COMPLEX INTERACTIONS AMONG ECOSYSTEM SERVICES, HUMAN WELL-BEING, AND THEIR LINKAGES TO TELECOUPLING PROCESSES

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ABSTRACT

COMPLEX INTERACTIONS AMONG ECOSYSTEM SERVICES, HUMAN WELL-BEING, AND THEIR LINKAGES TO TELECOUPLING PROCESSES

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With rapid economic and population growths, the increasing separation between where ecosystem services are needed and from where they are supplied makes managing multiple ecosystem services difficult. It is also challenging to strengthen the synergies between such ecosystem service flows and conservation activities as conservation activities can enhance human well-being through the improvements of ecosystem services. Increasing the demands for ecosystem services across regions may accelerate ecosystem service flows yet also damage the basic ability to provide ecosystem services in supply areas. However, little research has holistically examined the environmental and socioeconomic impacts of increasing separation between the supplies of and demands for ecosystem services. To fill these gaps in knowledge, the overall objectives of this dissertation are to examine the intricate interconnections among ecosystem service flows, natural systems, human well-being, and conservation policies simultaneously. This dissertation research applies the integrated framework of telecoupling (socioeconomic and environmental interactions over distances) to systematically uncover the agents, causes, and effects of dynamic ecosystem service flows across multiple coupled human and natural systems.

This research explores telecoupling processes regarding nature-based tourism (cultural service, Chapter 2 and 3), food (provisioning service, Chapter 4), and fresh water (provisioning and regulating service, Chapter 5) as well their interactions with biodiversity, human demands, and conservation policies. The spatial scale of this research is at the global level, as flows of

tourists, food, and fresh water occur across national and regional boundaries. Chapter 2 shows that protected areas managed strictly for biodiversity conservation have more visitors and species than those managed for mixed use. High population density surrounding protected areas and national income levels are also major socioeconomic factors related to nature-based tourism. Chapter 3 indicates that global tourism networks have become highly consolidated over time and that reduced transaction costs (e.g., language, distance, and visa policies) are more important in attracting international tourists than natural and cultural attractions. Furthermore, cost of living differences between countries decreased in importance over time. International tourist flows are resilient to political instability and terrorism risks. Chapter 4 investigates the effects of international food trade on biodiversity hotspots between developed and developing countries. My results show that international food trade may benefit global biodiversity due to the increasingly important role of developing countries without biodiversity hotspots in food exports. Chapter 5 explores how to integrate watershed conservation activities with built infrastructure approaches to sustain freshwater ecosystem services for global cities. My results indicate that wetlands in protected areas contribute to sustaining freshwater provisions to global cities. Forests in protected areas complement large dams for sediment reduction and hydropower production for cities, but cities mainly depend on dams for flood mitigation.

By assessing ecosystem service flows to people over distances, this research identifies how multiple ecosystem services are managed in order to provide benefits for distant beneficiaries and to whom subsidies (or payments) are paid for biodiversity and ecosystem service conservation. The integration of the telecoupling framework with ecosystem services provides new perspectives on global sustainability that help with the development of proactive strategies for biodiversity and ecosystem service conservation.

To Hana

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CHAPTER 1

INTRODUCTION

1.1. Background

The challenges of managing multiple ecosystem services include determining the complex interactions between ecosystem service supply and demand in the context of coupled human and natural systems (Bagstad et al., 2013; Burkhard et al., 2012; Chung and Kang, 2013; Liu et al., 2007). While rapid economic growth leads humans to become more dependent on ecosystem services (Guo et al., 2010), high ecosystem service demands may exceed the capacity of ecosystem service supplies worldwide (Burkhard et al., 2012; Liu et al., 2016a). The demands for ecosystem services in distant locations are met via processes such as food trade, water transfer, or traveling (Chung et al., 2018a; Chung et al., 2019; Liu et al., 2016a). The increasing separation between where ecosystem services are needed and where they are supplied from makes managing multiple ecosystem services difficult (Burkhard et al., 2014; Chung et al., 2018b; Wei et al., 2017). Increasing ecosystem service demand across regions may accelerate ecosystem service flows, but damage supply areas' basic abilities to provide those resources (Bagstad et al., 2014; Chung et al., 2018b). However, most of the last decade's ecosystem service assessments concentrate on either supply or demand separately.

In addition, it is a big challenge to strengthen the synergies between ecosystem services and biodiversity conservation (Adams, 2014). This is because people depend on key ecosystem services such as food production and water in areas that have high biodiversity (Brooks et al., 2014). Although biodiversity conservation efforts can enhance human well-being through the improvement of ecosystem services (Ferraro et al., 2015), increasing ecosystem service demand across regions may accelerate ecosystem service flows but damage the basic ability to provide ecosystem services in supply areas (Bagstad et al., 2014). Unequal distributions of ecosystem service benefits may also cause stakeholder conflict, institutional failure, and environmental degradation (Adams, 2014). A holistic approach is necessary to investigate complex interconnections between the ecosystem service supply and demand areas and to integrate human and natural systems using the framework of telecoupling (Liu et al., 2015b).

Telecoupling is defined as socioeconomic and environmental interactions over distances (Liu et al., 2013). The telecoupling framework allows the investigation of complex interconnections across distant coupled human and natural systems using five interrelated components: system, flow, cause, effect, and agent (Liu et al., 2013). Using the telecoupling framework helps determine causes and effects of flows (e.g., movement of energy, information, capital, and natural resources) between sending and receiving systems (Chung and Liu, 2019; Chung et al., 2018b). Ecosystem service supply, demand, and flow can be integrated with the telecoupling framework. In the telecoupling framework, ‘sending and receiving systems’ indicate ecosystem service supply and demand areas, respectively. ‘Causes’ are factors that affect the emergence and dynamics of telecouplings. ‘Effects’ are socioeconomic and environmental impacts triggered by telecoupling processes. ‘Flows’ are movements of ecosystem services, materials, energy, people, and information between systems. ‘Agents’ are entities that facilitate or prevent telecoupling processes. The integration of the telecoupling framework with ecosystem services can provide new perspectives on global sustainability and biodiversity that will help develop proactive strategies for global conservation. For example, the quantification of ecosystem service flows allows us to develop better management strategies, minimize trade-offs, and maximize synergies by determining the relationships between provisioning areas (e.g., source watersheds or sending systems) and beneficiaries (e.g., urban areas or receiving systems) (Serna-Chavez et al., 2014).

1.2. Goal and Objectives

The *main goal* of this dissertation is to analyze complex interconnections among biodiversity, ecosystem services, and human well-being, and their linkages to telecoupling processes. This dissertation research applies the integrated framework of *telecoupling* to systematically uncover the agents, causes, and effects of dynamic ecosystem services flows in telecoupled human and natural systems. The specific objectives are to:

(1) investigate global relationships between biodiversity conservation and nature-based tourism in protected areas (Chapter 2);

(2) examine international tourism dynamics in a globalized world using a social network analysis approach (Chapter 3);

(3) understand the impacts of international food trade on biodiversity hotspots (Chapter 4); and

(4) examine how to integrate built infrastructure and watershed conservation activities to sustain freshwater ecosystem services for world's cities (Chapter 5).

This research explores telecoupling processes regarding food (provisioning services), nature-based tourism (cultural services), and freshwater (provisioning and regulation services). The spatial scale of this research is at the global level because flows of food, tourism, and freshwater occur across national and regional boundaries. By assessing ecosystem service flows to people, this research can identify how multiple ecosystem services are managed to provide benefits for distant beneficiaries and to whom subsidies (or payments) are paid for ecosystem services conservation. This dissertation research contributes to a better foundation for future research and policy to enhance global sustainability.

CHAPTER 2

GLOBAL RELATIONSHIPS BETWEEN BIODIVERSITY AND NATURE-BASED TOURISM IN PROTECTED AREAS

In collaboration with

Thomas Dietz and Jianguo Liu

Abstract

The relationships between biodiversity conservation and ecosystem services (ES) are widely debated. However, it is still not clear how biodiversity conservation and ES interact with different strategies in and surrounding protected areas (PAs), the cornerstone for biodiversity conservation. Here, we present results from the interplay between biodiversity conservation and nature-based tourism (a cultural ES), while controlling for environmental and socioeconomic factors in and surrounding terrestrial PAs worldwide. Results indicate that nature-based tourism is more frequent in PAs that are of higher biodiversity, older, larger, more accessible from urban areas and at higher elevation. High population density surrounding PAs and national income levels are also major socioeconomic factors related to nature-based tourism. Furthermore, PAs managed mainly for biodiversity conservation have nearly 35% more visitors than those managed for mixed use. Strict management for biodiversity is also associated with increased biodiversity. These results show the importance of biodiversity in addressing nature-based tourism and suggest this interrelationship could be altered by different management strategies used by PAs.

2.1. Introduction

For more than a century, designating and managing protected areas (PAs) has been done with a goal of allowing current use of biodiversity, usually through tourism, while preserving resources for future generations (Beissinger et al., 2017). But since the first designation of PAs, there have been conflicts over the appropriate goals in managing such areas (Dietz, 2017b; Joppa and Pfaff, 2010; Liu et al., 2012; Mace, 2014; Tallis and Lubchenco, 2014; Watson et al., 2014). One goal emphasizes the protection of natural systems and biodiversity (nature for itself) (Mace, 2014). The other emphasizes the contribution of ecosystem services (ES) from PAs to human well-being (nature for people) (Mace, 2014). Some PAs are managed with a sharp focus on the sole goal of preserving biodiversity; others are managed with an intent to enhance the provision of multiple types of ES. Of course, preservation of natural systems and biodiversity can contribute to cultural ES, including nature-based tourism (Bayliss et al., 2014; Clements and Cumming, 2017). Additionally, biodiversity may enhance the production of a wide variety of ES beyond just cultural ES (Chung et al., 2015; Smith et al., 2017; Turner et al., 2012) but it is not necessarily the case that managing a PA for biodiversity will optimize overall provision of ES (Karp et al., 2015; Naidoo et al., 2008). Thus, understanding the relationship between ES and biodiversity is a major challenge for sustainability science (Carpenter et al., 2009; Chan et al., 2006; Graves et al., 2017; Ouyang et al., 2016; Turner et al., 2007).

Two further complexities emerge because PAs are not isolated from the rest of the world. First, PAs are often surrounded by a large "buffer zone" that is outside the direct management of the PA but that affects and is affected by what happens in the PA (DeFries, 2017). Further, PAs are telecoupled with non-adjacent systems in several ways that influence the supply of and demand for ES (Bagstad et al., 2013; Liu et al., 2016a). Most visitors to PAs have traveled from distant places to visit them (Liu et al., 2013; Xiao et al., 2017). PAs may provide water purification that have benefits to people hundreds or thousands of kilometers away, and in turn may be affected by upstream degradation of water quality (Watson et al., 2014). Agricultural activities surrounding PAs can negatively influence biodiversity conditions in PAs (Bailey et al., 2016; Palomo et al., 2013). The demand for agricultural products from the surrounding PAs may also be local, regional or global (Liu et al., 2015a). Finally, invasive species, which threaten many PAs may have their origins across the globe and climate change, a severe threat to many PAs, has its drivers distributed globally as well (Pimm et al., 2014; Zhong et al., 2015).

For many PAs, one of the most important ES is providing an attractive destination for nature-based tourism, which is both regional and global in origin. Such tourism may be influenced in complex ways by how PAs are managed (Graves et al., 2017; Karp et al., 2015). In some PAs, managing primarily for biodiversity might discourage nature-based tourism, while in others such management might be compatible with high demand for visits. Agricultural landscape surrounding PAs may provide additional attractions that could either increase or decrease demand for tourism at a PA (Baudron and Giller, 2014; Fleischer et al., 2018; Jie et al., 2013; Liu et al., 2012).

For individual PAs, we can trace plausible paths by which biodiversity conservation strategies change demand for nature-based tourism via environmental and socioeconomic changes in the PA and surrounding areas. But there is little empirical analysis of the overall effects of PA management on tourism demand and supply. To address this gap in the literature, we used data from PAs worldwide to examine the number of visitors to PAs as a function of the number of species in the PA and the management strategy being used, while controlling for environmental and socioeconomic factors. In addition, we investigated how different conservation strategies influence biodiversity and other factors both inside and outside PAs. Our analysis addresses two questions. First, how does biodiversity and nature-based tourism interact in PAs that may be governed by different conservation strategies? Second, which environmental and socioeconomic factors in and surrounding PAs influence visitation to PAs? Our analysis is based on terrestrial PAs that have visitation information between 2000 and 2014. Our results can contribute to a better understanding of how biodiversity and nature-based tourism interact in PAs and how these interactions may be altered by different conservation strategies used by PAs.

2.2. Materials and Methods

2.2.1. Data

The dataset was obtained by aggregating data from a number of international institutions, national statistical agencies, online datasets and the gray literature (Table S2.1). Our key dependent variable was the average annual visitor numbers for each PA. The final dataset contained 929 PAs in 50 countries with the annual visitor numbers at some point in the period 2000 to 2014 (Figure 2.1 and Table S2.2). We calculated visitation as the average annual visitor numbers in each PA over the 15-year period.

The two key independent variables are the management strategy being used at the PA and its biodiversity. Management strategy was operationalized as the IUCN management category. The IUCN management category is based on the primary management objectives of PAs, which should apply to more than 75% of the PA area (Dudley, 2008). The IUCN category facilitates global assessments across different countries by providing an international standard for classifying management strategies of PAs. The primary objective of categories II-IV is to protect biodiversity (PAs managed for biodiversity), while categories V-VI are to both protect nature and use natural resources sustainably (PAs managed for mixed use) (Baudron and Giller, 2014; Dudley, 2008; Joppa et al., 2008; Laurance et al., 2012). For example, Categories II-IV focus on minimizing human activities keeping the system in “as a natural state as possible”, but Categories V-VI allow sustainable use of natural resources (e.g., hunting and/or forestry) to balance interaction between people and nature (Dudley, 2008). Dividing all PAs into two groups helps to differentiate conservation management practices between those that manage for nature for itself (II-IV) and those that manage for nature and people (V-VI). We divided all 929 PAs into two groups (II- IV and V-VI): 677 PAs in Category II-IV were coded 1 and 252 PAs in Category V-VI were coded 0. We excluded marine PAs and PAs which had not been classified into one of the IUCN management categories. PAs in IUCN category Ia and Ib where visitor access is strictly limited were also excluded. To include active management PAs, we selected PAs that were designated and managed at the national or sub-national level. The designated PAs have a long-term commitment to conservation with legal means (IUCN and UNEP-WCMC, 2017).

Second, biodiversity was operationalized as the number of species of birds, mammals and amphibians within the PA (Jenkins et al., 2013; Pimm et al., 2014). The biodiversity mapping website (http://biodiversitymapping.org) provided a global map of species ranges for birds, mammals and amphibians based on data from IUCN (IUCN, 2014) and BirdLife International NatureServe (BirdLife International NatureServe, 2013). A species range polygon underlies these mapping efforts. We selected mammals, birds and amphibians because these species have most comprehensive data at a global level and because they seem likely to be the species that will influence visitors’ preferences (Hausmann et al., 2017a; Siikamäki et al., 2015). The species range maps provide current species native range “determined by using known occurrences of the species" as well as "the knowledge of habitat preferences, suitable habitat, elevation limited, and other expert knowledge of the species and its range (IUCN, 2014)." Although the species range maps are the best available global datasets, we note the maps may overestimate species richness as the range of potential distribution tends to be larger than the actual occurrences of the species (Willemen et al., 2015). All species maps have a spatial resolution of 10km by 10km, based on 2013 updated data. We only included native and extant species. We overlaid this species range map with the locations of PAs and extracted species number in each PA by using zonal statistics in ArcGIS (ESRI, 2015).

We included as control variables a number of characteristics of the PA that might influence its attractiveness for nature-based tourism: size, mean elevation, mean annual temperature, mean annual precipitation and age (years since formal designation). We also controlled for remoteness which was defined as travel time (in minutes) from the nearest major cities (population>50,000) and the percentage of total water supply originated in the PA. Higher percentage of water supply in the PA indicates that the PA has more freshwater resources (WRI, 2015). Finally our model included dummy variables for the continent in which the PA was located. Data on size, mean elevation, mean annual temperature, mean annual precipitation and travel time from the nearest urban area for each PA was extracted from the appropriate geographic data bases using PA boundaries to develop zonal statistics in ArcGIS.

In addition to the features of the PA itself, we have characterized buffer zones for each PA. Following previous research (Joppa and Pfaff, 2010; Wittemyer et al., 2008), we specified 10-km buffer zones around each PA. Capturing activity within the buffer zones is important because the PA and its management may influence conditions within the buffer zones and vice versa. For each 10-km buffer zone, we extracted population density, agricultural yield and the percentage of agricultural area. We selected only cases with valid values for all variables excluding those PAs for which data for relevant variables were missing.

We appreciated that specifying a 10-km buffer zone is somewhat arbitrary. To test the sensitivity of our analysis to the size of the buffer zone, we performed a multiple ring buffer analysis in ArcGIS and QGIS (ESRI, 2015; QGIS Development Team, 2014). We designated 10-km distance intervals from the PA boundary (0-km) to 50-km buffer zones. Then, we extracted numerical values from the PA boundary and each of five rings (0-10, 10-20, 20-30, 30-40 and 40-50 km) using the spatial dataset. In each PA boundary and ring, we obtained numerical values of environmental and socioeconomic factors (population density), agricultural factors (agricultural yield and agricultural area) and regulating ES (water supply originated in PAs). In the multiple ring buffer analysis, we did not consider agricultural factors within PA boundaries because many PAs prevent people from engaging in agricultural activities (Palomo et al., 2013).

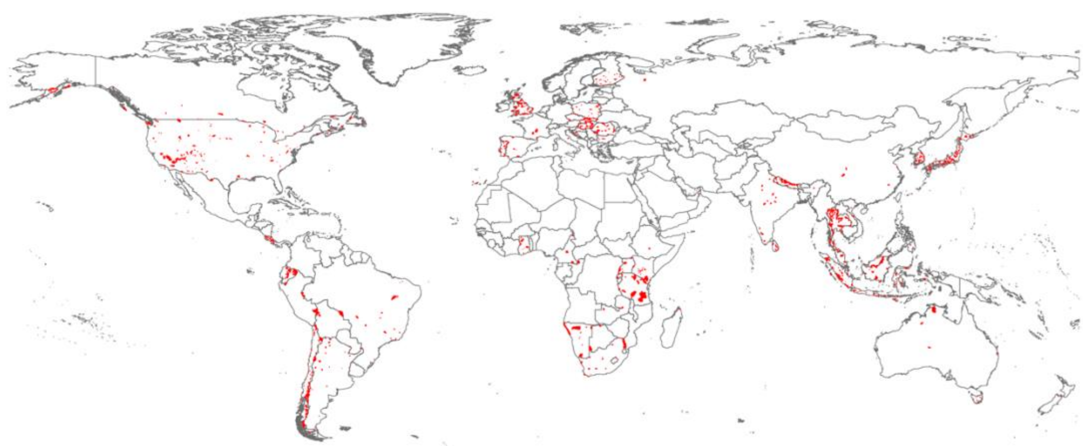

Figure 2.1. 929 PA locations in the world.

2.1.2. Modeling strategy

Our basic model predicts annual visits to each PA as a function of the species richness of the PA and the management strategy being used, with strategies ranging from strict emphasis on biodiversity protection to more mixed use. We also include a variety of control variables in our regressions to minimize the risk that the effects we estimate for biodiversity and management strategy are spurious. We control for features of the PA by including its size, mean elevation, annual mean temperature and precipitation, remoteness and age. We also control for population density within a 10 km buffer zone around the PA and for the affluence (gross domestic product per capita) of the nation in which the PA is located (reliable data on affluence cannot be obtained at a spatial scale corresponding to the 10 km buffer zone). Controls are also included for agriculture in the buffer zone and water supply originated in PAs (agricultural yield, % land area in agriculture and % total water supply originating in PAs). We provide a summary of variables regarding nature-based tourism hypothesis (Table 2.1). Finally, we include dummy variables for continent.

This model allows us to address our research questions by examining how biodiversity, management strategy and the characteristics of the PA itself and its buffer zone influence the popularity of a site for nature-based tourism.

Table 2.1. Summary of variables regarding nature-based tourism hypothesis

Variables	Relationships with nature-based tourism	Source(s)
Species richness	More species richness contributes to greater nature-based tourism value.	Arbieu et al. (2017); Hausmann et al. (2017a); Siikamäki et al. (2015); (Smith et al., 2017); Willemen et al. (2015)
Management strategies	PAs managed for biodiversity actively encourage visitors for nature-based tourism.	Dudley (2008)
Size of PAs	Larger size of PAs has more visitors	Balmford et al. (2015); Baum et al. (2017)
Elevation	Geographical attributes such as elevation may influence visitors' preferences	Hausmann et al. (2017b); Kumari et al. (2010)
Temperature and precipitation	Climate and weather are important factors for visitors (e.g., low humidity and heat stress)	Scott et al. (2008a); Verbos et al. (2017)
PA remoteness	Visitors are reluctant to go remote PAs	Balmford et al. (2015); Neuvonen et al. (2010)
PA age	Visitor numbers increase with PA age	Karanth and DeFries (2011); Neuvonen et al. (2010)
Population	Visitor numbers are higher when there is a higher population density surrounding PAs.	Balmford et al. (2015); Ghermandi and Nunes (2013)
GDP per capita	PAs in high-income countries have more visitor numbers	Balmford et al. (2015); Ghermandi and Nunes (2013)
Agricultural factor	Agricultural landscape surrounding PAs may provide additional attractions and/or food-related activities.	Baudron and Giller (2014); Fleischer et al. (2018); Hjalager and Johansen (2013); Jie et al. (2013)
Water supply in PAs	Plenty of water resources in PAs provide greater attractions (e.g., lakes, streams, waterfalls)	Cao et al. (2016); Nyaupane and Chhetri (2009); Reinius and Fredman (2007)

2.1.3. Regression model

The multiple regression equation for the nature-based tourism model is in the multiplicative form commonly used in the STIRPAT models (STochastic impacts by Regression on Population, Affluence and Technology) of human drivers of environmental change (Dietz, 2017a):

$$Y = aX_1^{b_1}X_2^{b_2}X_3^{b_3} \dots X_n^{b_n}E$$

For ease of estimation we used log base e of all except the binary variables, thus:

$$\log(Y) = a + b_1 logX_1 + b_2 logX_2 + b_3 logX_3 + \cdots + b_n logX_n + logE$$

Where Y stands for the average annual visitor numbers in each PA from 2000 to 2014, X_1 is the number of species, X_2 is IUCN management category, X_3 is the area of each PA, X_4 is mean elevation, X_5 is annual mean temperature, X_6 is annual precipitation, X_7 is PA remoteness from major cities, X_8 is PA age, X_9 is population density, X_{10} is per capita GDP at the national level, X_{11} is agricultural yield, X_{12} is the percentage of agricultural area, X_{13} is % water supply originated in PAs, X_{14} - X_{17} are dummy variables for each continent (Asia and Oceania, Africa, Europe and North America). E is the error term. Note that in this multiplicative form the unstandardized regression coefficients can be interpreted as elasticities. That is, our estimates indicated that a 1% change in an independent variable is associated with a b% change in the dependent variable, net of all other variables in the model. STIRPAT models have frequently been used to examine non-linearities beyond the log-log form and other specifications when there are theoretical arguments to do so. However, since our analysis is an initial exploration of factors related to visitation, we have kept to this rather well known functional form.

To account for model selection uncertainty, we used an information theoretic approach for model averaging. This approach provides robust parameter estimates based on model averaging across the best set of models by information theoretic criteria (e.g., Akaike Information Criterion (AIC)) rather a more traditional approach of selecting the best fitting model (Galipaud et al., 2014; Grueber et al., 2011). We first generated a candidate model set of 131,072 models to determine the model set for averaging. These models were then ranked based on AICc (AIC for small samples) to avoid overfitting (Grueber et al., 2011). Models with a smaller AICc are considered to have a better fit. We used a top 2AICc cut-off criterion which results in a set of three best models. The top 2AICc cut-off criterion indicates that AICc difference between model *i* and the top-ranked model is less than 2 ($\Delta_i = AICc_i - AICc_{top}$)

(Burnham and Anderson, 2002). Then, the parameter estimates of the top three models were averaged using Akaike weights (w_i). The Akaike weights (w_i) indicate the relative likelihood of the candidate models with a normalized scale (0-1) and provide a way to interpret Δ_i values as probabilities (Burnham and Anderson, 2002). Models with a bigger Δ_i have a smaller w_i. The percentage of water supply originating in PAs did not appear in the final model as this variable was not included in the top three models developed using the information theoretic approach.

Although some variables were not statistically significant, including all variables allow to identify indirect relationships on the annual visitations via biodiversity and guards against spurious relationships (Table 2.2). To formally test the indirect impacts of other factors on the annual visitations via biodiversity, we performed the regression of visitor numbers on all other variables except biodiversity. To capture the difference of the number of species between PAs primarily managed for biodiversity and PAs managed with more mixed objectives, we also modeled the number of species in PAs as a function of the same independent variables. Since this analysis is secondary to the analysis of tourism, we did not deploy the information theoretical approach to model selection.

According to the correlation matrix for the independent variables, 96% of 76 pairs had the value of r less than 0.5 (Figure S2.1). In addition to the correlation matrix, we examined collinearity using variance inflation factors (VIF) (O'Brien, 2007). All VIFs were less than 5, indicating no serious collinearity problems (Table S2.3). All statistical analyses were performed with R software (R Core Team, 2017). The information theoretical model averaging approach was deployed using MuMIn package in R. We used the procedures developed by Frank et al. (2013) to examine the robustness of our results. These procedures calculate what proportion of cases in the data set would have to be replaced with null hypothesis cases in order for the significance of a coefficient to drop below a threshold of interest. We used the conventional p=0.05 as our threshold. If a relatively modest proportion of cases would have to be replaced with null cases for a coefficient to fall below the p=0.05 threshold then the inference is rather fragile; if a high proportion of cases would have to be replaced the inference is robust.

Table 2.2. Descriptive statistics of dependent and independent variables, N=929.

Category	Variable	Mean	Std. Dev
Nature-based Tourism	Annual visitor numbers in PAs (persons)	367,405	1,793,697
Biodiversity	Total species (species)	326.88	172.540
Protected Area	IUCN category (II-IV=1)	0.729	0.445
	Size of PAs (km^2)	860.91	2,640.659
	Mean elevation (meter)	825.3	880.661
	Annual mean temperature (ºC)	14.449	8.063
	Annual precipitation (mm)	1298.898	827.042
	PA remoteness (minutes)	360.8	413.191
	PA age (year)	38.24	23.095
Demographic	Population density§ (persons/km^2)	140.012	471.987
Economic	GDP per capita¶ (2005 const. $ per capita)	16,127.7	16,342.57
Agricultural factor	Agricultural yields§ (tonne/km^2)	553.9	387.914
	Agricultural area§ (%)	30.051	24.773
Regulating ES	Water supply originated in PAs (%)	13.66	13.827
Region	Asia and Oceania	0.378	0.485
	Africa	0.097	0.296
	Europe	0.231	0.422
	North America	0.127	0.333
	Latin America	0.167	0.373

§ 10-km buffer zone
¶ Country level data, not PAs level

2.3. Results

2.3.1 Biodiversity and its conservation strategies had a positive relationship with nature-based tourism

Biodiversity has a positive relationship with the number of annual visitors to PAs (Table 2.3). Each 1% increase in the number of species is associated with an increase in annual visitors of about 0.87%, indicating that biodiversity is one of the strongest influences on tourism. IUCN management category also has a positive association with the annual visitors meaning that PAs managed strictly for biodiversity conservation attract more visitors than PAs for mixed use. Validation suggests that these results are relatively robust. To invalidate the inference of a positive relationship of the number of species with the annual visitors, 48% of the estimated effect would have to be due to bias (Frank et al., 2013). One can interpret this as 48% (or 446 PAs) of the cases in this study would have to be replaced with null hypothesis cases to invalidate the inference.

Table 2.3. Summary results of the model averaging predicting annual visitor numbers in PAs.

Category	Variable	Model 1	Model 2	Model 3	Model Averaging
Biodiversity	Total species (species)	0.879** (0.231)	0.870** (0.231)	0.868** (0.234)	0.874** (0.232)
Protected Area	IUCN category (II-IV=1)	0.351* (0.166)	0.347* (0.166)	0.348* (0.166)	0.349* (0.166)
	Size of PA (km^2)	0.309** (0.039)	0.309** (0.039)	0.310** (0.039)	0.309** (0.039)
	Mean elevation (meter)	0.329** (0.058)	0.343** (0.057)	0.331** (0.058)	0.334** (0.058)
	Annual mean temperature (ºC)	-0.378* (0.161)	-0.341* (0.160)	-0.383* (0.162)	-0.367* (0.162)
	Annual precipitation (mm)	-0.480** (0.118)	-0.469** (0.118)	-0.483** (0.118)	-0.477** (0.118)
	PA remoteness (minutes)	-0.236* (0.111)	-0.253* (0.111)	-0.240* (0.112)	-0.242* (0.112)
	PA age (year)	0.665** (0.117)	0.668** (0.117)	0.663** (0.117)	0.665** (0.117)
Demographic	Population density§ (persons/km^2)	0.455** (0.061)	0.469** (0.060)	0.448** (0.066)	0.458** (0.062)
Economic	GDP per capita¶ (2005 const. $ per capita)	1.262** (0.086)	1.279** (0.086)	1.266** (0.087)	1.268** (0.087)
Agricultural factor	Agricultural yields§ (tonne/km^2)	0.101 (0.059)	-	0.094 (0.063)	0.099 (0.060)
	Agricultural area§ (%)	-	-	0.027 (0.091)	0.027 (0.091)
Regulating ES	Water supply originated in PAs (%)	-	-	-	-
Region	Asia and Oceania	1.866** (0.223)	1.837** (0.223)	1.856** (0.226)	1.855** (0.224)
	Africa	0.967* (0.321)	0.867* (0.316)	0.966* (0.321)	0.935* (0.323)
	Europe	0.685* (0.267)	0.687* (0.268)	0.658* (0.282)	0.681* (0.271)
	North America	1.233** (0.304)	1.186** (0.303)	1.221** (0.306)	1.216** (0.305)
Intercept		-9.757** (1.797)	-9.451** (1.790)	-9.695** (1.810)	-9.648** (1.803)
R^2		0.478	0.476	0.478	
k		17	16	18	
AICc		3969.053	3969.936	3971.041	
Δ_i		0.000	0.882	1.988	
w_i		0.129	0.083	0.048	

* P<0.05, ** P<0.001
§ 10-km buffer zone
¶ Country level data, not PAs level
Values in parentheses are standard errors

2.3.2 Nature-based tourism was influenced by socioeconomic and environmental drivers

We find that agriculture surrounding PAs and water supply in PAs do not have a direct relationship with the annual visitor numbers in PAs at the p=0.05 level. Additionally, indirect associations of agriculture and water supply on visitor numbers via biodiversity are not significant (Table S2.4). Population density in 10-km buffer zones around PAs is positively associated with visitor numbers (P<0.001). We acknowledge that our cross-sectional data cannot disentangle causal direction: some people in the buffer zones may also visit the PA (the larger the population, the more visitors) but large numbers of visitors may also encourage local population growth. Per capita GDP has the strongest link to the number of visitors (P<0.001) presumably because in high-income countries there are more people who can afford nature-based tourism and because PAs in high-income nations may be more desirable destinations since there may be larger budgets for tourist infrastructures (e.g., visitor centers), all other things being equal.

The characteristics of PAs also influence the annual visitor numbers in PAs. The age and size of PAs positively affect the visitor numbers (P<0.001). Older PAs have had more time to gain recognition, often represent the most spectacular areas and may have been preserved in more pristine state than more recent PAs. In addition, PAs with larger sizes attract more nature-based tourists, presumably because large PAs have more natural attractions and habitats for species.

