UNIVERSITY OF SOUTHAMPTON

FACULTY OF NATURAL ENVIRONMENTAL SCIENCES

Centre for Biological Sciences

An ecosystem service approach to quantifying the role of freshwater biodiversity in supporting food security

by

Emma Grace Elizabeth Brooks

Thesis for the degree of Doctor of Philosophy

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Cover photo: Girl selling fish in the market at Stung Treng, Cambodia © William Darwall

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ABSTRACT

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AN ECOSYSTEM SERVICE APPROACH TO QUANTIFYING THE ROLE OF FRESHWATER BIODIVERSITY IN SUPPORTING FOOD SECURITY

Emma Brooks

There is increasing emphasis to consider ecosystem services in natural resource policy and management, which has the potential to provide win-wins for species and habitat conservation and human use of resources. Inland freshwater fisheries provide over 33% of the world's small scale fish catch and employ over 60 million people. However inland waters are the most threatened ecosystem in the world, which in turn threatens the livelihoods and sustenance of millions globally. This thesis assesses the role of freshwater species in providing food as an ecosystem service, particularly to poor and vulnerable groups, and how the needs of fisheries align and contrast with threatened species inhabiting the waterways. In Chapter 2 the value of fisheries was assessed alongside other ecosystem services provided by inland water systems, where it was shown that monetary and nonmonetary valuations suggest very different priorities across a suite of services. Disaggregation of beneficiaries also showed a mismatch in prioritisation between different stakeholders, and in particular that fishermen and women, who rely most directly on the water resources, value resources incompatibly from a standard monetary valuation. In Chapter 3 I examined the effect of biodiversity on fishery yields and variability to determine if there is a potential for a win-win for conservation and ecosystem service delivery, and showed that increased species richness provides a significant positive contribution. The study of inland water systems and fisheries is hampered by a lack of data and in order to map the benefits from fisheries a model was created for Chapter 4 to spatially predict the relative importance of inland waterways to fishery yields. Example output from this model was used in Chapter 5 to explore how fisheries and freshwater species hotspots overlap spatially, and how this information can be used to determine

potential areas of synergy where improved management could ensure benefits to humans while protecting wetland species, but equally to examine where there is potential conflict. With an increased understanding of freshwater ecosystems and their link to the resources they provide, there is potential for inland waters to be managed to benefit both people relying on food provision and the species living within.

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DECLARATION OF AUTHORSHIP

I, Emma Grace Elizabeth Brooks, declare that this thesis and the work presented in it are my own and has been generated by me as the result of my own original research.

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I confirm that:

1. This work was done wholly or mainly while in candidature for a research degree at this University;
2. Where any part of this thesis has previously been submitted for a degree or any other qualification at this University or any other institution, this has been clearly stated;
3. Where I have consulted the published work of others, this is always clearly attributed;
4. Where I have quoted from the work of others, the source is always given. With the exception of such quotations, this thesis is entirely my own work;
5. I have acknowledged all main sources of help;
6. Where the thesis is based on work done by myself jointly with others, I have made clear exactly what was done by others and what I have contributed myself;
7. Parts of this work have been published as:

Brooks, E.G.E., Smith, K.G., Holland, R.A., Poppy, G.M. & Eigenbrod, F. (2014) Effects of methodology and stakeholder disaggregation on ecosystem service valuation. *Ecology and Society*, 19, 18.

Brooks, E.G.E., Holland, R.A., Darwall, W.R.T. & Eigenbrod, F. (2014) Global evidence of positive impacts of freshwater biodiversity on fishery yields. *Global Ecology and Biogeography*, 25(5): 553-562.

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1 Introduction

"The ecosystem approach is a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way, and which recognises that people with their cultural and varied social needs, are an integral part of ecosystems" (Maltby 2000, adopted by the Convention on Biological Diversity)

1.1 Ecosystem services

The ecosystem approach has been widely promoted as a framework for achieving sustainable development through the management of environmental systems for the benefit of nature and people (Haines-Young & Potschin, 2007) and as such has been incorporated into policy agendas from local authorities to national programmes, to international agreements such as the Convention on Biological Diversity (CBD, 1992). Since the publication of the Millennium Ecosystem Assessment (MEA, 2005), recognition of the provision of goods, benefits and well-being to humans from ecosystems has increasingly been incorporated into environmental sciences and policy as ecosystem services (ES). The acknowledgement of ES focuses on the ecological structures and processes that result in benefits to humans to support their survival and well-being at the local, regional and global scale (Daily, 1997). As part of the MEA, ES were classified within four categories: provisioning, cultural, regulating and supporting services (MEA, 2005). Since then the concept has developed to identify the ecosystem functions that underpin processes, and concentrates instead on the final delivery of benefits and goods (Fisher & Turner, 2008; Fisher *et al.*, 2008).

One example of ES is the delivery of sustenance and nourishment. The provision of food and therefore food security is ubiquitous and multi scale, with local production and benefit, but also national and international trading worth many billions annually (FAO, 2014b). It is now well established that healthy ecosystems play a pivotal role in the provision of nutrition (Balmford *et al.*, 2002; Boody *et al.*, 2005; Brown, 2005; Nellemann *et al.*, 2009; Boelee, 2011; Hurni *et al.*, 2015), and indeed food security is one of the most essential ES. The influences of the environment on food security delivery is realised locally, but similarly it can also be affected at the regional or even global scale, for instance through the effects of climate change (Brown, 2005; Schmidhuber & Tubiello, 2007). Perpetuation of vital ecosystems services through the maintenance of healthy environments and ecosystems can be an essential contribution to the sustainable provision and improvement of food security (Nellemann *et al.*, 2009).

1.2 Food security

The world population is currently over 7 billion, and could be as high as 12 billion by the end of the century, predominantly due to population growth in developing countries (Gerland *et al.*, 2014; UN, 2015). As the world population continues to grow, the provision of food security is increasingly of global concern (Godfray *et al.*, 2010). While a number of definitions are in circulation, the prevailing definition agreed upon at the 2009 World Summit on Food Security is that:

> *"Food Security exists when all people, at all times, have physical, social and economic access to sufficient, safe and nutritious food to meet their dietary needs and food preferences for an active and healthy life."* (FAO, 2009a)

This not only alludes to access of calorific intake, but the temporal stability of a nutritionally sustaining and enriching diet (Barrett, 2010). The summit declaration went on to describe the key dimensions of this definition as the four pillars of food security, which comprise of availability, access, utilisation and stability (see Box 1.1). These outline the key variables necessary to define a nation, community or household as food secure, and fulfilment of all four pillars for all is necessary and vital for human well-being globally (Boelee, 2011).

1.3 Malnutrition

Although agricultural production and intensification is on the increase, food security is a real issue for millions of people around the world. Establishing the scale of global undernourishment is fraught with difficulty – not only is obtaining data difficult, but different indicators will provide discrepancies in estimates for the scale of malnourishment (Barrett, 2010). Nevertheless, it is widely recognised as a huge problem. The most widely cited figures come from the Food and Agriculture Organization of the United Nations (FAO), with current estimates of 795 million people (or 11% of the global population) suffering from chronic hunger. Three-quarters of these people live in rural areas, and 98% are from developing countries – the greatest number in Asia and the greatest proportion found in sub-Saharan Africa (FAO *et al.*, 2015). In many countries, undernutrition rates are even higher than undernourishment, resulting, among other things, in stunted growth of one in four children globally (FAO *et al.*, 2013). According to the UN's Standing Committee on Nutrition malnutrition is the largest single contributor to disease worldwide, and contributes to half of all child mortality.

Box 1.1: The four pillars of food security (from FAO 2006):

Availability: Sufficient quantities of food must be produced and be consistently available to individuals, either from domestic production or from importation.

Access: Ability of individuals to have the physical, social and economic access to enough food. This includes the knowledge and the ability to produce or procure the food, including the capacity to produce the total amount of food required, or having the purchasing power to buy food.

Utilisation: Food must meet the specific dietary and nutritional needs of individuals, as well as socio-cultural considerations. It also includes proper food processing and storage techniques, as well as adequate health and sanitation services. Food security integrates the notion of food safety.

Stability: The fourth pillar incorporates the need for temporal stability in the previous three pillars, both seasonally and year on year. This incorporates issues of price stability, as well as considering storage capacities or other means of insuring continued food supply year round.

Due to the expanding population, food production will need to as much as double by 2050 (Royal Society, 2009; Foley *et al.*, 2011), and food prices are predicted to rise by another 30-50% over the next several decades (Nellemann *et al.*, 2009). This is likely to most impact the poor, who already spend the majority of their income on food (FAO, 2011b). In addition climate change and variability will adversely affect food security particularly in developing countries. The potential effects of climate change on food security have been reviewed extensively (e.g. Field *et al.* 2014) but can be summarized as a change in the amount and variability of climatic elements such as temperature and precipitation. Additionally, more extreme events including hydrologic events such as floods and droughts will become more common, more intense, and appear in new places (Solomon *et al.*, 2007). It is indicated that the effects of climate change are likely to be most strongly felt in developing countries. For instance, it is estimated that Africa's potential agricultural output will reduce by 30-50% by 2100, which will adversely affect food security and exacerbate malnutrition (Parry *et al.*, 2007). Not only in terms of nourishment but also in terms of livelihoods, it is the poor, women and marginal groups who are most vulnerable to the effects of climate change and variability (Bohle *et al.*, 1994).

1.4 Inland fisheries

Food security principally considers the continued provision from agriculture, however extensive amounts of food are harvested directly from ecosystems. One such example is the harvesting of species, predominantly fishes, from aquatic sources. Fisheries were a US$217.5 billion industry in 2010 and are rapidly expanding, with aquaculture the fastest growing food-producing sector (FAO *et al.*, 2012). However aquaculture itself impacts freshwater use and habitat and is considered as a competitor to inland capture fisheries rather than a sub-sector within it (Youn *et al.*, 2014).

Much of the focus on capture fisheries is on marine harvests however inland waters are of huge importance, with hundreds of millions of people depending upon wetlands directly for the provision of food and livelihoods (Richter *et al.*, 2010). According to FAO the total worldwide yield from inland capture fisheries in 2013 was almost 12 million tonnes (FAO, 2015), although it is estimated that due to under-reporting, particularly of artisanal and subsistence catches, the true figure may be two or three times greater than this (FAO, 2003; Youn *et al.*, 2014). In fact if all harvest was truly accounted for, production from inland waters could rival that of marine (Welcomme, 2011a). For Europe, North America and Australia the primary importance of inland fisheries is for recreation, where recreational fishing is valued at many billions. However recreational fishing makes only a minor contribution to the total global harvest, where 67% of all reported catch was harvested in Asia and 25% from Africa (FAO, 2015).

Consumption of fish offers a vital contribution to the utilisation pillar of food security by providing specific nutritional merits. In many parts of the world inland waters are often the primary, if not only, source of protein (Dugan *et al.*, 2010). For instance, it is estimated that 60-70 million people in the lower Mekong Basin rely upon fish as their primary source of protein (Neiland & Béné, 2008), with 60% of the Cambodian population's protein being derived from the lake of Tonle Sap alone (Smith *et al.*, 2005). Fisheries also play an important role in providing essential micronutrients particularly vitamin A, calcium, iron and zinc through consumption of smaller fish (Gibson & Hotz, 2001; Roos *et al.*, 2007; Chamnan *et al.*, 2009).

Moreover small scale fisheries provide a vital lifeline to low earning rural populations. For many communities that lack access to formal financial systems, fishing serves as a ‘bank in the water’ by providing a year round commodity which can be sold or bartered, e.g. for medicine, education or for seeds or fertiliser (Béné *et al.*, 2009). Similarly, freshwater harvests can act as a ‘safety net’ in times of hardship, for instance in times of drought or crop failure (Jul-Larsen *et al.*, 2003).

Throughout the thesis there is a focus on the subsistence and artisanal scale fisheries given their significance to food security, particularly for the rural poor. With around 90% of recorded fishes classified as ‘small-scale’ and more than 100 million more people involved with the post-harvest sector of these small-scale fisheries (Béné *et al.*, 2007), freshwater taxa play a significant role in the livelihoods and food security at the local level, particularly those living in rural areas and many of whom are living below the poverty line. They are of particular importance in Africa and Asia, where inland fisheries make a significant contribution to food and nutrient security (Béné *et al.*, 2007; Dugan *et al.*, 2010; Joffre *et al.*, 2012). Furthermore, while inland fisheries in developing countries provide only 33% of small scale fishery catch globally, they play a disproportionally large role in delivering livelihoods, particularly to women (who spend more of their income on family needs including food and medicine, Weeratunge *et al.*, 2010); they provide employment to more than 60 million people, totalling 56% of all people and 55% of women in developing countries working in fisheries (Dugan *et al.*, 2010; Mills *et al.*, 2011). It is likely that a large proportion of under-reported catch comes from subsistence and artisanal fisheries harvested by the rural poor, and therefore the significance of these fisheries to vulnerable groups is under-valued. The rural poor rely heavily on local ecosystems for primary goods and services and consequently the importance of biodiversity to food security in the developing world cannot be overstated (Snel, 2004).

While the dominant contribution to food security from freshwater harvesting is fishes it is noteworthy that other taxa also make significant inputs, particularly at the local scale. Approximately half a million tonnes of freshwater crustaceans are recorded as harvested annually at a value of USD 1 billion, and 375,000 tonnes of molluscs (FAO, 2015). As with fisheries, it is likely that the true contribution to food security from these species is likely to be significantly more. While the major focus of freshwater food security provision in this thesis will be from fish harvests, catch from other freshwater taxa is also considered.

Box 1.2 Overview of main threats to freshwater systems and species

Dams	Alters temperature, timing, speed and quality of river flow, and nutrient and sediment transport. Results in loss of floodplains and other wetlands. Blocks movement, connectivity and migration. Result in decline in fisheries productivity (Richter *et al.*, 2010). Rivers have been fragmented by more than 1 million dams globally (Jackson *et al.*, 2001), and reservoirs trap approximately 25% of sediment (Vörösmarty & Sahagian, 2000).
Water abstraction	Reduces timing and quality of river flow, reduces volume of standing water. More than 50% of freshwater runoff is captured for anthropogenic use (Jackson *et al.*, 2001), yet demand for water for industrial, municipal and agricultural purposes is still on the increase. Competition with anthropogenic uses of water is estimated to threaten biodiversity in at least 65% of global river discharge (Vörösmarty *et al.*, 2010).
Overharvesting	Depletes populations, alters food chains and biodiversity, and shifts catch to smaller species and individuals (Allan *et al.*, 2005), which in turn can result in ecological changes which reduce the resilience of the ecosystem to external shocks (Holmlund & Hammer, 1999). Most inland capture fisheries that are not stocked are overfished, or are being fished at their biological limit (FAO, 2003).
Pollution	Classed as both the addition of toxic materials and eutrophication from excessive nutrient discharge. Changes the chemical and biological composition of waterbodies and reduces oxygen content of water. Historically pollution has been a major problem in developed countries, which have largely improved in recent decades. Therefore is increasing concern now for developing countries, where for instance as much as 90% of wastewater is discharged untreated directly into waterways (Vörösmarty *et al.*, 2010).
Invasive species	Alters biodiversity, food chains, production and nutrient cycling. Fisheries using introduced species can be impacted through

	environmental disturbance, predation, competition, introduction of disease, and genetic contamination or hybridisation (Welcomme, 2001).
Land use change	Direct effects (e.g. draining of wetlands, canalisation of waterways) result in loss of habitat and ecosystem, and change in quality and quantity of water flow. Changes in terrestrial land use (e.g. deforestation, agriculture) also have profound knock on effects in flow of water, sediment, nutrients and pollutants.
Climate change	Variability in temperature and precipitation affect flow regimes, affecting availability of breeding and feeding habitats (e.g. floodplains). Fisheries catch linked to water levels, with droughts causing massive declines in African fisheries (Laë, 1994; van Zwieten & Njaya, 2003). Increased temperatures can have direct physiological effects resulting in reduced growth, reproductive success, tolerance to pollution and disease, and overall survival (Halls, 2009; Welcomme *et al.*, 2010).

