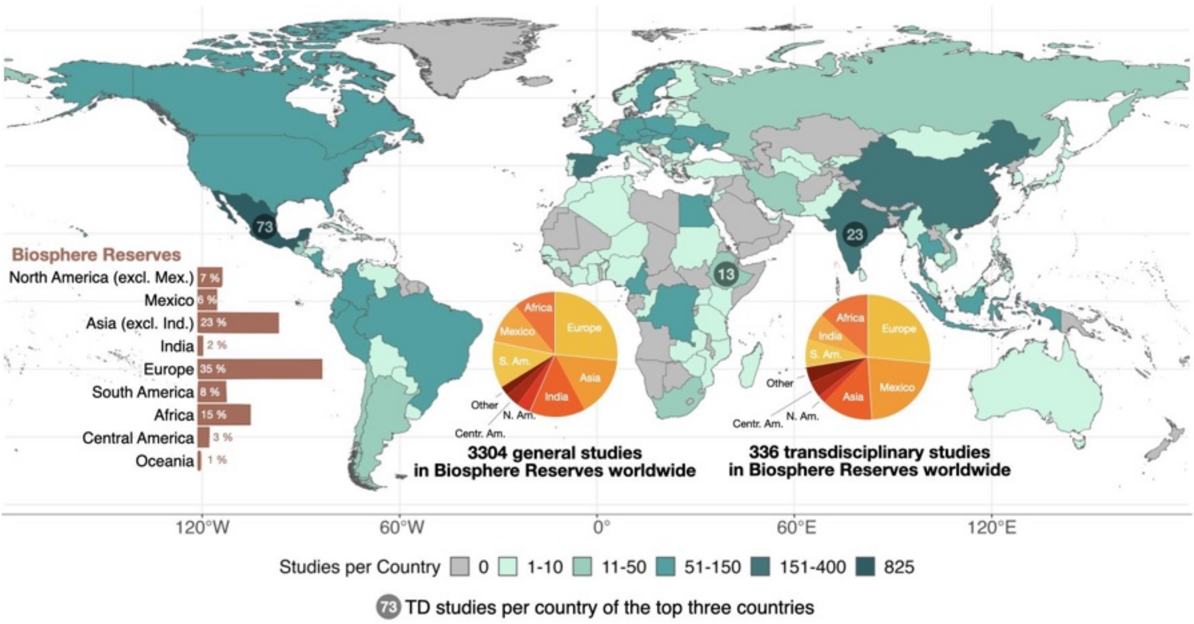


Fig. 1: World map of the general dataset with 3,304 publications of research in UNESCO Biosphere reserves. The color-coding illustrates the number of publications on Biosphere Reserves per country. The countries with the highest numbers of transdisciplinary studies are highlighted: Mexico 73, India 23 and Ethiopia 13. The share of publications per continent for 336 transdisciplinary publications and the general 3,304 publications is depicted

in the pie charts. Right bar plot: Share of Biosphere Reserves per continent considering all 748 designated Biosphere Reserves as of 2023.

Research in Biosphere Reserves is primarily conducted on the continent of the first author's institution (Fig. 2 A). Also, the institutions of the first and last author tend to be on the same continent. Even though the regional research focus applies to the transdisciplinary publications on Biosphere Reserves as well, a similar pattern can be observed (Fig. 2 B). If researchers from Europe and North America study Biosphere Reserves outside of the Global North, they focus on the Asian and African continent; hardly any researchers from there work in the Global North.

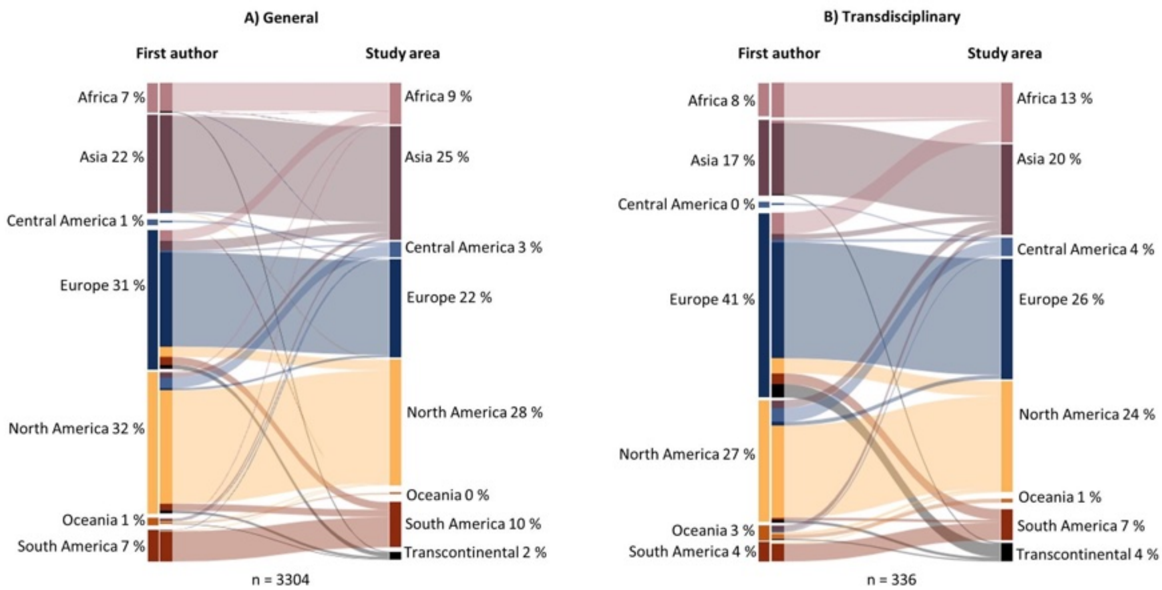


Fig. 2: Proportion of research institution of the first author at time of publication in relation to the study area. For A) the general dataset and B) the transdisciplinary research in Biosphere Reserves.

Temporal distribution

The annual number of published studies conducted in Biosphere Reserves increased from a few articles during the late 1970s to more than 300 in 2020 (Fig. 3 A). Generally, we observed an increasing trend of annually published studies compared to the number of designated Biosphere Reserves, which is highlighted by numbers on the logarithmic scale (Fig. 3 B, supplementary material Fig. S3). UNESCO designated substantially more Biosphere Reserves per year starting in the mid-1990s.

We also explored global trends at the level of the individual continents (Supplementary Material, Fig. S1). There have been designated Biosphere Reserves on all continents (excluding Antarctica) since the late 1970s. The number of studies from North America (including Mexico) increased from the mid-1990s and faster than on other continents, while the number of studies from Europe, Asia (including India), and South America started to notably increase only after 2000. Annual publications from African Biosphere Reserves saw only little increase until the mid-2010s after which they experienced a steep increase. First publications from Central America and Oceania also appeared during the late 1990s and early 2000s, but numbers remain relatively low until today. Overall, there is an over-proportional (compared to the share of designated Biosphere Reserves) amount of published studies originating from Asian, North American, South American, and lately African Biosphere Reserve, while European, Oceanian, and Central American Biosphere Reserves tend to be under-represented in published research.

Publications of transdisciplinary studies in Biosphere Reserves increased most notably after 2000 (Fig. 3 A in red). The share of transdisciplinary publications to all publications increased over time but fluctuates over the years (Fig. 3 B).