While the visitor numbers are positively associated with mean elevation, the visitor numbers are negatively associated with annual mean temperature and annual precipitation. This means that PAs with in a cooler temperature, lower precipitation and higher elevation have more visitors. People may visit PAs with high elevation areas to appreciate novel aesthetic views and natural habitats with high biodiversity because these PAs may avoid development pressures, maintain good natural habitat conditions and often have spectacular scenery. In addition, PA remoteness is negatively associated with the visitor numbers. PAs with good accessibility have more visitors. If PAs are located in the remote areas far from urban areas, people may not be able to afford the cost and/or time to visit the PAs even if the PAs provide good natural attractions.

All regional variables (Asia and Oceania, Africa, Europe and North America) have a significant p-value ($P<0.05$) when compared with Central and South America, the baseline continent. Net of the controls we have used, PAs in the other four continents have more visitors than those in Central and South America.

There are major variations in management goals in PAs, reflected in the IUCN categorization. We find that this categorization is capturing differences that are important in terms of the amount of biodiversity in a PA, with the PAs primarily managed for biodiversity having 1.05 times more species than the PAs managed with more mixed objectives (Table S2.4).

The nature of the buffer zone seems to have some correlation with number of visitors, with each 1% increase in population density associated with a 0.45% increase in visits. Agriculture in the buffer zone has no relationships with visitors to PAs. We tested the sensitivity of our analyses to the size of the buffer zones (Figure 2.2). Population and agricultural variables have the same pattern of effects when measured for larger buffer zones as they do in the 10-km buffer zone.

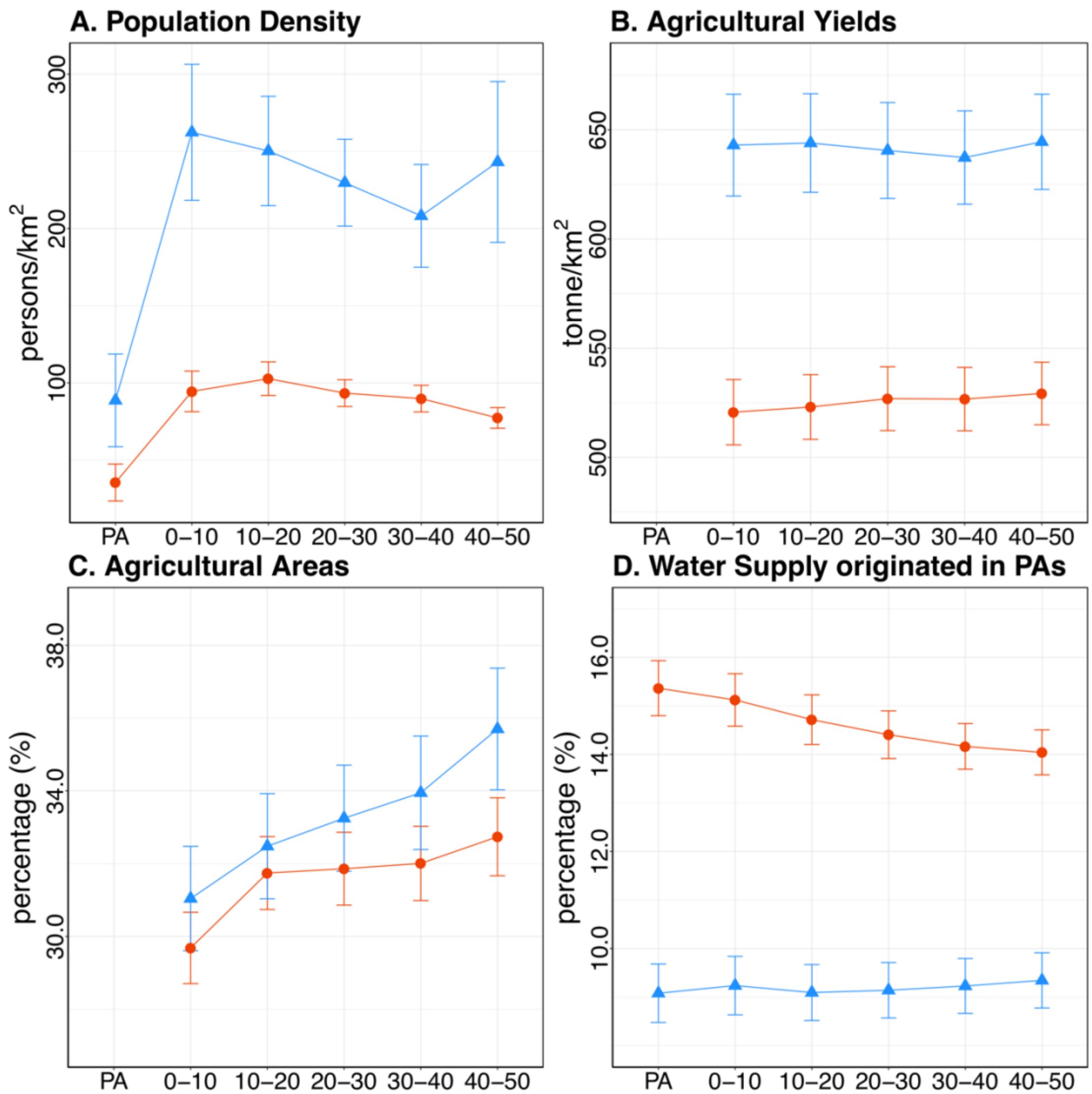

Figure 2.2. Buffer zone variables with 10-km distance increments across PAs boundaries. A. Population density, B. Agricultural yields, C. Agricultural areas, D. Water supply from upstream PAs

2.4. Discussion

2.4.1 The role of biodiversity in nature-based tourism

This study examines the relationships of biodiversity and other factors to nature-based tourism and the factors that are associated with biodiversity in PAs. The results demonstrate that biodiversity has a positive relationship with nature-based tourism even when a variety of other factors are controlled: with each 1% increase in biodiversity associates with a 0.87% increase in tourism. Furthermore, management strategies matter: PAs managed primarily for biodiversity protection have nearly 1.35 times the visits of those managed for mixed use. And management for biodiversity is associated with higher biodiversity, given the controls for other factors. Thus, we tentatively suggest that producing both biodiversity and nature-based tourism simultaneously is possible given appropriate conservation strategies. That is, biodiversity is compatible with economic development via tourism if proper strategies are deployed (Oldekop et al., 2016). More visitors can increase opportunities for local economic developments such as hotels, restaurants and employment opportunities for nature guides (Liu et al., 2012). Management plans that consider both biodiversity and local community participation could enhance economic development surrounding PAs and thus provide livelihood benefits to the local residents and reduce economic inequalities (Das and Chatterjee, 2015; Oldekop et al., 2016; Plummer and Fennell, 2009).

Because our data are cross-sectional, we cannot fully disentangle complex causal loops. Nevertheless, we feel our models capture the dominant interrelationships and lay the groundwork for further research. We have used an information theoretic approach to calculate the average of top models among the set of models. These models assume a linear in the logs functional form and specify no interactions of the form that allow effects to differ across subgroups in our data. But we note that results are fairly robust with regard to such specification errors—nearly half the cases would have to be invalidated to change our most important inferences and it seems unlikely that a missing specification that powerful has not been suggested in the literature. Of course, further work is required to overcome a lack of global biodiversity data. Although species richness is a crucial factor of nature-based tourism in PAs (Arbieu et al., 2017; Hausmann et al., 2017a; Siikamäki et al., 2015), the relationship of other aspects of biodiversity (e.g., evenness and abundance) to nature-based tourism in PAs warrants attention (Graves et al., 2017; Siikamäki et al., 2015).

Further research might fruitfully examine more complex causal feedbacks that we have been able to estimate. For example, it may be that higher biodiversity PAs are given more protective management strategies or that there is some feedback from high visitation rates to an emphasis on biodiversity protection policies. We also note that although we have used a well-accepted standard international classification of PA management strategies, we lack data that would allow for detailed comparisons of management strategies (e.g., targeted species, budgets for tourism). In particular, PAs in high-income countries may have better accessibility with larger budgets for tourist infrastructures (e.g., visitor centers, roads within PAs and campgrounds). The causal feedbacks can be complex. For instance, tourist infrastructure can increase the visitor numbers, but construction of tourist facilities, the footprint of the facilities and increased traffic can all be a threat to biodiversity (Daniel et al., 2012).

Several strategies would allow further research to expand on our analyses. There are ongoing efforts for improving global data sets by using social media (Hausmann et al., 2017b; Willemen et al., 2015) and developing global database of protected areas including visitor counts and biodiversity (Dubois et al., 2016; Schägner et al., 2017). These could all allow for more refined analyses. Data over time deployed as a panel would allow for strong causal inference. And detailed comparative case studies would allow a better understanding of how processes that link tourism, biodiversity and management strategy co-evolve.

2.4.2 Management implications

ES supply and demand change over temporal and spatial scales (Burkhard et al., 2014; Renard et al., 2015) and so do the interactions between biodiversity and nature-based tourism. Further, these changes are very context specific. It follows that effective plans for biodiversity protection would benefit from local community participation (Kovács et al., 2015; Liu et al., 2007; Pleasant et al., 2014). For example, with the rapid increases of human population and income in many parts of the globe, human demands for food have increased pressure on ecosystems including those in the buffer zone (Tilman and Clark, 2014). The increased human demands have caused unsustainable extraction of natural resources and biodiversity loss in many places (Liu et al., 2016b; Rands et al., 2010). We find PAs managed with mixed uses have higher agricultural yields in the buffer zones than those managed primarily for biodiversity conservation, while the proportion of agricultural areas in the buffer zones does not differ significantly across management strategies (Figure 2.2B and C). Population density surrounding PAs managed for mixed uses is also higher than those managed primarily for biodiversity conservation (Figure 2.2A). PAs managed primarily for biodiversity conservation have higher biodiversity and more water supply as well as lower anthropogenic pressures than those managed for mixed uses. Since anthropogenic pressures in the buffer zones mainly arise from the population density, land suitability for agriculture (e.g., slope, fertility and climate) and the demand for food production with urban development, these pressures could be reduced by more sustainable agricultural activities (Foley et al., 2011).

Because much of the demand for tourism comes from areas distant from PAs, applying integrated conceptual frameworks such as telecoupling (socioeconomic and environmental interactions over distances) can help develop a more holistic and refined analysis of changes in tourism supply and demand and their impacts on biodiversity over various temporal and spatial scales (Liu et al., 2013; Liu et al., 2016a). From the perspective of the telecoupling framework, nature-based tourism is a telecoupled system with complex interactions among local biodiversity, regional to global origins of nature-based tourism, international networks discussing and advocating management strategies for PAs and global changes in the supply and demand for ES (Liu et al., 2015a; Liu et al., 2015b). Disentangling these influences will require careful analysis of their dynamics over time. Here we have taken a first step by examining, in particular, how biodiversity, management strategy and the characteristics of the buffer zone surrounding a PA influences tourism and in turn how the buffer zone and management influence biodiversity.

CHAPTER 3

INTERNATIONAL TOURISM DYNAMICS IN A GLOBALIZED WORLD: A SOCIAL NETWORK ANALYSIS APPROACH

In collaboration with

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Abstract

A complex network of tourism has emerged in the globalized world, but there is little research on the dynamics of global tourism networks and the underlying forces that affect those dynamics. Using international tourism data for 124 countries between 2000 and 2013, we integrated cluster analyses and social network models to identify the structures of global tourism networks and uncover factors affecting changes in international tourist flows. Results indicate that global tourism networks have become highly consolidated over time and that reduced transaction costs (e.g., language, distance, and visa policies) are more important in attracting international tourists than natural and cultural attractions. Furthermore, cost of living differences between countries decreased in importance over time. Finally, international tourist flows are resilient to political instability and terrorism risks. Our approach and findings highlight the key strategic factors for decision-making to implement proactive tourism policies.

3.1. Introduction

Globally, tourism is booming, generating complex global networks with expanding economic power that consumes increasingly larger resources (Glaesser et al., 2017; Higham and Miller, 2018; Song et al., 2017). The globalization of tourism is increasing the interdependence between sending systems (supply areas, origins, departures) and receiving systems (demand areas, destinations, arrivals) worldwide, contributing to socioeconomic and environmental ties across regions (Dwyer, 2015; Glaesser et al., 2017; van der Zee and Vanneste, 2015; von Bergner and Lohmann, 2014). The proportion of the world economy occupied by tourism is rapidly increasing, accounting for approximately 10% of global GDP and employment in 2017 (Scott and Gössling, 2015; World Travel and Tourism Council, 2018). In addition, annual global tourism consumes approximately 16,700 PJ of energy, 138 km^3 of fresh water, 62,000 km^2 of land, 39.4 Mt of food, and leads to 4.5 Gt of CO_2 emissions (Gössling and Peeters, 2015; Lenzen et al., 2018). As tourism encourages extensive interactions between human and natural systems (Jones et al., 2016; Liu et al., 2015a), the tourism sector contains many opportunities to enhance global sustainability regarding job creation, economic growth, and environmental protection (Jones et al., 2016; Scheyvens, 2018; World Tourism Organization, 2018). These trends raise important questions about the impacts of the growing connectivity and interdependency of globalized tourism networks, yet research has not kept pace with these changes. A holistic conceptualization and quantification is therefore urgently needed.

In a globalized world, tourist flows fluctuate in response to a variety of socioeconomic and environmental factors across regions, which complicate tourism management by making supply and demand difficult to predict (Albrecht, 2013; Liu et al., 2015a; Song et al., 2017; van der Zee and Vanneste, 2015; von Bergner and Lohmann, 2014). Historically, international tourism mostly occurred between high-income countries, but in the mid-1990s international tourist arrivals increased rapidly in middle- and low-income countries (Scott and Gössling, 2015). Some of the tourism to middle and low income countries may have been nature-based and cultural tourism, but the effectiveness of conservation efforts (e.g., protected areas and World Heritage sites) in attracting more international tourists is uncertain (Cellini, 2011; Cuccia et al., 2016; Patuelli et al., 2013; Yang and Lin, 2011; Yang et al., 2010). There is also ongoing debate as to whether international tourism is resilient to political instability and terrorism risks (Liu and Pratt, 2017; Saha and Yap, 2013; van der Zee and Vanneste, 2015). Thus, tourism studies should explore the increased complexity of global tourism networks and how they respond to natural resources and social and political conditions.

Until now, quantitative research has been lacking to understand how the dynamics of global tourism networks have changed over time and how these networks affect, and are affected by, tourism supply and demand. Social network analysis is a sophisticated way to quantify the network structures of the tourism sector (Albrecht, 2013; Casanueva et al., 2016). Social network analysis also proves useful for uncovering the drivers of tourist flows in both sending and receiving systems (Albrecht, 2013; Merinero-Rodríguez and Pulido-Fernández, 2016). However, most tourism studies that use social network analysis concentrate on the structural characteristics of personal and organizational networks (e.g., density, centrality, and clusters) in the destinations (Casanueva et al., 2016; van der Zee and Vanneste, 2015). In addition, although many network models have been developed to estimate both network dependencies (e.g., reciprocation) and the drivers of network structures with statistical inference (e.g., standard errors, p-values, or posterior distributions) (Snijders, 2011), little tourism research applies network models to investigate the environmental and socioeconomic drivers of tourism.

To fill this research gap, we integrate a social network model with cluster analysis to uncover the network structure of international tourist flows and examine the factors influencing international tourism. Utilizing longitudinal data, the network model identifies the influence of environmental and socioeconomic factors on international tourism while accounting for statistical dependencies within global tourism networks. We answer two questions: (1) How has the network structure of international tourism changed over time?, and (2) Which factors contribute to increased international tourist flows over time? By establishing a theoretical foundation within a social network framework, we quantify the spatial and temporal changes of global tourism networks. On a practical front, measuring network dependencies and the factors involved in global tourism networks on both the supply and demand sides provides valuable insights for researchers, policymakers, and stakeholders implementing tourism development and destination management in a globalized world.

The next section begins with a literature review of social network analysis in tourism and factors that contribute to international tourism. The third section describes the data collection, processing, and network methods. The fourth section presents results from global-level network analyses. The paper concludes with a discussion of the theoretical and practical implications of employing these methods for future research and decision-making.

3.2. Literature Review

Our approach is based on an application of network science to describe international tourist flows as a network. This section is a narrative review that covers three topics: 1) the theoretical background of social network analysis in tourism, 2) the application of social network analysis to investigations of the dynamics of global tourism networks, and 3) the environmental and socioeconomic factors of international tourism used in this study.

3.2.1. Social network analysis

Social network analysis uses network and graph theory to investigate social structures (Baggio et al., 2010; Otte and Rousseau, 2002; Wasserman and Galaskiewicz, 1994). Social networks form a relational structure of ties (or edges) between actors (or nodes), such as friendships between individuals or trade between countries (Albrecht, 2013; Snijders, 2011). Similarly, international tourism forms a relational network by connecting the sending system (supply area, origin, departure) to attractions in the receiving system (demand area, destination, arrival) that is manifest in tourist flows (Albrecht, 2013; Sainaghi and Baggio, 2017).

The use of social network analysis to analyze tourism has grown rapidly over the last two decades (Baggio et al., 2010; Casanueva et al., 2016; Pulido-Fernández and Merinero-Rodríguez, 2018). Importantly, such approaches allow for the examination of both tourism supply perspectives (Pulido-Fernández and Merinero-Rodríguez, 2018; Sainaghi and Baggio, 2017) and tourism demand perspectives (Money, 2000; Tyler and Dinan, 2001). However, most tourism literature that uses social network analysis has focused on personal and organizational networks in tourism destinations (tourism supply-side) (Casanueva et al., 2016; van der Zee and Vanneste, 2015). For example, tourism studies have used social network analysis to investigate effects of collaborations among tourism stakeholders (Baggio, 2011; Pulido-Fernández and Merinero-Rodríguez, 2018), marketing (Bhat and Milne, 2008; Wang and Xiang, 2007), sustainable tourism (Albrecht, 2013), and geography (Jin et al., 2017; Lee et al., 2013) in tourism destinations.

Additionally, the most commonly used methods of social network analysis in tourism studies are concentrated on investigating static structural network properties (e.g., size, density, betweenness, and clusters) (Baggio et al., 2010; Benckendorff and Zehrer, 2013; Lee et al., 2013; Pulido-Fernández and Merinero-Rodríguez, 2018; Raisi et al., 2017; Scott et al., 2008b). Although tourism network properties may change significantly over time (Westveld and Hoff, 2011), few tourism studies have included any quantitative analysis of longitudinal datasets using a social network analysis approach (Baggio and Sainaghi, 2016; Jin et al., 2017). Recent exceptions include bibliometric network visualizations showing changes in tourism research output over time (Güzeller and Çeliker, 2018; Jiang et al., 2017; Li et al., 2017).

Social network analysis accounts for dependencies among ties between sets of actors (e.g., reciprocity and transitivity) (Snijders, 2011). For example, international tourism leads to dependence between sending and receiving countries if two countries have reciprocal tourism flows. Various statistical models have been developed to capture network dependencies between actors (Snijders, 2011). These statistical network models can estimate parameters to express network structures with statistical inference (e.g., standard errors, p-values, or posterior distributions).

The p_2 network model has been shown to yield a robust estimation procedure that accounts for network dependencies associated with common senders and receivers of network ties as well as potential reciprocal relationships between pairs of actors (Hoff, 2005; van Duijn et al., 2004). The p_2 model parameters are estimated with Bayesian inference based on a Markov Chain Monte Carlo (MCMC) algorithm (Hoff, 2005; van Duijn et al., 2004). Bayesian inference is a method for statistical inference used to compute the conditional probability of an event after taking into account new evidence or information that the event has occurred (Gamerman and

Lopes, 2006). The MCMC is a mathematical method for generating the probability distribution of a parameter by randomly sampling from a complex probabilistic space (Andrieu et al., 2003).

Social networks also contain temporal dependencies, wherein changes in network ties depend on the earlier structure of network ties (e.g., the evolution of international tourism networks) (Hoff, 2015; Snijders, 2011; Ward and Hoff, 2007). Longitudinal network data with regular temporal intervals are often referred to as network dynamics (Snijders, 2011). For longitudinal network data, statistical modeling approaches such as ordinary least squares and generalized linear models risk overestimating the significance of parameters by ignoring network and temporal dependencies with the assumption of independence (Westveld and Hoff, 2011). But Westveld and Hoff (2011) developed a mixed-effects model to account for both network and temporal dependencies as a stochastic process. The mixed-effects model extended the p_2 model of van Duijn et al. (2004) and Hoff (2005). This model (1) uses a latent space approach to produce visualizations of the network structure with the latent space positions, (2) develops a generalized linear modeling framework that allows for continuous data, and (3) outlines a general Bayesian estimation approach for model parameters with the MCMC algorithm (Westveld and Hoff, 2011).

Despite recent developments in social network models, these models have not been much used in tourism studies. With the social network model for longitudinal data, we provide a unique perspective on the dynamics of global tourism networks by quantifying both network and temporal dependencies. We also integrate social network modeling and cluster analysis to examine which environmental and socioeconomic factors influence changes in international tourist flows across countries. Thus, the application of social network models in tourism studies provides a better orientation to understand the processes of tourism development and destination management worldwide.

3.2.2. Hypothesized factors affecting international tourism

Following previous studies that investigated factors shaping tourism demand (Balmford et al., 2015; Lim, 1997, 1999; Marrocu and Paci, 2013; Peng et al., 2014; Song et al., 2012a; Song and Li, 2008; Witt and Witt, 1995), the most widely used factors affecting international tourism were considered for inclusion in the social network model regarding the characteristics of sending countries, receiving countries, and their pairs. These factors represent environmental, political, social, economic, and demographic features in both sending and receiving countries. We note that the factors used in tourism demand models may change extensively, depending on the research questions, time periods, methodologies, and selection of countries (Dogru et al., 2017). Based on the above literature review, we examine whether transaction costs (e.g., language, geographic distance, and visa policy) and demographic forces (e.g., population and income growth) are more important in attracting international tourists than natural and cultural attractions (e.g., protected areas and World Heritage sites) and political stability.

First, transaction costs of travel include visa-free status, national price-level difference, shared language, and proximity. International tourists prefer to travel to visa-free countries. Visa restrictions and requirements in destination countries can have a negative impact on the number of tourist arrivals (Balli et al., 2013; Cheng, 2012; Neumayer, 2010). Additionally, international tourists prefer to travel to countries that have advantageous prices relative to their home countries (Cheng, 2012; Dogru et al., 2017; Saha and Yap, 2013). There are two types of measurements for price level differences in the tourism demand model: 1) relative prices of the place of origin to the prices in the destination and 2) substitute prices of the destination to the prices in competing destinations (Dogru et al., 2017; Kronenberg et al., 2016). As a measurement of the price-level differences between countries, relative price standardized by exchange rates has been found to be more significant than the exchange rate alone (De Vita and Kyaw, 2013; Dogru et al., 2017). International tourists also prefer to travel to countries that use the same language as their home country. Thus, a shared language between sending and receiving countries plays an essential role in promoting tourist flows (Eilat and Einav, 2004; Khadaroo and Seetanah, 2008). Finally, international tourists prefer to travel to nearby countries. Greater distances between sending and receiving countries have a negative impact on international tourist flows (Lim, 1999; Patuelli et al., 2014; Yang et al., 2010). The number of direct flights between countries also contributes to increases in international tourist flows (Lohmann et al., 2009; Rehman Khan et al., 2017).

Second, demographic forces include population size and GDP per capita. Population and income per capita are important determinants for international tourist arrivals and departures. Tourism studies typically use real GDP per capita and population as proxies for relative income and market size (Lim, 1997; Peng et al., 2014; Witt and Witt, 1995). Higher per capita GDP in both sending and receiving countries positively affect international tourist flows (Lim, 1999; Saha and Yap, 2013; Song et al., 2010). International tourist flows also increase in sending and receiving countries with higher populations (Khadaroo and Seetanah, 2008; Llorca-Vivero, 2008; Yang et al., 2010).

Third, many tourism studies have investigated the role of conservation efforts (e.g., protected areas and World Heritage sites) for tourism demand (Song et al., 2012a). Larger protected areas have been found to attract more tourists (Balmford et al., 2015; Chung et al., 2018a). Protected areas are good at attracting nature-based tourists while conserving biodiversity (Balmford et al., 2015; Chung et al., 2015). Furthermore, nature-based tourism often contributes to the management and conservation of protected areas by providing financial resources (Buckley et al., 2015; Buckley et al., 2017). However, there is an ongoing debate regarding the effectiveness of World Heritage sites in promoting tourist arrivals (Cellini, 2011; Cuccia et al., 2016; Patuelli et al., 2013; Yang and Lin, 2011; Yang et al., 2010). While some studies show that the presence of World Heritage sites attracts more visitors due to proper management and accessibility (Richards, 2011; Su and Lin, 2014; Yang et al., 2010), others show that World Heritage sites do not affect the number of tourist arrivals (Cellini, 2011; Cuccia et al., 2016, 2017).

Fourth, empirical research lacks agreement regarding the effects of political instability and terrorism risks on both international tourist arrivals and departures (Liu and Pratt, 2017; Saha and Yap, 2013; van der Zee and Vanneste, 2015). Some studies have found that political instability and terrorism risks (e.g. public violence, riots, civil wars, and military coups) negatively influence international tourist arrivals (Eilat and Einav, 2004; Llorca-Vivero, 2008; Saha and Yap, 2013; Sönmez, 1998). But others have claimed that international tourists are resilient to political instability and terrorism risks (Liu and Pratt, 2017; van der Zee and Vanneste, 2015).

3.3. Materials and Methods

3.3.1. Data collection

Data on international tourist arrivals were obtained from the UN World Tourism Organization (UNWTO). This raw dataset covers over 200 countries from 1995–2013. UNWTO defines visitors to include both tourists (overnight visitors) and excursionists (same-day visitors) (World Tourism Organization, 2016a). Following UNWTO methods for estimating the number of international tourists, we excluded excursionists prior to selecting 124 countries over the period from 2000–2013 for analysis. The selected countries cover approximately 90% of international tourist arrivals in the specified time period.

Although the UNWTO data are the best available international tourist arrival datasets, the UNWTO dataset has some weaknesses inherent in how different jurisdictions collect visitor arrival data (World Tourism Organization, 2016a). When countries did not report international tourist arrivals at national borders (referred to as TF), we supplemented by using other datasets following UNWTO methods: international visitor arrivals at national borders (VF), international tourist arrivals at hotels and similar establishments (THS), or international tourist arrivals at collective tourism establishments (TCE) (World Tourism Organization, 2016b).

To test our hypotheses, we collected data regarding possible factors influencing international tourism: transaction costs of travel, environmental, political, and demographic factors. Transaction costs of travel included visa requirements for tourism, price level ratio to the market exchange rate, shared language, and geographic distances between sending and receiving countries. At the level of the pair of countries, the visa-free score is 1 if a receiving country waives visa requirements for a sending country, including both visa-free and visa-on-arrival entry (https://www.passportindex.org). The price level ratio measures the amount of a country's currency that is required to purchase a dollar's worth of goods relative to the United States (= 1) (The World Bank, 2017). The price-level differences between countries were calculated by subtracting the price level ratio of each receiving country from each sending country. Countries having a shared language was also included, where if two countries share an official language (e.g., Canada and the United Kingdom), their language factor was 1. Geographic distances between the centroids of pairs of countries were calculated using GeoDa (Anselin et al., 2006) and remained constant over the study period. The number of direct flights between countries was obtained from Openflights (https://openflights.org).

Environmental factors included the size of protected areas in receiving countries (IUCN and UNEP-WCMC, 2017), restricted to protected areas that are legally and officially designated at the national or sub-national level. Marine protected areas were excluded as well as the International Union for Conservation of Nature (IUCN) Category I protected areas, where tourism is prevented for strict conservation. Additionally, World Cultural Heritage sites were included as an environmental factor (UNESCO, 2017). World Natural Heritage sites were excluded to avoid double counting a site. Protected areas and World Cultural Heritage sites were used to represent a country's cultural ecosystem services (Balmford et al., 2015; Chung et al., 2018b; Yang et al., 2010).

Political factors included the index of political stability and the absence of violence and terrorism (The World Bank, 2017). The index of political stability and the absence of violence and terrorism measures the likelihood of political instability and politically motivated violence, ranging from -2.5 to 2.5 (The World Bank, 2017). In both sending and receiving countries, population size was a demographic factor (The World Bank, 2017). Per capita GDP was included as an additional economic factor (The World Bank, 2017).

3.3.2. Cluster analyses

We used Kliquefinder software to identify clusters of countries within global tourism networks (Frank, 1995, 1996). The raw data for this analysis consist of the total tourist flows between each pair of countries over a given interval. The algorithm maximizes the odds ratio of flows within clusters relative to between clusters by switching actors among clusters repeatedly. Because countries in the same cluster have a higher probability of sending tourists to each other than countries in different clusters, the Kliquefinder algorithm can identify clusters of countries that can then be investigated to see if they are focused around income level or other factors such as population or geographic location. To test the statistical significance of the clustering, Kliquefinder is applied to a random redistribution of flows. This is repeated 1,000 times, and the measure of fit is noted to generate a Monte Carlo sampling distribution under the null hypothesis of no clustering in the data (data are generated at random). The observed measure of fit is then compared to the Monte Carlo generated sampling distribution to obtain a p-value.

To perform the cluster analysis, we examined the number of international tourists in two different ways: by analyzing the average of the data from the first three years (2000–2002) and the last three years (2011–2013) in the UNWTO datasets, and by analyzing each year (from 2000–2013) separately. Although some significant political, social, and natural events occurred during the study period (e.g. the events of 9/11, the Indian Ocean tsunami, global financial crisis, the Arab Spring, and Olympics events), we consider our analyses to be valid because there are rarely multi-year periods in which a significant political, social, or natural event does not occur somewhere in the world. Furthermore, we believe that the use of both 3-year averages and single-year data accounts for such occurrences. We performed cluster analyses with Kliquefinder for each of the temporal periods, and tested for evidence of clusters in each period.

For the results from the analyses of both the 3-year periods and each individual year, we used the igraph package in R to visualize the cluster results. In the graphs, we used the number of international tourists to identify the core and peripheral countries in global tourism networks based on the k-core decomposition approach, an iterative approach that determines the most central nodes by consecutively cutting out the least connected nodes in a given network (Barberá et al., 2015). We also presented the cluster results by country on a global map using ArcGIS (ESRI, 2015).

3.3.3. Mixed-effects model

In addition to the cluster analyses, we used a mixed-effects (including random- and fixed-effects) model for longitudinal tourism network data with the number of international tourist arrivals from a sending country to a receiving country as the dependent variable. The mixed-effects model was developed by Westveld and Hoff (2011) to account for both network and temporal dependencies. Westveld and Hoff (2011) provided R code script that we deployed using the MCMCpack package in R. The results provide means and regression estimates of the factors affecting global tourism networks, as well as evidence of statistical dependencies. By using a generalized linear model framework, this model can adopt the gravity approach described in the next paragraph, which models the set of bilateral tourist flows (Khadaroo and Seetanah, 2008; Morley et al., 2014; Westveld and Hoff, 2011; Yang et al., 2010).

Since tourism is a type of trade in services, tourist flows can also be analyzed using the gravity approach for bilateral trade (Cheng, 2012; Eilat and Einav, 2004; Kimura and Lee, 2006; Morley et al., 2014). The gravity model has been widely applied on both the tourism supply and demand sides over the last decade (Marrocu and Paci, 2013; Morley et al., 2014). The gravity model of international trade can be derived from the Heckscher-Ohlin theory based on international differences in factor endowments (Deardorff, 2007). Furthermore, Morley et al.

(2014) derived a theoretical framework to support the gravity model for bilateral tourist flows by using consumer utility theory.