1.5 Threats and conservation of aquatic ecosystems

Freshwater biodiversity is vastly disproportionate in species diversity to the extent of freshwater globally. Freshwater habitats cover only 0.8% of the Earth’s surface, and contain just 0.01% of the planet’s water. Yet of all known species described to date, 10% of the global total are freshwater species, and freshwater fish alone account for a quarter of the world’s vertebrate diversity. Adding in amphibians, aquatic reptiles and mammals, as much as a third of all vertebrates are confined to freshwater habitats (Dudgeon et al. 2006). There is an exceptional level of endemism in many freshwater taxa due to the restricted habitats within the water bodies, and the limited connectivity defined by the nature of the hydrologic system. However the limited connectivity also restricts species’ ability to migrate in response to potential or realised threats, making them particularly vulnerable (Jaeger *et al.*, 2014).

Despite increasing recognition of the value of inland waters, over half of the world’s watersheds have been impacted by anthropogenic activities (Postel & Daily, 1996) and freshwater systems are the most threatened ecosystem type globally (Boon, 2000; Strayer

& Dudgeon, 2010; WWF, 2014). Some of the most pressing threats to global inland water systems are outlined in Box 1.2. Several of freshwaters most serious threats arise from the terrestrial land use surrounding the waters; for instance, the run-off of pollutions, eutrophication due to run-off of nitrogen and phosphorous from agricultural lands, changes in sediment and groundwater flow due to deforestation (Box 1.2). For this reason, when considering threats to freshwater systems the entire catchment or sub-catchment should be considered (Collares-Pereira & Cowx, 2004). This also corresponds to an appropriate management unit scale (Luck *et al.*, 2009).

It is rarely possible to distinguish the extent of impact any one of these threats may have on a system, however in much of the world's inland waters there are a combination of factors threatening the system and the species within (Dugan *et al.*, 2010). These processes not only impact wildlife but also ecosystem service delivery, including the health and productivity of fisheries. Conversely, ES delivery can be the cause of threats, e.g. through overharvesting. The overexploitation of one (or many) species leads inevitably to population declines, but also reduces the size of individuals. Paradoxically the collapse of particular species can be masked by an increase in overall yield, yet there are knock on trophic and ecological effects to other species (Allan *et al.*, 2005). Similarly, the extraction of water is vital to many ES, yet has a significant negative impact on the delivery of others, including fisheries and the provision of food security. For ES delivery to be sustainable it must not decrease capacity for future provision or for delivery of other services (Villamagna *et al.*, 2013). The ability to meet the human demands for water without compromising the flow of ES will be extremely challenging (Naiman & Dudgeon, 2010). Further to this are the conservation needs of freshwater taxa, regardless of how well their input into ES delivery is understood. The trade-off between the provision of goods and services, the human need for water and a reduction or prevention of biodiversity declines can only be addressed if waterways are recognised as legitimate users of water in their own right (Naiman *et al.*, 2002).

The ongoing delivery of food security from inland waters and the preservation of the freshwater systems on which they rely need not be mutually exclusive. While overharvesting is a direct cause of species declines, many of the threats facing freshwater systems endanger both the biodiversity and the provision of food security. It is therefore important to understand how food security may affect and be affected by conservation proposals (Fisher & Christopher, 2007). There is a need for better comprehension of the synergies between areas most important for biodiversity and those for delivering crucial ES (Larsen *et al.*, 2011; Durance *et al.*, 2016).

1.6 Thesis outline

The aim of this thesis is to explore the connection between freshwater taxa, food security and ecosystem services (particularly for poorer communities), and biodiversity conservation. The work has focused on the following research questions:

1. How important are fisheries as an ecosystem service to different stakeholders?
2. Is there a relationship between freshwater biodiversity and fishery yields?
3. Can the importance of fisheries as an ecosystem service be spatially predicted?
4. How can the relationship between freshwater biodiversity conservation and food security from fisheries be spatially assessed?

For the purposes of this thesis biodiversity is defined by species richness. While this does not incorporate the genetic and population level diversity which can be a key consideration in the preservation of ecological processes (Noss, 1990), it is an iconic measure, and has been chosen as a reflection of the availability of comprehensive data, and the applicability of the outputs to a wider audience.

The thesis begins with a case-study which examines the understated value of inland fisheries as an ecosystem service (Chapter 2). Comparison of a monetary assessment of aquatic ecosystem services versus a site specific stakeholder assessment of services highlights that inland fisheries may be far more important than standard ecosystem service assessments suggest, particularly for poorer groups. Chapter 3 explores the importance of biodiversity to fisheries provision at the global scale, both in terms of fisheries productivity and fisheries resilience. Chapter 4 designs a supply and demand model to globally map the importance of inland waters as a direct source of food security. This is then extended to focus in particular on areas of greatest importance to the most poor and vulnerable groups in Chapter 5, which also explores the spatial synergies and trade-offs of catchment management to benefit freshwater species conservation and/or maintain inland catch fisheries. Each chapter is written as a stand-alone manuscript, and Chapters 2 and 3 appear as they have been published in peer-reviewed journals (Brooks *et al.*, 2014, 2016).

Prior to starting this thesis I worked for the Freshwater Biodiversity Unit of the International Union for the Conservation of Nature (IUCN), the CASE partner for this project. During this time I was instrumental in the process of training experts, collating data to inform the species assessments and range maps, and facilitating peer review workshops to quality assure the final IUCN assessments and maps. It is the species data I worked on for Africa and Asia that is used in some parts of this thesis, alongside other open source datasets. Although two of the Chapters presented here have been published with co-authors, all analyses within this thesis have been carried out by me. For each of the published papers I was responsible for the majority of the design of the analyses and the writing of the manuscript, and I responded to the reviewers as lead and corresponding author.

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2 Effects of methodology and stakeholder disaggregation on ecosystem service valuation

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The original concept was discussed by EB and KS, with methodology designed primarily by EB with contributions from RH and FE. EB performed the research and analysed the data. EB wrote the manuscript, with contributory edits from all authors.

2.1 Abstract

Contingent valuation is one of the most commonly used methodologies utilized in ecosystem service valuation, thereby including a participatory approach to many such assessments. However, inclusion of non-monetary stakeholder priorities is still uncommon in ecosystem service valuations and disaggregation of stakeholders is all but absent from practice. Here, we look at four site-scale wetland ecosystem service valuations from Asia that used non-monetary participatory stated preference techniques from a range of stakeholders, and compare these prioritizations to those obtained from the largest economic assessments available globally – the Ecosystem Service Value Database (ESVD). Stakeholder assessment suggests very different priorities to those from economic assessments, yet priorities between different sites remained broadly consistent. Disaggregation of beneficiaries in one site showed marked differences in values between stakeholders. Economic values correlate positively with values held by government officers and business owners, but negatively with fishermen and women who are relying most directly on the wetland ecosystem services. Our findings emphasize that ecosystem service assessment – economic or otherwise – must capture the diversity of values present across stakeholder groups in order to incorporate site scale management issues, particularly in relation to poverty alleviation.

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2.2 Introduction

Since the publication of the Millennium Ecosystem Assessment (MEA, 2005) the provision of ecosystem services (ES) has been increasingly encouraged for consideration in policy and decision making (TEEB, 2012). This has particular relevance to much of the developing world, where human impact on the environment is expected to accelerate with increased impetus for development. With a focus on human well-being and social development, the ES concept has the potential to address poverty alleviation in conjunction with ecological concerns (Tallis *et al.*, 2008). ES assessment permits the environment to be placed within a development framework, and promotes the viewpoint that progression towards the Millennium Development Goals can be achieved alongside a context of conservation priorities (Sachs *et al.*, 2009). A major focus of the ES literature is on economic valuation and the methodologies to best appraise this (Hein *et al.*, 2006; Bateman *et al.*, 2011; TEEB, 2012). Economic values provide a commonly understood and comparable methodology to quantify the value of ES (Balmford *et al.*, 2002; Farber *et al.*, 2002), yet the use of monetary values in assessment also sets precedence to base policy on the economic outcomes. As the concept of ES accounting acknowledges that poorer households face disproportionate losses from the depletion of ES (TEEB, 2008), economic valuation could have negative consequences for those living in poverty if those ES on which they rely are not considered most financially valuable to society as a whole.

Participatory valuation techniques, which can be can be used to establish monetary or non-monetary values, are intended to overcome these issues to reflect not only the biophysical but the cultural and societal benefits from ES, and there is a growing literature that emphasizes the role of stakeholder participation in the assessment process (Cowling *et al.*, 2008; Reed, 2008). Non-monetary participatory techniques however are rarely seen to be used in valuation literature, with the most commonly applied mechanisms for stakeholder participation within ecosystem assessment are participatory ‘stated preference’ techniques such as contingent valuation or revealed preference techniques (TEEB, 2012). These are popular tools for providing economic values for non-direct goods; indeed contingent valuation is the most widely used methodology within ES valuation (Graves *et al.*, 2009).

The application and efficacy of these participatory valuation techniques is still debated (e.g. Skourtos *et al.*, 2010). Participatory approaches are largely applied to one or few ES (e.g. Hein *et al.*, 2006; Jenkins *et al.*, 2010), or for a combined-service scenario (e.g. Zander & Garnett, 2011; Kaffashi *et al.*, 2012), and therefore do not provide a stakeholder comparison across the full suite of ES of relevance. This provides little opportunity for stakeholders to contribute to often complex policy or management decisions beyond an individual monetary bid or preference for a limited question (Chee, 2004). Moreover, ES valuation tends to focus on assigning mean values derived for the affected society as a whole (Farber *et al.*, 2002). Considered as one aggregated group with no discrimination of different beneficiaries of different ES, this framework is likely to limit the contribution of ES consideration to poverty alleviation (Daw *et al.*, 2011).

Given the limited inclusion of stakeholders in most assessment processes, there is a lack of connection between the methodologies utilized (economic valuation) and the drivers of the framework (human well-being and poverty alleviation). However, to date there have been few direct comparisons of monetary and non-monetary methods, nor of the impact that disaggregating by different groups of stakeholders has on findings (Daw *et al.*, 2011). Here we examine the relationship between values of a non-monetary stated preference approach in four disparate sites in Asia and those derived from one of the largest databases of economic values of ES ever compiled, the Ecosystem Service Value Database (ESVD) (van der Ploeg *et al.*, 2010; de Groot *et al.*, 2012). We also examine the extent to which disaggregation of stakeholders at one site reflects the aggregated economic and non-economic valuations, and discuss the implications of these findings for the use of ES valuation in decision making.

2.3 Methods

2.3.1 Site description and collection of primary data

This research is part of a larger interdisciplinary project, Highland Aquatic Resources Conservation and Sustainable Development - HighARCS (www.higharcs.org). The project used an integrated approach to develop knowledge on the importance of aquatic resources at five highland sites in Asia. As part of the integrated site assessments, an ES valuation assessment was performed at each site. This information was used alongside further data including biodiversity surveys and socio-economic analyses to inform site-specific Integrated Action Plans designed to enhance poor livelihoods and contribute to highland aquatic resource conservation and sustainable use at each site (see website for more details).

The data presented here are a comparison of the ES valuation element of the project from four of the five sites. The project sites fall within Shaoguan, China; Buxa, West Bengal, India, and; Quang Tri and Son La, Vietnam (see Figure 2.1). An outline of each site is given in Appendix A.1.

Following centralized group training, a non-monetary participatory appraisal of ES was performed at each site by the in-country partner (see acknowledgments): focus groups were held to identify the ES used by stakeholders specific to the site, and to determine groups of stakeholders relevant to the site. Details of the full suite of ES selected for each site can be found in Table A1 in Appendix A.2. Individual interviews were then conducted across the range of stakeholders and participants were asked to give a score for each of the selected ES based on importance.

We compared the results of these non-monetary participatory approaches with mean economic values calculated from estimated pricings from the ESVD (van der Ploeg *et al.*, 2010). The ESVD has collated over 1350 value-estimates, and offers a useful tool to examine value importance of different ecosystem services in multiple biomes (de Groot *et al.*, 2012). It represents an extensive scope of values from the literature, collated from different studies conducted at a range of scales, including local scale. We used values from the global database to obtain estimates of the monetary values for the services considered in this analysis due to a paucity of regionally relevant data. Given that the sites are

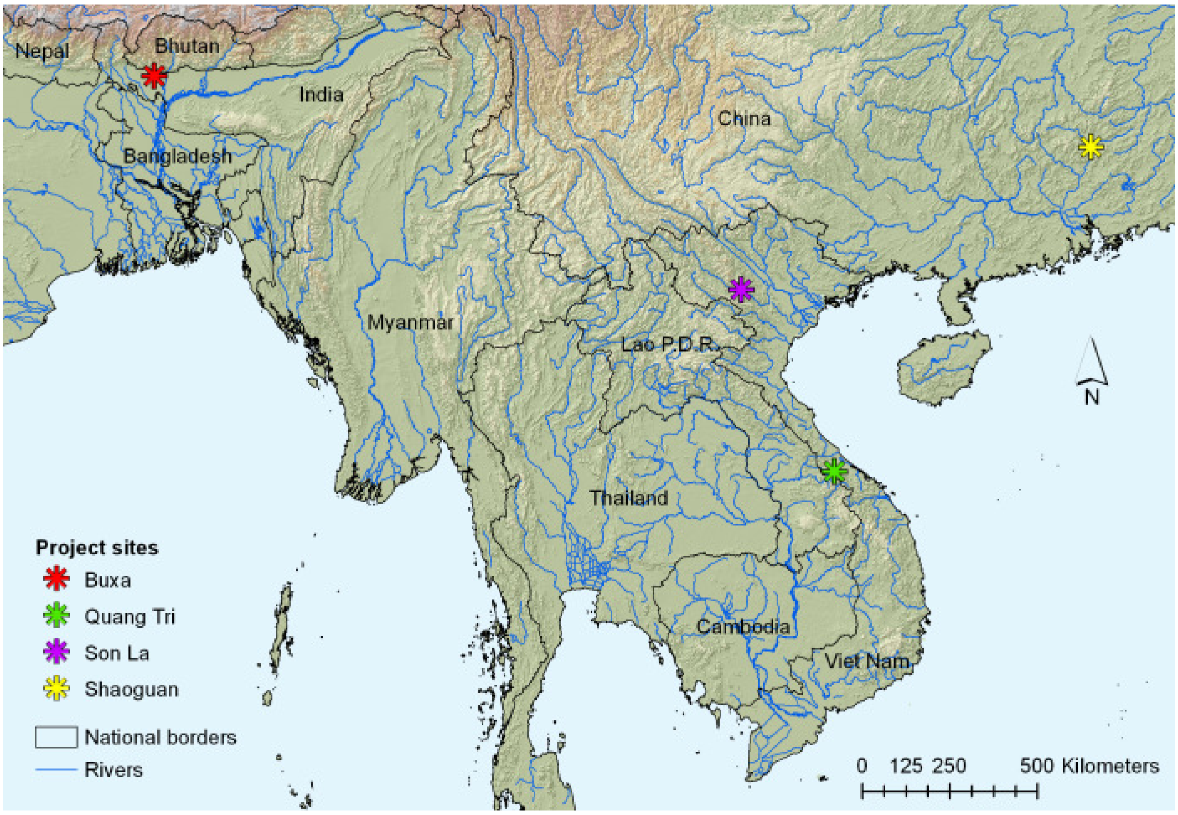

Figure 2.1 Map of HighARCS project sites.

geographically disparate across SE Asia this non-region specific estimate is a generalized value for comparison across all sites. Mean values were calculated from all values in the database for the specific freshwater ES, except those not given as price/ha/year. Those estimated from a benefits transfer methodology were also excluded as benefits transfer studies do not collect new data, but rather use existing studies as the basis of valuation estimates. In line with de Groot *et al.* (2012), a purchasing power parity (PPP) conversion factor was applied to all values for the relevant year (WorldBank, 2012), and adjusted for inflation to the 2010 rate (BLS, 2012).

In general, only ES identified and valued by all sites as part of the focus group process, and included within ESVD, were used in the comparison. In exception, Buxa did not consider hydropower or cultural value within their study. However, we included these services due to the high importance expressed at the other three sites.

Scores from the stakeholder interviews and values from monetary assessments were normalized onto the same 0-1 scale, where 0 indicates a low value and 1 a high stakeholder or economic value. At each site scores were normalized using min-max normalization; the minimum response score was subtracted from each response score, which was then divided by the range of scores for that site. As a result, the highest response score was converted to 1, and the lowest response score was converted to 0. A mean value per service was then calculated from the normalized scores to produce a non-monetary value for each ES at each site (hereafter referred to as participatory values). Economic values are a normalization of the mean monetary value for each service from the ESVD (hereafter referred to as economic values).