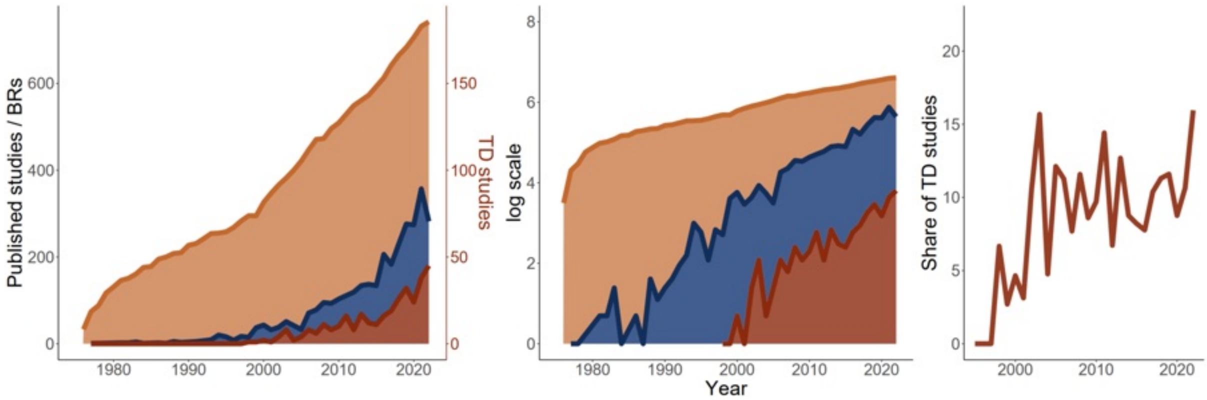


Fig. 3: Left: Cumulative number of designated Biosphere Reserves (BRs) in yellow and annual numbers of general publications in blue (n = 3,304) and transdisciplinary publications (TD) in red (n = 336) from 1975 to 2023, note the different y-axis-scales. Middle: The same on the log-scale. Right: Share of transdisciplinary publications from 1995 on.

<u>Research clusters</u>

We identified seven clusters best suitable for describing the thematic foci of the publications on Biosphere Reserves (Fig. 4 A). Studies with a social focus, including words such as participation, interviews or governance, cover similar research areas as studies on perspectives on people and cultural studies. Most of the publications of the transdisciplinary dataset belong to the participation group. Spatial studies, using words such as maps and pixel, were found close to the participation and culture clusters. Studies focusing on the biological environment, including words such as water and sediment, grouped close to microbiological studies, with words such as acid, bacteria or microbial. Botanical studies, represented by words such as vegetation and abundance, are grouped close to studies on genes. The share of studies with a focus on people, using words like rural, sustainable and cultural (Fig. 4 B and C , culture), decreased over time, while the share of studies with words such as participation, governance and actors increased (Fig. 4 B and C, participation).

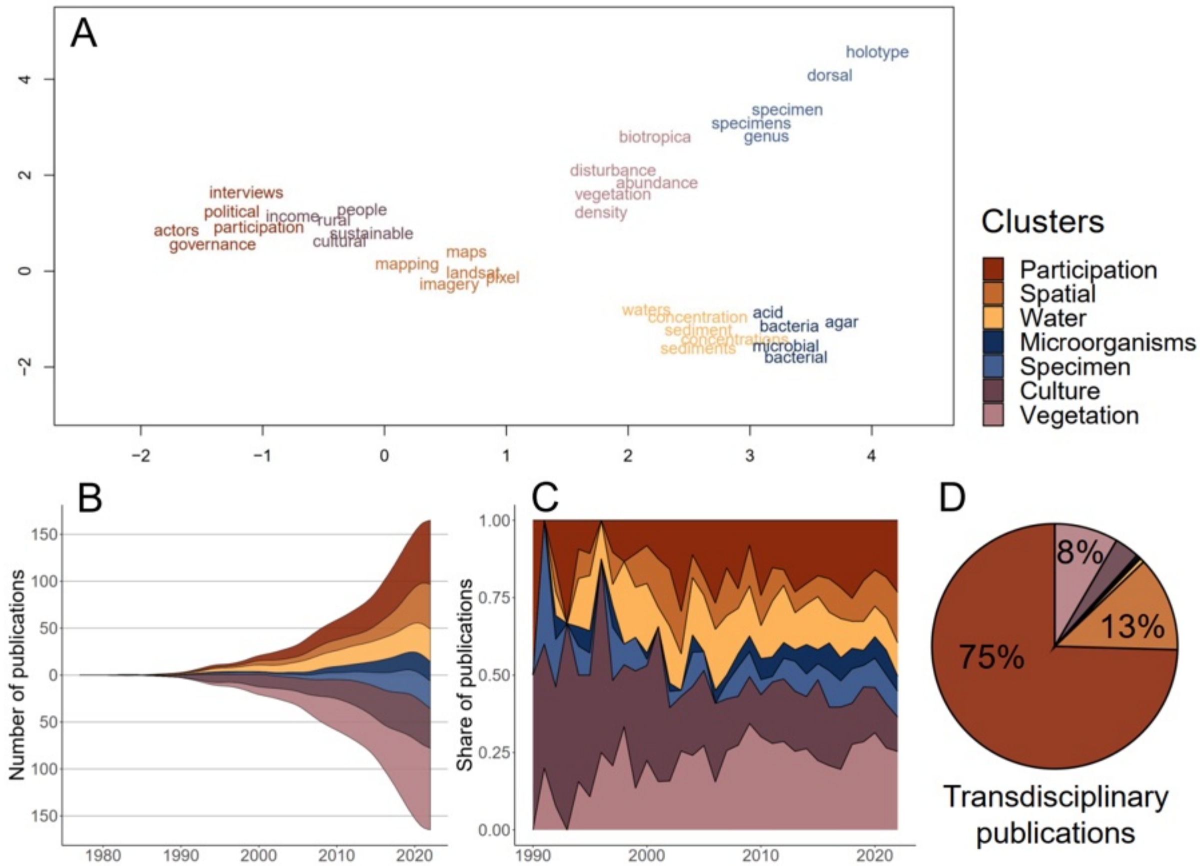


Fig. 4: Research clusters of 3,304 publications of research in Biosphere Reserves: A) Detrended component analysis results of the clusters. B) Annual total numbers, C) annual share and a D) pie-chart showing the representation of these groups in the transdisciplinary publication (n=336) only (top right).

We identified more groups with a focus on natural sciences for publications in the general data set than for transdisciplinary articles. The transdisciplinary studies were clustered in groups of social and political sciences in the groups of general publications. Generally, groups were more delineated and separated in the general studies than in the transdisciplinary studies.

We identified five natural groups best suitable to describe the topics of transdisciplinary publications on Biosphere Reserves (Fig. 5 A). Research clusters focusing on politics were grouped and studies on ethnobotany were most distant. Publications tackling deforestation and degradation increased over time (Fig. 5 B and C, degradation), as well as those focussing on collaborations and stakeholder engagement (Fig. 5 B and C, collaboration).