The gravity model assumes that international tourist flows between sending and receiving countries increase with a country's size (e.g., population and income) and decrease with transportation costs between countries (e.g., distance) (Eilat and Einav, 2004; Khadaroo and Seetanah, 2008; Witt and Witt, 1995). Some studies also include some dummy variables (e.g., visa requirements or shared language) in addition to the gravity model (Eilat and Einav, 2004; Neumayer, 2010), an approach we followed in our study. The basic gravity model for bilateral trade is shown as:

$$T_{ij} = B\,\frac{m_i^{\beta_1} m_j^{\beta_2}}{d_{ij}^{\beta_3}} E_{ij} \qquad (1)$$

where T_{ij} is the amount of trade flows between two regions i and j; m_i and m_j are the characteristics of each region's size (e.g., national income or population); d_{ij} is the geographical distance between region i and region j; E_{ij} is a normal distributed error term; and B, β_1, β_2, and β_3 are coefficients to be estimated. By taking the natural log transformation in equation (1), the basic gravity equation for estimation purposes can be expressed as follows:

$$\ln T_{ij} = \beta + \beta_1 \ln m_i + \beta_2 \ln m_j + \beta_3 \ln d_{ij} + \varepsilon_{ij} \qquad (2)$$

where ε_{ij} is a residual error term.

We applied this gravity model to annual international tourist arrivals between 124 countries for each year from 2000–2013. The model for longitudinal tourist flows is:

$$\ln \# \; of \; International \; Tourists_{i,j,t} = \beta_{0,t} + \beta_{1,t} \ln PA_{j,t} + \beta_{2,t} \ln World \; Heritage_{j,t} + \beta_{3,t} Political \; Stability_{i,t} + \beta_{4,t} Political \; Stability_{j,t} + \beta_{5,t} Visa \; Free_{i,j} + \beta_{6,t} Language_{i,j} + \beta_{7,t} \ln D_{i,j,t} + \beta_{8,t} \ln Price \; Level_{i,j,t} + \beta_{9,t} \ln GDP_{i,t} + \beta_{10,t} \ln GDP_{j,t} + \beta_{11,t} \ln Pop_{i,t} + \beta_{12,t} \ln Pop_{j,t} + s_{i,t} + r_{j,t} + g_{i,j,t} \quad (3)$$

where International $\text{Tourists}_{i,j,t}$ is the number of international tourist arrivals from sending country *i* to receiving country *j* at time *t*; $PA_{j,t}$ is the size of protected areas in the receiving country at time t; World $\text{Heritage}_{j,t}$ is the size of World Cultural Heritage sites in the receiving country at time t; Political $\text{Stability}_{i,t}$ and Political $\text{Stability}_{j,t}$ are the political stability and absence of violence and terrorism indices for the sending and receiving countries at time *t*, respectively; Visa $\text{Free}_{i,j}$ is the visa-free score between the sending and receiving country; $\text{Language}_{i,j}$ is shared language factor between the sending and receiving countries; $D_{i,j,t}$ is the geographic distance from the centroid of country *i* to the centroid of country *j*; Price $\text{Level}_{i,j,t}$ is the national price level difference between sending and receiving countries; $GDP_{i,t}$ and $GDP_{j,t}$ are the per capital GDP in the sending and receiving countries at time *t*, respectively; $Pop_{i,t}$ and $Pop_{j,t}$ denote the population size in the sending and receiving countries at time *t*, respectively; $s_{i,t}$ is a sender effect; $r_{j,t}$ is a receiver effect; and $g_{i,j,t}$ is a residual error term. The sender ($s_{i,t}$) and receiver ($r_{j,t}$) random effects measure the average deviations of the levels of tourist arrivals and departures in each country. With these effects, we can identify which countries are the most or least active in global tourism networks. In international tourism, International $\text{Tourists}_{i,j,t}$ is the directed flow from sending country *i* to receiving country *j* at time *t;* thus International $\text{Tourists}_{i,j,t}$ is not equal to International $\text{Tourists}_{j,i,t}$.

For the sake of clarity, we also used an alternative model, which compared the proportion of protected areas and World Cultural Heritage sites to the total land area of a country instead of compared to the absolute size of protected areas and World Cultural Heritage sites. This alternative model also included the number of direct flights between countries instead of the geographic distance.

To estimate both models, an MCMC algorithm iterated 11,000 times, and we dropped the first 1,000 iterations to allow convergence to the stationary distribution. Our model parameters were automatically saved every 10th scan. Then, we calculated means and 95% confidence regions of the parameters using the joint posterior distribution. For 95% confidence regions, we used Highest Posterior Density (HPD) interval.

3.4. Results

This section presents the results of our global-level network analyses in two parts: cluster analyses and the social network model. The first part of the analyses began by examining the network structure of international tourism in the two temporal periods (2000–2002 and 2011–2013) and in each year (from 2000–2013) individually. To test the sensitivity of our cluster results to the choice of the temporal periods, we examined the network structure of international tourism in each year from 2000–2013. The second part of the analyses determined which factors contributed to changes in international tourist flows over time and quantified network and temporal dependencies in global tourism networks.

3.4.1. Consolidated global tourism networks

While global tourism networks from 2000–2002 were divided into eight clusters (Figure 3.1A), the network structure from 2011–2013 had only two clusters (Figure 3.1B). Figure 3.1 also identified the core and peripheral countries in global tourism networks. The core countries (e.g., USA and western European countries) located in the center played active roles in both tourist arrivals and departures.

At the first time point (2000–2002), the largest cluster included 54 countries highlighted by yellow circles in Figure 3.1A. All high-income countries were located in this group. The remaining seven clusters included middle- and low-income countries, grouped by geographic locations (the Caribbean Sea, central and southern America, southern Africa, eastern and western Africa, central Asia, southern Asia, and eastern Europe) (Figure S3.1A). The dominant cluster sent a large number of tourists to countries within the same cluster (red lines in Figure 3.1A) and to the other seven clusters (gray lines in Figure 3.1A). Interestingly, over the period of 2011–2013, the dominant cluster expanded to include 121 countries. The consolidated cluster contained all countries in our dataset, excepting only Burkina Faso, Niger, and Togo in western Africa (Figure S3.1B).

The cluster results for each individual year from 2000–2013 also indicated the same pattern—that global tourism networks have become consolidated over time (Figure S3.1). Specifically, the number of clusters in 2009 was highest (12 clusters) over the 14-year period, followed by 2004 (11 clusters). These clusters were mainly based on geographic location (Figure S3.1). After 2009, the number of clusters decreased, from nine in 2010 to two in 2012 (Table S3.1). Informed by the Monte Carlo sampling distribution, we confirmed the existence of clusters in global tourism networks in each time period (Table 3.1 and Table S3.1, $P<0.001$).

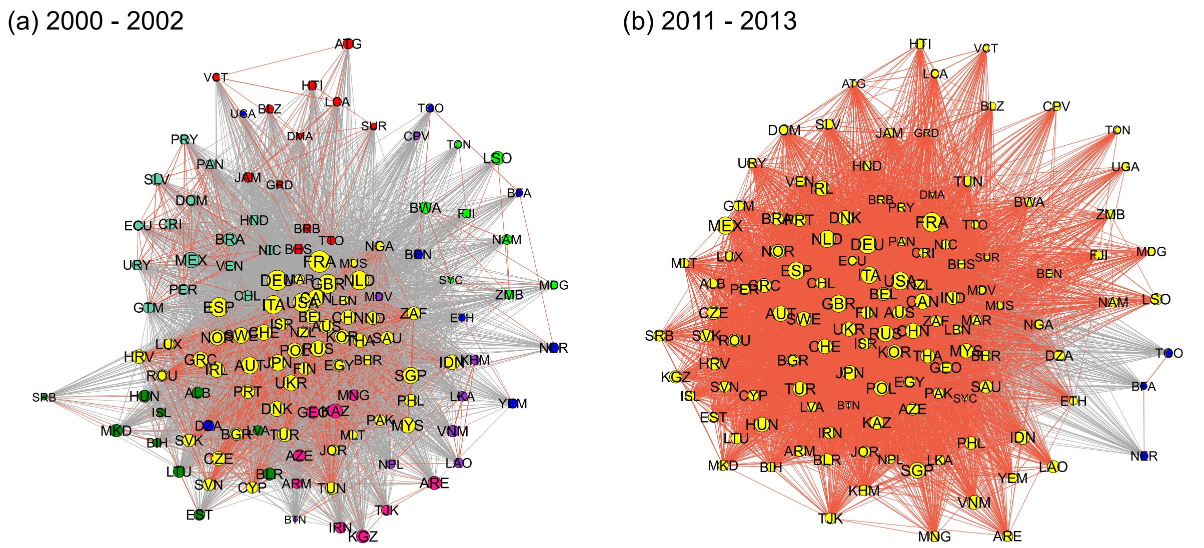

Figure 3.1. Clusters of global tourism networks in (a) 2000–2002 and (b) 2011–2013. The size of each node indicates the sum of international tourist arrivals and departures. Red ties indicate tourist flows within the same cluster, and gray ties indicate tourist flows between different clusters. The countries' locations in the cluster map represented the strength of the interactions between countries based on the number of international tourists. The core countries were located in the center of the cluster maps.

Table 3.1. Odds ratios for cluster analysis and p-value based on simulations followed by mean, median, and 95% Quantile interval of simulations.

	N	Odds ratio	p-value	Mean	median	2.5%	97.5%
2000–2002	8	0.792	<0.001	0.596	0.604	0.365	0.636
2011–2013	2	0.828	<0.001	0.561	0.563	0.532	0.591

3.4.2. Factors related to international tourism

By using a mixed-effects model, we were able to estimate the effect of each independent variable on international tourist arrivals, as well as of network and temporal dependencies within global tourism networks. Figure 3.2 shows the mean for each coefficient and its 95% HPD confidence intervals from 2000–2013. Regarding receiving countries, the size of protected areas and World Cultural Heritage sites did not have a significant relationship with international tourist flows. From 2000–2013, the coefficients for protected areas and World Cultural Heritage sites changed little and their confidence intervals contained zero (Figures 3.2A and 2B). In the alternative model, the proportions of protected areas and World Cultural Heritage sites to the total land area were also not statistically significant (their confidence intervals contained 0) between 2000 and 2013 (Figure S3.2).

Regarding sending countries, the coefficients for political stability and absence of violence and terrorism did not shift, and their intervals consistently contained zero (Figure 3.2C). However, with respect to receiving countries, the coefficients of political stability and absence of violence and terrorism declined from 2000–2011 and then shifted upward from 2011–2013 (Figure 3.2D).

Third, the coefficients for visa-free score and shared language were positive over the entire study period (Figures 3.2E and 2F). There was an increase in the coefficients for visa-free score from 2000–2013.

Fourth, international tourists prefer to travel to nearby countries. Geographic distance between sending and receiving countries was negatively associated with the number of international tourists from 2000–2013 (Figure 3.2G). In the alternative model, the number of direct flights between sending and receiving countries also was positively associated with the number of international tourists over time (Figure S3.2).

Fifth, the coefficients for price level difference between sending and receiving countries declined over the study period (Figure 3.2H). The confidence intervals were positive from 2000–2009 but contained zero from 2010–2013.

Sixth, in sending and receiving countries, higher income levels increase the number of both international tourist arrivals and departures. The coefficients for per capita GDP in sending countries increased over time (Figure 3.2I). In receiving countries, the confidence intervals for per capita GDP shifted upward (Figure 3.2J).

Finally, in both sending and receiving countries, population size was positively associated with the number of international tourists. Over the study period, all population coefficients were positive, and their intervals were consistently above zero (Figures 3.2K and L). This trend suggests that international tourism between countries with high per capita GDP and rapid population growth was above the global average. In receiving countries, the inferences we would make regarding per capita GDP and population size were more uncertain than for those in sending countries because of the larger confidence intervals over time.

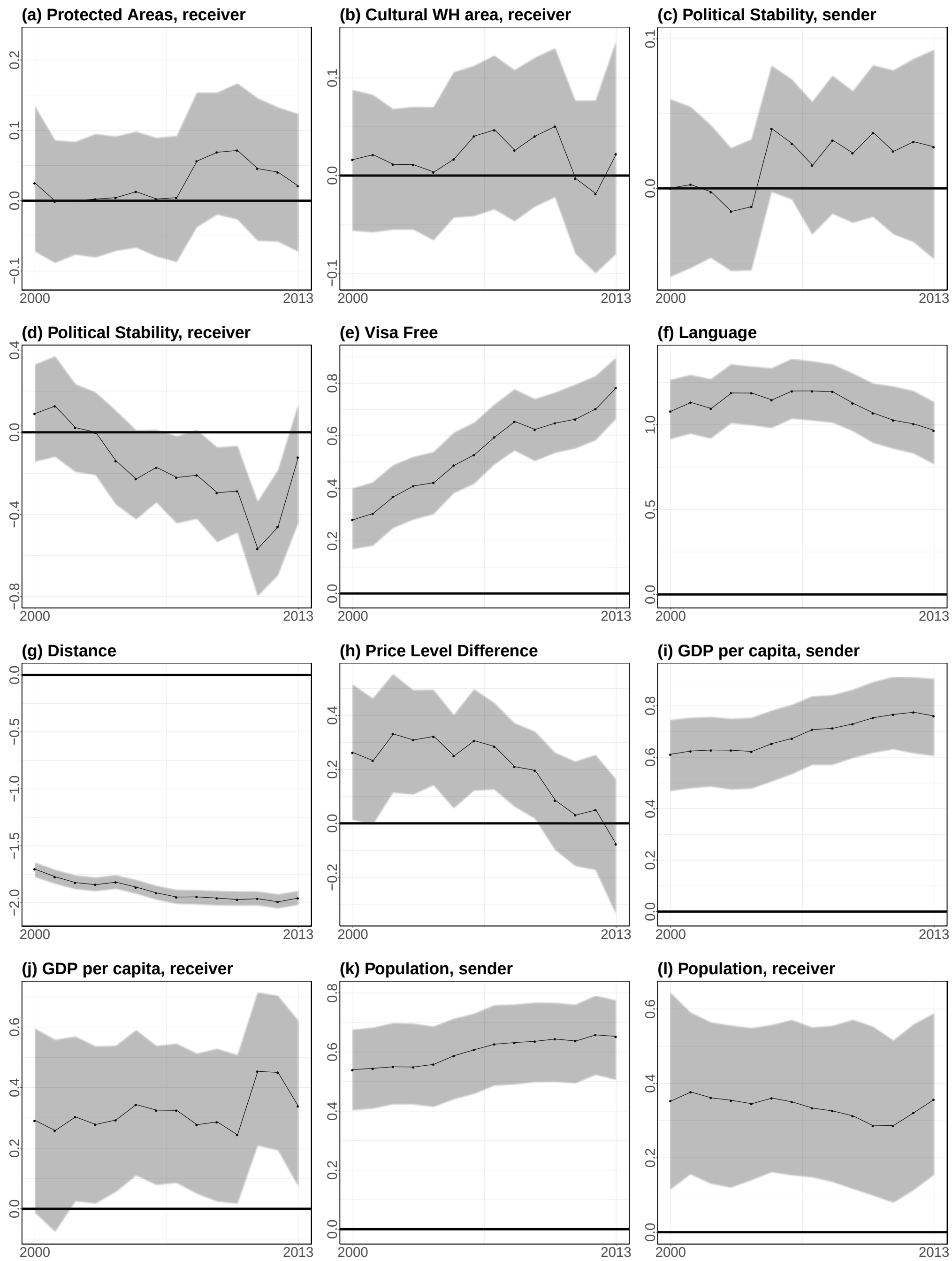

Figure 3.2. Mean and 95% Highest Posterior Density (HPD) confidence intervals of the coefficients from 2000–2013: (a) the size of protected areas in receiving countries (km^2), (b) the size of World Cultural Heritage sites in receiving countries (km^2), (c) political stability in sending countries (index), (d) political stability in receiving countries (index), (e) visa-free status between sending and receiving countries (visa-free=1), (f) shared language between sending and receiving countries (shared language=1), (g) distances between countries (km), (h) national price level difference between sending and receiving countries (price-level ratio), (i) per capita GDP in sending countries (constant 2010 US $), (j) per capita GDP in receiving countries (constant 2010 US $), (k) population size of sending countries (person), and (l) population size of receiving countries (person).

Phi parameter estimates identified the auto-regressive effect of the previous year on tourist arrivals, departures, and reciprocity of the current year (Table 3.2). The medians of the posterior distribution of Φ_s and Φ_{sr} were 0.998 and 0.003. This means that the number of international tourist departures in the current year highly depended on the level of international tourist departures from the previous year. Yet international tourist arrivals in the previous year did not have an impact on the current international tourist departures. In addition, the medians of Φ_r and Φ_{rs} are 0.967 and 0.004, respectively. When countries had a high number of international tourist arrivals in the previous year, they also tended to have large international tourist arrivals in the current year. However, the number of international tourist arrivals in the current year did not depend on the number of international tourist departures in the previous year. Finally, the median of Φ_{gg} was 0.014. This indicates that the level of reciprocity in the previous year may not explain the level of reciprocity in the current year.

Table 3.2. Phi parameter estimates with median and 95% Quantile intervals.

Parameter	Median	2.5%	97.5%
Φ_s	0.998	0.996	0.999
Φ_{sr}	0.003	0.000	0.005
Φ_{rs}	0.004	-0.009	0.018
Φ_r	0.967	0.958	0.975
Φ_{gg}	0.014	0.013	0.016

In 2000–2002 and 2011–2013, sender and receiver random effects were investigated at the country level (Figure 3.3). The random effects estimated the deviations of the number of international tourist arrivals from the predicted values by the mixed-effects model. The positions of the countries changed slightly from 2000–2002 to 2011–2013. USA, Canada, and Australia played crucial roles as both senders and receivers in global tourism networks, even after accounting for controls in the regression model. From 2000–2013, China, Spain, and Russia became active tourists-senders while South Africa, India, Malaysia, and Maldives became active tourists-receivers. Over the period of 2011–2013, China and Russia emerged as both important senders and receivers in global tourism networks.

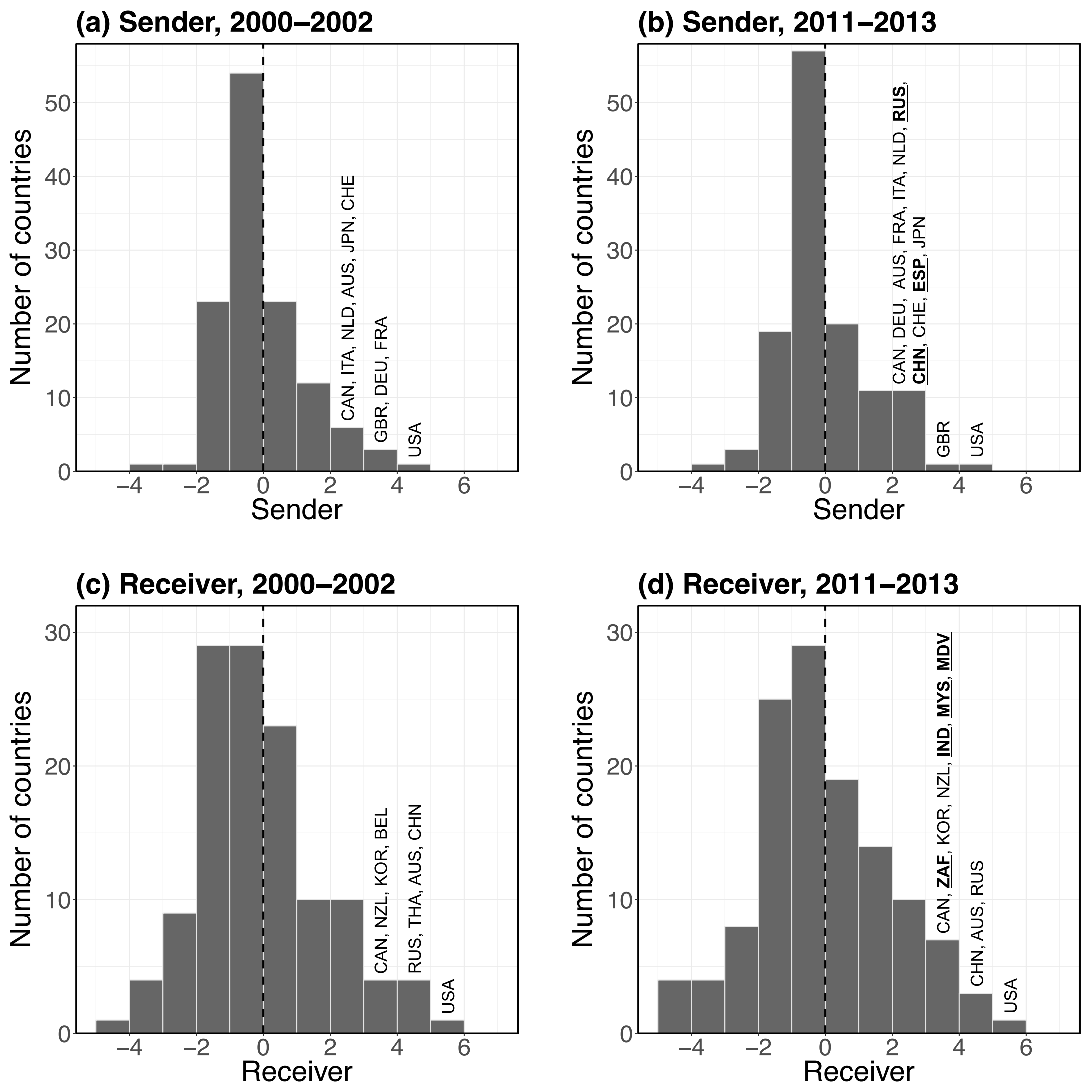

Figure 3.3. Distributions of (a) the sender effects in 2000–2002, (b) the sender effects in 2011–2013, (c) the receiver effects in 2000–2002, (d) the receiver effects in 2011–2013. Country abbreviations: Australia (AUS), Belgium (BEL), Canada (CAN), Switzerland (CHE), China (CHN), Germany (DEU), Spain (ESP), France (FRA), United Kingdom (GBR), India (IND), Italy (ITA), Japan (JPN), Republic of Korea (KOR), Maldives (MDV), Malaysia (MYS), Netherlands (NLD), New Zealand (NZL), Russian Federation (RUS), Thailand (THA), United States (USA), and South Africa (ZAF).

3.5. Discussion

3.5.1. Reasons behind consolidated global tourism networks

Using cluster analysis and a mixed-effects model for longitudinal network data, we investigated the flows and factors relating to international tourism. Social network analysis helped examine how international tourism connects regions and identify temporal changes in the network structure. Results of our cluster analysis show that international tourist flows form a consolidated network over time (Figure 3.1). Sender and receiver random effects from the mixed-effects model then revealed which countries played increasingly active roles in the consolidated networks (Figure 3.3).

Another finding of the mixed-effects model may indicate a causal relationship between the changes in global tourism networks in Figure 3.1 and Figure S3.1 and the factors in Figure 3.2. From 2000–2009, the price level difference between sending and receiving countries was a major factor of international tourist flows based on the law of demand. This result is consistent with previous studies (De Vita and Kyaw, 2013; Dogru et al., 2017). However, after 2010, the price level difference became a less important factor for international tourism. This result shows that middle- and low-income countries with rapid income and population growth, such as China, increasingly play an important role as sending countries (see also Buckley et al. (2015); Scott and Gössling (2015)). Despite the price level differences, developing countries send more tourists to both developed and developing countries.

In sending countries, per capita GDP and population size were the most significant factors for international tourism (Song et al., 2010; Song and Li, 2008; Yang et al., 2010). Per capita GDP and population size represent the effects of income level and market size differences between sending and receiving countries. In the consolidated networks, the roles of these factors in sending countries intensify over time (Lim, 1997; Peng et al., 2014; Witt and Witt, 1995). In receiving countries, although per capita GDP and population size are significant (Khadaroo and Seetanah, 2008; Saha and Yap, 2013), the uncertainty of the effects of these factors is high (i.e., large confidential intervals). While over half of international tourists visit high-income countries, increasing arrivals in new destinations such as Malaysia, a middle- to low-income country with a large population, led to the uncertainty of coefficients.

International tourist flows are complex and dynamic systems affected by many other factors that were not measured in our study. For example, global crisis events such as economic and financial downturns, political instability, terrorist attacks, and natural disasters can affect the size and frequency of international tourist flows (Hall, 2010). Our results may indicate that global crisis events have dispersed the consolidated global tourism networks, based on geographic locations (Figure S3.1). The global financial crisis from 2007–2010 may have caused the rapid increase in the number of clusters by weakening the interdependence between distant countries (see also Campos-Soria et al. (2015); Hall (2010)). In 2004, global tourism networks were separated into 11 clusters, in part because of outbreaks of severe acute respiratory syndrome (SARS) and the Indian Ocean tsunami (see also Hall (2010); Kuo et al. (2008)). These types of global events may also contribute to the uncertainty of some coefficients (e.g., political stability variable) in the mixed-effects model.

3.5.2. The role of conservation in international tourism

Although nature-based and cultural tourism are the fastest growing sectors in the tourism industry (Newsome et al., 2012; World Tourism Organization, 2015), the presented results show that efforts to conserve natural and cultural sites were not significant factors contributing to the number of international arrivals in receiving countries. The results from the mixed-effects model for the proportions of protected areas and World Cultural Heritage sites show that neither were significant factors between 2000 and 2013.

Within a given country, protected areas have varying success in attracting international tourists from different regions and over time (Diefendorf et al., 2012; Su and Lin, 2014). Some protected areas have higher levels of domestic tourist arrivals than international tourist arrivals (Chung et al., 2018b), whereas other protected areas attract more international tourists than domestic tourists (Baral et al., 2017). These varying patterns of international tourist arrivals may have led to an insignificant result in the mixed-effects model. In addition, many protected areas are located at high altitudes, far from the major urban areas from which most international tourism emanates (Chung et al., 2018a; Joppa and Pfaff, 2009). The remoteness of protected areas may prevent visits from international tourists (Chung et al., 2018a). Due to different numbers of international tourist arrivals, decision-makers may need to establish different management plans to increase tourism while protecting the environment effectively. For example, protected areas that successfully attract domestic tourists may lack the transportation infrastructure for international tourists. If decision-makers aim to increase international tourism, such protected areas will need additional infrastructure investment to increase accessibility from airports or train stations. However, further infrastructure development could have a negative environmental impact, and therefore should be considered as a part of management and conservation strategies.

Furthermore, World Cultural Heritage sites were not effective in attracting international tourists in accordance with the findings of Cellini (2011), Cuccia et al. (2016), and Cuccia et al. (2017). This is consistent with the main purpose of World Cultural Heritage sites, which is not to encourage tourism flows, but to "raise awareness" and "mobilize sustainable resources for long-term conservation" (Cellini, 2011; Cuccia et al., 2016; Su and Lin, 2014). In addition, the increase in international tourist arrivals in middle- and low-income countries that have few World Cultural Heritage sites may reduce the attraction of World Cultural Heritage sites for international tourists because over half of World Cultural Heritage sites are based in high-income European countries (Su and Lin, 2014). Although World Cultural Heritage sites are ineffective for international tourism, there are ongoing efforts to encourage cultural tourism to World Cultural Heritage sites. In the rapidly globalizing tourism network, one of the major challenges at World Cultural Heritage sites is how to encourage cooperation between the tourism and culture sectors. In 2015, UNWTO and UNESCO organized the first World Conference on Tourism and Culture to initiate the sustainable development of cultural tourism (World Tourism Organization, 2016c).

3.5.3. The impact of policies on international tourism

Visa-free policies can stimulate flows of international tourists. Between 1980 and 2015, visa openness in middle- and low-income countries increased, with fewer travel requirements than those of high-income countries (World Tourism Organization, 2016d). The increase in visa openness in middle- and low-income countries may attract more international tourists. Visa-free policies can also support sustainable economic growth because improving visa openness can contribute to an increase of tourism expenditures and create jobs without additional tourism development (Song et al., 2012b; World Tourism Organization, 2016d). To maximize the effects of visa openness, receiving countries need to prioritize relaxing their visa policies for citizens of sending countries with shared languages and short travel distances.

Further, international tourists are resilient to political instability and terrorism risks in both sending and receiving countries. This result is consistent with Liu and Pratt (2017) and van der Zee and Vanneste (2015). After 2007, international tourist arrivals in receiving countries show a complicated relationship with political instability and terrorism risks. From 2007–2011, international tourist arrivals were negatively associated with political stability and the absence of violence and terrorism index. Over the study period, European countries led this trend, as these European countries decreased in political stability and increased in violence and terrorism risks driven by the global financial crisis following the economic recession (Campos-Soria et al., 2015; The World Bank, 2017). The effect was a slight decrease in international tourist arrivals in European countries.

International tourist arrivals in high-income countries may be more resilient to political instability and terrorism risks than those of middle- and low-income countries (Liu and Pratt, 2017; Llorca-Vivero, 2008). In middle- and low-income countries, political instability and terrorism risks can lead to significant decreases in international tourism due to riots and wars (Sönmez, 1998). For example, in 2011, political changes in Middle Eastern and North African countries such as Egypt and Yemen led to decreases international tourist arrivals (Avraham, 2015). As a result, the Arab Spring contributed to the uncertainty of coefficients of political stability and absence of violence and terrorism index. The emergence of the Islamic State in Iraq and Syria (ISIS) and Syrian refugee crisis generated terrorism risks and political tensions in both the Middle East and the rest of the world (Khan and Ruiz Estrada, 2016). Countries that experience such events can have difficulties in tourism management and planning with unpredictable tourism demand (Issa and Altinay, 2006; Saha and Yap, 2013). Therefore, tourism policy makers should recognize the impacts of political instability and terrorism risks while planning crisis management strategies for the tourism industry (e.g., restoration of a positive image for international tourists) (Khan and Ruiz Estrada, 2016; Saha and Yap, 2013).

3.6. Conclusions

Our study is the first international tourism study to adopt a social network analysis approach that quantifies the complex structure of global tourism networks and examines underlying factors over time. The results of our global-level network analyses have several theoretical and practical implications, including identifying emerging countries that need tourism policies and providing key strategic factors for tourism development and destination management in each phase of global tourism networks. From a theoretical point of view, our global-level network analyses made a significant contribution to advancing the application of social network analysis approach in the tourism field since to date, a limited number of tourism studies have utilized a social network approach to perform a longitudinal quantitative study at a global level.

In drawing conclusions, we should also note the limitations of our study. The most compelling limitation regards the lack of data availability at the global level. For instance, due to the lack of time-series data for the visa-free score and for the number of direct flights between countries, we assumed the same visa policy and the number of direct flights over the period from 2000–2013. Additionally, although our cluster results may indicate that global tourism networks were dispersed following global crisis events (e.g., global financial crisis), we could not detect a causal relationship between global crisis events and changes of network structure in international tourism. Second, it is noted that when using longitudinal network data, it is difficult to discern the most important factors because the pattern of each factor is based on variation among years within a country and/or variation among countries. Third, we identified a few countries that were not predicted from the mixed-effects model. For example, although Australia has large geographic separation from other countries, Australia is the center of global tourism networks. This is because international tourism supply and demand have been influenced by many other factors across local, regional, and global levels. At the local and regional levels, different key factors for international tourism may require strategies different from our global implications, and therefore destination management should be flexible across regions. Future tourism network research will need to extend our methods to include hierarchical network models and examine hierarchical network structures from global to local levels. Furthermore, future tourism research should evaluate socioeconomic and environmental effects of international tourism as well as the agents that are involved in international tourism, in addition to the tourist flows and factors affecting tourism (causes) reported in this study. The new integrated framework of metacoupling (socioeconomic-environmental interactions within and between adjacent and distant systems such as countries) provides a good foundation for such future efforts as it integrates tourist flows, causes, agents, and effects across different systems (Liu, 2017).

Despite these limitations, on the practical front, quantifying the network structure of international tourism helps explore how international tourist flows are changing in the face of external social, economic, political, and environmental issues. Our cluster results confirm the consolidation of global tourism networks and identify which countries increasingly contribute to this trend over the past 14 years. Our results support that some global crisis events (e.g., global financial crisis and the Indian Ocean tsunami) may weaken the structure of international tourist flows from consolidated networks to separated networks based on geographic location. This result indicates that social, economic, political, and environmental changes in emerging countries may have more significant impacts on other countries in the same cluster than those in other clusters. Policy makers can utilize the results of our cluster analysis to understand the cross-border impacts of tourism development and destination management to attract more international tourists across countries.