2.3.2 Comparisons between sites

Due to the differences in methodologies of valuation, values between economic and participatory values were not compared directly but instead focused on how relative rankings varied between sites. Spearman's rho non-parametric correlations were used to quantify these relative rankings, both for inter-site comparisons, and to compare ranks of services between the economic values and the normalized non-monetary values for each site. This methodology follows practice of previous studies where mean scores have been compared directly (e.g. Rouquette *et al.*, 2009). However this alone does not account for the variation surrounding these means. Therefore 95% confidence intervals were generated for Spearman's rho by bootstrapping the data: each pair of site (and ESVD) data sets were randomly sampled with replacement to create corresponding sample data sets the same size as the originals. Spearman's rho was then calculated between each pair of sampled data sets. This was then repeated 10,000 times per pair of sites, from which 95% confidence intervals could be calculated. A correction for distribution bias from within the sample was applied using the bias-corrected and accelerated method (BCa) (Haukoos & Lewis, 2005). All calculations were carried out in R 3.0.1.

2.3.3 Disaggregation of ecosystem service valuation by stakeholder type

In addition to examining differences in ES values between sites, we also analyzed concordance in ES valuations between stakeholder groups in one site. As stated, each in-country partner identified their own stakeholders and did not divide stakeholder into the same groups. We therefore focused individual stakeholder analysis only on data from Shaoguan as they identified a good cross-section of widely applicable stakeholders, with a minimum sample size of 13 per stakeholder group. Participants represented four distinct stakeholder groups; government officers, business owners, farmers, and fishermen and fisherwomen (hereafter referred to as fishers). Farmers and fishers from the site are considered rural poor, with fishers being more dependent on direct resources from the land (Yiming *et al.*, 2010). Many of the fishers are described as subsistence communities, with the poorest households relying most directly on aquatic resources. ES scores from the stakeholder groups were compared against economic values as above.

2.4 Results

2.4.1 Comparisons between sites

The normalized responses of the importance of each ES to each site – aggregated across all groups of stakeholders - and the mean economic values for each ES are shown in Figure 2.2. There are marked differences between the range of scores from economic values, compared to the consistently high participatory values from any of the sites. The value of water supply is more than the double that of the next most economically valued service, flood control, which in turn is more than double of the next most valuable service, water treatment. In contrast, the participatory values assigned to different services are broadly similar. Difference in scores of ES between sites are relatively small, with a spread varying from 0.16 between sites for the value of fisheries, and 0.38 for the value of water supply.

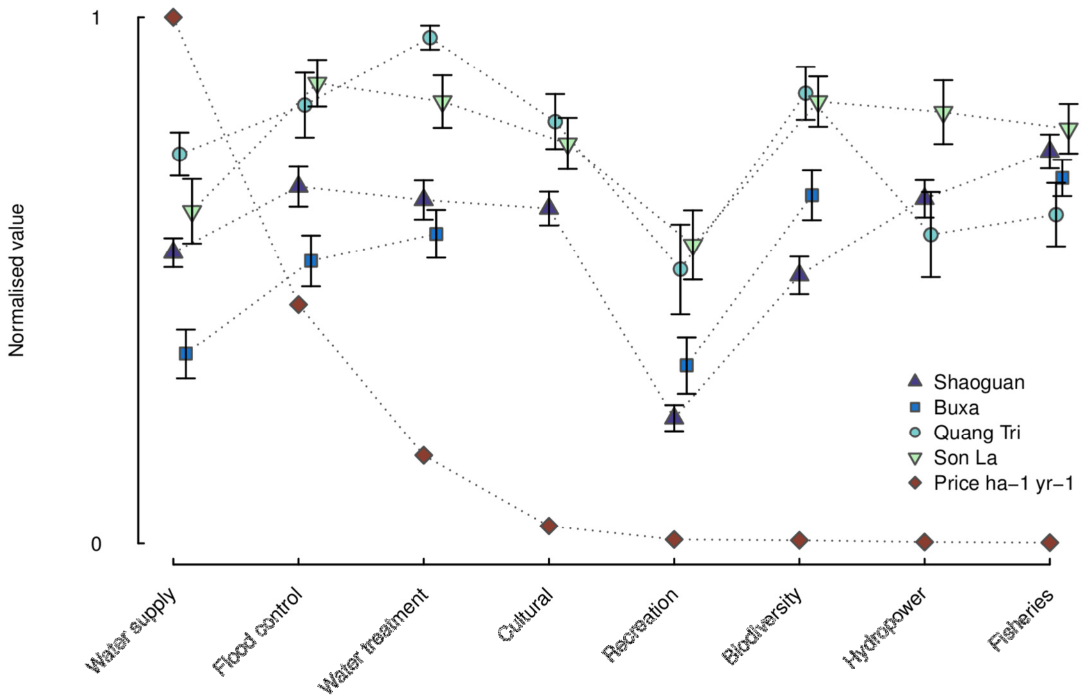

Figure 2.2 Normalized scores and standard error of ecosystem services of mean stakeholder perception values from four sites in Asia, and normalized mean monetary valuation. Lines have been added to help visualize the differences in rank of the services within each site, compared to the monetary valuation.

Correlations between participatory valuations of sites vary, but broadly correlate with each other exhibiting medium to strong correlations (Table 2.1). The exception to this is between Shaoguan and Quang Tri. By contrast, there is little congruence between the rankings of ES from participatory values at each site versus the economic values in all cases except Buxa, which shows a moderate negative correlation.

2.4.2 Disaggregation of ecosystem service valuation by stakeholder type

Disaggregation of beneficiaries in Shaoguan showed that there is large variation in the values that different groups of stakeholders place on ES. Male and female responses in Shaoguan were highly correlated (Spearman's Rho=0.78, df=21, P <0.001) and therefore not separated for the stakeholder disaggregation analyses.

Correlation between the rankings scored by different stakeholder groups and the economic values from ESVD show a high degree of variation (Table 2.2). Moderate to strong positive correlations are found between economic values from ESVD and the

Table 2.1 Spearman's rho correlations and BCa 95% confidence intervals between valuation scores of freshwater ecosystem services by monetary assessment and stakeholder values at four sites in Asia.

Valuation sources		Rho	BCa 95% CI
Economic	Shaoguan	-0.21	-0.55 – 0.02
Economic	Buxa	-0.66	-0.94 – -0.54
Economic	Quang Tri	0.40	-0.12 – 0.62
Economic	Son La	0.04	-0.38 – 0.60
Shaoguan	Buxa	0.60	0.26 – 0.83
Shaoguan	Quang Tri	0.07	-0.38 – 0.40
Shaoguan	Son La	0.49	0.19 – 0.90
Buxa	Quang Tri	0.37	0.06 – 0.77
Buxa	Son La	0.46	0.03 – 0.60
Quang Tri	Son La	0.71	0.48 – 0.98

Table 2.2 Spearman's rho correlations and BCa 95% confidence intervals between valuation scores of freshwater ecosystem services by monetary assessment and different stakeholder groups in Shaoguan, China.

Valuation sources		Rho	BCa 95% CI
Economic	Government	0.55	0.14 – 0.81
Economic	Business	0.72	0.45 – 0.93
Economic	Farmer	0.23	-0.31 – 0.63
Economic	Fisher	-0.52	-0.74 – -0.48
Government	Business	0.74	0.43 – 0.98
Government	Farmer	0.71	0.35 – 0.98
Government	Fisher	-0.07	-0.41 – 0.06
Business	Farmer	0.60	0.32 – 0.98
Business	Fisher	-0.07	-0.30 – 0.21
Farmer	Fisher	0.46	0.05 – 0.84

participatory values of government officers and business owners. ESVD values have no correlation with those of farmers, and there is a moderate negative correlation between economic values and participatory values of fishers. The participatory values of government officers also strongly positively correlate with business owners and farmers, whose values also positively correlate with each other. The values of farmers and fishers also positively correlate. Value scores of different stakeholder groups are presented in Figure 2.3, showing a marked difference again in breadth between economic scores and participatory scores for all ES excluding water supply. The similarity of the normalized

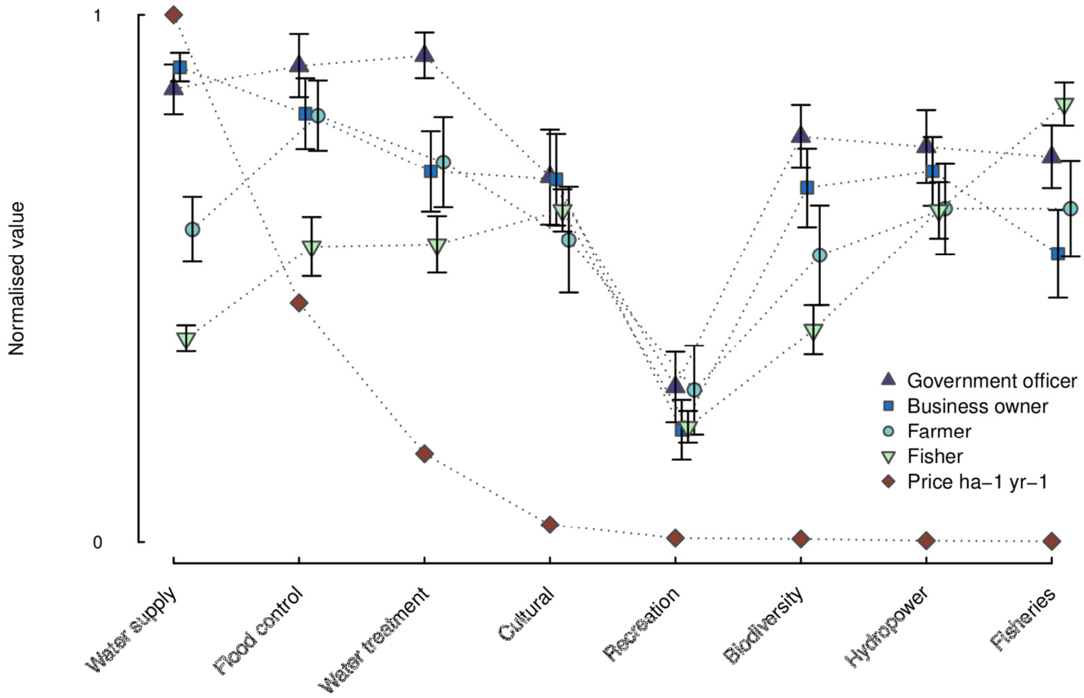

Figure 2.3 Normalized scores and standard error of ecosystem services of mean stakeholder perception values from Shaoguan, China, and normalized mean monetary valuation. Lines have been added to help visualize the differences in rank of the services within each site, compared to the monetary valuation.

scores between stakeholders differs by ES, with a spread varying from 0.08 between stakeholders for recreation value, and 0.51 for water supply.

2.5 Discussion

Values placed on freshwater ES across four disparate rural sites were broadly similar when measured by participatory assessments, but differed markedly when compared to globally derived economic values of the same services. Disaggregation of beneficiaries at one site shows that generalized economic valuation is likely to be a poor reflection of the values farmers and fishermen place on freshwater ES, but broadly correlate with the values placed on these resources by business owners and government officials.

Stakeholders with differing local sensitivities, cultural backgrounds and geographical locations might be expected to attach differing values to ES (Hein *et al.*, 2006), yet despite the situational differences there is remarkably little difference between the values scored at sites in the current study. Given that the sites considered here were asked to identify priority ES, it is unsurprising that all ES are scored reasonably highly, and a greater range of scores would be expected had a strict set of ES been tested. Nonetheless, the small interand intra-site variation found indicates a similarity of values for the ES considered held by stakeholders across a range of sites. This supports the strength and robustness of the participatory approach used here as a valuation method.

The implied similarities between sites are more strongly supported when comparing the priorities based on the rank order of participatory value means. Given that this work has been conducted as part of a large consortium project, the congruence between sites is encouraging. All project partners were given the same methodological training and instruction, but it has been carried out by different teams in different countries, on wetland sites that vary in their issues. Each site identified its own priorities and stakeholders, yet the ES valuations are remarkably concordant, showing that even across different countries, values for ES are similar at the community level.

A comparison of normalized values from an economic perspective suggests a disparity between values garnered by a non-monetary participatory approach and those projected from an economic approach. If multiple wetland ES were valued to a similar degree to each other in monetary terms it would be entirely plausible that they would well match the participatory scores once normalized. Instead we find quite a steep decline in economic values, indicating that a few wetland services have high economic value while the rest have little economic value, which is in contrast to the participatory values. Coupled with this the differences in rank orders suggest an incongruence of prioritizations of ES between the two methodologies. These results emphasize the need to understand and incorporate non-monetary values in order to fully inform environmental management and decision-making (Martín-López *et al.*, 2007).

Critically, the findings of the study show a disaggregation of stakeholder views and suggest that the values held by those relying most directly on wetland ES (here the fishers, followed by the farmers) differ most widely from those predicted by economic valuation. These results are perhaps unsurprising, yet are nonetheless extremely important given the implications on equity, in particular within a poverty alleviation framework, or when assessing ES whose beneficiaries include a range of stakeholders including vulnerable groups.

Coupled with the correlations of prioritizations expressed by both government officers and business owners with economic values, it is clear that it is vital to consider that the opinions of decision makers (here represented by government officers and business owners) may not concur with those stakeholders that are most directly dependent upon the

ES. Commonly in assessments where stakeholders are considered only a subset of stakeholders are included, or the perceptions of all stakeholders are amalgamated together, despite the expectation that different stakeholder groups would hold different values for the same ES (Vermeulen & Koziell, 2002; Daw *et al.*, 2011). The correlation in our study between the economic valuations and the stakeholders that are most likely to be influential in decision-making (here, government officers and business owners) highlights the disparity of representation by standard economic valuations for all stakeholders, and especially the poor. In particular the government officers would be expected to represent a general public value which may downplay interests of specific groups. While government officers in this study do correlate well with business owners and farmers, there is no correlation with the fishers, the most vulnerable group.

As stated, locally explicit participatory values have been compared to generalized economic rankings from multiple scales, and localized valuation are unlikely to reflect global values due to generalization error (Plummer, 2009). However, given the lack of congruence between stakeholder groups, it is unlikely that local economic valuation would suitably reflect all groups, and it is interesting to see that even non-region specific generalized economic rankings have such strong correlation with business owners, the stakeholder group expected to be most influenced by economic concerns. In turn this suggests that economic valuation supports and represents the interests of businesses over that of local communities, and while a potentially useful tool in some contexts may not be representative in a poverty alleviation framework, which should be carefully considered in any valuation process.

It is likely that the large disparity in values – economic or otherwise – that different groups of stakeholders in Shaoguan placed on ES represent differences in the spatial scales at which stakeholders value ES (Hein *et al.*, 2006; TEEB, 2012). The purpose of the majority of valuation assessments is to inform management plans but these are likely to be at a larger scale of consideration than site-specific issues such as those explored in this project. For instance for freshwater the basin or catchment level scale is increasingly considered the most useful management unit for inland waters (IUCN, 2000; Collares-Pereira & Cowx, 2004; Vigerstol & Aukema, 2011) which would not relate well to considerations at the community-level. In addition, a lack of data on most ES in most regions means that datasets such as the ESVD are likely to be used in valuation assessments, which – as we have shown – is unlikely to reflect the priorities of the poorest rural stakeholders.

Our results add to the literature showing that ES valuation assessments must disaggregate stakeholders not only to represent the entire range or dependencies and benefits from the system, but also to consider the benefits at multiple scales (Hein *et al.*, 2006; Martín-López *et al.*, 2007). For example, fisheries production is economically the least valuable of the ES considered, yet was consistently scored highly at the sites, with stakeholders at Shaoguan and Buxa scoring it as the most valuable ES provided. While in monetary terms fisheries have relatively little importance for society as a whole, for some that live close to these wetlands fish are paramount to their subsistence and livelihoods and often act as an emergency resource when other livelihoods options fail (Béné *et al.*, 2009). If decision-making is based on economic assessment to wider society alone, the importance of this ES at the local scale to the over 60 million people in developing countries dependent on freshwater fisheries (Dugan *et al.*, 2010) is potentially ignored. Government managed sites have been shown to be managed for higher economic value than community-managed sites (Hicks *et al.*, 2009), which is unlikely to suit the interests of local livelihood dependence. The comprehension of ES and natural capital currently struggles to incorporate the considerable worth of non-financial benefits, particularly where this benefit is felt by only a sub-set of stakeholders.