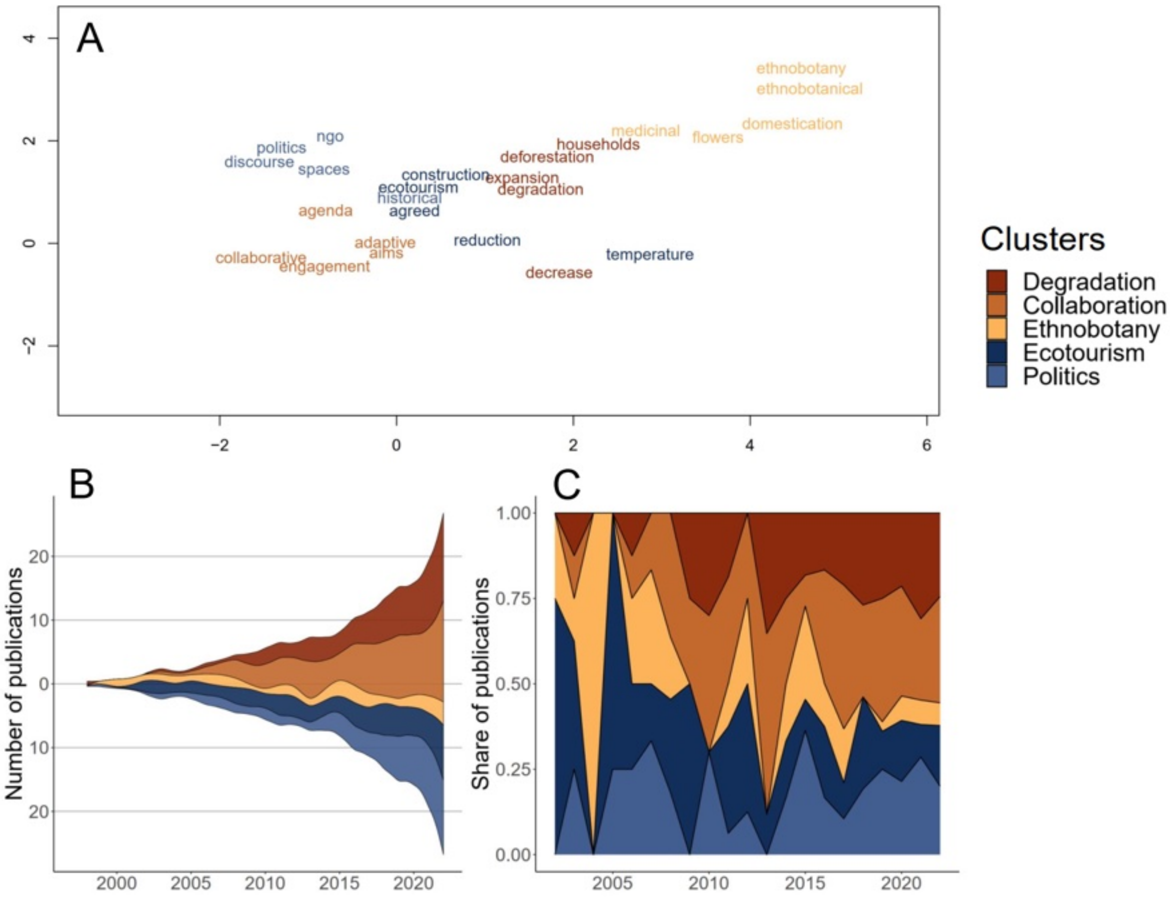


Fig. 5: Research clusters in transdisciplinary publications of research in Biosphere Reserves (n=336). A) Word cloud of research fields with five clusters, B) Annual total numbers and C) annual share.

<u>Diversity and participation</u>

Gender

We identified a higher percentage of male first authors (48%) and last authors (55%), compared to female first authors (31%) and last authors (24%) for publications studying Biosphere Reserves (Fig. 6). The pattern is also the same across continents, with South America and Asia having a higher number of female first authors. In South America there are even more female first authors than male first authors. With 43% male and 47% female first authors, the ratio for publications with a transdisciplinary approach is closer to parity, although 55% of the last authors are male and only 32% female. There is likely a geographic bias in these results due to a very high share of unknown gender for Asia, Africa and Oceania. This originates from genderize.io generally showing lower accuracies for non-western names. Female first authors work with male (51%) and female (41%) last authors, whereas male first authors mostly work with male last authors (70%) (see Supplementary Material, Fig. S2, A). The publications with a transdisciplinary approach show a more balanced ratio (see Supplementary Material, Fig. S2, B). Regardless of the continent, more female first and last authors were identified in the transdisciplinary studies than in the general studies. It shifted not exclusively from male to female author shares, but showed a lower share of unknown gender in the transdisciplinary studies. Still, for both types of studies the majority of last authors were male. For additional percentages see the interactive supplementary material of the online version of this article.

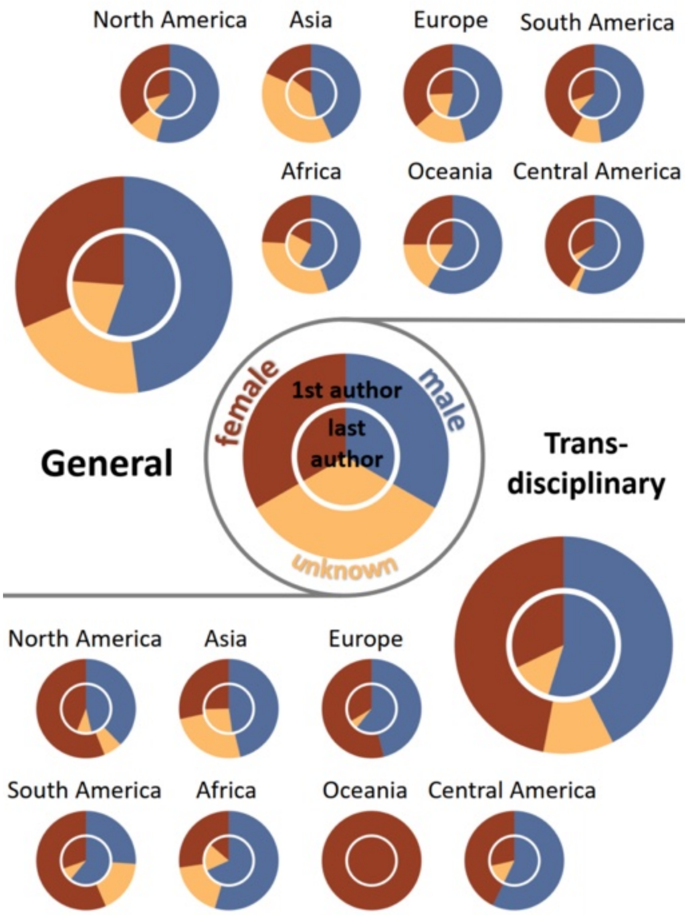


Fig. 6: Share of the gender (binary) of the first and last author for all publications of research in Biosphere Reserves (top) and for the publications with a transdisciplinary approach (bottom). Each dataset is visualized with its overall shares (large circles) and differentiated per continent of the study area (smaller circles). Absolute numbers (general/transdisciplinary): Overall (3,304/336), North America (924/82), Asia (834/67), Europe (720/89), South America (335/23), Africa (306/44), Oceania (12/3) and Central America (111/14). Transcontinental studies were excluded for this figure.

Actor participation in transdisciplinary research

In terms of actor types involved in transdisciplinary research in Biosphere Reserves, landusers, Indigenous People, government organizations and Biosphere Reserves management bodies have been most involved in publications on Biosphere reserves with a transdisciplinary approach. In comparison, Youth and Women groups were least involved (Fig. 7). Despite this general pattern, some differences can be identified among clusters of publications. Ethnobotany-related research seems to engage mainly only with indigenous and land use actors, but shows a higher percentage of involvement of Women groups.

Publications included in the politics cluster show, in general, a higher diversity of actors involved.

Regarding the extent of participation, actors were predominantly involved in a consultative role in transdisciplinary studies (98 %). Most studies (41%) built on collaboration with non-academic actors in the study design and only few publications (7%) reported about empowering actors. 53 of the 336 articles with a transdisciplinary approach (16%), were (co-)authored by the participating actors. Of all publications with a transdisciplinary approach, 51% acknowledged the actors, regardless of the type of participation.