Our mixed-effects model identifies key strategic factors for proper tourism development and destination management. In consolidated global tourism networks, results indicate that transaction costs (e.g., shared language, geographic distance, and visa policy) are more important in attracting international tourists than natural and cultural attractions (e.g., protected areas and World Cultural Heritage sites). We suggest that middle- and low-income countries that increasingly depend on the tourism industry should maintain their political stability and enhance visa-free policies to encourage more international tourist arrivals. In this situation, these countries have put more effort into tourism development such as transportation and accommodation. However, a high degree of tourism development traditionally conflicts with environmental protection. One of the best ways to balance between tourism development and environmental protection is to integrate tourism development plans into conservation policies. Our results show that conservation efforts (e.g., protected areas) may contribute to balancing the benefits and risks of tourism development for international tourism, and thus avoid over-development in the long run. In conclusion, the presented approach and findings provide a better foundation for evidence-based decision-making to implement proactive tourism policies.

CHAPTER 4

RETHINKING INTERNATIONAL FOOD TRADE FOR GLOBAL BIODIVERSITY CONSERVATION

In collaboration with

Jianguo Liu

Abstract

To achieve the United Nations Sustainable Development Goals such as food security and biodiversity, it is essential to identify their interrelationships. It is widely held that developed countries negatively affect biodiversity in developing countries through importing food. However, through examining comprehensive datasets comprising 300 food items across 160 countries during 2000–2015, our results show that developed countries exported more food to developing countries than they imported from developing countries, suggesting that biodiversity in developed countries is also negatively affected by production for exports to developing countries. This is especially the case when developed countries with biodiversity hotspots exported food to developing countries without biodiversity hotspots. Furthermore, most exports from developing countries, especially those with biodiversity hotspots, went to other developing countries instead of developed countries. On the other hand, because many developed countries with biodiversity hotspots imported food from developing countries without biodiversity hotspots, such imports might have actually benefited biodiversity in developed countries. Developing countries without biodiversity hotspots played an increasingly important role as net exporters in international food trade. With increasing attention to food security and biodiversity (e.g., the upcoming Fifteenth Meeting of the Conference of the Parties to the Convention on Biological Diversity), it is time to develop new approaches that help operationalize the post-2020 global biodiversity framework and achieve relevant UN Sustainable Development Goals by minimizing the negative impacts of global food production and trade on biodiversity hotspots worldwide.

4.1. Introduction

As the world pursues the ambitious Sustainable Development Goals (SDGs) including food security and biodiversity it is important to understand their interrelationships (Lu et al., 2015; Nilsson et al., 2016; United Nations, 2015). The 17 SDGs were adopted by the UN General Assembly in 2015 and are seen as “a to-do list for people and planet, and a blueprint for success” (United Nations, 2015). Quantitative information on the connections among SDGs is urgently needed to assess whether and how the multiple SDGs can be achieved simultaneously (Nilsson et al., 2016; Xu et al., 2020a; Xu et al., 2020b). Over the past few decades, increasing food availability (a key component of food security) while sustaining biodiversity is key factors for global sustainability (Carole and Ignacio, 2016; Crist et al., 2017; Delzeit et al., 2017; Foley et al., 2011; Wiedmann and Lenzen, 2018). Identifying the relationships between global food production and trade and biodiversity becomes essential to pursue multiple SDGs, which linked with SDGs 2 (food security) and 15 (biodiversity) (Nilsson et al., 2016; United Nations, 2015; Wiedmann and Lenzen, 2018).

With continuous population and income growth (Crist et al., 2017; Marques et al., 2019) as well as uneven distribution of food supply and demand, international food trade is essential for ensuring food availability (Porkka et al., 2013), improving nutrient access (Wood et al., 2018), and meeting rising food demands (Foley et al., 2011; MacDonald et al., 2015). Many countries depend on food imports to meet their growing demands (DeFries et al., 2010; Godfray et al., 2010; Moran and Kanemoto, 2017), but rapid increases of international food trade cause environmental consequences around the world (Crist et al., 2017; Dalin et al., 2017; Lenzen et al., 2012). Producing food for exports causes land use and land cover change (Chaudhary and Kastner, 2016; DeFries et al., 2010; Delzeit et al., 2017) and exerts pressure on biodiversity in exporting countries (Chaudhary and Kastner, 2016; Green et al., 2019; Lenzen et al., 2012; Moran and Kanemoto, 2017; Tilman et al., 2017).

Biodiversity is distributed unevenly across space (Brooks et al., 2014). Therefore, the impact of international food trade on biodiversity is highly dependent on the origins of food production (Carole and Ignacio, 2016; DeFries et al., 2010; Moran and Kanemoto, 2017). It is widely concluded that importing food from tropical, developing countries to developed countries is worsening biodiversity (Chaudhary and Kastner, 2016; Lenzen et al., 2012; Moran and Kanemoto, 2017). Although many studies have documented the negative impacts of international food trade on biodiversity in developing countries with rich biodiversity (Chaudhary and Kastner, 2016; DeFries et al., 2010; Lenzen et al., 2012; Moran and Kanemoto, 2017), little is known about biodiversity implications of food exports from developed countries to developing countries, despite the fact that some developed countries have biodiversity hotspots – areas with high concentration of biodiversity (Myers, 2003; Myers et al., 2000), while some developing countries do not. Failing to recognize developed countries with biodiversity hotspots and developing countries without biodiversity hotspots may lead to biased results about the impacts of food trade on biodiversity worldwide. Thus, understanding food trade among countries with and without biodiversity hotspots is crucial for uncovering the implications of international food trade for global biodiversity.

To address the fundamental knowledge gaps, we divided 160 countries with relevant data into three categories: high-hotspot, low-hotspot, and non-hotspot countries. Specifically, we identified 64 high-hotspot countries (countries where biodiversity hotspots account for more than 50% of terrestrial lands), 53 low-hotspot countries (biodiversity hotspots < 50%), and 43 non-hotspot countries (countries with no biodiversity hotspots) (Figure S4.1, Table S4.1). We also classified the countries in each category as developing (with low, low-middle, and upper-middle income) and developed (with high income) according to the World Bank's income classification (The World Bank, 2017). These classifications help analyze food trade among countries with different concentrations of biodiversity and levels of economic development. Our food dataset contains relevant annual information for 300 food items, including 203 crops from 2000–2015.

4.2. Materials and Methods

4.2.1. Biodiversity hotspot and non-hotspot countries

We divided all 160 countries with available data into 117 hotspot countries and 43 non-hotspot countries (Figure S4.1, Table S4.1) based on the relevant biodiversity hotspot information (Myers, 2003; Myers et al., 2000). Hotspot countries are those that contain at least part of a recognized global biodiversity hotspot (Liu et al., 2003; Myers, 2003; Myers et al., 2000). Biodiversity hotspots are areas with not only a high degree of species richness (hold >= 0.5% world's plants as endemics), but also a high degree of vulnerability (lost >= 70% of primary, native vegetation) to human disturbance (Myers, 2003; Myers et al., 2000). In contrast, non-hotspot countries do not include any part of a global biodiversity hotspot. Because hotspot countries vary substantially in terms of biodiversity hotspots (Figure S4.4), we classified hotspot countries into 64 high-hotspot countries and 53 low-hotspot countries, in which biodiversity hotspots account for more or less than 50% of terrestrial lands, respectively.

High-hotspot countries, low-hotspot countries, and non-hotspot countries also have a number of other differences. For example, in developed and developing high-hotspot countries, 85.2% and 93.8% of agricultural areas were located in biodiversity hotspots respectively, whereas in developed and developing low-hotspot countries, only 7.4% and 21.8% of total agriculture land were located in a biodiversity hotspot, respectively (Figure S4.3). Since non-hotspot countries have no hotspots, agricultural area in non-hotspot countries was of course not located in any biodiversity hotspots. Per capita GDP of developed non-hotspot countries in 2015 ($44,990 in 2010-constant USD) was roughly 1.6 times as high as that of developed high-hotspot countries ($26,103) and developed low-hotspot countries ($28,306). Developing countries had similar per capita GDP across high-hotspot ($4,060 in 2010-constant USD), low-hotspot ($3,720), and non-hotspot countries ($3,991). Developed high-hotspot countries had the lowest population growth rates (4.5%) during 2000–2015, followed by developed non-hotspot countries (9.3%) and developed low-hotspot countries (11.0%). Population size in developing high-hotspot countries, developing low-hotspot countries, and developing non-hotspot countries increased 24.5%, 23.0%, and 23.9% during 2000–2015, respectively. In addition, land size in developed low-hotspot countries in 2015 (3,648,610 km^2) was 21.2 times and 7.2 times as large as that of developed high-hotspot countries (172,263 km^2) and developed non-hotspot countries (506,390 km^2), respectively. Land size in developing low-hotspot countries in 2015 (1,124,795 km^2) was 5.1 times and 2.2 times larger than developing high-hotspot countries (219,132 km^2) and non-hotspot countries (516,284 km^2), respectively.

4.2.2. Data collection

Datasets were obtained from the UN FAO, the UN Data, and the World Bank (The World Bank, 2017; UN FAO, 2018; United Nations Statistics Division, 2015). Our database consisted of agricultural, environmental, and socioeconomic data. We selected the time period from 2000–2015 because of data availability. Agricultural datasets were obtained from the UN FAO(UN FAO, 2018) and included information about food production, food trade matrices, agricultural areas, agricultural intensification (fertilizer application, pesticide use, and water withdrawal), and average dietary energy supply adequacy (a percentage of average dietary energy requirement). The basic food trade unit in this research was the physical volume (metric tonne) of food produced, imported, and exported. This unit was chosen for two reasons. First, the number of countries in the volume dataset was much higher than in the monetary dataset. Second, using the volume of food trade is more appropriate for showing the extent to which food trade is linked with agricultural area because the monetary value varies with price fluctuations. Socioeconomic data such as population and per capita GDP came from the World Bank (The World Bank, 2017). We also used published spatial data for identifying hotspot countries. Hotspot countries were identified according to Conservation International (Myers, 2003; Myers et al., 2000).

4.2.3. Aggregate analysis

In the aggregate analysis, we divided the data collected from the individual countries into three groups of countries (high-hotspot countries, low-hotspot countries, and non-hotspot countries), which were further divided into developed and developing countries. We calculated agricultural intensification and agricultural area change from 2000–2015. By using agriculture and land use datasets, we were able to divide agricultural intensification values by agricultural area in each group. For example, we aggregated the amounts of fertilizer application and agricultural areas in each group. Then, we divided the amounts of fertilizer application by total agricultural areas. We also calculated food trade flows among high-hotspot countries, low-hotspot countries, and non-hotspot countries in each individual year from 2000–2015, as well as annual averages over the same time period. In the FAO food trade matrix dataset (UN FAO, 2018), we used food import matrix data. Food export matrix datasets were only used for filling in the data gaps in food import data. In addition, we used an origin-tracing algorithm to reduce data uncertainty regarding re-exports (Dalin et al., 2017; Kastner et al., 2011). For example, some countries such as the Netherlands import food products from exporting countries and re-export them to other importing countries. The origin-tracing algorithm developed by Kastner et al. (2011) has a basic assumption that food consumption in each country proportionally originates from their domestic production and other countries. The origin of food imported can be examined using the bilateral food trade data. This algorithm assigns re-export volumes from intermediate countries to the original exporting country of production (Dalin et al., 2017; Kastner et al., 2011).

We also estimated the amount of land saved due to food imports based on yield and quantity of those imports (Liu, 2014). The quantity of food imports (tonne) was divided by yield (tonne/km^2) in each country. The amount of land saved by food imports was aggregated with developed and developing countries in high-hotspot countries, low-hotspot countries, and non-hotspot countries.

4.2.4. Panel data analysis

To uncover factors affecting food production for domestic supply, food exports, and food imports, we performed panel data analyses in R (R Core Team, 2017). Panel data analysis allows control for variables in different entities (e.g., countries) over time (Torres-Reyna, 2010). We selected the random effects model because agricultural, socioeconomic, and environmental differences across countries have some influence on the quantity of food production and trade. The random effects model assumes that the each entity has its own error term that is random and not correlated with independent variables in the model (Torres-Reyna, 2010). An advantage of the random effects model over fixed effects is that time-invariant variables can be included as independent variables. We used the same value of biodiversity hotspots in our panel data analyses because of the lack of time-series data for biodiversity hotspots.

Since food production has two important purposes—domestic supply and export—we included the quantities of food production for domestic supply and export separately. This separation is essential for identifying responsible parties. For example, although the impact of food production on biodiversity is likely the same irrespective of whether the produced food is consumed locally or exported, the percentages of food exported out of food produced differed among countries. Exporting countries may shift some responsibilities for biodiversity loss caused by food production for exports to importing countries.

To identify the changes in significant factors for food production and trade over time, we constructed random effects models for three different periods (2000–2007, 2008–2015, and 2000–2015). The three random effects models had 160 countries and included 8, 8, and 16 temporal points in each panel, respectively. We estimated the amounts of food production for domestic supply, food exports, or food imports as a function of agricultural factors (average dietary energy supply adequacy and total agricultural area), socioeconomic factors (per capita GDP and total population), and environmental factors (percentage of biodiversity hotspots). We performed log transformation on all dependent and independent as the log-log transformation allows to interpret coefficients as an elasticity (Chung et al., 2018a).

We performed the Breusch-Pagan Lagrange multiplier (LM) test to choose between a random effects model and a simple ordinary least squares model (Torres-Reyna, 2010). The LM test concluded that there are significant differences across countries (existence of panel effects) and that our random effects models were more suitable. The random effects model allows to include time-invariant variables that preclude fixed effects. We also tested for heteroscedasticity using the Breusch-Pagan test. Since heteroscedasticity was detected in all random effects models, we controlled for heteroscedasticity using a robust covariance matrix estimation (also known as a sandwich estimator) (Torres-Reyna, 2010). We identified multicollinearity problems using variance inflation factors (VIF). A VIF of 10 indicates a severe multicollinearity problem (O'Brien, 2007). All VIF results in our models were less than 4.

4.3. Results

Results indicate that developed countries were net food exporters while developing countries were net food importers during 2000–2015 (Figure 4.1 and 4.2). Among the exports from developed countries to developing countries, almost all (97.0%) went to hotspot countries. Specifically, developing hotspot countries received 97.8% of the exports from developed hotspot countries to developing countries (Figure 4.2). Of exports from developed high-hotspot countries to developing countries, 34.1% and 52.2% went to developing high-hotspot and low-hotspot countries, respectively.

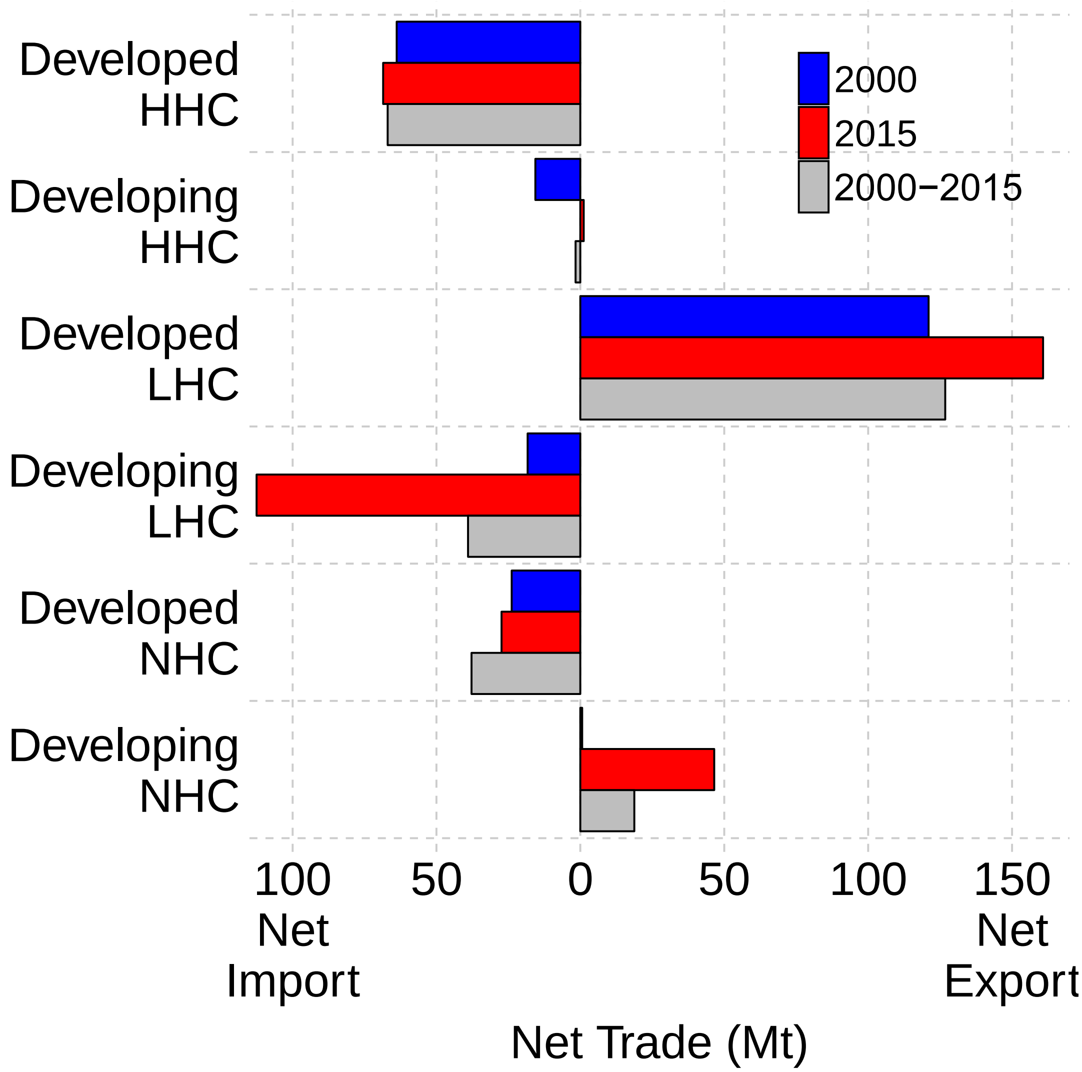

Figure 4.1. The quantity of net food trade between developed and developing high-hotspot countries (HHC), low-hotspot countries (LHC), and non-hotspot countries (NHC). Blue indicates net food trade in 2000, red indicates net food trade in 2015, and gray indicates average net annual food trade from 2000–2015. The net amounts of food trade in each group are not linearly increased or decreased over time. The net amounts of food trade in 2000 and 2015 can be lower or higher than those in other mid-years as shown in Figure S4.2.

Among the exports from developed low-hotspot countries to developing countries, 44.0% and 54.3% were destinated to high-hotspot and low-hotspot countries. Developed low-hotspot countries (e.g., USA and France) were the main contributors to international food trade as net exporters while developing low-hotspot countries (e.g., China and Egypt) played an increasingly important role as net food importers (Figure 4.1 and 4.2). The USA and France accounted for

77.9% (158.9 Mt/year during 2000–2015) of food exported from developed low-hotspot countries (203.9 Mt/year), but only 3.3% and 12.2% of terrestrial areas were biodiversity hotspots, respectively, and the agricultural area in biodiversity hotspots accounted for 1.8% and 8.2% of total agricultural area respectively (Figure S4.3). Agricultural areas in these countries also decreased from 2000–2015 by -2.1% in the USA and by -3.6% in France, suggesting less areal impact of agriculture on biodiversity.

Food imports from developed low-hotspot countries with the highest quantities of net food exports kept much land from food production in high-hotspot countries. In developing high-hotspot countries, an estimated 202,257 km^2 of agricultural area per year (as big as the combined territories of two high-hotspot countries—Cuba and Guatemala) was saved due to imports from developed low-hotspot countries during 2000–2015 (Table S4.3). In developed high-hotspot countries, food imports from developed low-hotspot countries accounted for a saving of 34,650 km^2 of agricultural area per year, larger than two thirds of Costa Rica's territory.

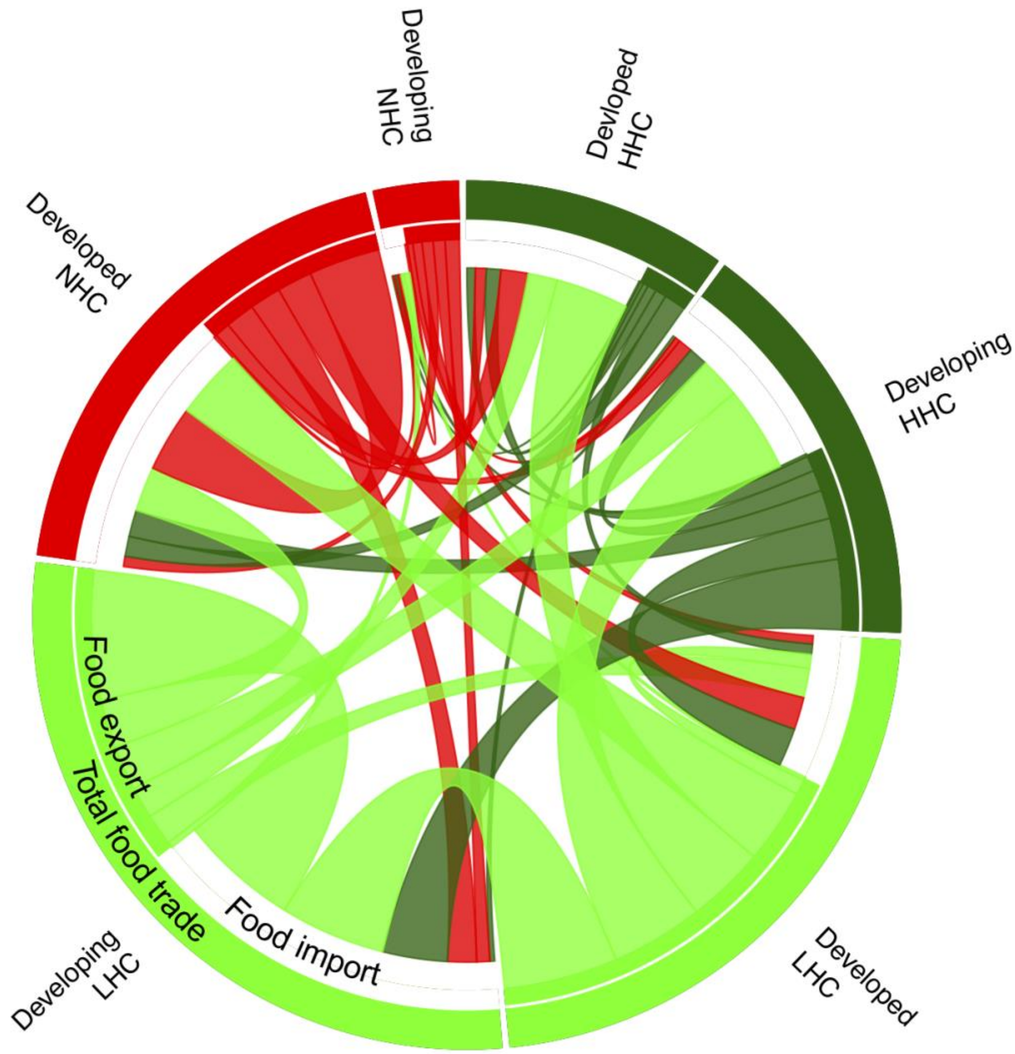

Figure 4.2. Average annual food flows (Mt/year) from 2000–2015. Food flows between developed and developing high-hotspot countries (HHC), low-hotspot countries (LHC), and non-hotspot countries (NHC). Non-hotspot countries are marked by red, high-hotspot countries by dark green, and low-hotspot countries by light green. The arc length of an outer circle indicates the sum of food exported and imported in each group. The arc length of a middle circle refers to the quantity of food exports. The inner arc length shows the quantity of food imports. Raw data from UN FAO (2018).

Developing high-hotspot countries were net food importers (1.7 Mt/year of average net annual food imported) from 2000–2015 (Figure 4.1 and 4.2). Such imports are particularly important to reduce agricultural impacts on biodiversity in high-hotspot countries because 78.1% of high-hotspot countries had over 90% of their terrestrial area as biodiversity hotspots (Figure S4.4), and 92.1% of high-hotspot countries' agricultural area was located in biodiversity hotspots (Figure S4.3). Over the same period, developing high-hotspot countries accounted for 50.3% (96.1 Mt/year during 2000–2015) of total food imports among all high-hotspot countries (190.9 Mt/year) (Table S4.2). Food imports from low-hotspot and non-hotspot countries to developing high-hotspot countries accounted for production in roughly 7.6% of annual agricultural area (340,428 km^2, larger than the territory of Malaysia) in developing high-hotspot countries.

Developing low-hotspot countries rapidly increased their net food imports during the study period by 514.4% (Figure S4.2, Table S4.2). Of exports from developed non-hotspot countries to developing countries, 93.8% went to developing hotspot counties (31.0% and 62.8% to high-hotspot and low-hotspot countries, respectively).

Most exports (56.3%) from developing countries went to other developing countries, rather than developed countries (Figure 4.2). Of which, the destinations of 52.3%, 61.7%, and 40.5% of exports from developing high-hotspot, low-hotspot, and non-hotspot countries were developing countries. In other words, developed countries received less than half of the exports from developing hotspot countries and more than half of the exports from developing non-hotspot countries. Among the exports from developing hotspot countries to developed countries, most of them (60.9%) were destinated to developed hotspot countries (Figure 4.2). Of exports from developing high-hotspot countries to developed countries, developed high-hotspot and low-hotspot countries received 18.2% and 48.2%, respectively. Among exports from developing low-hotspot countries to developed countries, almost equal amounts (28.5%, 28.2%) went to developed high-hotspot and low-hotspot countries.

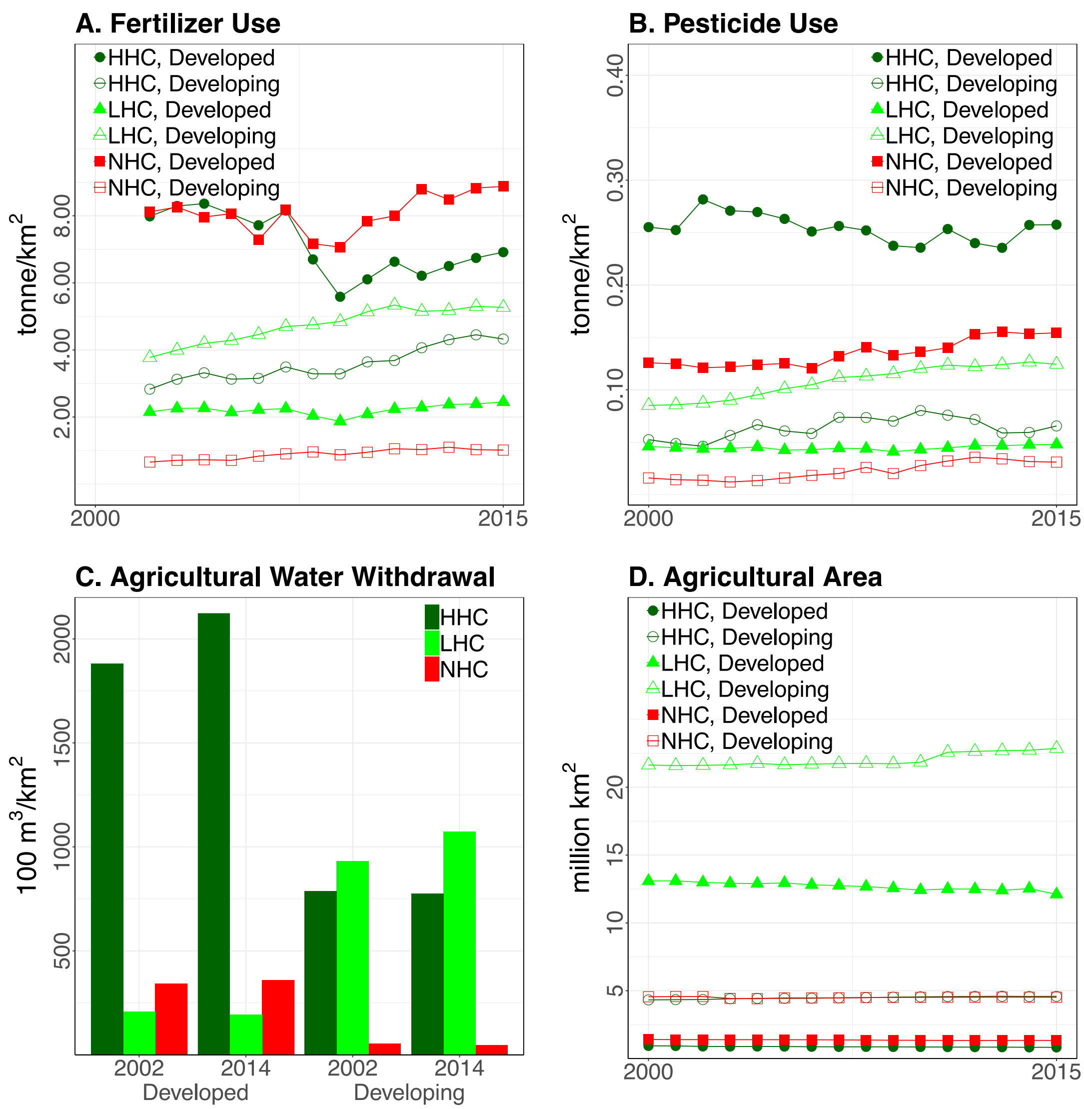

Figure 4.3. Changes in agricultural intensification and agricultural area in high-hotspot countries (HHC), low-hotspot countries (LHC), and non-hotspot countries (NHC), with each group subdivided into developed and developing countries. (A) Fertilizer use (tonne/km^2); (B) pesticide use (tonne/km^2); (C) agricultural water withdrawal (m^3/km^2); (D) agricultural area change (km^2). Raw data from UN FAO (2018).

Imports to developed high-hotspot countries (67.0 Mt/year of average net annual food imported during 2000–2015) can also help further reduce negative impacts on biodiversity because agricultural intensification in developed high-hotspot countries was higher than in other types of countries (Figure 4.3). For example, fertilizer use per unit in developed high-hotspot countries (6.9 tonne/km^2) was 60.5% higher than developing high-hotspot countries (4.3

tonne/km^2) in 2015 (Figure 4.3A). Pesticide use per unit in developed high-hotspot countries (0.258 tonne/km^2) was three times higher than in developing high-hotspot countries (0.066 tonne/km^2) in 2015 (Figure 4.3B). In 2014, freshwater withdrawal (212,333.8 m^3/km^2) for agricultural production in developed high-hotspot countries was about twice as high as in developing high-hotspot countries (77,599.7 m^3/km^2) (Figure 4.3C). Therefore, food imports may decrease biodiversity threats in developed high-hotspot countries as, without food imports, more agricultural land in developed high-hotspot countries would be used or intensified for domestic food production.