Participatory assessment can be cheaply and easily undertaken (Springate-Baginski *et al.*, 2009). Some limitations of other participatory approaches (e.g. Skourtos *et al.*, 2010) can be mitigated using a non-monetary methodology as used here, as it does not require a financial value input, and any and all ES can be considered. This approach can broaden the valuation process beyond easily mapped ES, such as water provision and carbon sequestration data, and allows for the consideration of the full range of ES relevant to the study, and at the same scale, without the bias of capital prioritization. The methodology incorporates participation into the valuation process which increases uptake and success of resulting recommendations within communities, particularly those in least developed countries (Christie *et al.*, 2012). Unsurprisingly, the slight differences not only in valuation but also in ranking between the sites (Figure 2.2, Table 2.1) indicate that benefits transfer is unlikely to be an appropriate application of this methodology, although it is of note that the two sites with the strongest correlation were between the two closest sites, Quang Tri and Son La, both located in Vietnam.

In summary, our results provide important additional evidence that conclusions on ES assessments for poverty alleviation cannot be drawn unless the considerations of target poor groups are incorporated (Daw *et al.*, 2011). It is unlikely that either stated preference or economic valuations in isolation are enough to fully inform an ES assessment which includes poorer households among its dependents, but an appropriate integration of both approaches would lead to better informed decision-making. To properly consider the importance of services and to inform policy decisions it is important to adopt a more comprehensive approach that disaggregates stakeholders and considers importance for societal groups whose interaction with systems operates at differing scales. Applying an assessment methodology that includes multiple stakeholder priorities will maximize the beneficiaries of an ecosystem service approach.

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3 Global evidence of positive impacts of freshwater biodiversity on fishery yields

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The original concept was discussed by all authors. EB designed the methodology and statistics and conducted all of the analyses. EB wrote the manuscript, with contributions from all authors.

3.1 Abstract

3.1.1 Aim

An often-invoked benefit of high biodiversity is the provision of ecosystem services. However, evidence for this is largely based on data from small-scale experimental studies of relationships between biodiversity and ecosystem function that may have little relevance to real-world systems. Here, large-scale biodiversity datasets are used to test the relationship between the yield of inland capture fisheries and species richness from 100 countries.

3.1.2 Location

Inland waters of Africa, Europe and parts of Asia.

3.1.3 Methods

A multimodel inference approach was used to assess inland fishery yields at the country level against species richness, waterside human population, area, elevation and various climatic variables, to determine the relative importance of species richness to fisheries yields compared with other major large-scale drivers. Secondly, the mean decadal variation

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in fishery yields at the country level for 1981–2010 was regressed against species richness to assess if greater diversity reduces the variability in yields over time.

3.1.4 Results

Despite a widespread reliance on targeting just a few species of fish, freshwater fish species richness is highly correlated with yield ($R^2 = 0.55$) and remains an important and statistically significant predictor of yield once other macroecological drivers are controlled for. Freshwater richness also has a significant negative relationship with variability of yield over time in Africa ($R^2 = 0.16$) but no effect in Europe.

3.1.5 Main conclusions

The management of inland waters should incorporate the protection of freshwater biodiversity, particularly in countries with the highest-yielding inland fisheries as these also tend to have high freshwater biodiversity. As these results suggest a link between biodiversity and stable, high-yielding fisheries, an important win–win outcome may be possible for food security and conservation of freshwater ecosystems. However, findings also highlight the urgent need for more data to fully understand and monitor the contribution of biodiversity to inland fisheries globally.

3.2 Introduction

The degree to which species diversity underpins ecosystem functioning, and ultimately ecosystem services, is a central question in ecology with significant implications for policy and conservation (Mace *et al.*, 2012). It is now well established that biodiversity frequently has a positive effect on ecosystem functioning (Balvanera *et al.*, 2006; Duffy, 2009; Cardinale *et al.*, 2011). However, not all studies support this conclusion (Hooper *et al.*, 2005). Observational studies appear to contradict results from experimental studies, and vary in clarity and the direction of any relationship (Naeem, 2002). It is therefore possible that biodiversity–ecosystem functioning experiments may not be indicative of realized differences in ecosystem functioning in natural systems. This disconnect may in some cases be due to differences in scale and system (Duffy, 2009), and there is a disparity between the small scales at which these experiments have been performed and the scale at which management and conservation decisions are made (Cardinale *et al.*, 2012).

The use of real-world datasets bypasses some of these issues, yet currently there are only a handful of sufficiently large-scale studies with which to examine the effect of biodiversity on ecosystem function in natural ecosystems. To date global studies have examined marine (Worm *et al.*, 2006) and botanical (Maestre *et al.*, 2012) systems. To understand whether there is a generalizable relationship between ecosystem function and biodiversity or whether a more idiosyncratic relationship exists there is a need to examine evidence across a range of scales, systems and taxa. Moreover, more work is needed to look beyond questions of generalized functionality and productivity and consider direct links to human wellbeing. Such work is important because there is a much poorer understanding of the links between biodiversity and the final ecosystem goods that actually confer benefits to humans (Mace *et al.*, 2012).

The question of the extent to which biodiversity underpins ecosystem services is relevant to a range of systems, but has perhaps the greatest policy relevance in terms of the provisioning services that underpin food security. Studies of the relationship between ecosystem functioning and biodiversity show that increased species richness (SR) may provide (1) a buffering effect in fluctuations of productivity and/or (2) an overall performance-enhancing effect (Yachi & Loreau, 1999). However, as these conclusions have been largely drawn from plant community experiments these mechanisms may have limited generality across systems (Pinto *et al.*, 2013). Although current focus in this area is on examining the impacts of biodiversity within human agricultural systems (e.g. as reviewed in Power, 2010), people also rely on natural habitats for the provision of food.

At least 2 billion people depend directly on inland freshwaters for the provision of food (Richter *et al.*, 2010), and in many parts of the world inland waters are often the primary source of protein and micronutrients (Béné *et al.*, 2007; Dugan *et al.*, 2010). In 2010, global inland capture fisheries yielded over 11 million tonnes, with inland aquaculture yielding up to four times that amount (FAO *et al.*, 2012). Globally there are hundreds, if not thousands, of freshwater species that contribute to food security, yet the relationship between species diversity and yield remains poorly understood in freshwater systems (Balmford *et al.*, 2008). Recent research suggests a performance-enhancing effect (Greene *et al.*, 2010; Carey & Wahl, 2011) and a buffering effect (Greene *et al.*, 2010; Franssen *et al.*, 2011) of biodiversity on yield associated with freshwater fish communities, although it is unclear how such results transfer to natural freshwater systems at larger scales (Carey & Wahl, 2011). Different species can make a disproportionate contribution to ecosystem functions (McIntyre *et al.*, 2007), but in practice most fisheries concentrate on maximizing biomass – which is highly affected by such factors as phosphorus levels and macrobenthos biomass in freshwater systems (Hanson & Leggett, 1982) – and have little interest in harvesting a diversity of species. As a consequence, there is no degree of certainty that higher freshwater biodiversity is linked to enhanced livelihoods and increased human wellbeing. Indeed most fishery managers would prefer the ease of managing a fishery based on fewer species for which stock assessment tools aiming at maximum sustainable yield are more easily applied. Therefore greater comprehension is needed of how the relationship between biodiversity and ecosystem functioning can influence our understanding of the implications of freshwater biodiversity loss, and contribute to defining management objectives for inland freshwater systems (Dudgeon, 2010).

Beyond food security, understanding the degree to which biodiversity underpins freshwater fisheries has particular policy relevance because freshwater systems are of major importance for the conservation of biodiversity. Freshwater habitats are disproportionately species rich given that they cover only 0.8% of the Earth's surface; 10% of species described to date and as many as a third of all vertebrates are confined to freshwater habitats (Dudgeon *et al.*, 2006). Freshwater systems are highly threatened, with many freshwater taxonomic groups facing a significantly higher extinction risk than terrestrial groups (Darwall *et al.*, 2008). As a result, if freshwater biodiversity is shown to generally underpin inland fisheries, the food security implications of this relationship would provide a powerful additional argument to conserve freshwater systems and the biodiversity contained within them above and beyond purely conservation objectives.

Here, datasets from the Food and Agriculture Organization of the United Nations (FAO) and the International Union for Conservation of Nature (IUCN) covering 100 countries are used to provide the first large-scale test of the hypotheses that high freshwater biodiversity has a positive effect on (1) fishery yields and (2) variability of yield over time. As ecosystem function is a result of more than just the target species (Hensel & Silliman, 2013), the analyses were conducted using fish SR and then repeated to include additional freshwater faunal groups, namely molluscs, odonates and decapods (see Appendix B.4).

3.3 Methods

All analyses, unless otherwise specified, were conducted in R, version 3.0.2 (R Core Team, 2014).

3.3.1 Freshwater biodiversity and yield

The relationship between yields of inland capture fisheries and biodiversity was examined using comprehensive datasets from IUCN (2012) and FAO (2011) along with other macroecological drivers (see Appendix B.7 for all data sources). Biodiversity analysis was based on species native range maps of 9075 freshwater species from the IUCN Red List of threatened species (IUCN, 2012), including 5203 species of fish, 1790 molluscs, 1329 odonates and 753 decapods. Range maps are compiled by experts in accordance with the IUCN Red List guidelines (available at www.iucnredlist.org) and derived from a combination of known and expected species localities. The IUCN spatial dataset is the most comprehensive continental-scale dataset available on the distribution of all known freshwater taxa from these groups mapped to the river/lake subcatchment scale. SR per country was calculated in ArcMap 10 (ESRI, Redlands, CA, USA) using the range maps from IUCN Red List assessments for fish alone and then all available freshwater taxa. Countries for inclusion in the analyses were restricted to those that have a complete suite of species range maps for the taxonomic groups considered and have been comprehensively assessed by IUCN, namely Africa, Europe and parts of Asia (see Figure 3.1B).

The FAO capture database FishStatJ is the most authoritative assessment of the status of world inland fisheries and reports national annual yield data since 1950 that can be filtered

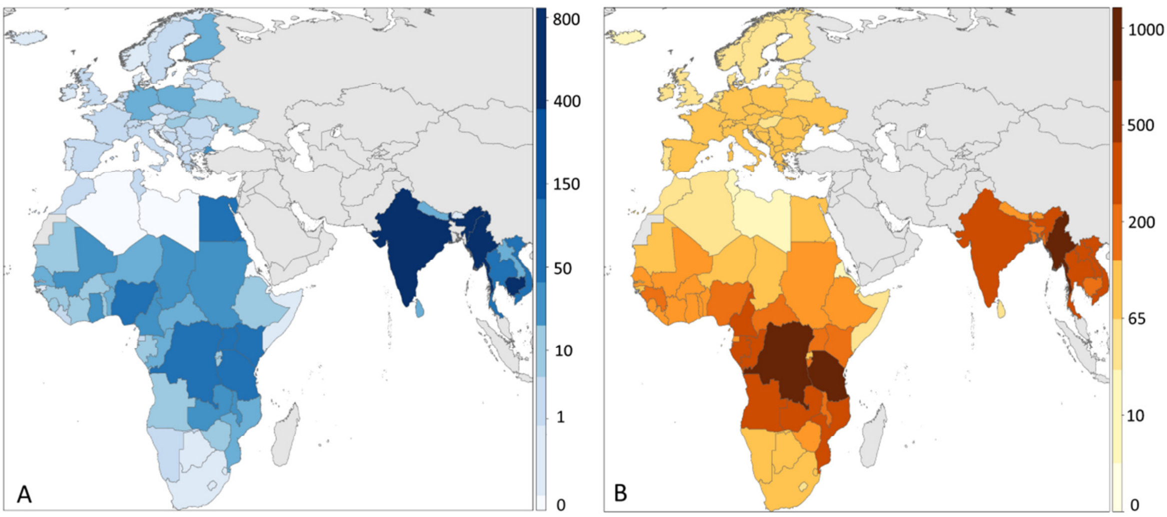

Figure 3.1 Data included within the study. (A) FAO inland water capture fisheries yield per country (thousands of tonnes) (axis quarter-root transformed). (B) Freshwater fish species richness per country (axis cube-root transformed).

by country, taxonomy, fishing area and yield measures (FAO, 2011a).

3.3.2 Macroecological and human drivers of fisheries yield

Fisheries yield is a product of biophysical drivers and human effort. As such, quantifying the effect of biodiversity (SR) on mean yield (t) was achieved by first building a predictive model of all putative large-scale drivers of fishery yield and then determining the relative importance of SR compared with the other drivers – area of surface water, fishing effort, productivity and elevation. Mean yield (t) was calculated at the country level from data for 2001–10 to best reflect the time period of the IUCN species assessments (conducted from 2004 to 2010) (Figure 3.1A). The sum of the surface area of inland waters (km^2) per country was extracted from the Global Lakes and Wetlands Database (GLWD) (Lehner & Döll, 2004) to account for available habitat for freshwater species. As no comprehensive data exist on per unit effort of fisheries or number of fishers at the country level, the human population within 10 km of inland waters for each country (hereafter the waterside population) was used as a proxy for fisheries effort. Population was extracted from a raster layer of the 2000 global rural population reported by FAO at 5-arcmin resolution (Salvatorre *et al.*, 2005), and a range of buffers around each of the waterbodies included in the GLWD were applied using ArcMap 10. Waterside population was validated against primary data on fishing effort available for 28 African lakes (see Appendix B.1), with the 10-km buffer yielding the strongest relationship ($r = 0.75$, $n = 28$, $P < 0.001$). Yield was not standardized by fishing effort or area (*sensu* Kantoussan *et al.*, 2014). This is because both fishing effort (waterside population) and area affect the yield relationship at the country level; considering both as predictor variables accounts for this shared effect. A more in-depth discussion of this issue is included in Appendix B.1.

Productivity, or energy available within the system (which is highly correlated with climate at the continental scale; Hawkins *et al.*, 2003), is a major driver of global freshwater biodiversity patterns (Tisseuil *et al.*, 2012). As no spatial data currently exist on global freshwater productivity, this factor was controlled for by using the principal components from a principal components analysis (PCA) carried out on 19 spatial climatic data layers including mean and seasonality for temperature and precipitation variables (Appendix B.2). A broken stick stopping model selected the first two principal components for inclusion in the model; these together account for 79.8% of climate variation. Use of PCA in this way reduces multidimensionality and eliminates collinearity between variables, and is an established approach for controlling for productivity across latitudes in continental-scale analyses (Hawkins *et al.*, 2003; Tisseuil *et al.*, 2012). Finally, mean elevation (m) was collated for each country.

3.3.3 Relationship between freshwater biodiversity and yield

All possible linear regression models were built using mean yield (t) as the dependent variable, with fish SR, waterside population, climatic PCA components, inland water surface area (km^2) and mean elevation (m) as explanatory variables. Testing the residuals of the models using Moran's *I* standard deviate test showed that there was spatial autocorrelation. Therefore a multimodel inference approach using simultaneous autoregressive spatial model (SAR) methods was conducted following Maestre *et al.*, (2012). The type of SAR used was the spatial simultaneous autoregressive error model (SAR_{err}) method, as this is robust to the type of spatial autocorrelation that is present in the data (Kissling & Carl, 2008), calculated within the spdep package in R. Spatial autocorrelation is accounted for by the inclusion of a spatial weighting matrix calculated based on distances between centroid points of countries. This spatial weighting matrix represents an additional term within the SAR model that describes relatedness between individual samples (countries) caused by spatial structure that is not fully accounted for by the other model parameters (Dormann *et al.*, 2007). Where necessary, Box–Cox transformations were used to normalize the distribution of the residuals, equalize the variance and improve the fit of the models (Osborne, 2010). The full SAR models are presented with the results tables (Table 3.1 in section 3.4.1 and Table B3 in Appendix B.6).