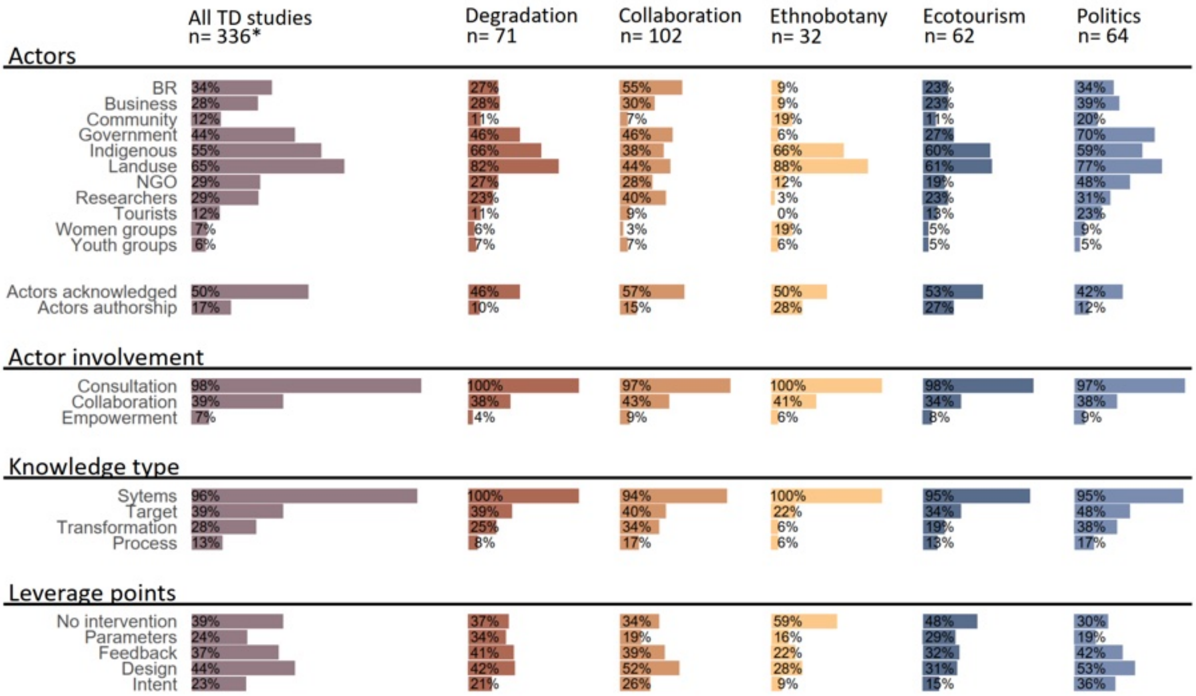


Fig. 7: Shares of actor groups, their way of involvement, generated knowledge types and leverage points in transdisciplinary studies, overall (far left) and for the five assigned research clusters. Shares for the categories of the individual variables do not add up to 100% as more than one or none of the categories could be valid for a single study. * 5 studies were not assigned to any group.

Transformative potential

We identified the creation of systems knowledge for most of the 336 publications with a transdisciplinary approach (Fig. 7). Few studies produced transformative (28 %) or process knowledge (28 %). This is reflected in the research clusters as well. Highest shares of target knowledge were identified in the (environmental) degradation, collaboration and politics research cluster. Process knowledge was mainly discussed in the collaboration and politics research cluster.

We found leverage points at a design level in 44 % and at a feedback level in 37 % of the publications with a transdisciplinary approach. Parameters and intent were least studied. A rather large number of studies did not specifically explore any interventions (39%). Patterns are again similar in all research clusters. The highest shares of design and intent leverage points were found in the collaboration and politics research cluster.

Discussion

Temporal and spatial trends in Biosphere Reserves research

Our results showed that scientific publications about Biosphere Reserves have increased steadily in the last decades, in line with recent reviews (Kratzer 2018, Ferreira et al. 2020). However, the number of articles on Biosphere Reserves can be expected to increase substantially in the near future, as the number of scientific articles published annually is generally increasing (Fire and Guestrin 2019). The share of transdisciplinary studies in Biosphere Reserves has increased slightly in comparison to all Biosphere Reserves research. This trend could be due to a general uptake of transdisciplinary research in sustainability science (Brandt et al. 2013, Ghodsvali et al. 2019). The Seville Strategy, in 1995, recognized the need for more social sciences and humanities in exploring good practice for the implementation of the MAB programme, and the most recent MAB strategy calls for biosphere reserves to operationalize sustainability science using transdisciplinary approaches (UNESCO 2017). While these strategies have set agendas and proposed relevant issues for governance and research, Biosphere Reserves are still widely dedicated to nature conservation (Reed 2016, Pool-Stanvliet and Coetzer 2020) - and Biosphere Reserves research to natural sciences rather than transdisciplinarity, as the total number of transdisciplinary studies represent a small fraction of existing biosphere reserves' research.

Spatial trends in Biosphere Reserves research revealed that Europe, North America (mostly Mexico) and Asia (mostly India) contributed most publications, both in the general and transdisciplinary data sets. The particularly high number of publications from Mexico and India has been pointed out in recent reviews (Kratzer 2018, Ferreira et al. 2020). These numerous publications are likely due to funding opportunities from dedicated governmental agencies and specific research institutions with long-standing research history in those areas (Box 2). Although most researchers studied areas on the same continent as their professional affiliation, researchers located in the Global North worked in Biosphere Reserves in the Global South more often than the other way around, as was identified in recent reviews in Biosphere Reserves research (Ferreira et al. 2020), and in transdisciplinary research in general (Brandt et al. 2013). In Africa, this pattern was even more pronounced for transdisciplinary research, with an even higher proportion of publications than in the general data set being produced by researchers affiliated to Europe. These results may concur with what has been identified as a neocolonial pattern in scientific publications across many disciplines (Dahdouh-Guebas et al. 2003), in particular in climate and sustainability science (Sultana 2022, 2023), as well as transdisciplinary sustainability research (Zonta et al. 2023). To address power imbalances and neocolonialism in sustainability science, scholars have, for example, proposed strategies to centre knowledge, philosophies and people from the Global South (Chilisa 2017, Sultana 2023) - or methodologies and practices to foster e.g reflexivity, safe spaces, respect and meaningful benefits for communities (Pereira et al. 2020, Thambinathan and Kinsella 2021, Reed et al. 2023). In this regard, we acknowledge that the authors team, albeit international and interdisciplinary, is mostly affiliated in Europe, and thus proposed a perspective from the Global North.

Research clusters

The research clusters in the general data set revealed a disciplinary gradient, from social sciences dedicated to participation and culture, to natural sciences dedicated to species-related studies. However, most clusters included publications investigating topics related to microorganisms, water, species, vegetation and spatial analysis. It is likely that this part of the research mostly contributes knowledge to the conservation mission of the MAB programme. Note however a slight increase in the number of publications dedicated to participation - which in turn might contribute to a better understanding of how to implement the human development mission of the MAB programme. While these results confirm recent findings showing that most research in Biosphere Reserves is still restricted to natural sciences (Kratzer 2018, Pool-Stanvliet and Coetzer 2020), the clusters also pinpointed a potential developing trend focussed on issues of participation and governance - a trend that would need confirmation by future reviews.

Box 2. Focus on Biosphere Reserves research in Mexico and India

With 25.1% (= 825 articles) and 22% (= 73 articles) respectively, scientists from Mexico publish the most in general and transdisciplinary studies on Biosphere Reserves as compared to all other countries. Of the 41 designated Mexican Biosphere Reserves, Calakmul, La Sepultura, El Viscaino, Sian Kaan and Mariposa Monarca are mostly studied with a transdisciplinary research design with six to ten studies each. There seem to be several catalysts for transdisciplinary research in Mexico: The National Council for Science and Technology in Mexico (CONACYT) subsidized 22 of the 73 transdisciplinary studies with grants and scholarships. 12 transdisciplinary papers were written by scientists of the National Autonomous University of Mexico. Additionally, “El Colegio de la Frontera Sur, Unidad Campeche (ECOSUR)” subsidized 10 of the transdisciplinary papers with knowledge, financial and logistical support.

India has 12 designated Biosphere Reserves and 11.8% of the general publications related to Biosphere Reserves come from there. Remarkably, the MAB programme in India was only launched in 1986 and the first Biosphere Reserve was established in 2000. All 23 transdisciplinary papers on Indian Biosphere Reserves were conducted in five Reserves, namely Nanda Devi (11), Khangchendzonga (5), Nilgiri (3), Sunderban (3) and Nokrek (1). The GB pant Institute of Himalayan Environment and Development alone supported one third of the transdisciplinary research papers and may therefore act as a catalyst for transdisciplinary research.

Publications on Biosphere Reserves in Mexico and India increased considerably in the last 10 years (Fig. Box 2). In Mexico, UNESCO designated 18 new Biosphere Reserves only in 2006. Both countries are characterized by distinct and highly vulnerable biodiversity, comprising priority regions for global conservation (Olson and Dinerstein 2002). Conservational efforts in both countries are high and research is supported. In Mexico, the National Commission of Natural Protected Areas (CONANP) manages and supports the Network of Biosphere Reserves, protecting in total more than 11,5% of Mexican land. In India government agencies, such as the Ministry of Environment, Forest and Climate Change, and the National Biodiversity Authority, provide support for scientific studies and conservation initiatives. The high research output in Mexico and India could be related to external funding as well. Mexico and India are amongst the highest recipients of biodiversity aid. From 1980-2008, India was first and Mexico 4th place with 9% and 3% respectively of worldwide biodiversity funding (Miller et al. 2013).