Developing non-hotspot countries (e.g., Ukraine and Romania) played an increasingly important role as net food exporters (with an average net food export of 18.8 Mt/year during 2000-2015) (Figure 4.1, Figure S4.5 and S4.6). They exported 1.6% (6.7 Mt) of their food production in 2000, but 9.7% (61.6 Mt) in 2015 (Table S4.4). Such exports freed much area for production in hotspot countries. For instance, developed high-hotspot countries saved agricultural areas of 24,339 km^2 (larger than the territory of Belize) per year from 2000–2015 (Table S4.3). Developing high-hotspot countries saved agricultural areas of 3,270 km^2 (over 60% larger than the territory of Mauritius) in 2000 and increasingly 72,309 km^2 (approximately the territory of Panama) in 2015 (Table S4.3). Developing non-hotspot countries had the lowest agricultural intensification and least agricultural area among all types of countries, which suggests food imports from developing non-hotspot countries further reduce biodiversity threats from food production in hotspot countries (Figure 4.3). Fertilizer use per unit land in developing non-hotspot countries (1.0 tonne/km^2) was 430% and 330% lower than developing low-hotspot countries (5.3 tonne/km^2) and developing high-hotspot countries (4.3 tonne/km^2) in 2015, respectively (Figure 4.3A). Pesticide use per unit in developing non-hotspot countries (0.031

tonne/km^2) was 300% and 113% lower than developing low-hotspot countries (0.124 tonne/km^2) and developing high-hotspot countries (0.066 tonne/km^2) in 2015 (Figure 4.3B). In 2014, freshwater withdrawal (4,680.5 m^3/km^2) for agricultural production in developing non-hotspot countries was 23 times and 17 times lower than developing low-hotspot countries (107,351.5 m^3/km^2) and developing high-hotspot countries (77,597.7 m^3/km^2) respectively (Figure 4.3C). While agricultural areas in developing non-hotspot countries decreased 0.7% from 2000–2015, agricultural areas in developing low-hotspot countries and developing high-hotspot countries increased 5.6% and 6.0% respectively (Figure 4.3D).

Panel data analyses for three different time periods (2000–2007, 2008–2015, and 2000–2015) identified factors that affect global food production and trade (Table 4.1). Countries with larger agricultural areas tended to produce more for both domestic consumption and exports, whereas countries with smaller agricultural areas tended to import more food. This result may indicate that food importers with smaller agricultural areas displaced agricultural land use to food exporters. For instance, developed high-hotspot countries had the smallest agricultural areas among the six types of countries (Figure 4.3D) and would have saved 97,177 km^2 of agricultural area (larger than the territory of Portugal) per year during 2000-2015 as net food importers, accounting for roughly 11.1% of their annual agricultural area (Table S4.3). In addition, the average dietary energy supply adequacy had a positive association with the quantity of food imported. Countries that had a higher average dietary energy supply imported more food from abroad to meet increases in per capita caloric and protein demands. Per capita GDP and population size drove all significant correlated results with food production and trade (Table 4.1).

Table 4.1. Coefficients of panel data analyses in three different periods: 2000–2007, 2008–2015 and 2000–2015.

Variable	2000–2007			2008–2015			2000–2015		
	Food production for domestic supply	Food export	Food import	Food production for domestic supply	Food export	Food import	Food production for domestic supply	Food export	Food import
Biodiversity hotspot (%)	0.055 (0.037)	0.222* (0.081)	0.085* (0.036)	0.043 (0.033)	0.121 (0.094)	0.096 (0.063)	0.076* (0.038)	0.275* (0.098)	0.123* (0.060)
Dietary energy supply adequacy (%)	1.522** (0.297)	2.415** (0.730)	1.197* (0.387)	0.680* (0.281)	1.164 (0.832)	1.693* (0.652)	1.393** (0.249)	1.146 (0.785)	1.861** (0.541)
Agriculture area (km_2)	0.324* (0.121)	0.486* (0.162)	-0.159* (0.061)	0.313** (0.062)	0.264 (0.136)	-0.259** (0.067)	0.455** (0.114)	0.683** (0.181)	-0.231** (0.064)
Population (1,000 persons)	0.719** (0.139)	0.518* (0.195)	1.086** (0.070)	0.718** (0.069)	0.753** (0.170)	1.193** (0.081)	0.577** (0.121)	0.316 (0.223)	1.165** (0.078)
GDP per capita (constant $)	0.137** (0.032)	0.673** (0.123)	0.621** (0.064)	0.155** (0.040)	0.595** (0.130)	0.597** (0.084)	0.155** (0.043)	0.849** (0.156)	0.621** (0.087)
Intercept	-1.918 (1.393)	-14.943** (3.170)	-5.725** (1.623)	2.056 (1.212)	-7.902* (3.283)	-7.827* (2.836)	-1.663 (1.247)	-10.893* (3.468)	-8.969** (2.163)
R-Squared	0.658	0.223	0.487	0.599	0.125	0.312	0.556	0.150	0.344
F-statistics	476.2	71.16	235.78	375.5	35.46	113.85	627.23	89.17	263.38

Values in parentheses are standard errors
All variables are log transformation variables
** P<0.001, * P<0.05

4.4. Discussion

Our research integrating biodiversity hotspots and economic development status provides a new perspective—international food imports may benefit developing and developed countries with biodiversity hotspots. By increasing the proportion of food production for exports, developing non-hotspot countries with lower intensification played an increasingly important role as net exporters in international food trade. As threats from agricultural activities vary among species and across space (Brooks et al., 2014; Moran and Kanemoto, 2017), species-specific analyses are needed within national boundaries. Worldwide, identifying species-specific relationships with international food trade items would be possible through the analysis of high-resolution data (Green et al., 2019; Moran and Kanemoto, 2017; Wiedmann and Lenzen, 2018). Future research efforts are needed to accurately determine causal relationships among global food production and trade and biodiversity based on high-resolution sub-national and local data over time (Carole and Ignacio, 2016; Green et al., 2019; Moran and Kanemoto, 2017; Wiedmann and Lenzen, 2018), and to identify the impacts of food production for exports in specific locations.

With increasing attention to food security and biodiversity conservation (e.g., global assessment of biodiversity and ecosystem services (Díaz et al., 2019), upcoming meeting of the Conference of the Parties to the Convention on Biological Diversity (UNEP)), it is time to rethink international food trade by creating more innovative approaches to minimize the negative impacts of global food production and trade on biodiversity in both developed and developing hotspot countries, especially high-hotspot countries. For instance, new international initiatives and agreements are necessary to reduce threats to biodiversity from food production and trade (Brooks et al., 2014; Ehrlich and Harte, 2015; Redford et al., 2015). Food prices should incorporate the biodiversity cost of producing food (Pe'er et al., 2014). Earnings from such a price hike could be used to mitigate negative impacts on biodiversity. Biodiversity can also benefit from the further development of techniques with low input but high yield, such as wildlife-friendly farming (Pywell et al., 2015). Both food importing and exporting countries working together to implement new policies and technologies can lower negative impacts on biodiversity while increasing food security. The approaches and findings in this paper provide a foundation for further work incorporating data with higher resolutions to quantify biodiversity impacts, operationalize post-2020 global biodiversity framework, and achieve United Nations Sustainable Development Goals (e.g., Goal 2 – food security and Goal 15 – biodiversity conservation) across multiple scales worldwide.

CHAPTER 5

INTEGRATING BUILT INFRASTRUCTURE AND WATERSHED CONSERVATION TO SUSTAIN FRESHWATER ECOSYSTEM SERVICES FOR GLOBAL CITIES

In collaboration with

Kenneth Frank, Yadu Pokhrel, Thomas Dietz, and Jianguo Liu

Abstract

Worldwide rapid urbanization demands more freshwater. This need is conventionally met through the construction of infrastructure. Watershed conservation activities have also increased to provide freshwater ecosystem services, but little research has examined the intricate relationships between built infrastructure and watershed conservation activities for provisioning freshwater ecosystem services to global cities. By using egocentric network analysis, this study examines how to integrate built infrastructure approaches with ongoing watershed conservation activities for sustaining four freshwater ecosystem services (i.e., freshwater provision, sediment regulation, flood mitigation, and hydropower production) to cities. Our results indicate that wetlands in protected areas contribute to sustaining freshwater provision to cities. Forest cover in protected areas can improve the capacity of large dams for sediment reduction and hydropower production, but cities mainly depend on dams for flood mitigation. Our findings suggest strategic approaches for integrating built infrastructure and watershed conservation activities to enhance urban water sustainability.

5.1. Introduction

Over the past few decades, rapid urbanization causes various water-related problems such as water shortage, water quality, floods, and energy for cities worldwide (McDonald and Shemie, 2014; McDonald et al., 2014; McDonald et al., 2016). With increased urban population and income levels, built (or gray) infrastructure has been rapidly constructed to meet the increased freshwater demands for cities (Grill et al., 2019; McDonald and Shemie, 2014; McDonald et al., 2014; Vorosmarty et al., 2010). Built infrastructure defines as the human-engineered construction for water resources such as dams and treatment facilities (Gartner et al., 2013). Modifications of natural river systems through built infrastructure increase water security for residential users (Tessler et al., 2015) but cause the loss of freshwater biodiversity, water quality, and habitat degradation (Grill et al., 2019; Michalak, 2016; Palmer, 2010; Vorosmarty et al., 2010). Since the early 20th century, almost 90% of watersheds providing water to cities have experienced a degradation of their water quality, including increases in nitrogen and phosphorous due to anthropogenic activities (e.g., changes in agricultural land use) (McDonald et al., 2016). This degraded water quality directly affects water for drinking and recreation in cities (Michalak, 2016). Furthermore, potential supplies of freshwater ecosystem services (ES) to cities have decreased over time and across regions (Dodds et al., 2013).

On the other hand, watershed conservation activities (e.g., protected areas (PAs) and investments in watershed services (IWS)) have continuously provided provisioning freshwater ES as a part of the Convention on Biological Diversity Aichi Targets and the United Nations Sustainable Development Goals (SDGs) (Harrison et al., 2016; Romulo et al., 2018; Tellman et al., 2018; Visconti et al., 2019). Watershed conservation activities can potentially help reduce the negative effects of built infrastructure that degrade freshwater biodiversity, damage fisheries, and displace local people (Liu et al., 2016a; Palmer et al., 2015; Ziv et al., 2012). Watershed conservation areas can provide various freshwater ES to humans, such as fisheries (Brennan et al., 2019; McIntyre et al., 2016), the improvement of water quantity and quality (Harrison et al., 2016; Mapulanga and Naito, 2019; Veldkamp et al., 2017; Vörösmarty et al., 2018), flood regulation (Russi et al., 2013; Tellman et al., 2018), recreational opportunities (Chung et al., 2018b), and carbon sequestration (Viña et al., 2016). Specifically, water storage from forests and wetlands in PAs may increase the capability for freshwater provision, flood protection, and hydropower production in addition to dam storage (Harrison et al., 2016; Moran et al., 2018; Tellman et al., 2018). The capacity of watershed conservation areas under IWS can also help meet the increased freshwater demands of cities by maintaining high freshwater ES and biodiversity (Adamowicz et al., 2019; Romulo et al., 2018; Zheng et al., 2013).

With the rapid increases of PAs and IWS worldwide, the networks of watershed conservation areas may complement built infrastructure by providing various freshwater ecosystem services. Yet, little is known about the relationships between built infrastructure and watershed conservation activities for sustaining freshwater ES to cities. The relationship warrants attention because maintaining the benefits of built infrastructure while conserving healthy freshwater ecosystems is a complex challenge (Grill et al., 2015; Poff and Schmidt, 2016). These relationships between built infrastructure and watershed conservation activities become more complicated as cities are increasingly reliant on not only surrounding watersheds but also distant watersheds through built infrastructure construction (e.g., dams and canals) (Liu and Yang, 2013; Liu et al., 2016a; McDonald et al., 2016). Thus, a new strategy is urgently needed to integrate a built infrastructure approach with ongoing watershed conservation activities (i.e., PAs and IWS) (Harrison et al., 2016; Romulo et al., 2018; Zheng et al., 2013).

To achieve sustainable freshwater ES supplies to cities, management strategies should consider a balance between human demands and ecosystem conservation (Lehner et al., 2011; Vorosmarty et al., 2010). Built infrastructure approach and watershed conservation activities can be combined because the supply of freshwater ES depends on both protected watersheds and traditional built infrastructure (Green et al., 2015; Vörösmarty et al., 2018). However, very limited studies to date have examined the approaches for integrating built infrastructure and watershed conservation activities in source watersheds for sustaining freshwater ES to cities. Therefore, the goal of this study is to fill this knowledge gap.

We seek to answer two questions: (1) Which built infrastructure and watershed conservation activities have a significant relationship with freshwater ES supplies for global cities? and (2) Which socioeconomic and environmental factors of source watersheds and cities contribute to the changes of freshwater ES supplies to cities? This study focuses on four freshwater ES—freshwater provision, sediment regulation, flood mitigation, and hydropower production in source watersheds—as these four ES flows have exponentially increased to meet cities' water demands. By using network analysis, we examine how built infrastructure, PAs, and IWS influence the provision of freshwater ES for global cities, while controlling for the net of geographical factors, watershed characteristics, and city characteristics. This study provides a new perspective on how to combine built infrastructure approaches and watershed conservation activities to sustain freshwater ES supplies for cities while protecting freshwater ecosystems and biodiversity.

5.2. Materials and Methods

5.2.1. City and watershed selection

We first identified global cities that mainly depend on surface water sources from the City Water Map database (McDonald et al., 2014). Each selected city has an average population of over 300,000 people from 2000 to 2010 according to the World Urbanization Prospects data (UNDP, 2015). For cities' urban extents, we used the Global Administrative Database (GADM) that defines urban administrative areas (Global Administrative Areas, 2018). For cities not defined in the GADM, we used the global urban extent map from Schneider et al. (2009) based on MODIS satellite data. In the USA, the Cartographic Boundary File for urban areas was used to define urban extents (United States Census Bureau, 2017).

For each city, we identified three types of source watersheds: (1) freshwater source watersheds (freshwater provision and sediment regulation), (2) flood watersheds, and (3) hydropower watersheds. As freshwater ES are produced in source watersheds and provide benefits to cities, source watersheds are directly and indirectly connected to cities through the flows of freshwater ES. Source watersheds were designated following the United States Geological Survey (USGS) HydroSheds database (Lehner et al., 2008).

Freshwater source watersheds provide surface water sources to cities. Cities depend on not only the surrounding watersheds but also distant watersheds for freshwater resources (Liu and Yang, 2013; McDonald et al., 2016). Surface water in freshwater source watersheds is transferred from water intake points to the city. Surface water intake points were obtained from the City Water Map database (McDonald et al., 2014). Freshwater source watersheds are also watersheds with sediment flows affecting freshwater quality in cities. Flood watersheds have a higher elevation than cities, can overlap with urban extent areas, and increase or reduce the flood risks of cities by directly draining surface water to the urban extent area. Hydropower watersheds generate and provide electricity from hydropower dams to cities and are connected with cities through high voltage power lines within 100 km from the urban extent. High voltage power lines linkages to the cities were obtained from OpenStreetMap (https://www.openstreetmap.org).

With the different locations and numbers of watersheds for each freshwater ES, the number of cities also varies across freshwater ES. For exploring freshwater provision and sediment regulation, we selected 333 cities and 1,198 freshwater source watersheds. We also analyzed a total of 665 flood watersheds across 200 cities for flood mitigation. Finally, we selected 197 cities and 469 hydropower watersheds for hydropower production (Figure S5.1).

5.2.2. Freshwater ecosystem services

Built infrastructure and watershed conservation activities have a variety of impacts on natural ecosystems and ES. For example, hydropower capacity may increase with the number of dams, but such expansion causes habitat loss and river fragmentation, and those changes in turn impact freshwater biodiversity and water quality (Grill et al., 2019; Palmer, 2010; Roy et al., 2018). Watershed conservation can benefit forest and wetland cover, positively contributing to fisheries and water quality (Bilotta et al., 2012; Brennan et al., 2019; McIntyre et al., 2016; Vörösmarty et al., 2018). Both built infrastructure and watershed conservation areas contribute to increasing water regulation (e.g., freshwater provisioning and flood mitigation) (Chen and Olden, 2017; Harrison et al., 2016; Mapulanga and Naito, 2019; Roy et al., 2018; Russi et al., 2013).

We examined four freshwater ES that are closely linked with cities' water-related demands: freshwater provision (Veldkamp et al., 2017), sediment regulation (Cohen et al., 2014), flood mitigation (Dottori et al., 2016), and hydropower production (Byers et al., 2018; Zarfl et al., 2015) (Table S5.1). These four ES have flows from source watersheds to cities and are divided into provisioning and regulating ES. Provisioning ES include freshwater provision and hydropower production. Regulating ES are comprised of sediment regulation and flood mitigation. We used global modeling data for freshwater ES, except for hydropower production. These datasets utilized local and regional observational data for to produce their output data. The resulting datasets have been widely used in peer-reviewed papers in high-impact journals (Best, 2019; Grill et al., 2019; Smith et al., 2019; Veldkamp et al., 2017; Vorosmarty et al., 2010).

5.2.2.1. Freshwater provision

In this study, freshwater provisioning that supplies cities refers to the annual average volumes of surface water flowing through a river channel. Surface water is extracted at water intake points and transferred to cities (McDonald et al., 2014). Freshwater provision data for 2001–2010 were obtained from phase 2 of the Inter-Sectoral Impact Model Intercomparison Project (ISIMIP2a, http://www.isimip.org), which provides the daily outputs from five global hydrological models (GHMs): H08 (Hanasaki et al., 2008a, 2008b), LPJmL (Bondeau et al., 2007; Schaphoff et al., 2013), MATSIRO (Pokhrel et al., 2012; Pokhrel et al., 2015), PCR-GLOBWB (van Beek et al., 2011; Wada et al., 2014), and Water Gap (Müller Schmied et al., 2016). With these five GHMs, driven by three historical climate forcing datasets (PGFv2 (Sheffield et al., 2006), GSWP3 (Dirmeyer et al., 2006), and WFDEI (Weedon et al., 2014)), we used 15 model combinations to quantify the volumes of surface water supplies from source watersheds to cities. We extracted the annual-averaged values of each of the 15 models' outputs in water intake points and calculated median values for the 15 model combinations in each source watershed.

The GHMs account for the most natural surface and sub-surface hydrologic processes relevant for the simulation of water resource availability (e.g., local run-off and upstream discharge) at 0.5° (~50 km) grid cells globally. Human water management activities are also represented by accounting for various sectoral water demands including those for the agriculture (irrigation and livestock), industry (manufacturing and thermal energy), and public (domestic use) sectors under time-varying socioeconomic conditions (e.g., population, GDP, and land-use) (Veldkamp et al., 2017).

5.2.2.2. Sediment regulation

We obtained results from a global suspended sediment flux model based on the WBMsed global hydrology model to represent the surface water quality of cities' freshwater provisioning (Cohen et al., 2014; Wisser et al., 2010). Cohen et al. (2014) provided the amounts of suspended sediment flux in a 6 arc-minute (~12 km or 0.1°) grid cell. We extracted the amounts of annual-averaged suspended sediments in water intake points for cities from 2000 to 2010. We concentrated on surface water sources, not groundwater, because built infrastructure and watershed conservation activities mainly contribute to changes in surface water quality (e.g., sediment flux and phosphorous pollution) (McDonald et al., 2016; Robin Abell et al., 2017). Although suspended sediments are crucial to sustain freshwater ecosystems in downstream areas (e.g., creating natural habitats) (Vercruysse et al., 2017), suspended sediments deteriorate water quality and therefore cause additional costs for urban water treatment (Bilotta and Brazier, 2008; Bilotta et al., 2012; McDonald et al., 2016).

5.2.2.3. Flood mitigation

We used global flood hazard maps with return periods of 100 years to identify the probability of river flood magnitudes over an urban area (Dottori et al., 2016). These flood hazard maps show flood extents and depths in a 30 arc-second (~1 km or 0.0083°) grid cell based on hydrological information from the Global Flood Awareness System (GloFAS) (Alfieri et al., 2018; Dottori et al., 2016). Based on this model, we calculated the proportion of flood extent areas to total urban extent areas in each flood watershed.

5.2.2.4. Hydropower production

The Global Power Plant Database provides the geolocation of operational hydropower dams above 1 megawatt (MW) capacity (Byers et al., 2018). This database covers approximately 89% of global installed capacity in the hydropower sector (Byers et al., 2018). This dataset provides point hydropower locations, and we aggregated the installed capacity of the hydropower dams in each hydropower watershed.

5.2.3. Source watershed and city characteristics

To examine which characteristics contribute to four freshwater ES flows from source watersheds to cities, we collected data regarding dams, watershed conservation activities, environmental factors, and socioeconomic factors in source watersheds and cities. These data were obtained from international organizations, online databases, and peer-reviewed papers (Table S5.1). Our indicators are dam density as a measure of built infrastructure, and watershed conservation activities included PAs in source watersheds (IUCN and UNEP-WCMC, 2017) and IWS programs in cities (Romulo et al., 2018).

For each of the three different types of source watersheds (freshwater source, flood, and hydropower), we obtained information on forest and wetland cover in PAs, dam density, irrigation areas, and geographic characteristics of the watersheds. The spatial boundaries and characteristics of PAs were obtained from the World Database on Protected Areas (IUCN and

UNEP-WCMC, 2017). We selected terrestrial PAs that are legally designated and actively managed at the national or sub-national level. We also included all PAs that were assigned, not reported, or not assigned to the IUCN management category because many countries do not consistently apply or use the IUCN management category (Bingham et al., 2019). Since many PAs spatially overlap each other (Deguignet et al., 2017; Jones et al., 2018), we dissolved PA boundaries to avoid double counting problems. Then, we intersected a single PA polygon with each watershed's boundary using ArcGIS (ESRI, 2015).

Forest cover data were obtained from global land cover data that provide the percentage of forest cover with 1 km resolution (Tuanmu and Jetz, 2014). Wetland cover data were collected from the Global Lakes and Wetlands database, which provides global wetland extents at 30 arc-second (~1 km) resolution (Lehner and Döll, 2004). Then, in each watershed, we calculated the proportion of forest and wetland cover in PAs to total watershed areas, respectively.

The attributes of dams were obtained from the Global Reservoir and Dam (GRanD) database (Lehner et al., 2011). This database includes the name, spatial location, construction year, and various characteristics of dams that are higher than 15 m and have a reservoir larger than 0.1 km^3. To estimate river length, river network data were obtained from the HydroSHEDS at 30 arc-second (~1 km) resolution (Lehner et al., 2008). With dam numbers and river lengths, we calculated dam density (dams per 100 km of river length) in each watershed. We also included irrigated croplands from the Global Food Security-Support Analysis Data with 1 km resolution (Thenkabail et al., 2016). Using the size of irrigated croplands, we calculated the proportion of irrigation areas to total watershed areas.

Geological characteristics of watersheds included the size of each watershed, geographic distances between cities and watersheds, elevation, and slope. We calculated the size of watersheds and geographic distances between the centroids of cities and source watersheds using ArcGIS (ESRI, 2015). Elevation and slope data in river networks were gathered from Domisch et al. (2015) at 1 km resolution.

Cities' characteristics consisted of the presence of IWS programs, population size, the size of the urban economy, and climatic factors. IWS program data were collected from Romulo et al. (2018) and Bennett and Ruef (2016). IWS, a kind of payments for ES, are broader conservation strategies to provide and enhance freshwater ES with incentive-based mechanisms between the beneficiary and provider of watershed services (Huber-Stearns et al., 2015; Romulo et al., 2018). We included IWS programs that provided freshwater resources to a city in the City Water Map database and had a specific goal for drinking water protection (Bennett and Ruef, 2016; Romulo et al., 2018).

Average annual populations size from 2000 to 2010 were obtained from the World Urbanization Prospects report (UNDP, 2015). Spatially explicit GDP data in 2010 were obtained from the global dataset of gridded GDP and population scenarios at 0.5° (~50 km) resolution (Murakami and Yamagata, 2019). Climatic factors (annual mean temperature and annual precipitation) came from the WorldClim database at 1 km resolution (Hijmans et al., 2005). Since our dataset included spatially explicit data, we extracted variables at the watershed or city level by using zonal statistics in R (R Core Team, 2017). For example, we extracted the numeric values of elevation and slope in watersheds and GDP and climatic factors in cities.

5.2.4. Egocentric network analysis

We used multi-level models applied to an egocentric network analysis to estimate the contribution of each independent variable to freshwater ES supplies from source watersheds to cities (Wellman and Frank, 2001). Because cities usually have more than one source watershed, they form an egocentric network: ego is the city and alters are the source watersheds. Each tie and source watershed (alter) at the end of that tie is nested in each urban water network and the city (ego) to which that network belongs (Figure 5.1). Cities (egos) form an egocentric network by environmentally and socioeconomically interacting with source watersheds (alters) that supply freshwater ES to cities. Multi-level network models help examine the effects of the characteristics of each ego, its alters, and their ties to freshwater ES flows. We investigated how the characteristics of source watersheds and cities contribute to freshwater ES flows.

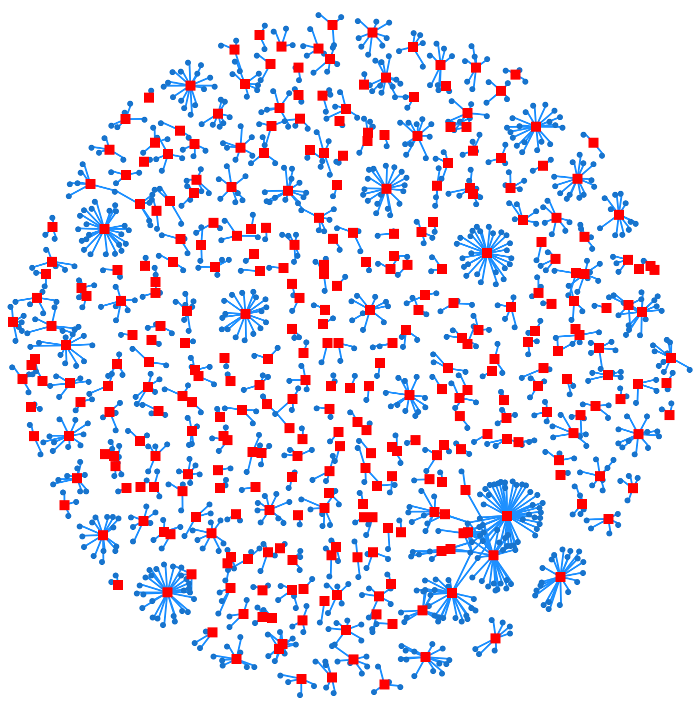

Figure 5.1. Egocentric networks between cities (egos) and freshwater source watersheds (alters). Red squares indicate cities, and blue circles indicate freshwater source watersheds. Each city has more than one source watershed, and thus they form an egocentric network.

The level 1 model includes the effects of the characteristics of alter (i) and tie (i, j), and the level 2 model includes the effects of the characteristics of ego (j). At level 1, we modeled changes in freshwater ES flows as a function of forest cover in PAs, wetland cover in PAs, dam density, irrigation area, watershed areas, distance from city to watershed, elevation, and slope. To estimate the effects of individual egos (j) on freshwater ES flows, the level one model's coefficients linking changes in flows to characteristics, β_{0j} , are used as an outcome in the level two model. At level 2, we modeled the intercept in the level 1 model as a function of the IWS program's presence, urban population size, urban GDP, temperature, and precipitation. The multi-level model for the flows of freshwater ES between alter (i) and ego (j) is as follows:

Level 1 (Alter and tie):

$$
\begin{aligned}
Freshwater\ ES_{ij} \\
&= \beta_{0j} + \beta_1 forest\ cover\ in\ PA_i + \beta_2 wetland\ cover\ in\ PA_i + \beta_3 dam\ density_i \\
&+ \beta_4 irrigation\ area_i + \beta_5 size\ of\ watersheds_i \\
&+ \beta_6 distance\ from\ urban\ area_{ij} + \beta_7 elevation_i + \beta_8 slope_i + e_i
\end{aligned}
$$

Level 2 (Ego):

$$
\begin{aligned}
\beta_{0j} = \gamma_{00} + \gamma_{01} IWS\ program_j + \gamma_{02} urban\ population_j + \gamma_{03} urban\ GDP_j \\
+ \gamma_{04} temperature_j + \gamma_{05} precipitation_j + u_{0j}
\end{aligned}
$$

For example, γ_{01} represents the effect of the presence of the IWS program. The errors at level 1, e_i, are assumed to follow a normal distribution (0, σ^2), and the level 2 errors, u_{0j}, are assumed to follow a normal distribution (0, τ_{00}). To pursue linearity and normality, we carried out natural log transformations on all variables. Then, we estimated multi-level models in R using the restricted maximum likelihood method (Bates et al., 2015; R Core Team, 2017). We also measured variance inflation factors (VIF) to check the multicollinearity of our multi-level models. Our VIF results showed that independent variables of multi-level models had no serious multicollinearity problems (Table S5.2).

5.3. Results

Forest cover in PAs of source watersheds had a negative relationship with the amount of sediment flux but was positively associated with hydropower production (Table 5.1). Watersheds with larger wetland cover in PAs had larger freshwater provisioning. The extent of forests and wetlands in PAs increased sustaining freshwater ES to cities except for flood mitigation. Watersheds with high dam density had low sediment flows and flood risks while having high hydropower production. However, dam density did not have a statistically significant effect on freshwater provisioning to cities. In fact, the estimate of dam density was considerably less than its standard error. Our results indicate that forest covers in PAs complemented dams for sediment reduction and hydropower production. The proportion of irrigation areas in source watersheds was negatively associated with freshwater provisioning to cities.

Geological characteristics in watersheds also contributed to the flows of freshwater ES to cities. Watersheds with larger watersheds and greater distances between watersheds and cities had a positive relationship with more freshwater provisioning, sediment flows, and hydropower production. Watersheds at lower elevations provided fewer freshwater and sediments. Steeper watersheds had larger sediment flows and hydropower production but lower flood risks.

City population size was positively associated with freshwater provisioning and sediment flows, while urban GDP was negatively associated with these two ES. In other words, cities with high population size and low GDP had not only high freshwater provision but also high sediment flows. Cities with higher average temperatures had larger sediment flows and hydropower production. Cities' precipitation had a positive association with freshwater provision. The presence of IWS programs in cities was not statistically significant with all four freshwater ES, but with a p-value less than 0.1, freshwater provisioning was at the margins of statistical significance ($P = 0.058$).

In addition, we identified the spatial locations of new large dams and PAs from 2000 to 2016 at the watershed level (Figure 5.2). We concentrated on freshwater source watersheds and hydropower watersheds, as both dams and PAs positively contributed to sediment reduction and hydropower production. From 2000 to 2016, new PAs were designated in 34.1% of freshwater source watersheds and 56.1% of hydropower watersheds without new large dam constructions. These watersheds were mainly located in North America and Europe (Figure 5.2). In the same period, areas in 4.8% of freshwater source watersheds and 2.8% of hydropower watersheds not only received new PA designations but also constructed new large dams worldwide. However, 2.9% of freshwater source watersheds and 3.8% of hydropower watersheds constructed large dams without new PA designations, of which approximately two-thirds were located in China and India (Figure 5.2). China and India did not designate new PAs in 97.3% of freshwater source watersheds and all hydropower watersheds over the period 2000 to 2016. In these two countries, 11.9% and 15.6% of their freshwater source watersheds and hydropower watersheds, respectively, constructed large dams without any new PA designations.

Table 5.1. Multi-level coefficients predicting four freshwater ecosystem services.