From all possible models, minimized second-order Akaike information criteria corrected for small sample size (AIC_c) were used to select the best fitting models. The AIC_c of all models selected that included SR as a predictor were compared with those of the same models not including SR. Where the AIC_c of models differs by less than 2, the models are considered to be indistinguishable (Burnham & Anderson, 2002). The Akaike weights of each model were calculated based on the ΔAIC_c, i.e. the difference between the AIC_c of each model and that of the best model (Burnham & Anderson, 2002), and therefore a set was created from all models where ΔAIC_c was different by less than two from the best model, hereafter known as the top model set. Multimodel-averaged parameter estimates of the analysis were calculated using the top model set. The relative importance of each predictor variable was calculated as the sum of the Akaike weights of all models that included the predictor of interest (Burnham & Anderson, 2002). Commonality analysis was then conducted to determine the unique, common and total effects of each of the variables within each of the top model sets (Nimon & Reio, 2011). The variance inflation factor (VIF) was calculated for the top models to check for collinearity between predictor variables.

3.3.4 Relationship between freshwater biodiversity and variability of yield

The coefficient of variation (CV) is a measure commonly used to quantify variation within a system (e.g. Pinto *et al.*, 2013). The CV of yield (t) was calculated for each country for three decadal increments from the years 1981–2010, and the mean CV was compared with linear regression to fish SR per country. Records prior to 1981 were excluded due to the higher chance of inaccuracies and extrapolated figures with older data (Garibaldi, 2012). As before, Box–Cox methodology was used to determine the most appropriate transformation to ensure that data fitted modelling assumptions. The analysis was repeated for country data subset by continent for comprehensive datasets (Africa and Europe).

When examining the link between biodiversity and variation in fisheries yield, CV would not differentiate between a yield which is steadily increasing or decreasing and one which is unstable but fluctuating in similar increments around the mean (see Figure B3 in Appendix B.5). Therefore a variation measure was adapted to consider differences of year on year yield (see Appendix B.3) and calculated alongside CV for comparison.

3.4 Results

3.4.1 Freshwater biodiversity and yield

Considered in isolation, there is a strong positive relationship between fish SR and mean annual yield (t) ($R^2 = 0.55$, $F = 122.6$, $P < 0.001$). The relationship between mean annual yield (t) and each of the macroecological drivers used in the models is shown in Figure 3.2.

Fish SR is an important predictor of overall fisheries yield in the global SAR models that include it and the other major macroecological drivers of yield. These global models – which explain most of the variation in fisheries yield ($R^2 = 0.76$; Table 3.1) – shows that all the variables considered here are important for predicting fisheries yield with the exception of the second principal component of the climatic variables. SR is present in both of the best-fitting models (Table 3.1A) among the 64 possible models. These top models have the smallest AICc and fewest variables for models with comparable AICc. The residual difference per country for the full model is shown in Figure B2 Appendix in B.5).

Table 3.1 Simultaneous autoregressive spatial models of country-level inland water fisheries yield (t) (quarter root transformed). A) Two best fitting models B) Same models repeated excluding species richness as a variable. Shaded cells indicate which of the biodiversity, climatic and geographic variables were included in the model. SR = species richness of fishes (cubic root transformed), P = human population living within 10km of inland waterbodies (quarter root transformed), C1 = first principal component of climatic variables, C2 = second principal component of climatic variables, A = inland water area in km^2 (quarter root transformed), E = mean elevation (m) (cubic root transformed). ΔAICc = difference between the AICc of each model and that of the best model, Wi = Akaike weights. Full model as calculated in SAR_{err}: $\sqrt[4]{Yield} = \sqrt[3]{SR} + \sqrt[4]{P} + C1 + C2 + \sqrt[4]{A} + \sqrt[3]{E}$

A)	SR	P	C1	C2	A	E	Pseudo R^2	AIC_c	ΔAIC_c	W_i
	■	■	■		■	■	0.76	545.50	0	0.39
	■	■	■	■	■	■	0.76	547.35	1.85	0.15

B)	SR	P	C1	C2	A	E	Pseudo R^2	AIC_c	ΔAIC_c	W_i
		■	■		■	■	0.74	552.31	6.81	0.01
		■	■	■	■	■	0.74	552.72	7.22	0.01

Excluding SR from the models resulted in a reduction in mean adjusted R^2 of 0.02 (Table 3.1B). SR made a contribution of between 0.03 and 0.05 in unique effects for each of the top models (Table 3.2), while area contributed 0.03, and waterside population contributed between 0.06 and 0.07 in unique effects for each model. When shared variation was also considered, SR contributed a total effect of 0.55 (accounting for 72% of pseudo R^2), far greater than the other variables except for waterside population, which contributed 0.60 (79%) (Table 3.2). The overall contributions of each of the variables within the models are shown in Table 3.2 and Table B2 in Appendix B.6. With the VIFs between predictor variables in all of the best models being well below 10 there was no suggestion of undue collinearity between variables (Table 3.3).

Finally, river fisheries are known to be as much as three times more productive than lakes (Randall *et al.*, 1995) and therefore as a type of sensitivity analysis the models were repeated with river area weighted higher than lake area. This made no difference to the relative importance of fish SR or the total effects of fish SR upon the models, and is therefore not considered further in this study.

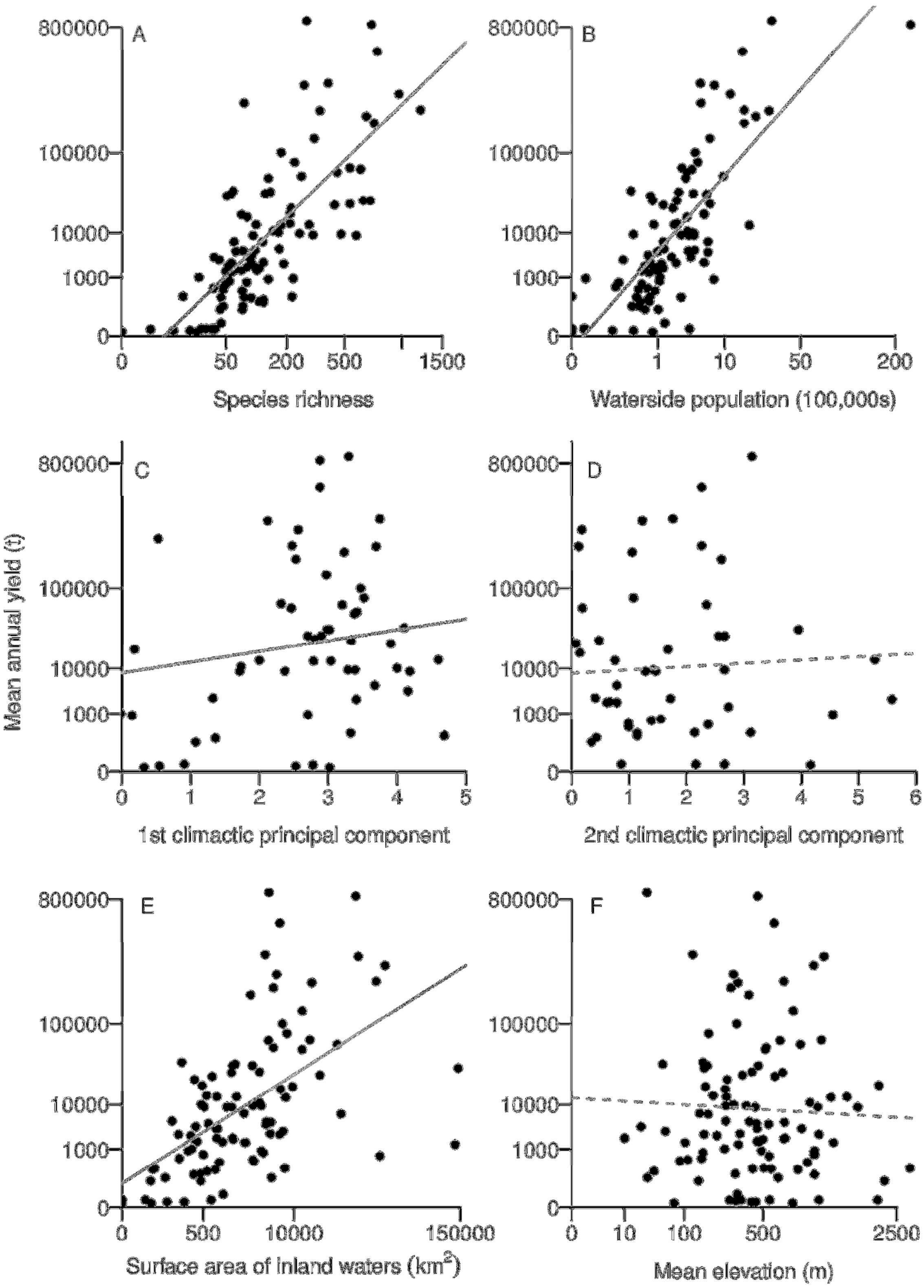

Figure 3.2 Relationship between inland water capture fisheries mean annual yield (t) (axes quarter-root transformed) and model predictor variables at the country level (n = 100): axes for fish species richness and mean elevation (m) are cubic-root transformed. Details of the climatic principal components are given in Table S1 in Appendix S6. Solid lines show $P < 0.05$, dashed lines are non-significant.

Table 3.2 Commonality coefficients of top model set of SAR models of country-level inland water fisheries yield (t). Abbreviations as in Table 3.1.

Model	Variable (x)	Unique	Common	Total	% of R^2
1	SR	0.05	0.50	0.55	72%
	P	0.06	0.54	0.60	79%
	C1	0.01	0.23	0.24	32%
	A	0.03	0.32	0.34	45%
	E	0.01	-0.01	0.002	0.3%
2	SR	0.03	0.52	0.55	72%
	P	0.07	0.53	0.60	79%
	C1	0.01	0.23	0.24	32%
	C2	0.003	0.007	0.01	1%
	A	0.03	0.32	0.34	45%
	E	0.01	-0.001	0.002	0.3%

Note: *Unique* = unique effect of x. *Common* = Σcommon effects of x. *Total=Unique+Common*. % of R^2=*Total/Adj.* R^2

Table 3.3 Variance inflation factors of predictor variables of top model set of SAR models of country-level inland water fisheries yield (t). Shaded cells indicate which of the biodiversity, climatic and geographic variables were included in the model. Abbreviations as in Table 3.1.

SR	P	C1	C2	A	E
2.48	2.19	1.81		1.86	1.13
3.16	2.36	1.89	1.29	1.90	1.15

3.4.2 Freshwater biodiversity and yield variability

Based on data for all countries included in this study there is no significant relationship between SR and CV of fisheries yield (t) ($R^2 = 0.02$, $F = 4.14$, $P = 0.07$; Figure 3.3A). However, independent examination of continent-scale data found a significant negative relationship between SR and CV when only African country data are examined ($R^2 = 0.16$, $F = 9.81$, $P = 0.003$; Figure 3.3B); this is not present in European data ($R^2 = -0.02$, $F = 0.22$, $P = 0.65$, Figure 3.3C). When variability was analysed using an adapted metric which examines year-to-year differences the negative relationship for all countries is

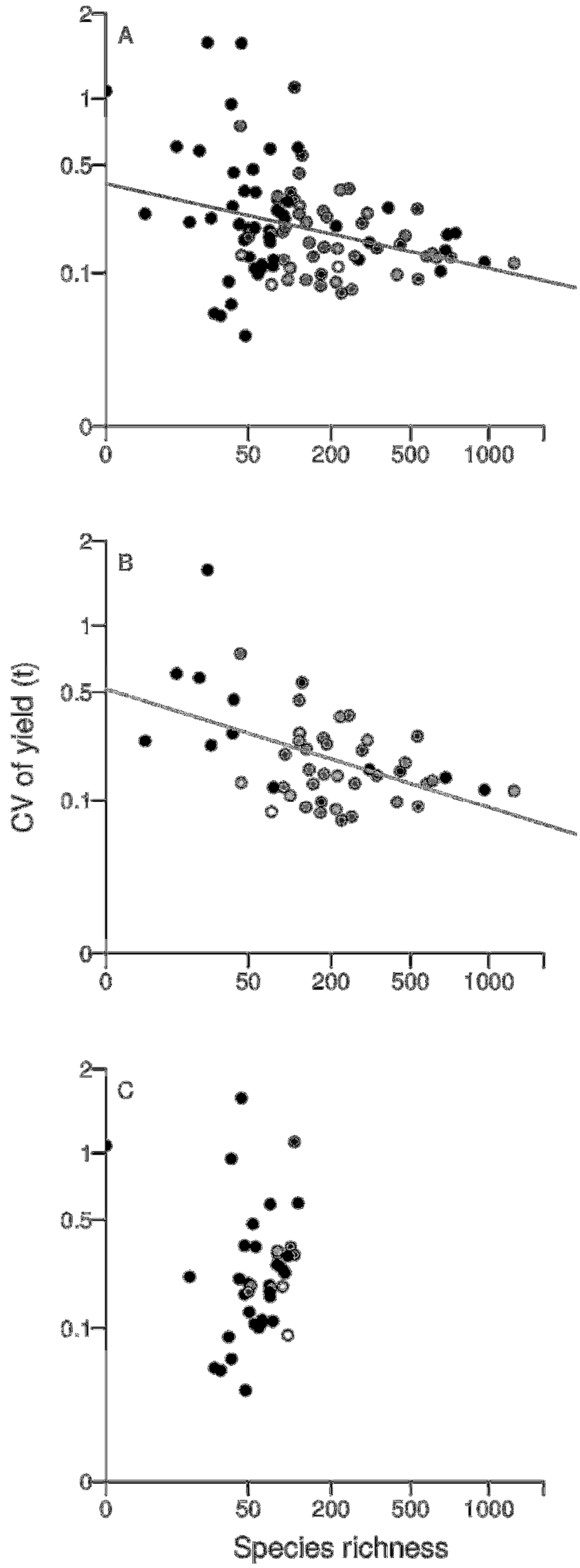

Figure 3.3 Relationship between fish species richness and mean coefficient of variation of yield (t) (both axes cubic-root transformed). (A) All countries within the boundaries of this study (*n* = 100). (B) African countries (*n* = 48). (C) European countries (*n* = 41). The proportion of FAO yield data per country that has been estimated or extrapolated by FAO is graded from white (all years estimated) to black (all actual data).

significant (Table B5 in Appendix B.6). At the continental scale, relationships are similar to those found using CV, although they are slightly weaker for the African data.

3.5 Discussion

This work provides the first large-scale analysis of the relationship between freshwater biodiversity and inland fisheries. In showing that there is positive effect of freshwater biodiversity on fishery yield at the global scale, these results extend the growing body of work that shows a positive effect of biodiversity on increased productivity (e.g. see reviews in Cardinale *et al.*, 2011, 2012) examining a final ecosystem good using large-scale, real-world data. Countries with higher freshwater fish SR report a higher mean yield – a finding that mirrors smaller-scale work on freshwater fish in mesocosms (Carey & Wahl, 2011) and reservoirs in the American Midwest (Carlander, 1955) as well as on marine fisheries (Worm *et al.*, 2006). Importantly, this finding holds after accounting for other macroecological and human drivers. Given the scale of the analysis and the number of covariates considered it is unsurprising that the independent effect of fish SR on yield is small (3–5%), and the effect is comparable to similar studies that have drawn analogous conclusions in different systems (e.g. Maestre *et al.*, 2012).

Previous theoretical and empirical studies have suggested that there are both positive and negative relationships between biomass in fish communities and biodiversity (Hugueny *et al.*, 2010). The two most prominent mechanisms generally responsible for positive relationships between yield and biodiversity are (1) the sampling effect, where dominant species increase productivity, and (2) the complementarity effect, where productivity is higher than would be suggested by consideration of individual species alone due to niche partitioning and facilitation (Loreau *et al.*, 2001). Negative relationships between biodiversity and yield can also occur if one species is able to exploit a limiting resource to such an extent that it is less available to other species (density compensation; MacArthur *et al.*, 1972).

The positive relationship between mean yield and SR revealed in the current analysis provides evidence for either a sampling or a complementarity effect. Data on the proportional contribution of species to total yield from the FAO (Table B6 in Appendix B.6) indicate that for many countries over 50% of total yield is attributed to fewer than five species. This might suggest that the sampling effect is acting as a mechanism causing performance enhancement. If this was the sole mechanism then it cannot be concluded that biodiversity per se is responsible (Loreau *et al.*, 2001) for the relationship described in the current study. However, inland fisheries are not entirely dominated by a handful of species in all countries (Table B6 in Appendix B.6), and socioeconomic factors (e.g. targeted fishing of the most economically valuable species) rather than ecological community structure could contribute to the dominance of a few species in catch statistics. As such, complementarity effects of biodiversity are still possible, even for a fishery whose yields are largely dependent on a few exploited species. Indeed, previous studies have demonstrated that complementarity and sampling effects may not be mutually exclusive (Loreau *et al.*, 2001), complicating our understanding of the underlying mechanisms.