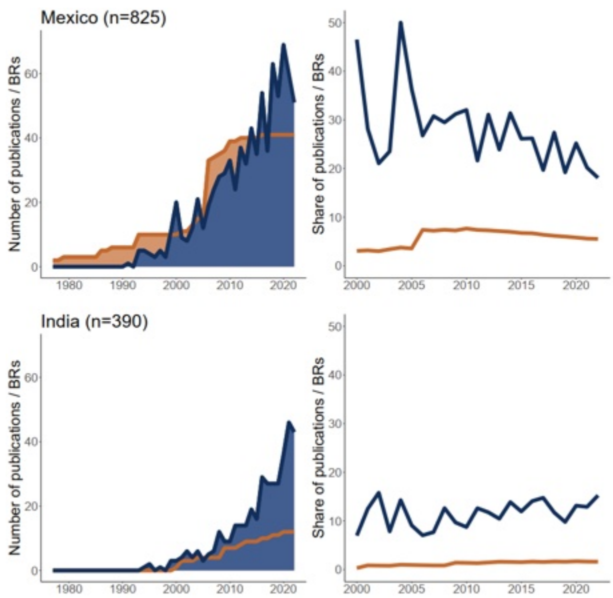


Fig. Box 2: Number of annual publications in blue and cumulative number of designated Biosphere Reserves in yellow from 1975 to 2023 for Mexico (top-left) and India (bottom-left). Share of annual publications and designated Biosphere Reserves after 2000 for Mexico (top-right) and India (bottom-right).

Transdisciplinary research in Biosphere Reserves also revealed a gradient from publications with a governance focus (politics, discourse, agenda) to social-ecological and ecological studies (ethnobotany, domestication, medicinal). Topic-wise, current transdisciplinary research in Biosphere Reserves could be classified in five clusters: ethnobotany, degradation, ecotourism, politics and collaboration. These transdisciplinary clusters were less differentiated than those in the general data set, suggesting that there are yet no clear schools within transdisciplinary research in Biosphere Reserves. The politics and collaboration clusters accounted for a majority of publications with a shared focus on governance.

Therefore, this review suggests that Biosphere Reserves seem to be used merely as interesting and logistically attractive sites to carry out research, rather than an object of research per se. Notwithstanding, the analyses also highlighted a coherent, albeit developing, literature bundle aiming to address issues related to Biosphere Reserves governance and management, and to the successful implementation of the MAB Programme and Agenda 2030 for Sustainable Development. Hence, there is still much room to explore conditions for successful Biosphere Reserves governance - and to highlight the contributions of the World Network of Biosphere Reserves to support place-based knowledge co-production in sustainability science (Barraclough et al. 2023).

Diversity and participation

In order to analyze diversity in Biosphere Reserves researchers, we recorded the gender of first and last authors. We found that research on biosphere reserves involved vastly fewer female than male authors across all continents. In particular, there were significantly fewer female last authors, often considered as Principal Investigators (PIs) - this discrepancy between the numbers of female first and last authors was particularly strong in Europe and South America. These results correspond to recent reviews showing that men still largely dominate the scientific system, especially when it comes to senior positions and e.g. co-authoring as PIs (Huang et al. 2020, Hofstra et al. 2020, Ross et al. 2022). In transdisciplinary research in Biosphere Reserves, the share of female authors increased remarkably in comparison to the general data set, while the share of authors categorized under unknown gender was reduced. In North and South America, Europe and Oceania, we observed even more female than male first authors. Although the share of female last authors was higher than in the general data set, it remained well below parity. Why Women proportionally authored more transdisciplinary publications than publications in the general data set remains unclear. Global reviews have shown that Women are better represented in specific disciplines, e.g. political science and psychology (Huang et al. 2020), or brain science and jurisprudence (Holman et al. 2018). However, there is no clear evidence so far, and to our knowledge, about the representation of Women in sustainability science, nor in transdisciplinary research. Our results call for a much stronger commitment to gender equality in (transdisciplinary) Biosphere Reserves research. Many guidelines have been provided to address gender inequalities in science. Examples include, but are not limited to, feminist and slow scholarship (Mountz et al. 2015), mother-friendly measures within research labs (Leventon et al. 2019), policies against early-career drop out (Cardel et al. 2020) and a feminist ethos of care in transdisciplinary sustainability science (Staffa et al. 2022).

To better understand diversity and participation in Biosphere Reserve research, we also analyzed which actor groups were involved in transdisciplinary publications and to what extent. Our analysis revealed that land users, governmental and Biosphere Reserve representatives, but also Indigenous People were involved in several studies, regardless of research clusters. It has been shown that transdisciplinary research too often relied on elite participants, e.g. government or large NGOs, while underrepresented groups are often least involved (Turnhout et al. 2020). While this holds true in our study for Women and the Youth, it is remarkable that Indigenous People were involved in more than half of the transdisciplinary studies. Note that Indigenous People, Women, the Youth and local communities are mentioned as target groups for effective and equitable participatory planning in the most recent MAB strategy (UNESCO 2017). Nonetheless, participation was very limited in most studies and usually restrained to extracting information through e.g. interviews, questionnaires or surveys. This transdisciplinary theory-practice gap has been identified in former reviews (Brandt et al. 2013, Jahn et al. 2021). Barriers to collaborative and empowering practices include funding contexts that e.g. require short-term results (Jahn et al. 2021), difficulties in ensuring participation of various actors (Lang et al. 2012, Lawrence et al. 2022) or underlying power relations and conflicts that fail to be addressed (Turnhout et al. 2020, Pereira et al. 2020). The aspirational character of transdisciplinarity has been criticized as an extractive, power-ladden and often neocolonial pattern that should be addressed more stringently in research (Zonta et al. 2023). To address this theory-practice gap, guidelines and recommendations have been provided - notably calling for a radical engagement with power relations, conflicts and discomfort within transdisciplinary research (Ghosh 2020, Turnhout et al. 2020, Pereira et al. 2020, Fritz and Binder 2020, Barraclough et al. 2023). An example study reporting about empowering processes was found in Rivera-Arriaga et al. (2021), who reported about participatory governance processes in collaboration with governmental, scientific and Mayan community representatives, in order to prevent ecological degradation and address local socio-political issues in Los Petenes Biosphere Reserve (Mexico). The results included building local capacities and co-creating a place-based, innovative management scheme aiming to ensure community wellbeing and environmental health.

Transformative potential

Our analysis showed that most transdisciplinary studies were limited in their transformative potential. Most studies were restricted to producing systems knowledge, i.e. helping to understand the current state and root causes of a specific system or issue. This held true for all research clusters. The Ethnobotany cluster produced even less target, transformation or process knowledge than all other clusters. This cluster seemed to build on citizen science to collect e.g. botanical data, which can explain the strong focus on systems knowledge. On the contrary, the collaboration and politics clusters featured a stronger transformative potential, with more studies producing target and transformation knowledge. Yet, these results confirm the aspirational character of transdisciplinary research (Brandt et al. 2013, Zscheischler and Rogga 2015, Turnhout et al. 2020) and call for a stronger engagement to produce transformative and solution-oriented knowledge, notably to support Biosphere Reserves management and the successful implementation of the MAB programme (Barraclough et al. 2023). An example of a study producing target knowledge, or knowledge about potential solutions and visions for a Biosphere Reserve, can be found in Choudhary et al. (2021), in which strategies and recommendations are developed for Community-based Tourism, with the goal to ensure conservation and rural development in the Majang Forest Biosphere Reserve (Ethiopia).

This review showed that transdisciplinary studies had mixed results in addressing concrete interventions for transformative change at different leverage points. For instance, a large part of the publications (regardless of research clusters) did not address any particular intervention, and only a quarter of all studies addressed interventions at intent level, or the deepest leverage (Abson et al. 2017). These mixed results mirror literature reviews on leverage points in research about food and energy systems (Dorninger et al. 2020), marine and coastal pollution (Riechers et al. 2021a) or in Arctic Indigenous food systems (Zimmermann et al. 2023). The collaboration and politics clusters seemed more impactful than all others, with many studies addressing deep leverage points at design and intent level. While the Lima Action Plan (UNESCO 2017) calls for inter- and transdisciplinary research to better understand how to improve the management and governance of Biosphere Reserves, there is much room to address potential interventions on deep leverages for this purpose. In this regard, strengthening research that addresses issues of collaboration, politics, and governance could support in leveraging this transformative potential and help bridge the gap between the concept of Biosphere Reserves and its implementation. An example of a study addressing deep leverage points can be found in Sharip et al. (2018). The study builds on focus group discussions with local actors to identify management challenges and to formulate recommendations for improved local communication and coordination of the several organizations responsible for environmental protection and governance in the Tasik Chini Biosphere Reserve (Malaysia).