	Variable	Water Supply	Sediment Flow	Flood Risk	Hydro-power
Watershed (Level 1)	Forest cover in PAs (%)	-0.005 (0.026)	-0.094** (0.034)	0.085 (0.077)	0.241** (0.087)
	Wetland cover in PAs (%)	0.094* (0.044)	0.112. (0.057)	0.006 (0.104)	0.060 (0.168)
	Dam density (#/100 km of river length)	0.005 (0.040)	-0.154** (0.052)	-0.606* (0.271)	1.549** (0.340)
	Irrigation area (%)	-0.073* (0.029)	0.058 (0.037)	0.014 (0.045)	-0.053 (0.082)
	Watershed area (km^2)	0.148** (0.023)	0.309** (0.030)	-0.048 (0.059)	0.268* (0.114)
	Urban-watershed distance (km)	0.253** (0.053)	0.151* (0.069)	-0.107 (0.115)	0.530* (0.208)
	Elevation (meter)	-0.114** (0.042)	-0.245** (0.054)	-0.166 (0.135)	-0.275 (0.191)
	Slope (degree)	-0.032 (0.040)	0.138** (0.051)	-0.214** (0.061)	0.340** (0.121)
Urban (Level 2)	IWS program (0, 1)	-0.634. (0.333)	0.190 (0.355)	-0.115 (0.347)	0.443 (0.413)
	Urban population (1,000 persons)	0.218* (0.089)	0.219* (0.096)	0.149. (0.077)	0.008 (0.123)
	Urban GDP-PPP (2005 const. billion USD)	-0.166* (0.084)	-0.300** (0.090)	-0.055 (0.079)	-0.122 (0.130)
	Temperature (°C)	0.126 (0.220)	1.326** (0.239)	0.410* (0.183)	0.852* (0.380)
	Precipitation (mm)	0.980** (0.121)	0.074 (0.131)	0.081 (0.118)	-0.024 (0.177)
	Intercept	-5.009** (1.072)	-5.487** (1.175)	1.903 (1.509)	-2.240 (2.379)
	Random effect				
	City (Intercept)	1.651**	1.736**	0.620**	1.179**
	Residual	0.683**	1.211**	1.040**	1.890**
	N	1,323	1,323	767	497

Standard errors in parentheses: ** P < 0.01, * P < 0.05, . P < 0.1

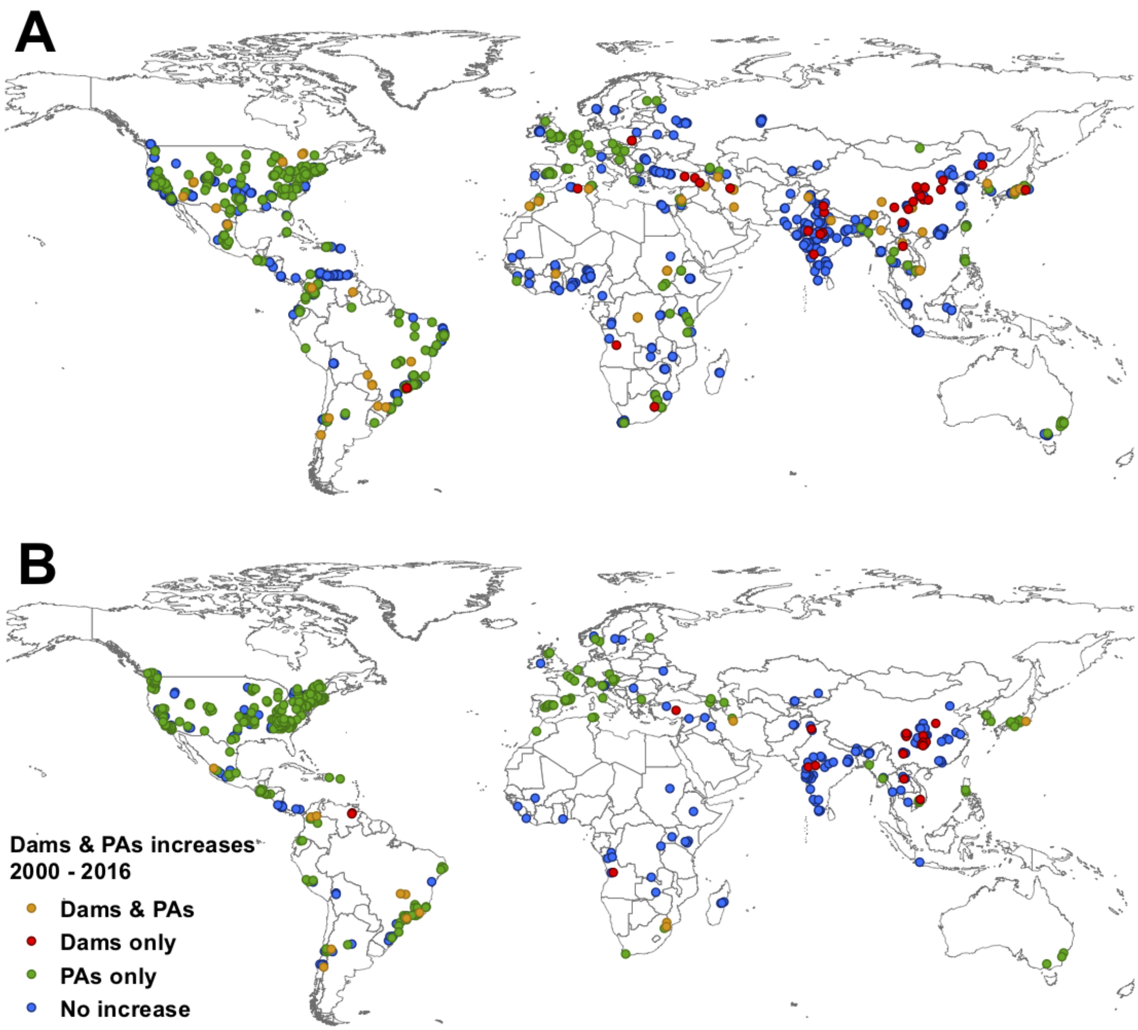

Figure 5.2. Spatial changes in numbers of dams and sizes of PAs from 2000 to 2016 in (A) freshwater source watersheds and (B) hydropower watersheds. Orange indicates an increase in the numbers of dams and size of PAs, red indicates an increase in only dams, green indicates an increase in only PAs, and blue indicates no increases in dams' numbers or PAs' sizes in each watershed from 2000 to 2016.

5.4. Discussion

5.4.1. The role of forests and wetlands in protected areas

This study examines a possible way to integrate watershed conservation activities (PA and IWS) with built infrastructure for sustainably providing four freshwater ES for cities. Forest cover in PAs and dams both provide sediment reduction. In addition, forest PAs increase hydropower production over what would be expected from dams alone. Protected forests in source watersheds help decrease sediment flows because forest covers reduce soil erosion with tree root systems, high infiltration rates, and low overland flows (Blumenfeld et al., 2009; Gartner et al., 2013; Vercruysse et al., 2017). High evapotranspiration rates in protected forests can reduce overland runoff and therefore sediment generation and transport (Edwards et al., 2014; Wang et al., 2011; Wu et al., 2014). Upstream protected forests may also enhance the longevity of dams with the reduction of sediment flows to a reservoir (Bilotta and Brazier, 2008). Furthermore, protected forests can provide sustainable water sources for hydropower production by influencing river discharge via rainfall and soil moisture (Moran et al., 2018; Stickler et al., 2013). Wetland PAs help sustain the amounts of surface water supplies to cities. Protected wetlands retain water in wetland soils and vegetation, and the water gradually flows into streams and rivers (Maltby and Acreman, 2011). But cities mainly depend on dams for flood mitigation.

Our results show that PAs can enhance various freshwater ES supplies for cities, in addition to their main purpose of biodiversity conservation. Thus, integrating built infrastructure with PAs contributes to maintaining freshwater ES supplies to meet cities' demands while conserving natural habitats for biodiversity (Visconti et al., 2019; Vörösmarty et al., 2018; Yang et al., 2019). Watershed conservation activities for freshwater ES flows could also support the global sustainable development agenda (Vörösmarty et al., 2018). Integrating the two approaches can have co-benefits for multiple SDGs simultaneously, including freshwater sources (SDG 6, clean water and sanitation), hydropower production (SDG 7, affordable & clean energy), dams (SDG 9, industry, innovation of infrastructure), cities (SDG 11, sustainable cities & communities), and biodiversity (SDG 15, life on land) (Bhaduri et al., 2016; Garrick et al., 2017; United Nations, 2015; Vörösmarty et al., 2018).

5.4.2. Conservation strategies for freshwater ES

This study highlights important implications for new PA designations that help sustain freshwater ES supplies to cities. As expected, increased urban populations had a positive relationship with the amounts of freshwater provision and sediment loads from source watersheds. However, increased affluence in cities was negatively associated with both freshwater provision and sediment loads, suggesting an ameliorating impact of affluence. Cities with low affluence may have to use low quality of freshwater, partly because of the lack of water infrastructure and conservation activities in their source watersheds (McDonald et al., 2014; Romulo et al., 2018). New PA designations in such cities’ freshwater source watersheds may be crucial to reduce sediment flows, because high sediments in source watersheds cause additional costs for urban water treatment (Bilotta and Brazier, 2008; McDonald et al., 2016; Vercruysse et al., 2017). Such PA designations, however, need to consider other social, economic, and political contexts to avoid potential conflicts with local communities (Dinerstein et al., 2019; Kremen and Merenlender, 2018; Symes et al., 2016). In addition, increased temperature tended to increase sediment load and flood risk for global cities by affecting the availability of water resources. For instance, high temperature may increase sediment flows from source watersheds with the reduction of vegetation cover as well as the loss of ground aggregates (Achite and Ouillon, 2007; Haritashya et al., 2006). A warmer climate with high temperatures may also raise flood risks worldwide (Hirabayashi et al., 2013). If this cross-sectional relationship holds with temporal changes, climate changes could exacerbate these problems.

Our analysis can also help target areas where conservation actions might improve the flows of multiple freshwater ES. Source watersheds, particularly in China and India, have largely focused on dam construction without new PA designations from 2000 to 2016. While PAs appear not to change the flood protection services from dams, PA designations in the source watersheds of these dams could add to sediment reduction and hydropower production. Additionally, although our network analyses included geological characteristics (i.e., watershed size, distance to cities, elevation, and slope) largely to avoid spurious effects, results indicate that these geological characteristics played an important role in freshwater ES supplies. Source watersheds with a larger size, greater distance from cities, and steeper slope have less sediment flow and more hydropower production. Source watersheds with a larger size, greater distance from cities, and lower elevation have more freshwater provision while reducing sediment flows.

Other types of watershed conservation activities would also work for sustaining freshwater ES supplies in highly developed watersheds. Our results show that there was a conflict for freshwater resources between irrigated croplands in source watersheds and cities' water demand. In the context of water resource conflicts, IWS programs might be an alternative to PAs for freshwater provisioning and could possibly balance the negative effects of irrigated croplands to meet the freshwater demand that comes from population and affluence (Romulo et al., 2018; Zheng et al., 2013). Many cities adopted IWS programs after experiencing low freshwater provisioning, partly because of high irrigation withdrawals in source watersheds (McDonald et al., 2016; Romulo et al., 2018; Veldkamp et al., 2017; Zheng et al., 2013). In source watersheds, irrigation demands from agriculture are driven by food demand beyond city and watershed boundaries (Dalin et al., 2012; Dalin et al., 2017; Soligno et al., 2019). For instance, in India, which accounts for 6.8% of the global net increase in green leaf areas from 2000 to 2017, cropland increases contributed to 82% of that net increase (Chen et al., 2019). Thus, future watershed conservation activities in India might find the IWS approach a useful tool for improving freshwater provisioning, although further research is clearly needed to establish more firmly the effects of IWS programs.

In some countries, other types of payments for ES may also have indirect effects on freshwater ES supplies to cities. For example, in 2000, China implemented the world's largest forest conservation program—the Natural Forest Conservation Program (NFCP)—to conserve and restore forests. The NFCP in China has significantly contributed to net increases in forest cover over the past two decades (Chen et al., 2019; Viña et al., 2016). Since the NFCP bans and monitors illegal logging and harvesting in natural forests (Viña et al., 2016), conservation and restoration of forests under this program may provide additional freshwater ES to cities (Ouyang et al., 2016). These watershed conservation activities can be expanded to other regions that experience rapid dam construction and high levels of human intervention without any watershed conservation efforts.

Our study has several limitations. We note that our multi-level models correlate rather than estimate causal directions among dams, conservation activities, and freshwater ES flows because of the lack of time-series data. These interrelationships may be altered with seasonal changes of freshwater ES supplies and different changes between dam numbers and conservation activities over time.

5.5. Conclusions

This study determines the relationships of built infrastructure and watershed conservation activities with freshwater ES for cities. Our findings and approach provide a new perspective to the link between cities that demand freshwater ES and the watersheds that provide them. From a practical point of view, our analyses suggest ways that watershed conservation activities can enhance the function of built infrastructure in providing sustainable freshwater ES supplies in urban water systems. One of the best ways is to integrate dam construction and conservation policies. Our results may indicate that forest cover in PAs improves the capacity of large dams in sediment reduction and hydropower production. Wetland cover in PAs helps provide further freshwater provisioning to cities. In our analysis, dams mainly contribute to flood mitigation in cities, although of course this global pattern undoubtedly has many regional exceptions. In conclusion, our findings capture the role of watershed conservation activities for freshwater ES supplies to global cities as well as their potential provision of additional ES that can enhance the function of large dams. Therefore, we hope this study sets groundwork for future research.

CHAPTER 6

CONCLUSIONS

By integrating the telecoupling framework with ecosystem services, this dissertation uncovers complex telecoupling processes of tourism, food trade, and flows of fresh water with biodiversity, conservation activities, and human well-being across distant places. My research includes these important telecoupling processes regarding the dynamic flows of ecosystem services at the global level. Some of the greatest challenges revolve around increasing pressures on food production for exports in tropical countries, water demands in agriculture and urban areas, and tourism in conservation areas that have high cultural heritages and ecological hotspots.

The second chapter of this dissertation examines how different conservation strategies affect nature-based tourism in terrestrial protected areas. My results show that protected areas strictly managed for biodiversity protection have 35% more visitations than those managed for mixed use. In addition, biodiversity is positively associated with the number of nature-based tourists even when other environmental and socioeconomic factors are controlled. Management for biodiversity has a positive relationship with the number of species. Therefore, the results imply that enhancing both biodiversity and nature-based tourism together is feasible given suitable conservation strategies.

The third chapter of this dissertation adopts a network analysis approach that determines the dynamics of global tourism networks and identifies underlying socioeconomic and environmental factors over time. My results confirm the consolidation of global tourism networks and identify which countries have rapidly contributed to this consolidation over the past two decades. In consolidated global tourism networks, my mixed-effects model provides key strategic factors for proper tourism development and destination management. For example, results indicate that transaction costs (e.g., shared language, geographic distance, and visa policy) are more important in attracting international tourists than natural and cultural attractions (e.g., protected areas and World Cultural Heritage sites). My network approach and findings provide a science-based evidence for decision-making to implement proactive tourism policies.

The fourth chapter of this dissertation performs comprehensive global analyses to reveal unexpected food trade among developed and developing countries with or without biodiversity hotspots. This research combines biodiversity hotspots and economic development status and provides a new perspective that international food trade may benefit biodiversity conservation in both developing and developed countries with biodiversity hotspots. Additionally, my results indicate that developing countries without biodiversity hotspots played an increasingly important role as net exporters in international food trade. With rising attention to biodiversity conservation and food security, it is time to develop innovative approaches to minimize the negative impacts of global food production and trade on biodiversity hotspots.

The fifth chapter of this dissertation determines a feasible way to integrate watershed conservation activities with built infrastructure approaches to sustain essential freshwater ecosystem services for global cities. This research adopts an egocentric network analysis to investigate the complex relationships of watershed conservation activities and built infrastructure on the flows of key freshwater ecosystem services from source watersheds to cities worldwide. My results show that forest cover in protected areas complement large dams for sediment reduction and hydropower production. In addition, wetland cover in protected areas contributes to provide further freshwater provisioning to cities. However, global cities mainly depend on large dams for flood mitigation. My network approach and findings capture the role of watershed conservation activities for freshwater ecosystem services to integrate built infrastructure with watershed conservation activities to enhance urban water sustainability.

To sum up, this dissertation research can help reduce the negative impacts of ecosystem service flows on sustainable development as my results provide science-based knowledge for implementing cross-border policies and landscape planning that works to achieve environmental and socioeconomic sustainability. My approaches and findings also suggest the potential benefits of key ecosystem services with different conservation activities to inform policymakers, land managers, and other stakeholders, such as ecotourists, farmers, and urban residents. The dissertation's results are transformative for promoting benefits for various beneficiaries as this dissertation demonstrates how ecosystem service flows benefit different beneficiaries and those paying subsidies for global ecosystem service conservation.

Future research beyond this dissertation can uncover the complex interactions of multiple telecoupling processes (e.g., food-water-energy nexus trade) with diverse ecosystem services across the world. Integrating multiple telecouplings simultaneously is more challenging than focusing on a single telecoupling because telecoupling processes interact with each other across spatial and temporal scales (Liu et al., 2018; Liu et al., 2015b). Quantifying the dynamics of multiple telecouplings with key ecosystem services is essential to develop practical policies and land management strategies for global sustainable development. For example, food production for exports surrounding protected areas may threaten charismatic species that play an important role to attract nature-based tourists. Also, the increasing water uses for irrigation in source watersheds may decrease freshwater provisioning to urban areas. Establishing protected areas in source watersheds may enhance the retention of fresh water, while providing natural attractions for tourists. Transforming the understanding of multiple telecoupling processes and their cross-sectoral interactions with ecosystem services can help decision-makers formulate adaptive management priorities for natural lands in ways that promote target biodiversity while provisioning essential ecosystem services to beneficiaries. This dissertation provides a foundation for future research by integrating the telecoupling framework with ecosystem services at the global level. This dissertation can help with the development of win-win proactive strategies for both biodiversity and ecosystem service conservation to enhance global sustainability.

APPENDICES

APPENDIX A

SUPPORTING INFORMATION FOR CHAPTER 2

Table S2.1. The descriptions of dependent and independent variables.

	Variable	Dataset	Unit of Measure	Time Period	Spatial Extent	Reference	Link
Nature-based Tourism	PAs visitor numbers	Annual visitation data for PAs	person	2000-2014	Global	Balmford et al. (2015), National Statistics, Gray literatures	http://journals.plos.org/plosbiology/article?id=10.1371/journal.pbio.1002074
Demographic	Population	LandScan	person (30arc second~1km^2)	2002-2012	Global, raster	Bright et al. (2013)	http://web.ornl.gov/sci/landscan
Biodiversity	The number of species	Birds, Mammals, Amphibians	species	2000s	Global, raster	Pimm et al. (2014)	http://biodiversitymapping.org
Agricultural factor	Agricultural Yields	Yields for 175 crops	tonne/km^2 (5arc minute~10km^2)	2000	Global, raster	(Monfreda et al., 2008)	http://www.earthstat.org/data-download
	Agricultural area	Cultivated and managed vegetation area	proportion (0-100, 30arc second~1km^2)	the early 2000s	Global, raster	Tuanmu and Jetz (2014)	http://www.earthenv.org/landcover.html
Regulating ES	Upstream Protected Land	% total water supply originated in protected land	% of total water supply originated in PAs	2000s	Global, watershed	WRI (2015)	http://www.wri.org/resources/data-sets/aqueduct-global-maps-21-data
Economic	Income level	GDP per capita	2005 USD const.	2000-2014	Country level	United Nations Statistics Division (2015)	http://data.un.org
Protected Area	PAs management status	IUCN PAs management category	Category, II-VI	2014	Global	IUCN and UNEP-WCMC (2017)	http://www.protectedplanet.net
	PAs age	Subtraction of PAs establishment year from 2014	year	2014	Global	IUCN and UNEP-WCMC (2017)	http://www.protectedplanet.net
	Size of PAs	PAs areas	square kilometers	2014	Global	IUCN and UNEP-WCMC (2017)	http://www.protectedplanet.net

Table S2.1 (cont'd)

	Variable	Dataset	Unit of Measure	Time Period	Spatial Extent	Reference	Link
Protected Area	Elevation	Global Multi-resolution Terrain Elevation (GMTED 2010)	Elevation (30arc second~1km2)	2010	Global, raster	EROS Data Center (2015)	http://earthexplorer.usgs.gov https://lta.cr.usgs.gov/GMTED2010
	Temperature	Annual mean temperature	°C (30 arc second~1km2)	1950-2000	Global, raster	Hijmans et al. (2005)	http://www.worldclim.org
	Precipitation	Annual mean precipitation	millimeter (30 arc second~1km2)	1950-2000	Global, raster	Hijmans et al. (2005)	http://www.worldclim.org
	PAs remoteness from major cities (>50,000)	Travel time to major cities: A global map of Accessibility	Time (minutes) (30 arc second~1km2)	2000	Global, raster	Nelson (2008)	http://forobs.jrc.ec.europa.eu/products/gam/download.php

Table S2.2. List of PAs (N=929).

Africa (N=90)	
Cameroon	Mbam et Djerem (II), Nki (II), Waza (II)
Republic of the Congo	Nouabale-Ndoki (II)
Ethiopia	Bale Mountains (II)
Ghana	Bomfobiri (IV), Bui (II), Digya (II), Gbele (VI), Kakum (II), Kalakpa (VI), Mole (II), Owabi (IV), Shai Hills (VI)
Kenya	Aberdare (II), Amboseli (II), Arabuko Sokoke (II), Hell's Gate (II), Lake Bogoria (II), Lake Nakuru (II), Longonot (II), Meru (II), Mount Kenya (II), Nairobi (II), Samburu (II), Shimba Hills (II), Tsavo East (II), Tsavo West (II)
Madagascar	Analamerana (IV), Montagne d'Ambre (II)
Namibia	Ai-Ais Hot Springs (II), Cape Cross Seal Reserve (IV), Daan Viljoen Game Park (II), Etosha (II), Gross Barmen Hot Springs (III), Hardap Recreation Resort (V), Khaudum (II), Mangetti (II), Mudumu (II), Nkasa Rupara (II), Popa Game Park (III), Skeleton Coast Park (II), Von Bach Recreation Resort (V), Waterberg Plateau Park (II)
Rwanda	Akagera (II), Nyungwe (IV), Volcans (II)
Tanzania	Arusha (II), Gombe (II), Katavi (II), Kilimanjaro (II), Kitulo (II), Lake Manyara (II), Mahale (II), Mikumi (II), Mkomazi (IV), Ruaha (II), Rubondo (II), Selous (IV), Serengeti (II), Tarangire (II), Udzungwa Mountains (II)
Uganda	Bwindi Impenetrable (II), Katonga (III), Kidepo Valley (II), Lake Mburo (II), Mgahinga Gorilla (II), Mount Elgon (II), Murchison Falls (II), Queen Elizabeth (II), Rwenzori Mountains (II), Semuliki (II)
South Africa	Agulhas National Park (II), Augrabies Falls National Park (II), Bontebok National Park (II), Golden Gate Highlands National Park (II), Kalahari Gemsbok National Park (II), Karoo National Park (II), Kruger National Park (II), Mapungupwe National Park (II), Marakele National Park (II), Mokala National Park (II), Mountain Zebra National Park (II), Namaqua National Park (II), Richtersveld National Park (II), Table Mountain National Park (II), Tankwa-Karoo National Park (II), Vaalbos National Park (II)
Zambia	Kasanka (II), Lavushi Manda (II)

Table S2.2. (cont'd)

Asia and Oceania (N=351)	
UAE	Dubai Desert Conservation Reserve (II)
Australia	Ben Lomond (II), Douglas-Apsley (II), Hartz Mountains (II), Hastings Caves (III), Kakadu National Park (II), Mole Creek Karst (II), Moreton Island (II), Mount Field (II), Purnululu (II), Uluru-Kata Tjuta National Park (II)
China	Dafengmilu (Jiangsu) (V), Huanglongsi (V), Jiuzhaigou (V), Wolong (V), Wuyishan (V)
Indonesia	Alas Purwo (II), Baning (V), Bantimurung Bulusaraung (II), Batang Gadis (II), Batu Angus (V), Batu Putih (V), Berbak (II), Betung Kerihun (II), Bogani Nani Wartabone (II), Bromo Tengger Semeru (II), Bukit Baka - Bukit Raya (II), Bukit Barisan Selatan (II), Bukit Dua Belas (II), Bukit Kaba (V), Bukit Serelo (V), Bukit Tiga Puluh (II), Bunder (VI), Camplong (V), Cani Sirenreng (V), Carita (VI), Cimanggu (V), D. Sicikeh-cikeh (V), Danau Matano (V), Danau Sentarum (II), Grojogan Sewu (V), Gunung Baung (V), Gunung Ciremai (II), Gunung Gede - Pangrango (II), Gunung Guntur (V), Gunung Halimun - Salak (II), Gunung Kelam (V), Gunung Leuser (II), Gunung Meja (V), Gunung Merapi (II), Gunung Merbabu (II), Gunung Palung (II), Gunung Pancar (V), Gunung Rinjani (II), Gunung Tampomas (V), Holiday Resort (V), Jember (V), Kawah Ijen (V), Kawah Kamojang (V), Kayan Mentarang (II), Kelimutu (II), Kerandangan (V), Kerinci Seblat (II), Klamono (V), Kutai (II), Laiwangi Wanggameti (II), Lejja (V), Lore Lindu (II), Madapangga (V), Malino (V), Mangolo (V), Manupeu Tanadaru (II), Manusela (II), Meru Betiri (II), Minas (Sultan Sarif Hasyim) (VI), Muka Kuning (V), Nanggala III (V), Papandayan (V), Pulau Kembang (V), Punti Kayu (V), Rawa Aopa Watumohai (II), Rimbo Panti (V), Ruteng (V), Sebangau (II), Seblat (V), Semongkat (V), Siberut (II), Sidrap (V), Sorong (V), Sultan Adam (VI), Suranadi (V), Tahura Ir. H. Juanda (VI), Talaga Bodas (V), Telaga Patengan (V), Telogo Warno Pengilon (V), Tesso Nilo (II), Tretes (V), Way Kambas (II)
India	Bandhavgarh (II), Bandipur (II), Bhadra (IV), Corbett (II), Jaldapara (IV), Kalakad (IV), Kanha (II), Kaziranga (II), Ken Gharial (IV), Melghat (IV), Mudumalai (IV), Panna (II), Pench (II), Periyar (IV), Rajiv Gandhi (Nagarhole) (II), Ranthambhore (II), Sariska (IV), Satpura (II), Valley Of Flowers (II)
Japan	Aichikogen (V), Akan (II), Akiyoshidai (V), Aso kuju (V), Bandai asahi (II), Biwako (V), Chichibu tama kai (V), Chubusangaku (II), Daisetsuzan (II), Echigosanzan-Tadami (II), Hakusan (II), Hayachine (II), Hiba-Dogo-Taishaku (V), Hida-Kisogawa (V), Hyonosen-Ushiroyama-Nagisan (V), Ibi-Sekigahara-Yoro (V), Iriomote (IV), Ishizuchi (V), Kitakyushu (V), Kongo-Ikoma-Kisen (V), Koya-Ryujin (V), Kurikoma (II), Kushiroshitsugen (II), Kyushuchuosanchi (V), Meiji Memorial Forest Minoo (V), Meiji Memorial Forest Takao (V), Minami alps (II), Muroo-Akame-Aoyama (V), Myogi-Arafune-Sakukogen (V), Nikko (V), Nishichugokusanchi (V), Onuma (V), Shikotsu toya (II), Shiretoko (IV), Sobo-Katamuki (V), Suzuka (V), Tanzawa-Oyama (V), Tenryu-Okumikawa (V), Towada hachimantai (II), Tsurugisan (V), Yaba-Hita-Hikosan (V), Yamato-Aogaki (V), Yatsugatake-Chushinkogen (V), Zao (V)

Table S2.2. (cont'd)

South Korea	Biseulsan County Park (V), Bogyeongsa County Park (V), Bongmyeongsan County Park (V), Bukhansan (V), Bullyeonggyegok County Park (V), Cheongnyangsan Provincial Park (V), Cheongwansan Provincial Park (V), Cheonmasan County Park (V), Chiaksan (II), Chilgapsan Provincial Park (V), Daedunsan Provincial Park (V), Daeiri County Park (V), Deogyusan (V), Duryunsan Provincial Park (V), Gajisan Provincial Park (V), Gangcheonsan County Park (V), Gayasan (II), Geumosan Provincial Park (V), Gibaeksan County Park (V), Gobok Provincial Park (V), Gwangyang Baegunsan (IV), Gyoungpo Provincial Park (V), Hwangmaesan County Park (V), Hwawangsan County Park (V), Ipgok County Park (V), Jangansan County Park (V), Jirisan (II), Juwangsan (II), Maisan Provincial Park (V), Moaksan Provincial Park (V), Mungyeongsaejae Provincial Park (V), Myeongjisan County Park (V), Naejangsan (II), Naksan Provincial Park (V), Namhansanseong Provincial Park (V), Odaesan (II), Sangnim Woods in Hamyang (IV), Seonunsan Provincial Park (V), Seoraksan (II), Sobaeksan (II), Songnisan (II), Taebaeksan Provincial Park (V), Unmunsan County Park (V), Upo Wetland (IV), Valley of Bulyeongsa Temple in Uljin (V), Wolchulsan (II), Woraksan (II)
Sri Lanka	Bundala (II), Gal Oya (II), Galway's Land (IV), Horton Plains (II), Kaudulla (II), Lahugala (II), Lunugamwehera (II), Maduru Oya (II), Minneriya (II), Uda Walawe (II), Wasgamuwa (II), Wilpattu (II), Yala East(Kumana) (II)
Malaysia	Batang Ai (II), Endau Rompin (Johor) (II), Gunong Gading (II), Kubah (II), Lambir Hills (II), Loagan Bunut (II), Niah (II), Taman Negara (II), Tanjong Datu (II), Tawau Hill Park (II)
Thailand	Bang Lang (II), Budo-Sungai Padi (II), Chae Son (II), Chalearm Rattanakosin (II), Doi Inthanon (II), Doi Khuntan (II), Doi Luang (II), Doi Phaklong (II), Doi Phukha (II), Doi Suthep-Pui (II), Erawan (II), Huai Nam Dang (II), Kaeng Krung (II), Kaeng Tana (II), Kaengkrachan Forest Complex (II), Khao Chamao-Khao Wong (II), Khao Khitchakut (II), Khao Laem (II), Khao Luang (II), Khao Nam Khang (II), Khao Nan (II), Khao Phanom Bencha (II), Khao Phravihan (II), Khao Pu - Khao Ya (II), Khao Sib Ha Chan (II), Khao Sok (II), Khao Yai (II), Khlong Lamngu (II), Khlong Lan (II), Khlong Wang Chao (II), Khuen Si Nakarin (II), Khun Chae (II), Khun Pra Vor (II), Klong Phanom (II), Kuiburi (II), Lansaang (II), Mae Charim (II), Mae Moei (II), Mae Phang (II), Mae Ping (II), Mae Puem (II), Mae Wa (II), Mae Wang (II), Mae Wong (II), Mae Yom (II), Mukdahan (II), Nam Nao (II), Nam Phong (II), Namtok Chat Trakan (II), Namtok Huai Yang (II), Namtok Klong Kaew (II), Namtok Mae Surin (II), Namtok Ngao (II), Namtok Phleiw (II), Namtok Sai Khao (II), Namtok Si khid (II), Namtok Yong (II), Ob Luang (II), Pa Hin Ngam (II), Pang Sida (II), Pha Tam (II), Phu Chong - Na Yoi (II), Phu Hin Rong Kla (II), Phu Kao - Phu Phan Kham (II), Phu Kradueng (II), Phu Lan Ka (II), Phu Langka (II), Phu Pa - Yol (Huai Huat) (II), Phu Pha Lek (II), Phu Pha Man (II), Phu Phan (II), Phu Rua (II), Phu Sa Dokbua (II), Phu Soi Dao (II), Phu Toei (II), Phu Wiang (II), Phu Zang (II), Ramkamhaeng (II), Sai Thong (II), Sai Yok (II), Salawin (II), Si Nan (II), Si Phangnga (II), Sri Lanna (II), Sri Satchanalai (II), Ta Phraya (II), Taad Moak (II), Taad Ton (II), Tai Romyen (II), Taksin Maharat (II), Thaleban (II), Tham Pla - Pha Seu (II), Thap Lan (II), Thong Pha Phum (II), Thung Salaeng Luang (II), Wiang Kosai (II)

Table S2.2. (cont'd)