The analysis here also provides qualified evidence of a positive effect of SR on the stability of yield over time, contributing to evidence of the role of biodiversity in regulating aggregate community properties (e.g. Cottingham *et al.*, 2001; Worm *et al.*, 2006). Analysis focused on freshwater systems in Africa demonstrates that the stability of yield decreases as SR decreases (Figure 3.3). These results add to the evidence from previous, smaller-scale studies that suggest that increased fish SR can lead to an increase in productivity and the stability of yields (Franssen *et al.*, 2011; Cardinale *et al.*, 2012). In particular this study suggests that findings from studies of sockeye salmon in Alaska, which show that diversity in the life history of populations increases productivity and buffers population fluctuations, particularly over long time periods (Greene *et al.*, 2010), may also apply to the diversity of fish species. However, these findings do not extend to data covering Europe or to aggregate data across all African and European countries. European freshwater systems have been heavily degraded and suffered dramatic changes and species extirpations (Freyhof & Brooks, 2011); such changes may be the reason for the lack of a relationship between richness and variability of yield observed for Europe, which in turn drives the aggregate pattern across all countries.

The contribution of fish SR (despite a frequent reliance on a limited number of targeted harvest species) to yield and stability of yield (in Africa) in this study highlights the likely importance of non-exploited species in freshwater systems globally. This is probably due to a number of functional processes carried out by species not directly harvested for consumption – such as nutrient cycling, habitat creation, water filtration and their role in the trophic web – all of which work to support the harvested species (Hensel & Silliman, 2013). Although the conclusions drawn here are based on fish SR, it is very likely that not just fish but also other components of freshwater biodiversity are important for fisheries. Unfortunately, it is not possible to disentangle the effects of fish SR and overall freshwater

SR in this study due to the extremely high collinearity (88%) between these two variables [see Appendix B.4 for detailed methods and results for additional analyses based on overall freshwater SR (fish, odonates, molluscs, decapods)].

The very good explanatory power of the global model ($R^2 = 0.77$) indicates that the results for biodiversity are very unlikely to be an artefact of another macroscale driver not considered in these analyses, and the residual variation of the model at the country level does not show any striking spatial pattern (Figure B2 in Appendix B.5). However, as with any large-scale analysis of existing datasets, the findings of the current study are dependent on both the completeness and the accuracy of the data underpinning it, and the findings come with a number of important caveats. Firstly, there are no primary datasets for two key drivers of fishery yields (fishing effort and freshwater productivity), meaning that proxy measures which may be imperfect representations of such drivers were utilized. It is therefore possible that some of the effect attributed to biodiversity is actually due to fishing effort or productivity.

Analysis of the variability in fisheries yield over time could also be influenced by a range of factors for which there are limited data. Principal amongst these is a lack of data on variation in fishing effort, which may vary in order to stabilize catches through time. In addition, there is no way to differentiate between types of, or scales of, fisheries; indeed, subsistence catches are vastly unreported (Béné *et al.*, 2007), which may in part explain the high unexplained variance in these analyses. European fisheries in particular may experience more intense management than their African counterparts (such as yield regulations and artificial stocking) and may be expected to provide more accurate reporting. However, differential reporting would not inflate the relationships between yield and fish SR reported in this study, as there is no reason to suspect a systematic bias of better recording of yields in countries with high than with low fish SR. If anything, biases in management and recording effort will have reduced the observed effects, as in general better management and recording of yields would be expected in countries with relatively low biodiversity (i.e. those in Europe) than in countries with high biodiversity (i.e. Africa).

There are also a number of issues with both the FAO and IUCN datasets. In many cases the FAO has had to rely on estimation or extrapolation to determine likely yield sizes. However, if only measured yield data are used for Africa (d.f. = 9), the R^2 for the effects of biodiversity on variability in yield increases from 0.16 to 0.21, suggesting that more accurate data could indicate an even stronger relationship. FAO yield data are currently only widely available at the country level but it would be beneficial to examine the relationships discussed here at multiple scales (e.g. catchment and subcatchment levels). Matching catchments to fisheries yields will facilitate the exploration of the link between the health of the river system and the productivity and variability of the yield in further detail. Examining the relationship at a finer resolution would also help to elucidate the role of fish SR versus the SR of other freshwater species, as the diversity of the taxonomic groups is not found to correlate at the smaller catchment scale (Darwall *et al.*, 2011). Although the IUCN data are the most comprehensive freshwater data available, and indeed could be used for analyses at a finer resolution than country level, they do not provide complete global coverage because they omit important fishing regions such as China and South America.

The findings, but also the limitations, of this study have major management implications for freshwater ecosystems, for three main reasons. Firstly, as the countries with the most important inland capture fisheries also generally have the highest freshwater biodiversity, it is clear that management of these key fisheries must be sustainable in terms of both yield and conservation. This study therefore provides strong support for efforts to promote multifunctional watersheds, with a focus on sustainable fisheries management and fish conservation initiatives (Dudgeon, 2010; Cowx & Portocarrero Aya, 2011). Secondly, results suggest that fish diversity may deliver benefits for human wellbeing – particularly in terms of maintaining constant yields over time. Capture fisheries are a critical part of food security and livelihoods, particularly in developing countries, where fisheries provide a major source of protein and micronutrients, and where they are used as a safety net in times of hardship, such as due to crop failure (Béné *et al.*, 2007; Dugan *et al.*, 2010). As such, these results provide a powerful argument for placing biodiversity conservation centrally within fisheries management. Finally, this study makes it clear that there is a paucity of data for freshwaters, including a thorough understanding of species compositions and distributions worldwide, and for major ecosystem-specific macroecological drivers such as productivity measures. Equally, a concentrated effort is required to increase reporting not only of inland fishery yields, but also of fishing efforts (see De Graaf *et al.*, 2012). Only by doing this will we be able to fully understand the extent of the role that biodiversity plays in underpinning inland fisheries.

Inland waters are the most threatened systems globally, with dams, water extraction, pollution and invasive species recognized as some of the biggest threats to freshwater systems and to fisheries, as well as overharvesting of the fisheries themselves (Dudgeon *et al.*, 2006). It is imperative that the relationships explored here should be considered within freshwater and fisheries management; the protection and conservation of species diversity in freshwater systems is a win–win outcome for human food delivery and conservation efforts to preserve freshwater ecosystems.

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4 Mapping the importance of freshwater species in food security

4.1 Abstract

Despite the pivotal role inland fisheries play for food security and livelihoods globally, to date there is limited spatial data available for this critical ecosystem service, and none at a landscape or larger scale. This chapter introduces a validated, bottom-up, globally applicable predictive model of fisheries provided at sub-catchment scale. The Freshwater Food Security (FFS) model presented here incorporates both supply (inland water area) and demand (waterside population and availability of livestock protein) to produce a predictive map of the relative importance of freshwater fisheries globally at a much finer resolution than official catch statistics, and can be used in management decisions across a range of extents and resolutions. Rank analyses indicate the FFS model correlates well (Spearman's rho = 0.67-0.86) against data from known fisheries data at three scales: global, regional (key African lakes and state level for India) and local (district level of Kerala in India), and at different resolutions (sub-catchment sizes). Sensitivity analyses show that these results are robust to the exact parameter values incorporated into the FFS, and that inland water area and waterside population alone are sufficient to predict fishery capture statistics. The FFS model provides vital insight into the importance of this resource for livelihoods, and so can facilitate the design of management programmes to protect freshwater systems, and the important food sources that are provided by them.

4.2 Introduction

Resources on the planet are in limited supply, and nowhere is this felt more than in global food security (FS). The human population passed 7 billion in 2011 and is expected to increase to 9.7 billion by 2050 (UN, 2015). Although agricultural production and intensification are constantly increasing, FS is still a real issue for millions of people around the world. To this end, one of the Millennium Development goals was to halve hunger (compared to 1990) by 2015. Improvements have been made as there has not been a rise in number of hungry, however approximately 870 million people are chronically undernourished, three-quarters of whom live in rural areas, and 98% of whom are in developing countries (FAO *et al.*, 2012). As such an advancement of the pledge has been made in the Sustainable Development Goals, to end hunger and ensure year round access to food to all people by 2030 (https://sustainabledevelopment.un.org/).

Inland water fisheries provide a pivotal role to FS in certain locations, particularly in developing countries. Healthy fisheries are capable of providing all four pillars of food security to local populations by ensuring availability, access and stability of food, plus utilisation by supplying important dietary and nutritional needs (FAO, 2006). Official FAO landings records show that over 11 million tonnes of fish are harvested globally each year from inland capture fisheries, with the greatest catches yielding from Asia (66% in 2012) and Africa (25%) (FAO, 2014a). Apart from notable exceptions (e.g. Lake Victoria, Amazon and Mekong fisheries) the vast majority of inland fisheries are small-scale, dominated by artisanal and subsistence fishers, and catches are marketed and consumed locally (Welcomme, 2011b). Although aquaculture is now the fastest growing food production sector globally, it has been shown that for countries that are most dependent on fish for FS, wild capture fisheries remain the most important source of supply (Hall *et al.*, 2013). In many areas fish are an essential source of animal protein and are particularly important in the diet of poorer households. Smaller fish, most commonly consumed by poorer households, are critical for supplying micronutrients and contributing to the health and mental development of children (Dugan *et al.*, 2010). Critically they are also important for livelihoods, providing employment to over 60 million people in developing countries (97% of the global inland fisheries workforce), and 55% of whom are women (Mills *et al.*, 2011), who spend a greater proportion of their earnings on family needs than men (Weeratunge *et al.*, 2010).

In order to assess the role of freshwater species to FS a spatial picture of their importance should be obtained to best understand where freshwater systems are playing a vital role in providing a nutrition and income source. It is essential to be able to map ecosystem services such as food provision for their management (Crossman *et al.*, 2013). Spatially explicit information can help inform decisions that input into such important international targets such as Millennium Development Goals (and its successor, the Sustainable Development Goals (UN, 2014)) and Aichi targets (see Box 4.1). Although food provision is the third most commonly mapped ecosystem service (Crossman *et al.*, 2013), food from any wild sources, let alone from inland waters, are rarely considered. The only known exception is based on benefits transfer (Schulp *et al.*, 2012, 2014b). While benefits transfer is commonly used in ecosystem service valuations constrained by time and budgets, its use often has low levels of validity and reliability and may introduce a bias which could undermine the integration of ecosystem service values into policy unless study and policy sites share identical characteristics (Richardson *et al.*, 2015). Currently, data on the

Box 4.1 Example of international policy targets.

Millennium Development Goal 1: Eradicate extreme poverty and hunger

Millennium Development Goal 7: Ensure environmental sustainability

Sustainable Development Goal 2: End hunger, achieve food security and improved nutrition and promote sustainable agriculture.

Sustainable Development Goal 15.1: By 2020, ensure the conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems and their services, in particular forests, wetlands, mountains and drylands, in line with obligations under international agreements.

Aichi Target 6: By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe ecological limits.

Aichi Target 14: By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable.

importance of inland fisheries relies heavily on reported catches which are known to undervalue the importance of subsistence fisheries (FAO, 2003; Youn *et al.*, 2014), and due to a lack of consistency are not spatially explicit beyond country level for much of the globe. The importance of freshwater species in food security should be modelled at the sub-catchment level, as this is increasingly considered the most useful management unit for inland waters (e.g. IUCN 2000, Collares-Pereira and Cowx 2004, Vigerstol and Aukema 2011, Syrbe and Walz 2012).

Spatially explicit consideration of ecosystem services is important in order to consider both supply and demand (Crossman *et al.*, 2013; Schulp *et al.*, 2014a; Stürck *et al.*, 2014). Previous efforts to incorporate supply and demand into ES mapping have largely been limited to local and regional spatial scales (Stürck *et al.*, 2014). The main challenge to mapping ecosystem services in general is either a lack of appropriate data at a suitable resolution and appropriate scale (Feld *et al.*, 2009; Stoll *et al.*, 2014), and/or a lack of resources and expertise to apply currently advocated mapping techniques (Martínez-Harms & Balvanera, 2012).

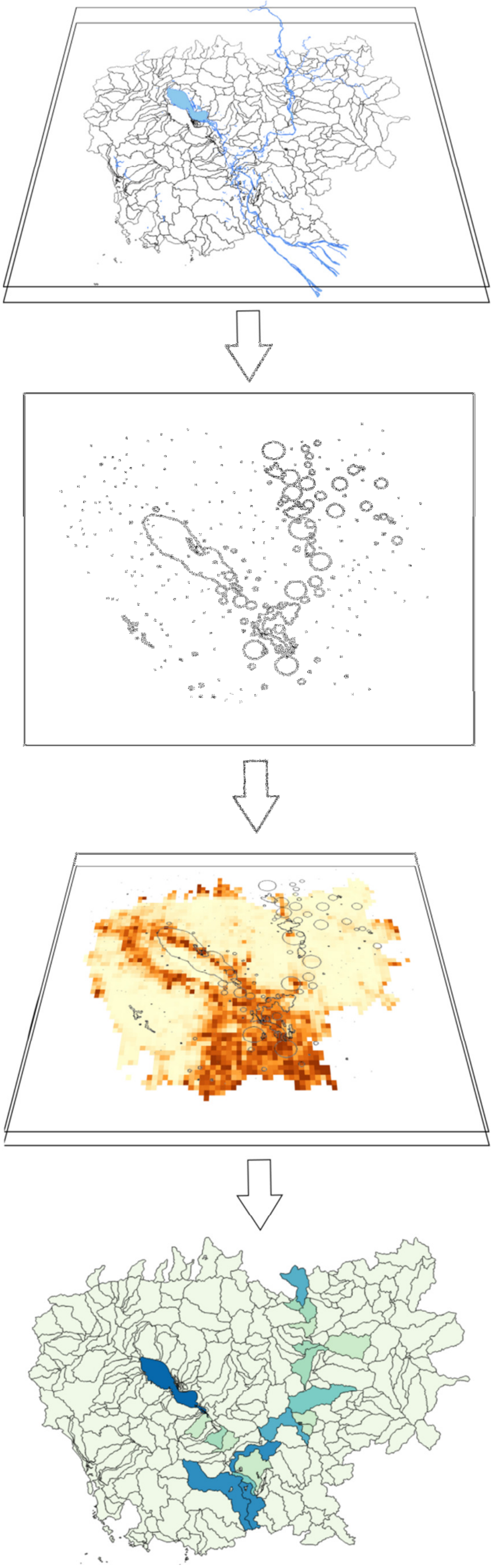
This chapter introduces a validated, bottom-up, globally applicable predictive model of fisheries provided at sub-catchment scale. The model maps the importance of freshwater fishes for food at a much finer resolution than official catch statistics, and can be used in management decisions across a range of extents and resolutions. By incorporating both supply and demand metrics in tandem the model presented here can predict and map the level of contribution freshwater species are making upon FS. The aim of this model is to overcome many common data (or lack thereof) problems by providing a simple tool to accurately produce a spatial overview of the relative scale of importance of each sub-catchment for providing a freshwater food source, within any given region of interest. This can identify areas of high potential for the provision of fisheries to dependent populations and thus inform management decisions of freshwater catchments, overcoming general data limitations and without the need for specialist technical skills. Publically available datasets have been used to apply and test the model, which is demonstrated and validated here at a range of resolutions and extents.

4.3 Methods

To approximate the importance of freshwater species for FS the model needs to simulate two major factors – supply and demand (Stürck *et al.*, 2014). The supply metric of the Freshwater Food Security (FFS) model is based on surface area of water within each sub-catchment, as a proxy for the size of the fishery. The principal demand driver for inland fisheries is for food, with the majority of harvests consumed locally (Welcomme *et al.*, 2010). Additionally, fish protein from inland waters is known to be the main source of protein in some developing countries (Dugan *et al.*, 2010), and fish supplies a higher proportion of animal protein consumed where livestock is more scarce (for instance it accounts for less than 10% of consumed animal protein in North America and Europe, but 26% in Asia and 17% in Africa (FAO, 2000)). It is surmised that there is likely to be a greater reliance on freshwater food security where agricultural protein per capita is lower. Therefore demand is based on population and livestock availability. Values for these demands are extracted by the model using sub-catchments rescaled in proportion to the extent of their supply, where the rescaling factor is defined by the surface area of water within the sub-catchment, in order to produce an overall FFS value per sub-catchment (see Figure 4.1).