Methodological challenges

Systematic literature reviews face common methodological challenges. Following recent reviews about Biosphere Reserves research (Kratzer 2018, Ferreira et al. 2020) and transdisciplinary research (Brandt et al. 2013, Zscheischler and Rogga 2015, Ghodsvali et al. 2019), we concentrated on articles available to a broad international readership, i.e. written in

English in peer-reviewed scientific journals. Although the proportion of other languages in the databases we used was small, we acknowledge that reviewing literature published in other languages might be relevant to give a complete overview of global literature to date, notably for transdisciplinary research.

To study diversity in authorship, we followed recent reviews (Fox et al. 2019, Hofstra et al. 2020, Ross et al. 2022) in categorizing authors as female, male or unknown gender based on their first names, using the genderize.io algorithm. Further empirical data would be necessary for a better understanding of authorship diversity and intersectionality. Furthermore, we acknowledge the need for more gender-sensitive and less biased tools to assess diversity in authorship globally. The algorithm we used was based on a binary understanding of gender and did not account for e.g. non-binary and fluid gender identities. This algorithm revealed a geographic bias, as first names of authors affiliated in Asia and Africa were significantly more often categorized as unknown than in Europe or North America. Finally, a more fine-tuned information on all co-authors (rather than only first and last authors) would give a more accurate overview, for example for fields where the second author is usually the PI.

Finally, we encountered common challenges in capturing contributions of transdisciplinary studies. Transdisciplinary research still rarely monitors and outlines societal impacts in scientific publications (Newig et al. 2019, Jahn et al. 2021, Schäfer et al. 2021). Societal impacts may become visible only in the long-term, after scientific publications (Pereira et al. 2020, Chambers et al. 2021). Following, it appeared difficult to appraise the transformative potential of transdisciplinary research through literature review of scientific publications only. Although we acknowledge that grey literature could address this gap, such in-depth studies have been conducted on a limited number of cases only (Jahn et al. 2021, Schäfer et al. 2021, Chambers et al. 2021). Hence, it remains a common procedure for global reviews of transdisciplinary research to focus on peer-reviewed articles only (Brandt et al. 2013, Ghodsvali et al. 2019). Our results analyze this research landscape and consequently are restricted to this branch of science.

Conclusion

The World Network of Biosphere Reserves provides ample opportunities for knowledge co-production about a wide array of sustainability issues - and for contributing place-based insights to global scientific debates. Yet, this review showed that a large portion of Biosphere Reserves research is located in a few specific continents with a focus on natural sciences. Definitely, transdisciplinary research has contributed to exploring conditions for successful Biosphere Reserves governance. However, there is scope for enhancing the transformative potential of Biosphere Reserves research. In that regard, a stronger commitment to gender equality, empowering forms of participation and knowledge integration about a broader range of topics are necessary. This would be essential to transform research in Biosphere Reserves into research about Biosphere Reserves and to highlight these areas as model regions for sustainability transformations.

References

Abson DJ, Fischer J, Leventon J, Newig J, Schomerus T, Vilsmaier U, Wehrden H von, Abernethy P, Ives CD, Jager NW, Lang DJ (2017) Leverage points for sustainability transformation. Ambio 46(1): 30–39

Abson DJ, Wehrden H von, Baumgärtner S, Fischer J, Hanspach J, Härdtle W, Heinrichs H, Klein AM, Lang DJ, Martens P, Walmsley D (2014) Ecosystem services as a boundary object for sustainability. Ecological Economics 103: 29–37

Barraclough AD, Reed MG, Coetzer K, Price MF, Schultz L, Moreira-Muñoz A, Måren I (2023) Global knowledge–action networks at the frontlines of sustainability: Insights from five decades of science for action in UNESCO 's World Network of biosphere reserves. People Nat

Barraclough AD, Schultz L, Måren IE (2021) Voices of young biosphere stewards on the strengths, weaknesses, and ways forward for 74 UNESCO Biosphere Reserves across 83 countries. Global Environmental Change 68: 102273

Bartlett C, Marshall M, Marshall A (2012) Two-Eyed Seeing and other lessons learned within a co-learning journey of bringing together indigenous and mainstream knowledges and ways of knowing. J Environ Stud Sci 2(4): 331–340

Brandt P, Ernst A, Gralla F, Luederitz C, Lang DJ, Newig J, Reinert F, Abson DJ, Wehrden H von (2013) A review of transdisciplinary research in sustainability science. Ecological Economics 92: 1–15

Caniglia G, Vogel C (2023) On being oriented: Strengthening transgressive orientations in transdisciplinary sustainability research through queer theory. GAIA - Ecological Perspectives for Science and Society 32(1): 167–171

Cardel MI, Dean N, Montoya-Williams D (2020) Preventing a Secondary Epidemic of Lost Early Career Scientists. Effects of COVID-19 Pandemic on Women with Children. Annals ATS 17(11): 1366–1370

Chambers JM, Wyborn C, Ryan ME, Reid RS, Riechers M, Serban A, Bennett NJ, Cvitanovic C, Fernández-Giménez ME, Galvin KA, Goldstein BE, Klenk NL, Tengö M, Brennan R, Cockburn JJ, Hill R, Munera C, Nel JL, Österblom H, Bednarek AT, Bennett EM, Brandeis A, Charli-Joseph L, Chatterton P, Curran K, Dumrongrojwatthana P, Durán AP, Fada SJ, Gerber J-D, Green JMH, Guerrero AM, Haller T, Horcea-Milcu A-I, Leimona B, Montana J, Rondeau R, Spierenburg M, Steyaert P, Zaehringer JG, Gruby R, Hutton J, Pickering T (2021) Six modes of co-production for sustainability. Nat Sustain 4(11): 983–996

Chilisa B (2017) Decolonising transdisciplinary research approaches: an African perspective for enhancing knowledge integration in sustainability science. Sustain Sci 12(5): 813–827

Choudhary CK, Fakana Selemon T, Mengist AB (2021) Conservation of Majang Forest Biosphere Reserve: An opportunity Conservation of Majang Forest Biosphere Reserve: South West Ethiopia. Asian Journal of Conservation Biology 10(2): 280–296

Coetzer KL, Witkowski ETF, Erasmus BFN (2014) Reviewing Biosphere Reserves globally: effective conservation action or bureaucratic label? Biological reviews of the Cambridge Philosophical Society 89(1): 82–104

Dahdouh-Guebas F, Ahimbisibwe J, van Moll R, Koedam N (2003) Neo-colonial science by the most industrialised upon the least developed countries in peer-reviewed publishing. Scientometrics 56(3): 329–343

Dorninger C, Abson DJ, Apetrei CI, Derwort P, Ives CD, Klaniecki K, Lam DP, Langsenlehner M, Riechers M, Spittler N, Wehrden H von (2020) Leverage points for sustainability transformation: a review on interventions in food and energy systems. Ecological Economics 171: 106570

Dufrene M, Legendre P (1997) Species Assemblages and Indicator Species: The Need for a Flexible Asymmetrical Approach. Ecological Monographs 67(3): 345

Ferreira AF, Zimmermann H, Santos R, Wehrden H von (2020) Biosphere Reserves' Management Effectiveness—A Systematic Literature Review and a Research Agenda. Sustainability 12(14): 5497

Fire M, Guestrin C (2019) Over-optimization of academic publishing metrics: observing Goodhart's Law in action. GigaScience 8(6)

Fischer J, Riechers M (2019) A leverage points perspective on sustainability. People Nat 1(1): 115–120