Nepal	Annapurna (VI), Api - Nampa (VI), Bardia (II), Chitwan (II), Dhorpatan (VI), Gauri-Shankar (VI), Kanchanjunga (VI), Khaptad (II), Koshi Tappu (IV), Krishnasar (VI), Langtang (II), Makalu-Barun (II), Manaslu (VI), Parsa (IV), Rara (II), Shey-Phoksundo (II), Shivapuri-Nagarjun (II), Suklaphanta (IV)
New Zealand	Abel Tasman (II)
Philippines	Mount Kitanglad Range (II), Mt. Pulag National Park (II), Puerto Princesa Subterranean River (III)
Vietnam	Cuc Phuong (II), Phong Nha-Ke Bang (II)
Europe (N=215)	
Bulgaria	Centralen Balkan (II), Vitosha (V)
Czech Republic	Česke Švycarsko (II), Krkonošsky narodni park (V), Šumava (II)
Finland	Helvetinjarven kansallispuisto (II), Hiidenportin kansallispuisto (II), Isojarven kansallispuisto (II), Kauhanevan-Pohjankankaan kansallispuisto (II), Kolin kansallispuisto (II), Koloveden kansallispuisto (II), Kurjenrahkan kansallispuisto (II), Lauhanvuoren kansallispuisto (II), Leivonmaen kansallispuisto (II), Liesjarven kansallispuisto (II), Linnansaaren kansallispuisto (II), Nuuksion kansallispuisto (II), Paijanteen kansallispuisto (II), Patvinsuon kansallispuisto (II), Petkeljarven kansallispuisto (II), Puurijarven ja Isonsuon kansallispuisto (II), Pyha-Hakin kansallispuisto (II), Repoveden kansallispuisto (II), Rokuan kansallispuisto (II), Salamajarven kansallispuisto (II), Seitsemisen kansallispuisto (II), Sipoonkorven kansallispuisto (II), Tiilikkajarven kansallispuisto (II), Torronsuon kansallispuisto (II), Valkmusan kansallispuisto (II)
UK	Arundel Park (IV), Attenborough Gravel Pits (IV), Aylesbeare Common (IV), Berney Marshes & Breydon Water (IV), Blean Woods (IV), Brampton Wood (IV), Brandon Marsh (IV), Brecon Beacons (V), Broads (V), Cairngorms (V), Castle Eden Dene (IV), Clifton Country Park (IV), Clumber Park (V), Coombe Valley Woods (IV), Danbury and Lingwood Commons (V), Dartmoor (V), Dungeness (IV), Elmley (IV), Epping Forest (IV), Exe Estuary (IV), Exmoor (V), Fairburn Ings (IV), Fowlmere (IV), Frampton Pools (IV), Gamlingay Wood (IV), Garston Wood (IV), Geltsdale (IV), Gibside (V), Ham Wall (IV), Havergate Island & Boyton Marshes (IV), Haweswater (IV), Hodbarrow (IV), Ken-Dee Marshes (IV), Lake District (V), Leighton Moss (IV), Loch Lomond and The Trossachs (V), Lochwinnoch (IV), Marshside (IV), Mere Sands Wood (IV), Mid Yare Valley (IV), Minsmere (IV), Nagshead (IV), Nene Washes (IV), New Forest (V), North Warren (IV), North York Moors (V), Northumberland (V), Northward Hill (IV), Oare Marshes (IV), Ogden Water (IV), Orford Ness (V), Otmoor (IV), Ouse Washes (IV), Parndon Woods & Common (IV), Peak District (V), Pembrokeshire Coast (V), Poole's Cavern and Grin Low Wood (IV), Pulborough Brooks (IV), Queenswood (IV), Radipole Lake (IV), Rye Meads (IV), Sherwood Forest (IV), Snettisham Carstone Quarry (IV), Snowdonia (V), South Downs (V), The Lodge (IV), Titchfield Haven (IV), Titchmarsh (IV), Tudeley Woods (IV), West Sedgemoor (IV), Wolves Wood Reserves (IV), Wood Of Cree (IV), Yorkshire Dales (V)

Table S2.2. (cont'd)

Spain	Aiguestortes i Estany de Sant Maurici (II), Cabaneros (II), Donana (II), El Teide (II), Garajonay (II), La Caldera de Taburiente (II), Ordesa y Monte Perdido (II), Parque Nacional de Timanfaya (II), Picos de Europa (II), Sierra Nevada (II), Tablas de Daimiel (II)
France	Causses du Quercy (V), La Narbonnaise en Mediterranee (V), Volcans d'Auvergne (V)
Croatia	Krka (II), Paklenica (II), Plitvicka jezera (II), Risnjak (II), Sjeverni Velebit (II)
Hungary	Aggteleki (II), Balaton-felvideki (V), Bukki (II), Duna-Drava (V), Duna-Ipoly (V), Ferto-Hansagi (II), Hortobagyi (II), Kiskunsagi (II), Koros-Maros (V), Orsegi (V)
Italy	Parco nazionale dei Monti Sibillini (II), Parco regionale La Mandria (V)
Poland	Babiogorski Park Narodowy (II), Białowieski Park Narodowy (II), Biebrzański Park Narodowy (II), Bieszczadzki Park Narodowy (II), Drawieński Park Narodowy (II), Gorczański Park Narodowy (II), Kampinoski Park Narodowy (II), Karkonoski Park Narodowy (II), Łomżyński Park Krajobrazowy Doliny Narwi (V), Magurski Park Narodowy (II), Narwiański Park Narodowy (II), Ojcowski Park Narodowy (V), Park Narodowy "Bory Tucholskie" (II), Park Narodowy "Ujście Warty" (II), Park Narodowy Gor Stołowych (II), Pieniński Park Narodowy (II), Poleski Park Narodowy (II), Roztoczański Park Narodowy (II), Świętokrzyski Park Narodowy (II), Tatrzański Park Narodowy (II), Wielkopolski Park Narodowy (II), Wigierski Park Narodowy (V)
Portugal	Alvao (V), Arriba Fossil da Costa da Caparica (V), Douro Internacional (V), Estuario do Sado (IV), Estuario do Tejo (IV), Montesinho (V), Paul de Arzila (IV), Paul do Boquilobo (IV), Peneda-Geres (II), Ria Formosa (V), Sapal de Castro Marim e Vila Real de Santo Antonio (IV), Serra da Estrela (V), Serra da Malcata (IV), Serra de Sao Mamede (V), Serra do Acor (V), Serras de Aire e Candeeiros (V), Sintra-Cascais (V), Tejo Internacional (V), Vale do Guadiana (V)
Romania	Balta Mica a Brailei (V), Bucegi (V), Buila - Vanturarita (II), Calimani (II), Ceahlau (II), Cheile Bicazului - Hasmas (II), Cheile Nerei - Beusnita (II), Comana (V), Cozia (II), Defileul Jiului (II), Domogled - Valea Cernei (II), Geoparcul Dinozaurilor Tara Hategului (V), Geoparcul Platoul Mehedinti (V), Gradistea Muncelului - Cioclovina (V), Lunca Joasa a Prutului Inferior (V), Lunca Muresului (V), Muntii Apuseni (V), Muntii Macinului (II), Muntii Maramuresului (V), Piatra Craiului (II), Portile de Fier (V), Putna - Vrancea (V), Retezat (II), Rodna (II), Semenic - Cheile Carasului (II), Vanatori Neamt (V)
Russia	Kenozersky (II)
Slovakia	Biele Karpaty (V), Cerova vrchovina (V), Horna Orava (V), Kysuce (V), Mala Fatra (II), Muranska planina (II), Pieninsky (II), Polana (V), Poloniny (V), Slovensky raj (II), Stiavnicke vrchy (V), Strazovske vrchy (V), Tatransky (II)

Table S2.2. (cont'd)

Central and South America (N=155)	
Argentina	Baritu (II), Bosques Petrificados (III), Calilegua (II), Campo de los Alisos (II), Chaco (II), El Leoncito (II), El Palmar (II), El Rey (II), Iguazu (II), Lago Puelo (II), Laguna Blanca (II), Laguna de los Pozuelos (III), Lanin (II), Lihue Calel (II), Los Alerces (II), Los Cardones (II), Los Glaciares (II), Mburucuya (II), Nahuel Huapi (II), Perito Moreno (II), Pre-Delta (II), Quebrada del Condorito (II), Rio Pilcomayo (II), San Guillermo (II), Sierra de las Quijadas (II), Talampaya (II)
Belize	Bermudian Landing Community Baboon Sanctuary (IV)
Bolivia	Eduardo Avaroa (IV), Madidi (II), Noel Kempff Mercado (II)
Brazil	Area De Protecao Ambiental Da Chapada Dos Guimaraes (V), Floresta Nacional De Brasilia (VI), Parque Nacional Da Chapada Dos Veadeiros (II), Parque Nacional Da Serra Da Canastra (II), Parque Nacional Da Serra Da Capivara (II), Parque Nacional Da Serra Da Cipo (II), Parque Nacional Da Serra Do Divisor (II), Parque Nacional Da Serra Dos Orgaos (II), Parque Nacional Da Tijuca (II), Parque Nacional Das Emas (II), Parque Nacional De Aparados Da Serra (II), Parque Nacional De Caparao (II), Parque Nacional De Sete Cidades (II), Parque Nacional De Ubajara (II), Parque Nacional Do Itatiaia (II), Parque Nacional Serra Das Confusoes (II)
Chile	Alerce Andino (II), Alerce Costero (III), Alto Biobio (IV), Altos de Lircay (IV), Altos de Pemehue (IV), Bellotos El Melado (IV), Bosque Fray Jorge (II), Cerro Castillo (IV), Cerro Nielol (III), Chiloe (II), Conguillio (II), Contulmo (III), Coyhaique (IV), Cueva Del Milodon (III), Dos Lagunas (III), El Morado (III), El Yali (IV), Federico Albert (IV), Futaleufu (IV), Hornopiren (II), Huemules de Niblinto (IV), Huerquehue (II), La Campana (II), Lago Cochrane (IV), Lago Jeinemeni (IV), Lago Las Torres (IV), Lago Penuelas (IV), Laguna del Laja (II), Laguna El Peral (IV), Laguna Parrillar (IV), Laguna Torca (IV), Lahuen Nadi (III), Las Chinchillas (IV), Las Vicunas (IV), Lauca (II), Llanos del Challe (II), Llanquihue (IV), Los Flamencos (IV), Los Queules (IV), Los Ruiles (IV), Magallanes (IV), Malalcahuello (IV), Malleco (IV), Mocho - Choshuenco (IV), Nahuelbuta (II), Nalcas (IV), Nevado Tres Cruces (II), Nuble (IV), Pali Aike (II), Pampa del Tamarugal (IV), Pan De Azucar (II), Pichasca (III), Puyehue (II), Queulat (II), Radal Siete Tazas (IV), Ralco (IV), Rio Clarillo (IV), Rio Los Cipreses (IV), Rio Simpsom (IV), Robleria Cobre Loncha (IV), Salar De Surire (III), Tolhuaca (II), Torres del Paine (II), Vicente Perez Rosales (II), Villarrica (II), Volcan Isluga (II), Yerba Loca (IV)
Costa Rica	Arenal (II), Bahia Junquillal (estatal) (IV), Barbilla (II), Barra del Colorado (mixto) (IV), Barra Honda (II), Braulio Carrillo (II), Cano Negro (mixto) (IV), Carara (II), Chirripo (II), Golfito (mixto) (IV), Grecia (VI), Guanacaste (II), Iguanita (estatal) (IV), Internacional La Amistad (II), Juan Castro Blanco (II), Las Tablas (VI), Los Santos (VI), Mata Redonda (estatal) (IV), Palo Verde (II), Rincon de la Vieja (II), Rio Macho (VI), Taboga (VI), Volcan Irazu (II), Volcan Poas (II), Volcan Tenorio (II)

Table S2.2. (cont'd)

Equador	Cajas (II), Cayambe Coca (VI), Chimborazo (IV), Cotacachi Cayapas (VI), Cotopaxi (II), Cuyabeno (VI), El Boliche (V), Limoncocha (VI), Llanganates (II), Parque Lago (V), Podocarpus (II), Sangay (II), Sumaco Napo-Galeras (II), Yacuri (II), Yasuni (II)
Peru	Bahuaja Sonene (II), Calipuy (III)
North America (N=118)	
Canada	Bruce Peninsula National Park of Canada (II), Cape Breton Highlands National Park of Canada (II), Elk Island National Park of Canada (II), Fathom Five National Marine Park of Canada (VI), Fundy National Park of Canada (II), Georgian Bay Islands National Park of Canada (II), Grasslands (III), Gros Morne National Park of Canada (II), Gwaii Haanas National Park Reserve and Haida Heritage site (II), Kejimkujik National Park and National Historic site of Canada (II), Mount Revelstoke National Park of Canada (II), Parc National du Canada de la Mauricie (II), Point Pelee National Park of Canada (II), Prince Edward Island National Park of Canada (II), Pukaskwa National Park of Canada (II), Reserve de Parc National du Canada de l'Archipel-de-Mingan (II), Riding Mountain National Park of Canada (II), Terra Nova National Park of Canada (II), Thousand Islands National Park of Canada (II), Waterton Lakes National Park of Canada (II)
USA	Acadia (IV), Agate Fossil Beds (V), Alibates Flint Quarries (V), Allegheny Portage Railroad (II), Aniakchak (V), Apostle Islands (V), Arches (II), Badlands (II), Bandelier (V), Big Bend (II), Big Thicket (V), Black Canyon of the Gunnison (II), Bluestone (V), Bryce Canyon (II), Canyonlands (II), Capitol Reef (II), Carlsbad Caverns (II), Casa Grande Ruins (V), Catoctin Mountain (II), Cedar Breaks (III), Chiricahua (V), City of Rocks (V), Colorado (III), Congaree (II), Coronado (III), Cowpens (III), Crater Lake (II), Craters of the Moon (III), Cumberland Gap (III), Cuyahoga Valley (II), Death Valley (II), Dinosaur (III), El Malpais (III), El Morro (V), Fort Bowie (III), Fort Pulaski (V), Fort Union Trading Post (III), Fort Washington (V), Gila Cliff Dwellings (V), Glacier (II), Grand Canyon (II), Grand Portage (V), Grand Teton (II), Great Basin (II), Great Sand Dunes (II), Great Smoky Mountains (II), Guadalupe Mountains (II), Hovenweep (V), Indiana Dunes (V), Jean Lafitte National Historical Park and Preserve, Barataria (V), Jewel Cave (V), John Day Fossil Beds (III), Johnstown Flood (III), Joshua Tree (II), Katmai (II), Kenai Fjords (II), Kings Canyon / Sequoia (II), Klondike Gold Rush (V), Lake Chelan (V), Lake Mead (V), Lava Beds (III), Little Bighorn Battlefield (V), Mammoth Cave (II), Mesa Verde (II), Missouri (V), Mojave (V), Montezuma Castle (V), Mount Rainier (II), Mount Rushmore (V), Natural Bridges (III), Niobrara (V), North Cascades (II), Olympic (II), Oregon Caves (V), Organ Pipe Cactus (III), Ozark (V), Pecos (V), Pictured Rocks (V), Pipe Spring (V), Rio Grande (V), Ross Lake (V), Saguaro (II), Santa Monica Mountains (V), Scotts Bluff (V), Shenandoah (II), Sleeping Bear Dunes (V), Theodore Roosevelt (II), Theodore Roosevelt Island (II), Timpanogos Cave (V), Tonto (V), Tumacacori (V), Tuzigoot (V), Voyageurs (II), Washington Monument (V), White Sands (V), Wupatki (III), Yosemite (II), Zion (II)

(in parentheses) IUCN management categories

Table S2.3. Variance Inflation Factor (VIF) for variables used in linear regression.

Category	Variable	VIF
Biodiversity	Total species (species)	3.244
Protected Area	IUCN category (I-IV=1)	1.244
	Size of PA (km^2)	1.734
	Mean elevation (meter)	1.763
	Annual mean temperature (ºC)	2.507
	Annual precipitation (mm)	1.908
	PA remoteness (minutes)	3.625
	PA age (year)	1.262
Demographic	Population density§ (persons/km^2)	3.373
Economic	GDP per capita¶ (2005 const. $ per capita)	3.690
Agricultural factor	Agricultural yields§ (tonne/km^2)	1.440
	Agricultural area§ (%)	2.635
Regulating ES	Water supply originated in PAs (%)	1.344
Region	Africa	2.699
	Europe	2.035
	North America	3.193
	Latin America	2.351

§ 10-km buffer zone

¶ Country level data, not PAs level

Table S2.4. Unstandardized coefficients from multiple regression model predicting annual visitor numbers except biodiversity and total species in PAs.

Category	Variable	Annual visitors	Total species
Biodiversity	Total species (species)	-	-
Protected Area	IUCN category (II-IV=1)	0.392* (0.168)	0.052* (0.023)
	Size of PA (km^2)	0.316** (0.040)	0.008 (0.006)
	Mean elevation (meter)	0.362** (0.058)	0.035** (0.008)
	Annual mean temperature (ºC)	-0.132 (0.148)	0.289** (0.020)
	Annual precipitation (mm)	-0.329* (0.111)	0.178** (0.015)
	PA remoteness (minutes)	-0.225* (0.113)	0.018 (0.016)
	PA age (year)	0.651** (0.118)	-0.013 (0.016)
Demographic	Population density§ (persons/km^2)	0.447** (0.066)	-0.0002 (0.009)
Economic	GDP per capita¶ (2005 const. $ per capita)	1.176** (0.085)	-0.104** (0.012)
Agricultural factor	Agricultural yields§ (tonne/km^2)	0.076 (0.063)	-0.020* (0.009)
	Agricultural area§ (%)	0.077 (0.091)	0.058** (0.013)
Regulating ES	Water supply originated in PAs (%)	0.059 (0.081)	0.060** (0.011)
Region	Asia and Oceania	1.894** (0.227)	0.044 (0.031)
	Africa	1.232** (0.315)	0.306** (0.044)
	Europe	0.756* (0.283)	0.113* (0.039)
	North America	1.529** (0.297)	0.357** (0.041)
Intercept		-6.251**	3.978**
R^2		0.470	0.692
F-statistic		50.61	127.9
DF		912	912

* P<0.05, ** P<0.001

§ 10-km buffer zone

¶ Country level data, not PAs level

Values in parentheses are standard errors

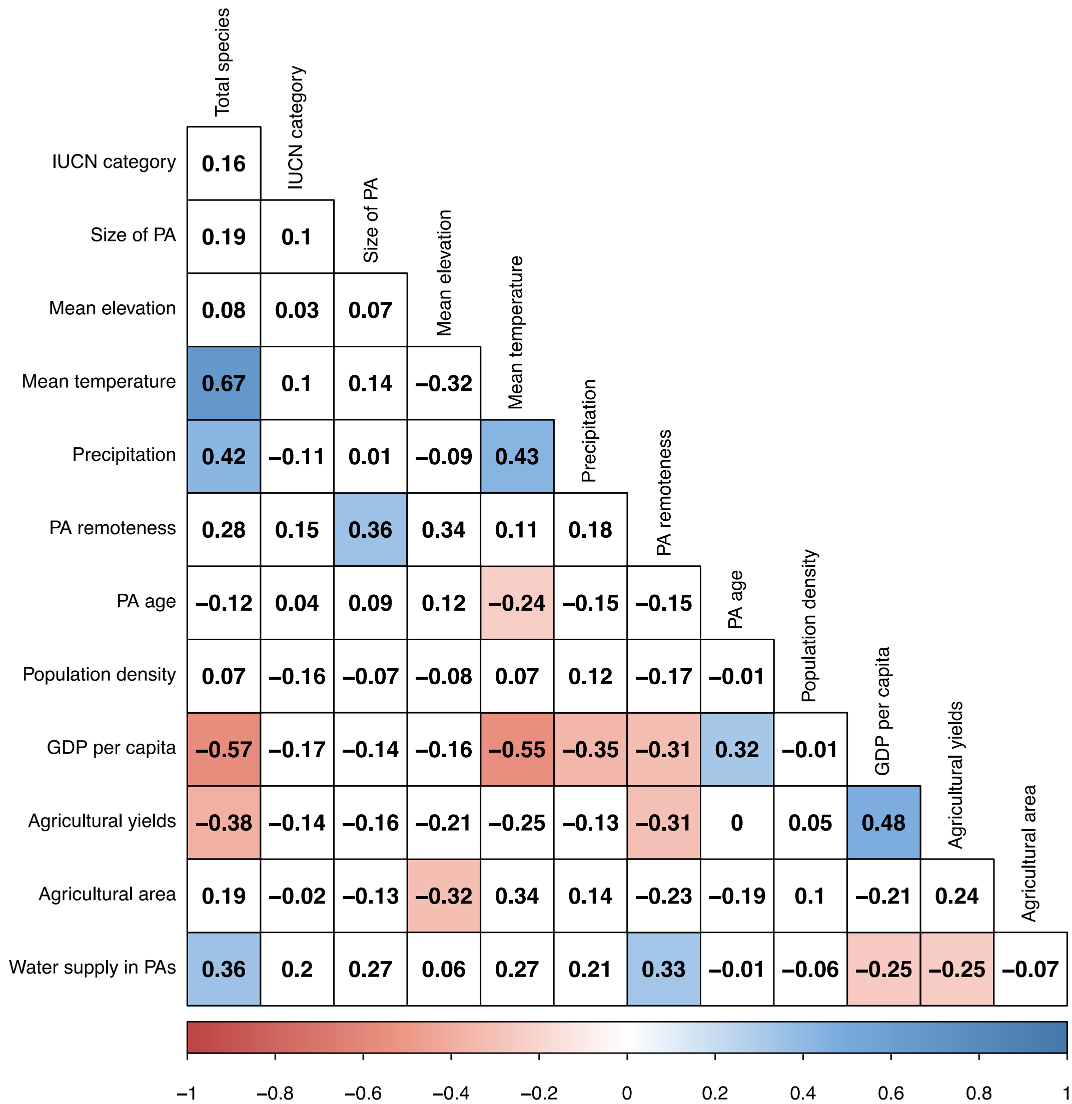

Figure S2.1. The Pearson's correlation matrix for the independent variables. Blue indicates positive correlation for a given pair, and red indicates negative correlation. Colored correlation coefficients are significant at the p=0.05 level.

APPENDIX B

SUPPORTING INFORMATION FOR CHAPTER 3

Table S3.1. Odds ratios for cluster analysis and p-value based on simulations followed by mean, median, and 95% Quantile interval of simulations.

	N	Odds ratio	p-value	mean	median	2.5%	97.5%
2000	2	0.852	<0.001	0.592	0.604	0.346	0.638
2001	7	0.804	<0.001	0.594	0.604	0.345	0.641
2002	9	0.778	<0.001	0.598	0.608	0.351	0.640
2003	3	0.715	<0.001	0.597	0.609	0.348	0.641
2004	11	0.822	<0.001	0.600	0.604	0.546	0.642
2005	2	0.856	<0.001	0.592	0.598	0.539	0.629
2006	2	0.851	<0.001	0.588	0.599	0.347	0.630
2007	9	0.765	<0.001	0.588	0.597	0.344	0.626
2008	9	0.789	<0.001	0.586	0.585	0.555	0.617
2009	12	0.819	<0.001	0.579	0.586	0.370	0.615
2010	9	0.779	<0.001	0.571	0.579	0.357	0.605
2011	2	0.822	<0.001	0.561	0.568	0.466	0.591
2012	2	0.845	<0.001	0.567	0.570	0.537	0.597
2013	5	0.713	<0.001	0.566	0.569	0.533	0.598

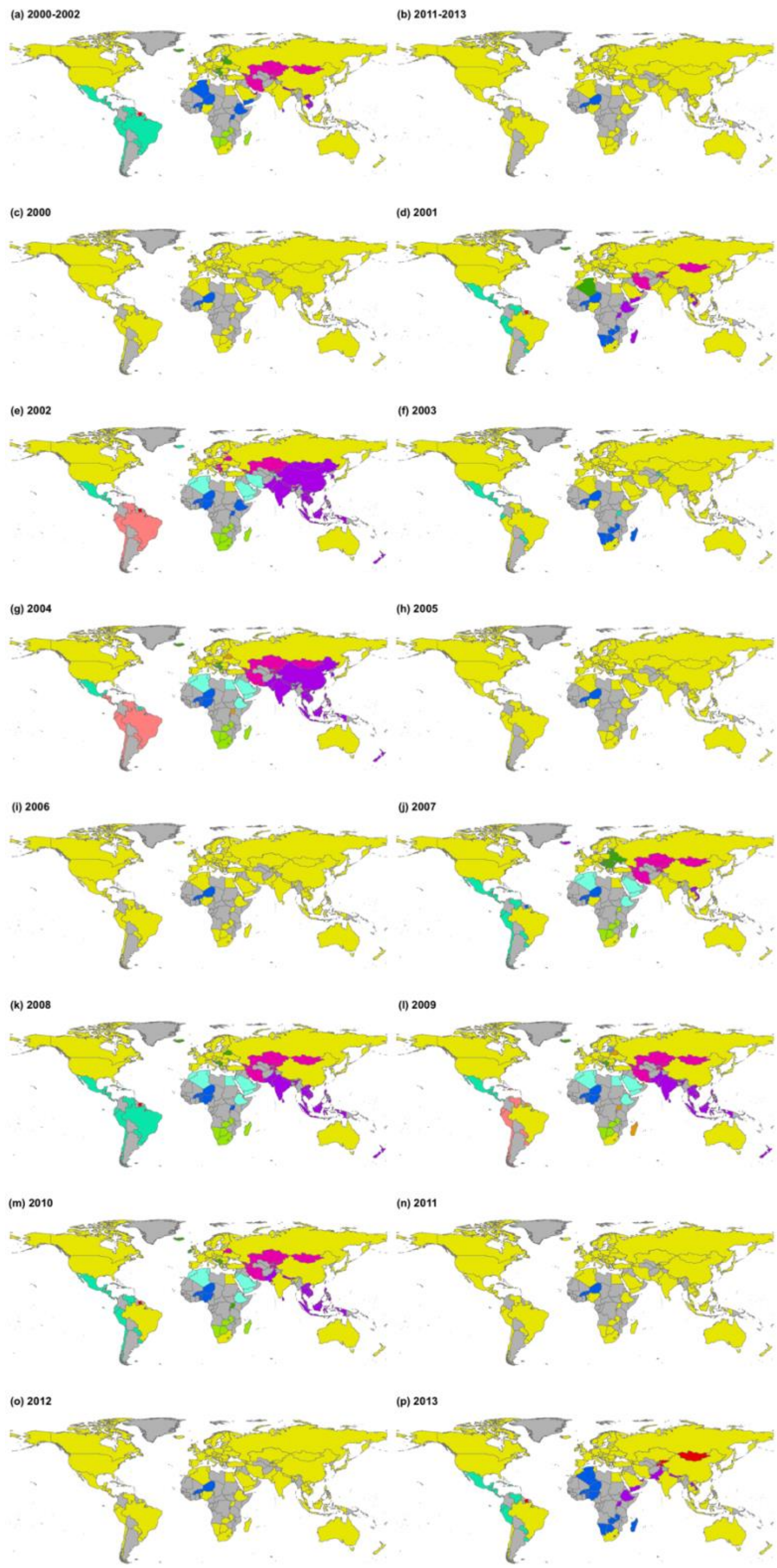

Figure S3.1. Clusters of global tourism networks by country: (a) 2000–2002, (b) 2011–2013, (c) 2000, (d) 2001, (e) 2002, (f) 2003, (g) 2004, (h) 2005, (i) 2006, (j) 2007, (k) 2008, (l) 2009, (m) 2010, (n) 2011, (o) 2012, and (p) 2013.

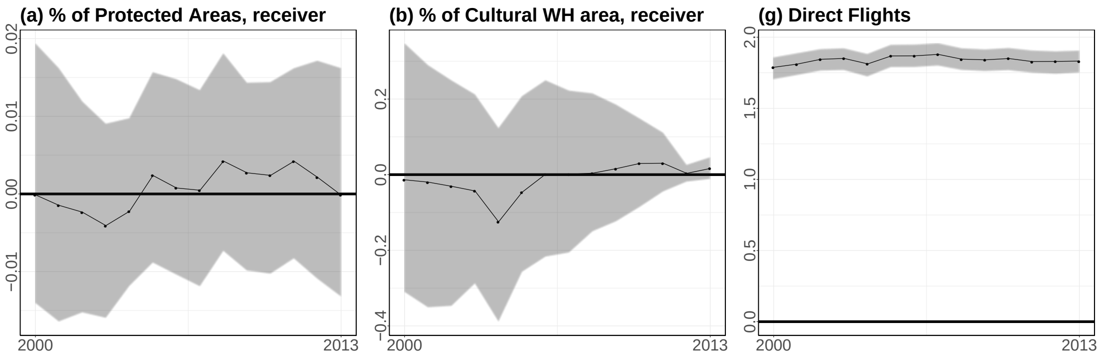

Figure S3.2. Mean and 95% Highest Posterior Density (HPD) confidence intervals of the coefficients from 2000–2013 in the alternative model: (a) the proportion of protected areas in receiving countries, (b) the proportion of World Cultural Heritage sites in receiving countries, and (g) the number of direct flights between countries.

APPENDIX C

SUPPORTING INFORMATION FOR CHAPTER 4

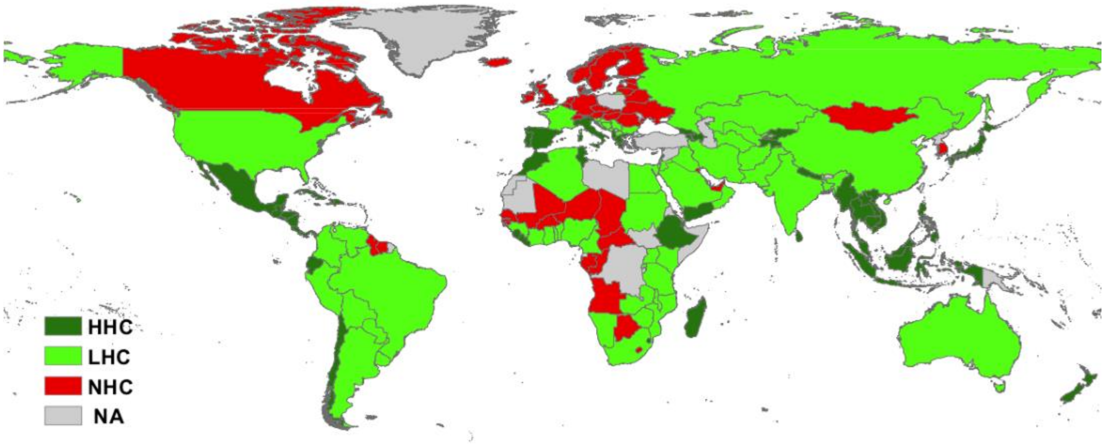

Figure S4.1. Global distribution of hotspot countries (high-hotspot countries [HHC], low-hotspot countries [LHC] and non-hotspot countries [NHC]). NA = countries with missing data. Raw data from Myers et al. (2000) and Myers (2003).

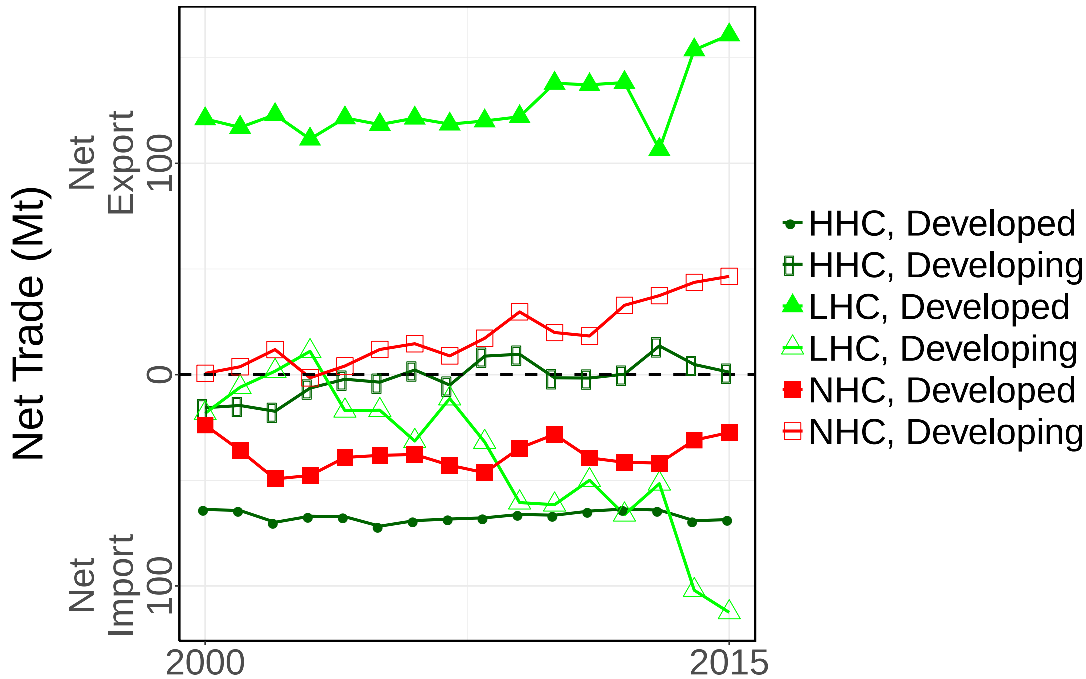

Figure S4.2. The amounts of net food trade between developed and developing countries in high-hotspot countries (HHC), low-hotspot countries (LHC), and non-hotspot countries (NHC) from 2000–2015. Non-hotspot countries are indicated by red, high-hotspot countries dark green, and low-hotspot countries by light green.