4.3.1 Potential Supply

Potential supply of inland water fisheries is represented by the area of water available within each sub-catchment, which is a strong predictor of potential fishery yields (Halls, 1999). Previously fish productivity in lakes has been modelled using a range of factors that



Each sub-catchment is weighted by surface area of inland waters within. In this example the Cambodian sub-catchment layer overlays the water layer (depicted left) to extract the surface area (km^2) of water per sub-catchment, and this value is used to inform the calculation of a new sub-catchment area. (Details given in main text).

The new area of each sub-catchment, representing its extent of influence, is used to rescale sub-catchments, while also maintaining their original shape where possible. Depending on the water surface area weighting, this may result in an enlarging or shrinking of the original sub-catchment. Depicted left you can see the new larger sub-catchments in the place of the highly important lake of Tonle Sap, and the Mekong River and tributaries, while sub-catchments with minimal water area have all but disappeared.

The rescaled sub-catchments are used to extract data from the demand layers. Here, the sub-catchments have been overlaid onto a population data layer. The enlarged sub-catchments around large bodies of water now not only overlap with any populations occurring within the original sub-catchment, but also those populations in surrounding areas that are likely to be making use the resource. This is then repeated for each layer, i.e. available livestock.

The values for each extracted layer are then combined to create an overall freshwater food security score, here depicted as low (pale green) to high (dark blue) – sub-catchments with a higher score are likely to be more important for freshwater food provision. In Cambodia, the lake of Tonle Sap is the most important for freshwater food security, but you can see the predicted increasing importance of the Mekong River as it flows downstream.

Figure 4.1 Example of how the freshwater food security model works using data for Cambodia.

include mean depth, water conductivity, nitrogen, phosphorous or a combination via the morpho-edaphic index and annual phytoplankton production (Downing *et al.*, 1990). Given the lack of data available for most lakes, only coarse estimates could be made for these metrics such as using climate variables as a proxy for primary productivity (as used in the previous chapter), which would be inappropriate to discern local differences. In addition these studies predominantly concentrate on discrete lakes, and there is little evidence to suggest how similar metrics can be applied to rivers. The FFS model therefore is based on the surface area of inland waterbodies which are globally available data. This defines the extent of influence for each sub-catchment within the model: the importance of a waterbody to local populations will be dependent upon the size of the waterbody and its expected returns.

The HydroBASINS Format 2 database (Lehner & Grill, 2013) was used to define the global inland water sub-catchments. From this database different levels of sub-catchments have been designated depending on the appropriate scale for analysis, from level 1 (containing 8,515 global sub-catchments, predominantly defined by land mass) to level 12 (defining 1,269,912 sub-catchments). For example, globally the database identifies 236,969 sub-catchments at level 8, with an area range of 0.002 km^2 to 370,211 km^2 and a mean and median area of 1,372 km^2 and 610 km^2 respectively. To calculate the surface area of water per sub-catchment, the total area of lakes, wetlands and major rivers was extracted per sub-catchment from the Global Lakes and Wetlands Database (GLWD) (Lehner & Döll, 2004) using ArcMap 10. The total surface area of rivers not included in GLWD were based on HYDRO1k layer of global rivers (USGS, 2000): for each stream order as defined within the HYDRO1k line feature layer attributes of the river shapefile, 100 random points were generated. Google Earth v7.1.2.2041 was used to accurately measure the width (m) of the river at those coordinates. It is assumed that seasonality of rivers is accounted for by the random seasonality of the images comprising Google Earth. A mean river breadth was then calculated from these measurements for each stream order size and applied to the HYDRO1k river line layer to create a river polygon layer, with area data associated with each polygon. These river area values were then multiplied by three to account for the greater level of fish productivity in rivers compared to lakes (Randall *et al.*, 1995). Total water area per sub-catchment was then calculated from the combined GLWD and HYDRO1k river area data.

For each sub-catchment, a new area value is calculated, defined by the area of a circle with a radius equalling the square root of the sub-catchment's total water surface area multiplied by 1,000. A linear relationship between the surface area and the productivity of a waterbody is not expected, not least because water bodies are three dimensional spaces and thus fisheries productivity would be expected to increase to a certain extent with depth. The square root term within the algorithm was selected as a first estimate for this non-linear relationship, and visual inspection of the resulting buffers indicated that this was a reasonable approximation of the extent of influence expected (see example in Figure C1 in Appendix C). Alternative buffer algorithms were also tested but had little effect on the results (see section 4.3.4.1). This new area, representing the sub-catchments extent of influence, is then mapped by enlarging or shrinking the original sub-catchment shape to match the new area. This rescaled sub-catchment may be larger or smaller than the original depending on the amount of water surface area within it (see example for Cambodia in Figure 4.1). Where it is not possible to maintain the original shape (i.e. where narrow sub-catchments mean that negatively buffering the original shape to match the new area will artificially cause the polygon to disappear entirely), sub-catchments are represented by a circle matching the new area positioned at their centroid instead. This rescaling approach enables the model to keep the true shape of sub-catchments wherever feasible, therefore keeping the geographic position of areas of influence as accurate as possible. For example, the lake of Tonle Sap in Cambodia is an important fishery (Lamberts, 2006). Replacing its original shape with a purely circular equivalent (scaled in proportion to its area of influence) would not reflect the elongated shape of the lake and would risk underestimating its importance to people living to the north-west and south-east of the lake, while overestimating its importance to those living in other areas (see Figure 4.1).

4.3.2 Demand

4.3.2.1 Population

A raster layer of the 2000 global rural population reported by FAO at the 5 arc-minute resolution was used (Salvatorre *et al.*, 2005). Total population was extracted from the weighted sums of population raster per rescaled sub-catchment (i.e. where rescaled sub-catchment boundaries pass through population pixels and do not fully enclose them, the proportional total of that pixel included within the sub-catchment boundary is added to the sum total of all pixels within the rescaled sub-catchment), and normalised on a 0-1 scale.

4.3.2.2 Agriculture

Livestock abundance layers representing total numbers of cattle, goats, pigs, poultry and sheep for 2005 at the 5 arc-minute resolution were collated from the Gridded Livestock of the World v2.0 (Robinson *et al.*, 2014). These were each multiplied by mean global mass for the relevant species derived from Thornton (2010), and summed to create a layer depicting kg livestock available per pixel. This layer was then divided by the population raster to represent agricultural protein available per capita per pixel, and then inverted (each pixel value subtracted from the maximum pixel value of the total extent) so that a lower output per capita would weight higher than a high output per capita (i.e. an area with lower agricultural protein per capita available would be expected to place a higher importance on fish resources). The resulting raster layer was used to extract weighted sums per rescaled sub-catchment, and normalised to a 0-1 scale.

Overall freshwater food security importance is calculated as:

$$FFS = Pop * Agr$$

where

$$Pop = \frac{Weighted\ sum\ of\ population\ per\ rescaled\ sub-catchment}{Max\ Population\ across\ all\ sub-catchments}$$

$$Agr = \frac{Inverse\ livestock\ production\ per\ capita\ per\ rescaled\ sub-catchment}{Max\ inverse\ protein\ per\ capita\ across\ all\ sub-catchments}$$

Dividing the summed component scores by the maximum score within the given dataset maintains a 0-1 scaling across the study region (with high scoring areas reflecting highly important areas of freshwater species upon FS), in order to produce a unitless relative input measure. The R code for the model is available in the Supporting Information (Appendix C.3).

4.3.3 Model validation

In order to illustrate and empirically validate the FFS model, it has been run at a range of extents and sub-catchment resolutions:

1) Global, HydroBASINS level 5
2) Global, HydroBASINS level 8
3) India, HydroBASINS level 8
4) India, HydroBASINS level 12
5) Kerala, HydroBASINS level 8
6) Kerala, HydroBASINS level 12
7) African lakes from GLWD

Validation of the FFS model occurred at 3 scales: global, regional (state level for India and for key African lakes) and local (district level of Kerala in India). These represent a range of scales for which official fisheries catch data exists. For the global maps, summed FFS scores were extracted for each country and compared to country level inland fisheries data from FAO (FAO, 2014a). For sub-global scales, the FFS model was run to incorporate all sub-catchments which fall in any part within the political boundaries. Within India, FFS has been compared to state level fisheries data (Latha, 2010), and Kerala state has been compared to district level fisheries landings data (Latha, 2010). A set of 31 African lakes were chosen based on the availability of fisheries data from the literature or from FAO country profiles (Table C1 in Appendix C). In each case, summed FFS scores within the appropriate administrative boundary were compared to fisheries data using Spearman's rho correlations. Spatial autocorrelation was accounted for by using a modified t test in the SpatialPack package in R (Osorio & Vallejos, 2014), which adjusts the degrees of freedom based on an estimate of the effective sample size (Clifford *et al.*, 1989). The variance in the rank differences between the fisheries catch data and amalgamated FFS scores for each scale and extent were calculated on normalised ranks (where each rank difference was divided by the maximum rank difference), to make comparison of the variance between different sizes possible.

4.3.4 Sensitivity analysis

The sensitivity of the model to each of the parameters was tested to ensure confidence and rigour within the model. The sensitivity analysis results were compared to each of the tested resolutions above (see 4.3.3).

4.3.4.1 Supply

To test the sensitivity of the model to the assumption that river fisheries are three times greater than lakes, the analyses were repeated without multiplying the river areas by three.

It is not expected that water surface area would have a linear relationship with fisheries productivity, not least because water bodies are three dimensional spaces and thus fisheries productivity would be expected to increase to a certain extent with depth. With improvements in data, for example the completion of the lakes and river volume spatial data layers (Lehner & Grill, 2013), it may be possible to more accurately represent the waterbody ratio to fisheries productivity, although it is worth noting that productivity in freshwater is lower at deeper depths, so it is unlikely that even volume data would provide an accurate proxy for fisheries productivity. To test the effect of choosing to represent the productivity by the square root of the surface area of water within each HydroBASINS, the analyses were compared to the same model based on a linear buffer, a cube root buffer and a log +1 buffer (an example of the differences in size of the resulting rescaled sub-catchments is shown in Figure C1 in Appendix C). Additionally the model was run using no buffer algorithm but simply using the original HydroBASINS sub-catchments to extract data from the demand layers.

4.3.4.2 Demand

Population and the inverse of the quantity of livestock per capita have been included within this model as predictive layers for the importance of freshwater fish for food provision. The model has been re-run using each layer independently, and again by changing the model algorithm to

$$FFS = \frac{Pop + Agr}{2}$$

To examine possible co-variation a cross-correlation matrix was derived of the model input datasets (inland water surface area, population and inverse livestock) and output values. This would highlight the internal correlations and possible redundancies within the model. This was tested using total values extracted for each layer for the Indian level 12 HydroBASINS, and then repeated using the total found within the rescaled sub-catchments as derived by the model.

Finally, in order to test the strength of water area alone as a predictor of fisheries data, the correlation between the fisheries landing validation data and inland water surface area was tested by extracting total surface area of water from within each of the administrative boundaries used in the model validation process (see 4.3.3), and comparing them with the fisheries data for each using Spearman's rho correlations corrected for spatial autocorrelation as above.

4.4 Results

Graphical representations of the FFS model output at multiple scales and extents are shown in Figure 4.2. The global FFS model predictions correlated strongly with the FAO capture fisheries data at the country level when the ranks were analysed at the level 5 HydroBASINS resolution ($r_{85.24}$=0.80) and at the level 8 HydroBASINS resolution ($r_{88.07}$=0.78). When the country of India was examined the FFS model correlated strongly with recorded fish productivity at the state level when ranks were analysed at the level 8 HydroBASINS resolution ($r_{19.94}$=0.76) and at the level 12 HydroBASINS resolution ($r_{21.55}$=0.86). Similarly, Kerala state FFS model correlated strongly with fisheries landings reported at the district level at the level 8 HydroBASINS resolution ($r_{4.90}$=0.84) although at the level 12 HydroBASINS resolution the strong correlation was no longer found to be significant once corrected for spatial autocorrelation ($r_{5.41}$=0.71, p=0.06). The African lakes also correlated strongly with reported fisheries landings for each lake ($r_{29.91}$=0.67). Heatmaps comparing the fisheries catch data with amalgamated FFS scores per administrative boundary are shown in Figure C3-C6, while the differences in rankings between the FFS model and the fisheries data per administrative boundary for each map are shown in Figure 4.3. Normalised variance of the rank differences are broadly similar across the range of scales and extents tested here (Table C5).

4.4.1 Sensitivity analysis

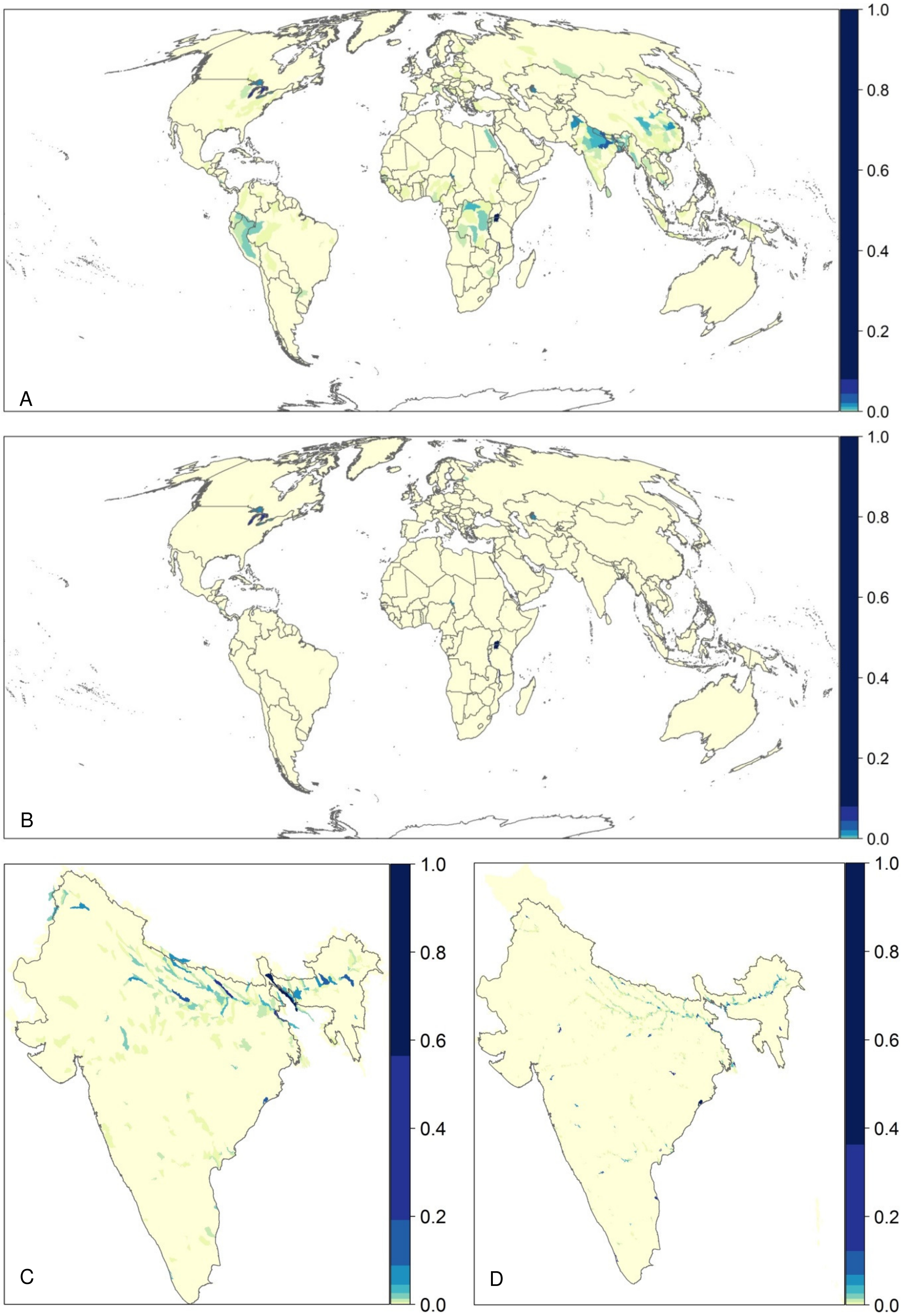
4.4.1.1 Supply

The model is robust to the formulations used. The calculated FFS scores had a comparably strong positive ranked correlation with fisheries data when river areas were multiplied by three versus when they were not (Table C2). River areas multiplied by three had slightly stronger ranked correlations for the global model using HydroBASINS level 8, India HydroBASINS level 12 and Kerala HydroBASINS 8, but slightly weaker for India HydroBASINS at level 8 and Kerala HydroBASINS level 12. Results for the global model at HydroBASINS level 5 and the African lakes were identical whether rivers were multiplied by three or not (Table C2).