Fox CW, Duffy MA, Fairbairn DJ, Meyer JA (2019) Gender diversity of editorial boards and gender differences in the peer review process at six journals of ecology and evolution. Ecology and Evolution 9(24): 13636–13649

Fritz L, Binder CR (2020) Whose knowledge, whose values? An empirical analysis of power in transdisciplinary sustainability research. Eur J Futures Res 8(1)

George C, Reed MG (2017) Operationalising just sustainability: towards a model for place-based governance. Local Environment 22(9): 1105–1123

Ghodsvali M, Krishnamurthy S, Vries B de (2019) Review of transdisciplinary approaches to food-water-energy nexus: A guide towards sustainable development. Environmental Science & Policy 101: 266–278

Ghosh A (2020) Deconstructing a 2-year long transdisciplinary sustainability project in Northern universities: is rhetorical nobility obscuring procedural and political discords? Sustain Sci 15(4): 1111–1127

Hirsch Hadorn G, Bradley D, Pohl C, Rist S, Wiesmann U (2006) Implications of transdisciplinarity for sustainability research. Ecological Economics 60(1): 119–128

Hofstra B, Kulkarni VV, Munoz-Najar Galvez S, He B, Jurafsky D, McFarland DA (2020) The Diversity-Innovation Paradox in Science. Proceedings of the National Academy of Sciences of the United States of America 117(17): 9284–9291

Holman L, Stuart-Fox D, Hauser CE (2018) The gender gap in science: How long until women are equally represented? PLoS biology 16(4): e2004956

Huang J, Gates AJ, Sinatra R, Barabási A-L (2020) Historical comparison of gender inequality in scientific careers across countries and disciplines. Proc. Natl. Acad. Sci. U.S.A. 117(9): 4609–4616

Ishwaran N, Persic A, Tri NH (2008) Concept and practice. The case of UNESCO biosphere reserves. IJESD 7(2): 118

Jahn S, Newig J, Lang DJ, Kahle J, Bergmann M (2021) Demarcating transdisciplinary research in sustainability science—Five clusters of research modes based on evidence from 59 research projects. Sustainable Development 30(2): 343–357

Kates RW, Clark WC, Corell R, Hall JM, Jaeger CC, Lowe I, McCarthy JJ, Schellnhuber HJ, Bolin B, Dickson NM, Faucheux S, Gallopin GC, Grübler A, Huntley B, Jäger J, Jodha NS, Kasperson RE, Mabogunje A, Matson P, Mooney H, Moore B, O'Riordan T, Svedlin U (2001) Environment and development. Sustainability science. Science 292(5517): 641–642

Knapp CN, Reid RS, Fernández-Giménez ME, Klein JA, Galvin KA (2019) Placing Transdisciplinarity in Context: A Review of Approaches to Connect Scholars, Society and Action. Sustainability 11(18): 4899

Kratzer A (2018) Biosphere Reserves research: a bibliometric analysis. ecomont 10(2): 36–49

Lang DJ, Wiek A, Bergmann M, Stauffacher M, Martens P, Moll P, Swilling M, Thomas CJ (2012) Transdisciplinary research in sustainability science: practice, principles, and challenges. Sustain Sci 7(S1): 25–43
Lawrence MG, Williams S, Nanz P, Renn O (2022) Characteristics, potentials, and challenges of transdisciplinary research. One Earth 5(1): 44–61
Leventon J, Roelich K, Middlemiss L (2019) An academic mother's wish list: 12 things universities need. Nature
Malmborg K, Wallin I, Brukas V, Do T, Lodin I, Neset T-S, Norström AV, Powell N, Tonderski K (2022) Knowledge co-production in the Helge å catchment: a comparative analysis. Ecosystems and People 18(1): 565–582
Meadows D (1999) Meadows, D. 1999. Leverage points: Places to intervene in a system., The Sustainability Institute., Hartland, USA
Meadows DH (2012) Thinking in Systems. A Primer. Taylor and Francis, Hoboken
Miller DC, Agrawal A, Roberts JT (2013) Biodiversity, Governance, and the Allocation of International Aid for Conservation. Conservation Letters 6(1): 12–20
Mountz A, Bonds A, Mansfield B, Loyd J, Hyndman J, Walton-Roberts M, Basu R, Whitson R, Hawkins R, Hamilton T, Curran W (2015) For Slow Scholarship: A Feminist Politics of Resistance through Collective Action in the Neoliberal University. ACME 14(4): 1235–1259
Nakagawa S, Ivimey-Cook ER, Grainger MJ, O'Dea RE, Burke S, Drobniak SM, Gould E, Macartney EL, Martinig AR, Morrison K, Paquet M, Pick JL, Pottier P, Ricolfi L, Wilkinson DP, Willcox A, Williams C, Wilson LAB, Windecker SM, Yang Y, Lagisz M (2023) Method Reporting with Initials for Transparency (MeRIT) promotes more granularity and accountability for author contributions. Nature communications 14(1): 1788
Newig J, Jahn S, Lang DJ, Kahle J, Bergmann M (2019) Linking modes of research to their scientific and societal outcomes. Evidence from 81 sustainability-oriented research projects. Environmental Science & Policy 101: 147–155
Norström AV, Cvitanovic C, Löf MF, West S, Wyborn C, Balvanera P, Bednarek AT, Bennett EM, Biggs R, Bremond A de, Campbell BM, Canadell JG, Carpenter SR, Folke C, Fulton EA, Gaffney O, Gelcich S, Jouffray J-B, Leach M, Le Tissier M, Martín-López B, Louder E, Loutre M-F, Meadow AM, Nagendra H, Payne D, Peterson GD, Reyers B, Scholes R, Speranza CI, Spierenburg M, Stafford-Smith M, Tengö M, van der Hel S, van Putten I, Österblom H (2020) Principles for knowledge co-production in sustainability research. Nat Sustain 3(3): 182–190
Olson DM, Dinerstein E (2002) The Global 200: Priority Ecoregions for Global Conservation. Annals of the Missouri Botanical Garden 89(2): 199
Olsson P, Folke C, Galaz V, Hahn T, Schultz L (2007) Enhancing the Fit through Adaptive Co-management: Creating and Maintaining Bridging Functions for Matching Scales in the Kristianstads Vattenrike Biosphere Reserve, Sweden. E&S 12(1)
Pereira L, Frantzeskaki N, Hebinck A, Charli-Joseph L, Drimie S, Dyer M, Eakin H, Galafassi D, Karpouzoglou T, Marshall F, Moore M-L, Olsson P, Siqueiros-García JM, van Zwanenberg P, Vervoort JM (2020) Transformative spaces in the making: key lessons from nine cases in the Global South. Sustain Sci 15(1): 161–178
Plummer R, Baird J, Dzyundzyak A, Armitage D, Bodin Ö, Schultz L (2017) Is Adaptive Co-management Delivering? Examining Relationships Between Collaboration, Learning and Outcomes in UNESCO Biosphere Reserves. Ecological Economics 140: 79–88
Pohl C, Hirsch Hadorn G (2008) Methodological challenges of transdisciplinary research. Natures Sciences Sociétés 16(2): 111–121
Pool-Stanvliet R, Coetzer K (2020) The scientific value of UNESCO biosphere reserves. S. Afr. J. Sci. 116(1/2)
Pool-Stanvliet R, Stoll-Kleemann S, Giliomee JH (2018) Criteria for selection and evaluation of biosphere reserves in support of the UNESCO MAB programme in South Africa. Land Use Policy 76: 654–663