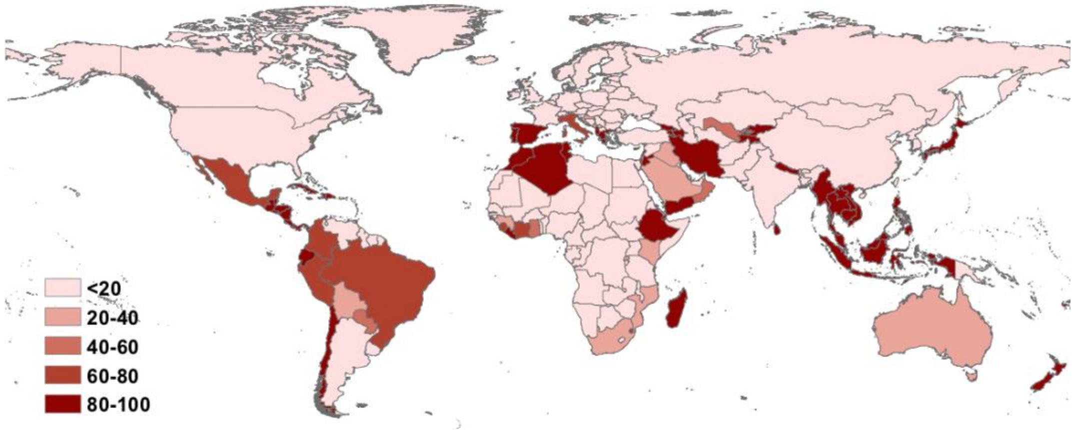

Figure S4.3. The percentage of agricultural area in biodiversity hotspots out of total agricultural area. Raw data from Myers et al. (2000), Myers (2003), and Tuanmu and Jetz (2014).

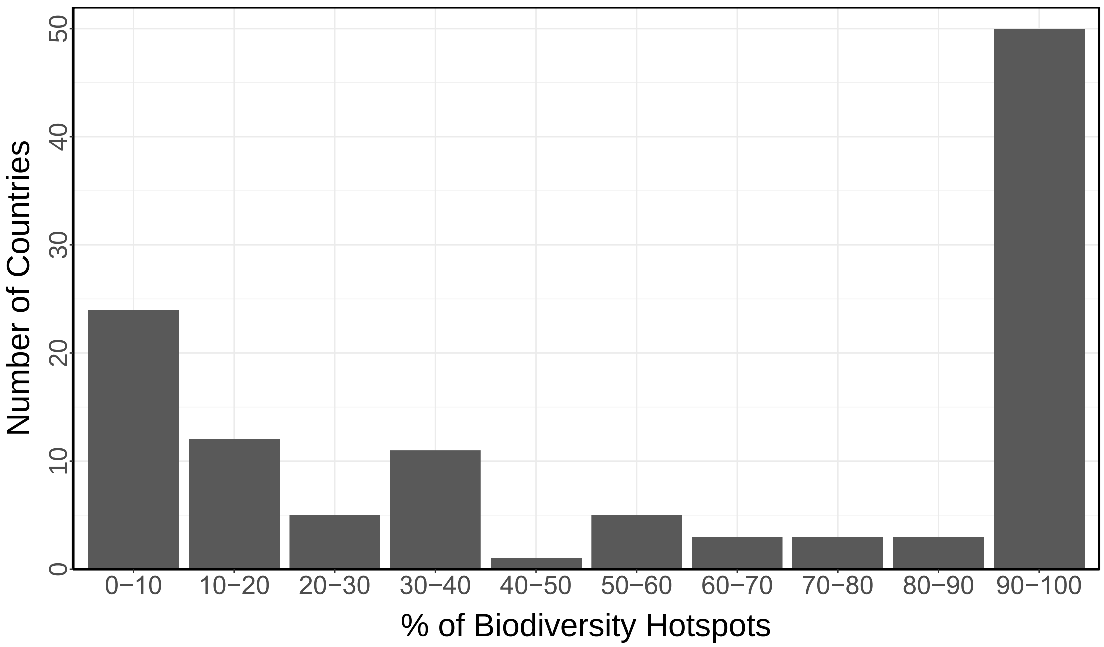

Figure S4.4. Number of countries with different percentages of biodiversity hotspots (land area with biodiversity hotspots out of total terrestrial land area). Raw data from Myers et al. (2000) and Myers (2003).

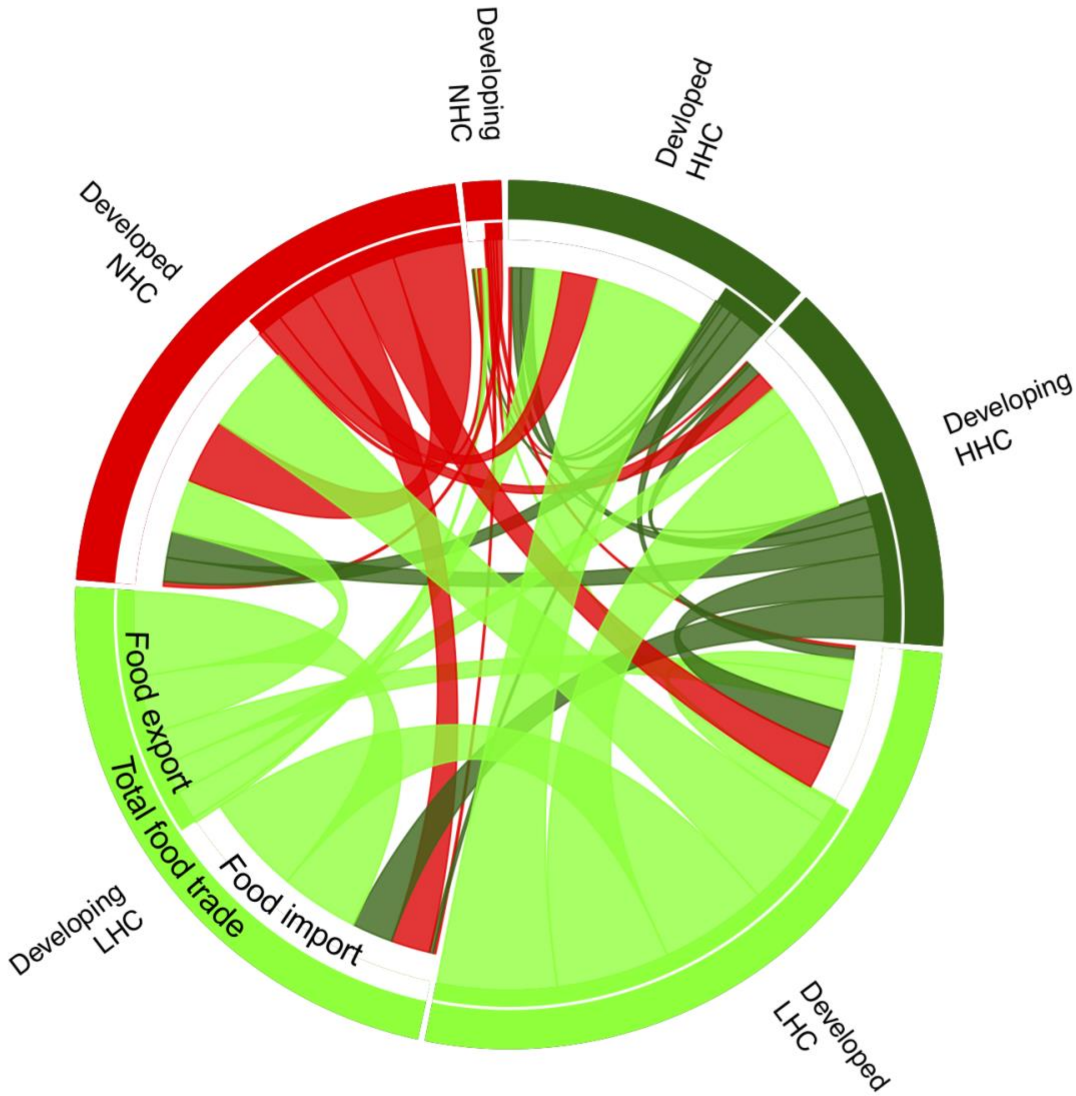

Figure S4.5. Annual food flows (Mt) in 2000. Food flows between developed and developing countries in high-hotspot countries (HHC), low-hotspot countries (LHC), and non-hotspot countries (NHC). Non-hotspot countries are indicated by red, high-hotspot countries by dark green, and low-hotspot countries by light green. The arc length of an outer circle indicates the sum of food exported and imported in each group. The arc length of a middle circle refers to the amounts of food exported. The inner arc length shows the amounts of food imported. Raw data from UN FAO (2018).

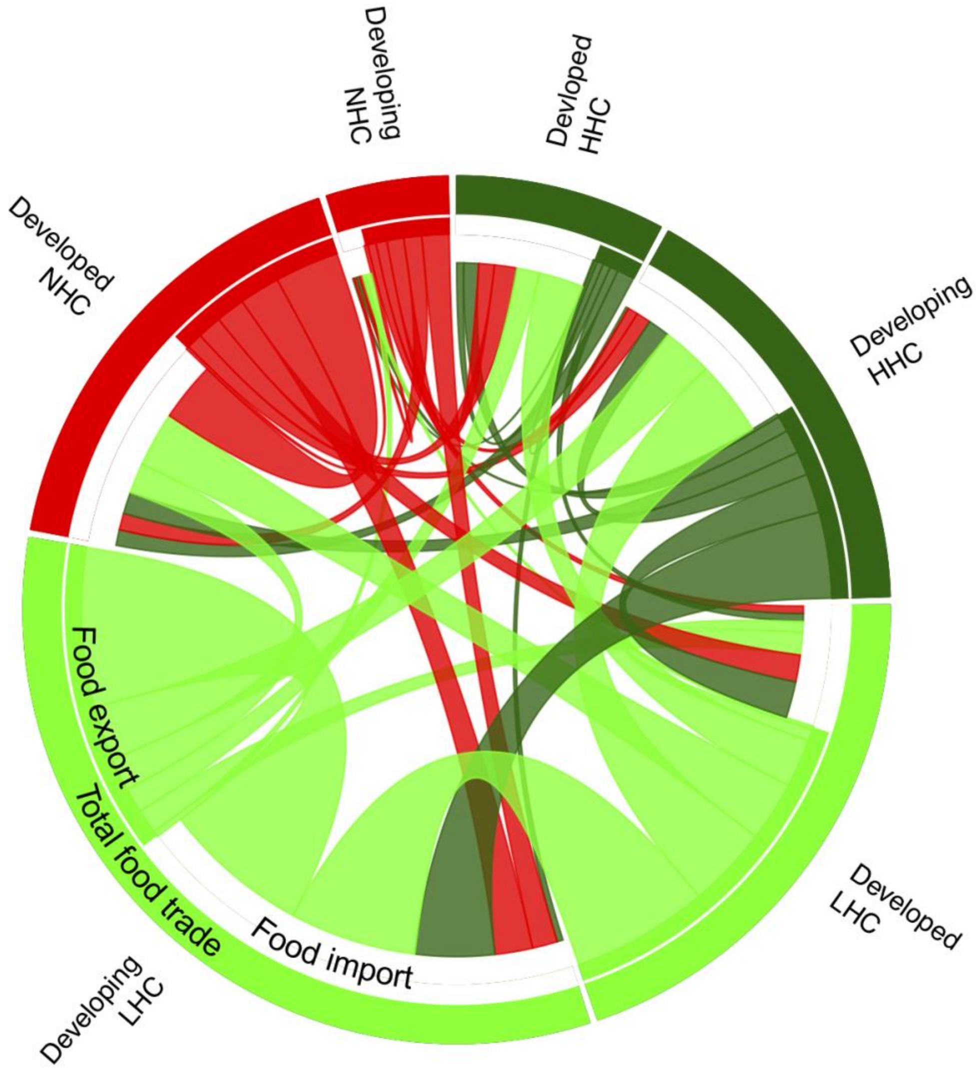

Figure S4.6. Annual food flows (Mt) in 2015. Food flows between developed and developing countries in high-hotspot countries (HHC), low-hotspot countries (LHC), and non-hotspot countries (NHC). Non-hotspot countries are indicated by red, high-hotspot countries by dark green, and low-hotspot countries by light green. The arc length of an outer circle indicates the sum of food exported and imported in each group. The arc length of a middle circle refers to the amounts of food exported. The inner arc length shows the amounts of food imported. Raw data from UN FAO (2018).

Table S4.1. List of hotspot and non-hotspot countries, subdivided into developed and developing countries groups.

High-hotspot Countries (HHC), N=64	
Developed, N=14	Antigua and Barbuda, Bahamas, Barbados, Brunei Darussalam, Chile, Cyprus, Greece, Italy, Japan, Malta, New Zealand, Portugal, Saint Kitts and Nevis, Spain
Developing, N=50	Albania, Armenia, Azerbaijan, Belize, Cabo Verde, Cambodia, Costa Rica, Cuba, Djibouti, Dominica, Dominican Republic, Ecuador, El Salvador, Ethiopia, Fiji, Georgia, Grenada, Guatemala, Haiti, Honduras, Indonesia, Jamaica, Kyrgyzstan, Lao People's Democratic Republic, Lebanon, Liberia, Madagascar, Malaysia, Mauritius, Mexico, Morocco, Myanmar, Nepal, Nicaragua, Panama, Philippines, Saint Lucia, Saint Vincent and the Grenadines, Samoa, Sao Tome and Principe, Sierra Leone, Solomon Islands, Sri Lanka, Swaziland, Tajikistan, Thailand, Tunisia, Vanuatu, Viet Nam, Yemen
Low-hotspot Countries (LHC), N=53	
Developed, N=10	Australia, Croatia, France, Israel, Oman, Russian Federation, Saudi Arabia, Slovenia, Uruguay, United States of America
Developing, N=43	Afghanistan, Algeria, Argentina, Bangladesh, Benin, Bolivia, Bosnia and Herzegovina, Brazil, Bulgaria, Cameroon, Cote d'Ivoire, China, Colombia, Egypt, Ghana, Guinea, India, Iran, Iraq, Jordan, Kazakhstan, Kenya, Macedonia, Malawi, Montenegro, Mozambique, Namibia, Nigeria, Pakistan, Paraguay, Peru, Rwanda, Serbia, South Africa, Sudan, Togo, Turkmenistan, Uganda, United Republic of Tanzania, Uzbekistan, Venezuela, Zambia, Zimbabwe
Non-hotspot Countries (NHC), N=43	
Developed, N=23	Austria, Belgium, Canada, Czechia, Denmark, Estonia, Finland, Germany, Iceland, Ireland, Kuwait, Latvia, Lithuania, Luxembourg, Netherlands, Norway, Republic of Korea, Slovakia, Sweden, Switzerland, Trinidad and Tobago, United Arab Emirates, United Kingdom
Developing, N=20	Angola, Belarus, Botswana, Burkina Faso, Central African Republic, Chad, Congo, Gabon, Gambia, Guinea-Bissau, Guyana, Hungary, Lesotho, Mali, Mongolia, Niger, Romania, Senegal, Suriname, Ukraine

Table S4.2. The quantity of food exports and imports for developed and developing countries in 2000, 2015, and 2000–2015 (average annual).

Year	Income level	Variables	HC		NHC	Total
			HHC	LHC		
2000						
	Developed	Food Export (Mt)	22.41	184.45	88.89	295.75
		Food Import (Mt)	86.22	63.43	112.76	262.41
		Export-Import (Mt)	-63.81	121.02	-23.87	33.34
	Developing	Food Export (Mt)	57.92	95.97	6.71	160.60
		Food Import (Mt)	73.57	114.28	6.09	193.94
		Export-Import (Mt)	-15.65	-18.31	0.62	-33.34
2015						
	Developed	Food Export (Mt)	31.75	248.65	128.91	409.31
		Food Import (Mt)	100.30	87.90	156.30	344.50
		Export-Import (Mt)	-68.55	160.75	-27.39	64.82
	Developing	Food Export (Mt)	139.19	219.64	61.58	420.41
		Food Import (Mt)	138.03	332.14	15.05	485.23
		Export-Import (Mt)	1.16	-112.5	46.53	-64.82
2000-2015						
	Developed	Food Export (Mt)	27.89	203.86	99.85	331.60
		Food Import (Mt)	94.86	77.05	137.69	309.60
		Export-Import (Mt)	-66.97	126.81	-37.84	21.99
	Developing	Food Export (Mt)	96.05	156.81	29.01	281.87
		Food Import (Mt)	97.75	195.84	10.26	303.86
		Export-Import (Mt)	-1.7	-39.04	18.75	-21.99

Table S4.3. The quantity of agricultural area saved (km2) due to food imports.

Year	Food Importers	Food Exporters							
		HC				NHC, Developed	NHC, Developing	Total	Total agricultural area
		HHC, Developed	HHC, Developing	LHC, Developed	LHC, Developing				
2000									
	HHC, Developed	4,673	5,616	37,847	17,101	15,702	1,518	82,457	869,410
	HHC, Developing	4,261	30,590	220,232	44,496	39,997	3,270	342,846	4,479,316
	LHC, Developed	30,135	129,833	313,423	216,360	470,589	15,625	1,175,964	12,696,596
	LHC, Developing	11,285	110,620	219,536	195,603	127,711	8,132	672,888	23,110,023
	NHC, Developed	8,171	9,638	34,090	14,142	20,186	1,466	87,693	1,365,563
	NHC, Developing	3,655	6,325	46,512	121,634	17,623	3,516	199,264	4,503,655
2015									
	HHC, Developed	7,642	13,362	33,726	29,595	20,468	16,766	121,558	822,858
	HHC, Developing	6,940	127,651	264,281	108,924	43,986	72,309	624,090	4,584,203
	LHC, Developed	60,598	227,096	521,931	562,989	334,566	249,063	1,956,244	12,104,780
	LHC, Developing	13,999	131,471	252,578	408,302	102,763	39,972	949,085	22,857,369
	NHC, Developed	11,777	16,167	36,956	21,896	40,557	8,837	136,189	1,336,014
	NHC, Developing	7,320	36,897	46,679	159,617	14,337	7,709	272,560	4,520,668

Table S4.3. (cont'd)

Year	Food Importers	Food Exporters							
		HC				NHC, Developed	NHC, Developing	Total	Total agricultural area
		HHC, Developed	HHC, Developing	LHC, Developed	LHC, Developing				
2000-2015									
	HHC, Developed	6,082	7,903	34,650	24,339	16,270	7,933	97,177	869,410
	HHC, Developing	6,005	57,936	202,257	89,437	31,090	18,943	405,668	4,479,316
	LHC, Developed	41,103	152,669	362,588	410,813	231,437	213,194	1,411,804	12,696,596
	LHC, Developing	10,954	120,458	192,947	259,794	66,367	19,602	670,121	22,907,336
	NHC, Developed	9,694	13,464	34,875	21,140	29,522	4,003	112,698	1,365,563
	NHC, Developing	6,075	14,651	47,734	109,958	11,456	4,066	193,941	4,503,655

Table S4.4. The percentages of population, food production, and food trade among high-hotspot countries (HHC), low-hotspot countries (LHC), and non-hotspot countries (NHC), with each group subdivided into developed and developing countries, for 2000, 2015, and 2000–2015 (average annual).

Year	Variables	HC				NHC, Developed	NHC, Developing
		HHC, Developed	HHC, Developing	LHC, Developed	LHC, Developing		
2000							
	Population (1000 persons)	268,720	928,744	545,837	3,588,769	317,928	177,626
	Population percentages	4.6%	15.9%	9.4%	61.6%	5.5%	3.1%
	Food Production (Mt)	655.0	1,704.0	2,852.9	6,832.6	1,171.5	433.5
	Food Production percentages	4.8%	12.5%	20.9%	50.1%	8.6%	3.2%
	Food Export (Mt)	22.4	57.9	184.5	96.0	88.9	6.7
	Food Export percentages	4.9%	12.7%	40.4%	21.0%	19.5%	1.5%
	Food Export/Food Production percentages	3.4%	3.4%	6.5%	1.4%	7.6%	1.6%
	Food Import (Mt)	86.2	73.6	63.4	114.3	112.8	6.1
	Food Import percentages	18.9%	16.1%	13.9%	25.0%	24.7%	1.3%
2015							
	Population (1000 persons)	280,722	1,156,289	605,638	4,366,135	347,337	220,028
	Population percentages	4.0%	16.6%	8.7%	62.6%	5.0%	3.2%
	Food Production (Mt)	524.1	2,330.9	2,952.2	9,596.2	1,023.0	638.5
	Food Production percentages	3.1%	13.7%	17.3%	56.2%	6.0%	3.7%
	Food Export (Mt)	31.7	139.2	248.7	219.6	128.9	61.6
	Food Export percentages	3.8%	16.8%	30.0%	26.5%	15.5%	7.4%

Table S4.4. (cont'd)

Year	Variables	HC				NHC, Developed	NHC, Developing
		HHC, Developed	HHC, Developing	LHC, Developed	LHC, Developing		
2015	Food Export/Food Production percentages	6.1%	6.0%	8.4%	2.3%	12.6%	9.7%
	Food Import (Mt)	100.3	138.0	87.9	332.1	156.3	15.1
	Food Import percentages	12.1%	16.6%	10.6%	40.0%	18.8%	1.8%
2000-2015							
	Population (1000 persons)	272,292	1,040,098	574,445	3,973,213	331,919	196,526
	Population percentages	4.3%	16.3%	9.0%	62.1%	5.2%	3.1%
	Food Production (Mt)	616.4	2,254.1	3,052.6	8,747.8	1,144.0	603.3
	Food Production percentages	3.8%	13.7%	18.6%	53.3%	7.0%	3.7%
	Food Export (Mt)	27.9	96.1	203.9	156.8	99.8	29.0
	Food Export percentages	4.6%	15.7%	33.2%	25.6%	16.3%	4.7%
	Food Export/Food Production Percentages	4.5%	4.3%	6.7%	1.8%	8.7%	4.8%
	Food Import (Mt)	94.9	97.8	77.1	195.8	137.7	10.3
	Food Import percentages	15.5%	15.9%	12.6%	31.9%	22.4%	1.7%

APPENDIX D

SUPPORTING INFORMATION FOR CHAPTER 5

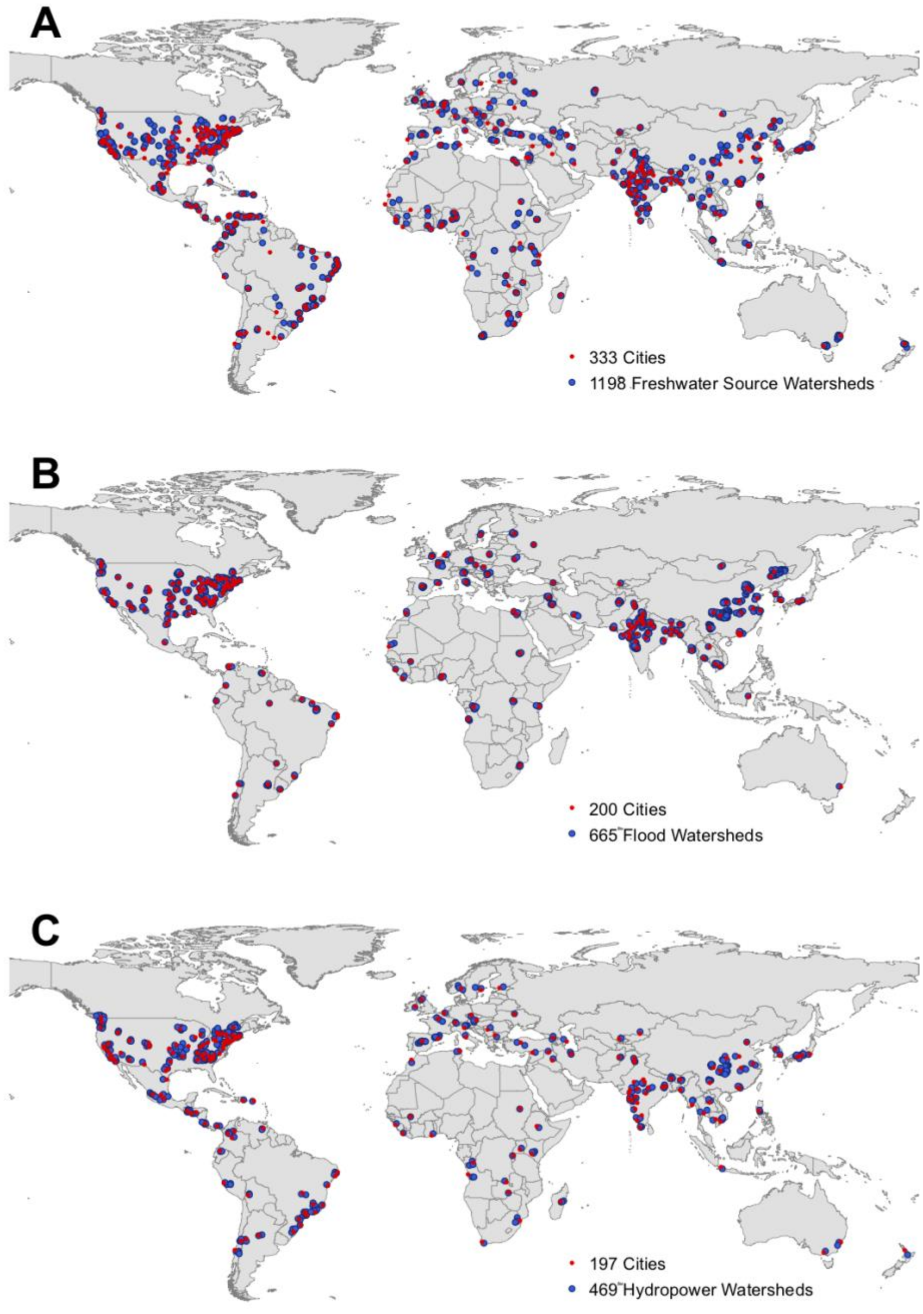

Figure S5.1. The locations of cities and watersheds in (A) freshwater source watersheds, (B) flood watersheds, and (C) hydropower watersheds. Red indicates cities, and blue indicates watersheds.

Table S5.1. Descriptions of dependent and independent variables.

Category	Variable	Dataset	Unit of measure	Time period	References	Link
System boundary		Global Administrative database (GADM)	polygon	2018	Global Administrative Areas (2018)	http://gadm.org
	Urban extents	Global urban extents	polygon	mid-2000	Schneider et al. (2009)	https://www.naturalearthdata.com/downloads/10m-cultural-vectors/10m-urban-area/
		Urban extents in the USA	polygon	2016	United States Census Bureau (2017)	https://www.census.gov/geo/maps-data/data/cbf/cbf_ua.html
	Watershed boundaries	HydroSHEDS	polygon	mid-2000	Lehner et al. (2008)	https://www.hydrosheds.org/page/hydrobasins
Flow	River length	The City Water Map v2.2	point and polygon	2016	McDonald et al. (2014)	http://doi.org/10.5063/F1J67DWR
		HydroSHEDS	30 arc-sec	mid-2000	Lehner et al. (2008)	http://www.hydrosheds.org/
	Power lines	OpenStreetMap – power networks	line	2000s	OpenStreetMap	https://www.openstreetmap.org
Freshwater ES	Freshwater availability	Global Water Modeling, ISMIP	m^3/s, 30 arc-min	2001-2010	Veldkamp et al. (2017)	https://www.isimip.org/outputdata
	Sediment flow	Modeled Suspended Sediment in Global Rivers	kg/m^3, 6 arc-min	2000-2010	Cohen et al. (2014)	http://sdml.ua.edu/datasets-2
	Hydropower production	A Global Database of Power Plants	MW, point	2000-2016	Byers et al. (2018)	http://datasets.wri.org/dataset/globalpowerplantdatabase
	Flood risk	Flood Hazard Maps at Global Scale	meter, 30 arc-sec	2000s	Dottori et al. (2016)	http://data.jrc.ec.europa.eu/collection/floods
Watershed	Protected area	ProtectedPlanet	polygon	-2016	IUCN and UNEP-WCMC (2017)	https://www.protectedplanet.net
	Forest	Global 1-km Consensus Land Cover	proportion (0-100), 30 arc-sec	2000	Tuanmu and Jetz (2014)	http://www.earthenv.org/landcover.html
	Wetland	Global lakes and wetlands database (GLWD)	30 arc-sec	2000s	Lehner and Döll (2004)	https://www.worldwildlife.org/pages/global-lakes-and-wetlands-database
	Dam	Global Reservoir and Dam database (GRanD)	point	-2016	Lehner et al. (2011)	http://globaldamwatch.org/grand/

Table S5.1. (cont'd)

Category	Variable	Dataset	Unit of measure	Time period	References	Link
Watershed	Irrigation areas	The Global Food Security-Support Analysis Data (GFSAD)	class, 30 arc-sec	2010	Thenkabail et al. (2016)	https://croplands.org/home
	Elevation and Slope	Freshwater environmental variables in EarthEnv	meter and degree, 30 arc-sec	2000s	Domisch et al. (2015)	http://www.earthenv.org/streams
City	Urban population	The World Urbanization Prospects	persons	2000-2010	UNDP (2015)	https://esa.un.org/unpd/wup/CD-ROM/
	Urban GDP	Global dataset of gridded GDP and population scenarios	2005 const. billion US$, 30 arc-min	2010	Murakami and Yamagata (2019)	http://www.cger.nies.go.jp/gcp/population-and-gdp.html
	IWS program	Investments or Payments in Watersheds Science Program	binary (0,1)	2000s	Romulo et al. (2018)	https://doi.org/10.1038/s41467-018-06538-x
	Temperature and precipitation	WorldClim v2	°C and mm, 30 arc-sec	1970-2000	Hijmans et al. (2005)	http://worldclim.org/version2

Table S5.2. Variance Inflation Factor (VIF) for variables used in multi-level models

	Variable	Water Supply	Sediment Flow	Flood Risk	Hydro-power
Watershed	Forest cover in PAs (%)	1.166	1.173	1.611	1.462
	Wetland cover in PAs (%)	1.160	1.160	1.363	1.202
	Dam density (#/100 km of river length)	1.125	1.121	1.105	1.203
	Irrigation area (%)	1.088	1.096	1.138	1.149
	Watershed area (km^2)	4.214	4.352	2.342	1.780
	Urban-watershed distance (km)	3.841	4.009	2.226	1.540
	Elevation (meter)	4.635	4.446	1.960	1.922
	Slope (degree)	3.501	3.397	1.814	1.880
Urban	IWS program (0, 1)	1.069	1.075	1.138	1.107
	Urban population (1,000 persons)	1.417	1.428	1.424	1.529
	Urban GDP-PPP (2005 const. billion USD)	1.519	1.539	1.695	1.815
	Temperature (°C)	1.207	1.212	1.155	1.423
	Precipitation (mm)	1.111	1.127	1.163	1.277

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