Reed MG (2016) Conservation (In)Action: Renewing the Relevance of UNESCO Biosphere Reserves. CONSERVATION LETTERS 9(6): 448–456
Reed MG, Abernathy P (2018) Facilitating Co-Production of Transdisciplinary Knowledge for Sustainability: Working with Canadian Biosphere Reserve Practitioners. Society & Natural Resources 31(1): 39–56
Reed MG, Godmaire H, Abernethy P, Guertin M-A (2014) Building a community of practice for sustainability: strengthening learning and collective action of Canadian biosphere reserves through a national partnership. Journal of environmental management 145: 230–239
Reed MG, Robson JP, Campos Rivera M, Chapela F, Davidson-Hunt I, Friedrichsen P, Haine E, Johnston ABD, Lichtenstein G, Lynes LS, Oloko M, Sánchez Luja M, Shackleton S, Soriano M, Sosa Peréz F, Vasseur L (2023) Guiding principles for transdisciplinary sustainability research and practice. People Nat
Riechers M, Brunner BP, Dajka J-C, Dușe IA, Lübker HM, Manlosa AO, Sala JE, Schaal T, Weidlich S (2021a) Leverage points for addressing marine and coastal pollution: A review. Marine pollution bulletin 167: 112263
Riechers M, Fischer J, Manlosa AO, Ortiz-Przychodzka S, Sala JE (2022) Operationalising the leverage points perspective for empirical research. Current Opinion in Environmental Sustainability 57: 101206
Riechers M, Loos J, Balázsi Á, García-Llorente M, Bieling C, Burgos-Ayala A, Chakroun L, Mattijssen TJ, Muhr MM, Pérez-Ramírez I, Raatikainen KJ, Rana S, Richardson M, Rosengren L, West S (2021b) Key advantages of the leverage points perspective to shape human-nature relations. Ecosystems and People 17(1): 205–214
Rivera-Arriaga E, Williams-Beck L, Vidal Hernández LE, García Arjona ME (2021) Crafting grassroots' socio-environmental governance for a coastal biosphere rural community in Campeche, Mexico. Ocean & Coastal Management 204: 105518
Ross MB, Glennon BM, Murciano-Goroff R, Berkes EG, Weinberg BA, Lane JI (2022) Women are credited less in science than men. Nature 608(7921): 135–145
Schäfer M, Bergmann M, Theiler L (2021) Systematizing societal effects of transdisciplinary research. Research Evaluation
Schultz L, Duit A, Folke C (2011) Participation, Adaptive Co-management, and Management Performance in the World Network of Biosphere Reserves. World Development 39(4): 662–671
Schultz L, Lundholm C (2010) Learning for resilience? Exploring learning opportunities in biosphere reserves. Environmental Education Research 16(5-6): 645–663
Sharip Z, Majizat A, Suratman S (2018) Socio-economic and institutional assessment of Malaysia's first biosphere reserve: Chini Lake. Lakes & Reservoirs 23(2): 104–116
Staffa RK, Riechers M, Martín-López B (2022) A feminist ethos for caring knowledge production in transdisciplinary sustainability science. Sustain Sci 17(1): 45–63
Stanvliet R, Jackson J, Davis G, Swardt C de, Mokhoele J, Thom Q, Lane BD (2004) The UNESCO biosphere reserve concept as a tool for urban sustainability: the CUBES Cape Town case study. Annals of the New York Academy of Sciences 1023: 80–104
Stoll-Kleeman S, La Vega-Leinert AC de, Schultz L (2010) The role of community participation in the effectiveness of UNESCO Biosphere Reserve management: evidence and reflections from two parallel global surveys. Envir. Conserv. 37(3): 227–238
Sultana F (2022) The unbearable heaviness of climate coloniality. Political Geography 99: 102638
Sultana F (2023) Whose growth in whose planetary boundaries? Decolonising planetary justice in the Anthropocene. Geography and Environment 10(2)
Talwar S, Wiek A, Robinson J (2011) User engagement in sustainability research. Sci. and Pub. Pol. 38(5): 379–390
Thambinathan V, Kinsella EA (2021) Decolonizing Methodologies in Qualitative Research: Creating Spaces for Transformative Praxis. International Journal of Qualitative Methods 20: 160940692110147

Turnhout E, Metze T, Wyborn C, Klenk N, Louder E (2020) The politics of co-production: participation, power, and transformation. Current Opinion in Environmental Sustainability 42: 15–21

UNESCO (1996) Biosphere Reserves: The Seville Strategy and the Statutory Framework of the World Network, Paris, France

UNESCO (2017) A New Roadmap for the Man and Biosphere (MAB) Programme and its World Network of Biosphere Reserves. MAB Strategy (2015-2025), Lima Action Plan (2016-2025), Lima Declaration, Paris, France

van Cuong C, Dart P, Hockings M (2017) Biosphere reserves: Attributes for success. Journal of environmental management 188: 9–17

Vogel C, O'Brien K (2022) Getting to the heart of transformation. Sustain Sci 17(2): 653–659

Walk H, Luthardt V, Nölting B (2020) Participatory learning for transdisciplinary science in biosphere reserves. In: Reed MG, Price MF (eds) UNESCO biosphere reserves. Supporting biocultural diversity, sustainability and society. Routledge, Abingdon, Oxon, New York, NY, pp 297–308

Zimmermann S, Dermody BJ, Theunissen B, Wassen MJ, Divine LM, Padula VM, Wehrden H von, Dorresteijn I (2023) A leverage points perspective on Arctic Indigenous food systems research: a systematic review. Sustain Sci

Zonta AL, Jacobi J, Mukhovi SM, Birachi E, Groote P von, Abad CR (2023) The role of transdisciplinarity in building a decolonial bridge between science, policy, and practice. GAIA - Ecological Perspectives for Science and Society 32(1): 107–114

Zscheischler J, Rogga S (2015) Transdisciplinarity in land use science – A review of concepts, empirical findings and current practices. Futures 65: 28–44

Acknowledgements for Dabard et al. 2024

We thank Martin Balas for contributing to initial concept development and article selection trials at an earlier stage of this paper. We thank Jahia Ifill Knobloch and Parker Myers for contributing to the coding of articles.

A 5 Positionality Statement

The research output and my approach can be put into reference. All maps and figures produced are not objective truth. The basis of all work is spatially explicit data yet processed and filtered by my understanding of Earth observation. I am a white, abled-bodied woman, both of my parents have a university degree. I was educated in Central Europe under the substantial influence and imprint of the Western science community and had access to funding money and institutional support in Europe and the USA. I am aware of these privileges that shaped to a certain extent my perspectives. Yet, the open-access directive of Earth observation institutions changes the field of remote sensing analysis substantially as ever more people have access to raw satellite data and the necessary learning platforms and materials to analyze them. I wish to further contribute and learn to broaden perspectives and share knowledge. I am hopeful that diverse perspectives will transform remote sensing analyses in sustainability science.

A 6 Acknowledgements

With some, it was a single inspiring coffee break, others contributed their scientific and emotional knowledge over all the years. Here, I list all their names from A to Z. I mention my supervisors and mentors, colleagues, family, friends, co-authors, and background co-authors. Thanks to everyone for jointly experiencing excursions, for encouraging me to test new research, theory and methods grounds, for online meetings during covid lockdowns, for giving me the opportunity to join TV interviews, for practicing sports and wellness, and for sharing music. Thank you for guiding me through pitfalls and for jointly celebrating achievements. Thank you for introducing me to each other, for supporting my research stay in Los Angeles, for re-reading my manuscripts, for positive affirmations, and for sharing constructive feedback, and critical perspectives.

"It looks like a solitary sport, but it takes a team."
Swimmer Diana Nyad

Ana Filipa Ferreira
Angela Dichte
Beate & Friedemann
Bernadette & Marlene
Caro, Yulia, Olli, Moritz & Chris
Caroline Dabard
Caroline Meier
Celina & Mischo
Christian Lotz
Deepika Mann, Nadine & Yojana
Douglas Sheil
Fabian Faßnacht
Fabian Schneider
Fabio Weiß & Martin Balas
Felix May
Flávia de Souza Mendes
Hans & Tilmann
Heike Zimmermann
Henrik von Wehrden
Jarne Jargow
Jeanette Blumröder
Jenny Hammerich, Laura
Joss & Helen
Lars Schmidt
Lea, Marie, Paula, Richard, Lila
Lennart
Maria Chiara & Serhii
Micha Edlich
Nathalie Pettorelli
Paty, Kirsten & Maria
Pierre L. Ibisch
Sabine & Rüdiger, Helmut
Samuel Favrichon
Sassan Saatchi
Solomiia, Frederike, Bruno
Vera Luthardt