Changing the buffer algorithm had different effects on the ranked correlations with known fisheries data across the ranges and extents tested, but in general did not produce notably different results (Table C3). All ranked correlations were strong regardless of algorithm used, with the exception of testing the African Lakes, which showed a moderate correlation with a linear algorithm but no correlation with a cubic root or log based buffer algorithm (Table C3). The ranked correlations were not significant for Kerala at HydroBASINS level 12. Using no buffer had mixed results, with Indian data still showing a slightly reduced but strong correlation, while Kerala level results did not correlate with fisheries data (Table C3).

4.4.1.2 Demand

At all sub-catchment resolutions except for Kerala (at both HydroBASINS level 8 and 12), using population data alone in the FFS model did not markedly differ from the original model, however it did slightly improve the strength of the ranked correlations with available fisheries data (Table C4). Within Kerala, including population *x* livestock remained the strongest ranked correlation with known fishery landings (Table C4).



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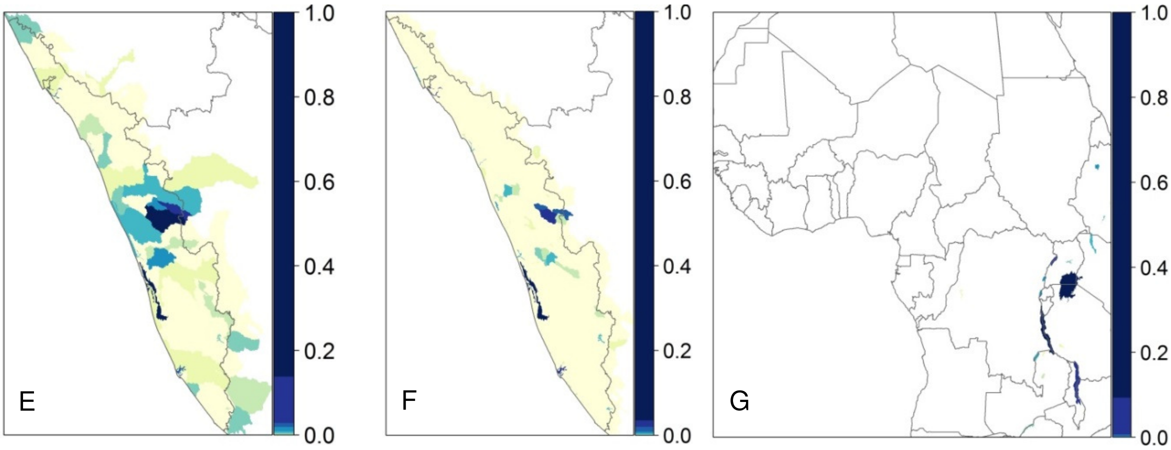


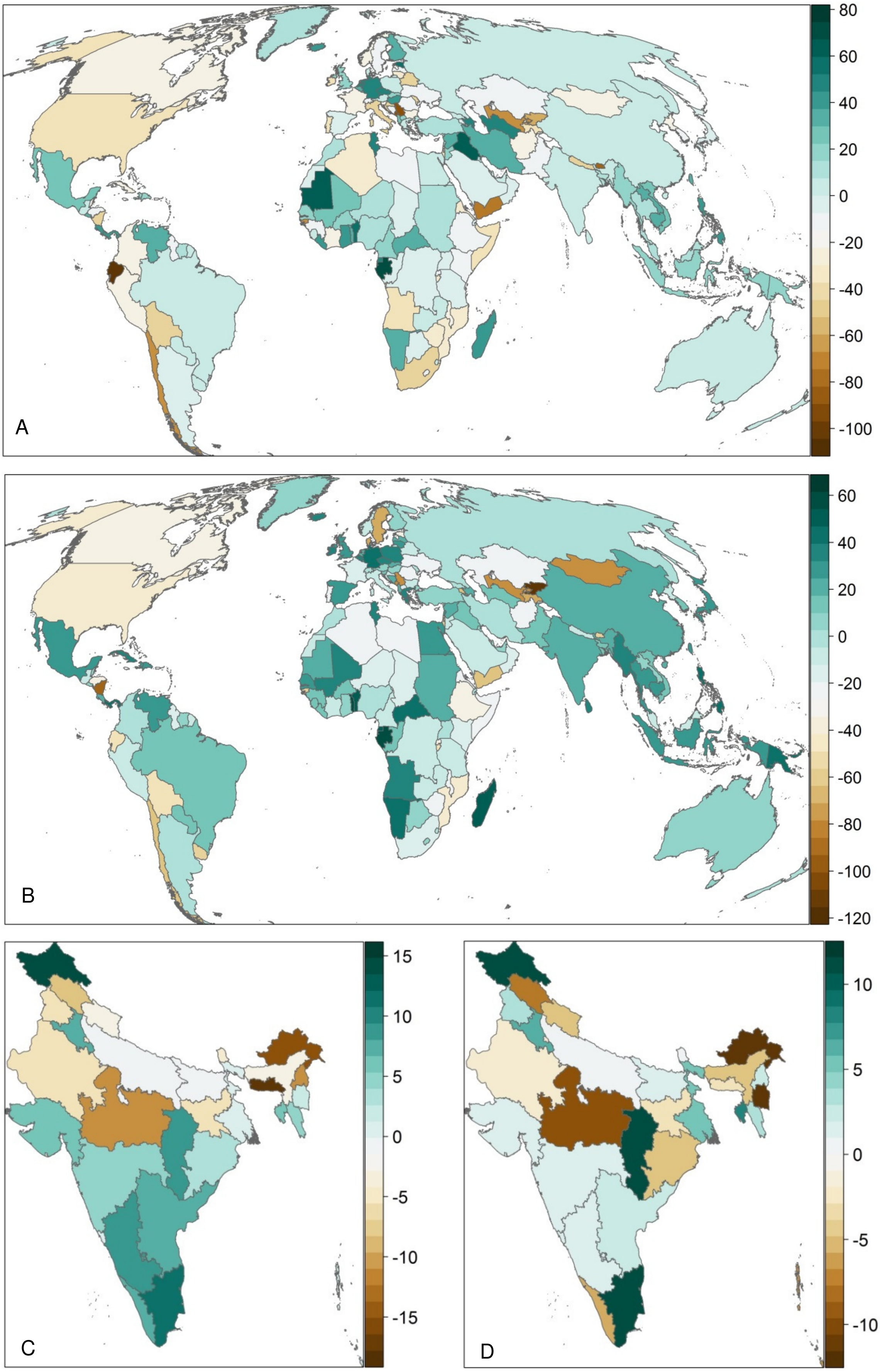
Figure 4.2 Importance of freshwater food security (FFS). A) Global data, using level 5 HydroBASINS sub-catchments. B) Global, level 8 HydroBASINS. c) Sub-catchments which fall within India, level 8 HydroBASINS. D) Sub-catchments which fall within India, level 12 HydroBASINS. E) Sub-catchments which fall within Kerala, level 8 HydroBASINS. F) Sub-catchments which fall within Kerala, level 12 HydroBASINS. G) African lakes from GLWD. Note that values are relative and therefore maps are not directly comparable.

There is a moderate correlation between population and the inverse of the amount of livestock per level 12 HydroBASINS in India, but a weak negative relationship between these components and the surface area of inland water available (Figure C7). There is a strong ranked correlation between area and the final FFS score produced by the model, a very weak positive correlation between population and FFS score, and a weak negative relationship between inverse livestock and FFS score at the HydroBASINS level (Figure C7). When calculated using the rescaled model sub-catchments, there is a strong ranked correlation between all model input and output components (Figure C8).

The correlation between the fisheries landings validation data and the total surface area of inland waters within each administrative boundary was weaker in all cases than the FFS score correlation, except for India at the HydroBASINS level 12 resolution, which was comparable (Table 4.1), confirming that the strength of the model is increased with the inclusion of demand metrics.

4.5 Discussion

Increasingly land cover is recognised as a suitable proxy to map ecosystem services (Dick *et al.*, 2014; Stoll *et al.*, 2014) assuming practitioners have accounted for the vulnerability to errors (Eigenbrod *et al.*, 2010); however these maps are based on terrestrial data. The FFS model is able to apply ecosystem service mapping principles to freshwater areas. The strong rank correlation of the output maps against known fisheries data supports the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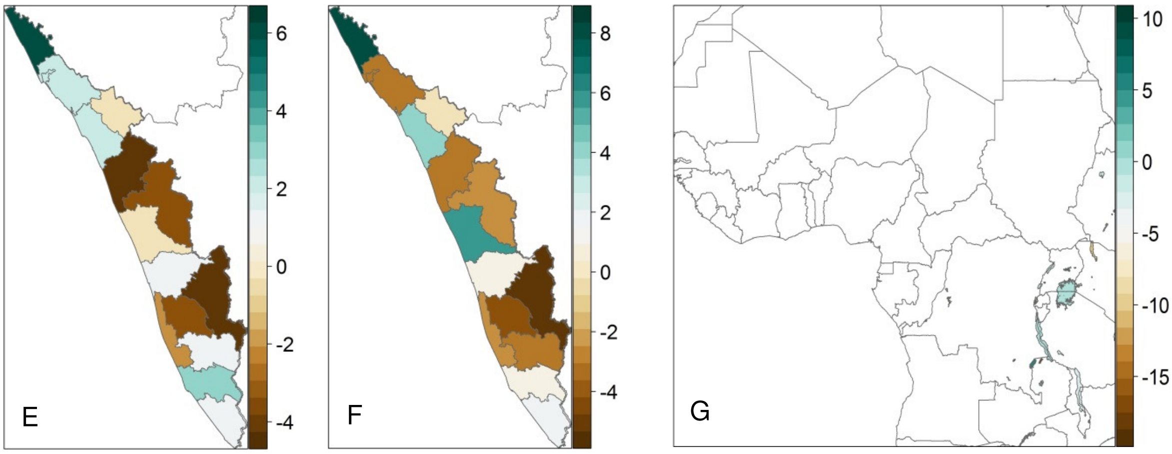

Figure 4.3 Difference in ranks between FFS model predictions and fisheries landing data, where rank no. 1 is for the highest or most important yield. Positive values denote where fisheries data ranked more highly than FFS predicts, negative values denote where FFS scores predict a higher rank than suggested by fisheries data. A) Global data at the country level, from FFS calculated using level 5 HydroBASINS. B) Global data at the country level, from FFS calculated using level 8 HydroBASINS. C) India at the state level, level 8 HydroBASINS. D) India at the state level, level 12 HydroBASINS. E) Kerala at the district level, level 8 HydroBASINS. F) Kerala at the district level, level 12 HydroBASINS. G) African lakes. Note differences in legends.

increasing emphasis of incorporating both supply and demand considerations to most accurately represent ecosystem services in mapping (Burkhard *et al.*, 2012; Schulp *et al.*, 2014a; Stürck *et al.*, 2014). The inclusion of livestock data did not necessarily increase the predictive power of the model, which was strongest based on population data alone (maps based on population alone are shown in Figure C2).

The FFS model provides a simple tool to spatially evaluate the importance of areas for the provision of food from freshwater sources, which can be achieved easily and cheaply through the freely available R software (R Core Team, 2014) and publically available

Table 4.1 Correlation between fisheries landing data and inland water surface area, corrected for spatial autocorrelation.

Region	HydroBASINS level	rho	df
Global	5	0.63	90.34
Global	8	0.63	82.81
India	8	0.83	20.84
India	12	0.86	21.30
Kerala	8	0.74	4.55
Kerala	12	0.68	5.69
African Lakes	GLWD	0.66	26.70

datasets such as those used here. It can incorporate any appropriate data and therefore can be made more relevant to specific research questions by using regionally specific high resolution data, or updating to more recent data such as using newly developed population data (e.g. www.worldpop.org.uk). Two thirds of ecosystem service maps are not validated with observed data or undergo sensitivity analysis (Seppelt *et al.*, 2011; Schägner *et al.*, 2013), yet testing of this model has shown strong rank correlations with known fisheries landing data, indicating a good level of accuracy across a range of scales and resolutions despite requiring few inputs, and that the model is robust to the assumptions made. By including the potential supply of the ecosystem service (here, potential habitat for food fish) to define the extent of influence within the model, it seeks to address a major limitation to previous ecosystem service models which have struggled to map the direct harvesting of a resource from an area that does not coincide with human populations, i.e. within waterbodies (Brauman *et al.*, 2007).

It is unsurprising that the resulting maps from the model are dominated by major lakes at each of the scales. Of the total liquid freshwater on the surface of the planet, 87% is believed to be found in lakes, with only 2% in rivers and the remainder in swampy land (Shiklomanov, 1993). For instance, Lake Baikal alone, the largest lake in the world by volume, contains 20% of the world's liquid fresh water, and summing the ten largest lakes globally accounts for approximately 70% of global unfrozen surface water. Although calculations here are based on surface area of waterbodies and not volume, eight of the ten largest lakes by volume are also in the top ten by surface area. In India the map is dominated by the Ganges River, the third largest river by discharge.

The resolution at which the model is run can make notable differences to the output and emphasises that models should be run at a scale appropriate for decision making (Turner *et al.*, 2001). In this case the larger disparities are a result of the difference between high resolution sub-catchments highlighting only larger (and therefore important) waterbodies, and lower resolution sub-catchments also highlighting catchments that cumulatively contain a large sum of water potentially from multiple waterbodies. The similarity in normalised variance of the rank differences between FFS scores and reported fisheries data suggests that there is not an optimum scale that the model is best suited to but is equally accurate across a range of resolutions and extents. Deciding on an appropriate scale would be a critical part of this tool, and it may well be that it would be useful to consider multiple resolutions in order to incorporate regional (top-down) and local (bottom-up) considerations simultaneously in order to address management and ecological needs across multiple scales (Huber *et al.*, 2010; Holland *et al.*, 2011).

It is worth noting that Spearman's rho correlations were used due to the non-normality of the data. However, using rank correlation measures can lead to a loss of information on the goodness of the fit of the model by converting the interval data to ordinal data. The Pearson's product-moment correlation coefficient could be calculated in tandem in order to better describe the proportion of the total variance in the observed data that can be explained by the model and therefore further test the accuracy of the model. More information could be gleaned about the model efficiency using alternative goodness-of-fit measures such as the coefficient of efficiency or the index of agreement, although caution should be taken in the interpretation of these measures in isolation (Legates & McCabe, 1999).

A strong focus of current ecosystem service mapping literature is to be able to assess multiple services at a range of scales in order to inform policy (Stoll *et al.*, 2014). It has been argued however that the creation of one classification scheme is impossible (Zhang *et al.*, 2010), and mapping techniques should reflect the context and purpose of the study (Schägner *et al.*, 2013). Those models that do attempt to have highly integrated models tend to be highly complex and involve large amounts of data (e.g. InVest (Sharp *et al.*, 2014) and ARIES (Villa *et al.*, 2014)), which in turn reduces the usability of the model to a wide audience (Schägner *et al.*, 2013). The FFS model is a parsimonious tool designed to easily spatially assess one ES, which can inform a specific ecosystem service question regarding the importance of areas for inland fisheries. However the resulting spatial information could be incorporated into more complex models to inform one of a suite or bundle of ecosystem services. In addition, the FFS model allows the flexibility of users incorporating their own data sources, which may be at a more appropriate scale, resolution and accuracy. Improvements to data such as the completion of inland water volume layers in the near future (Lehner & Grill, 2013) could be tested and incorporated if it improved the model. However, it is clear from the strong correlations at each scale that this simple model as it currently stands is a good fit for fishery yields.

The model does not consider negative drivers which would reduce expected yield, such as water abstraction (Youn *et al.*, 2014), water quality (Kolding *et al.*, 2008) and overexploitation (Zhang *et al.*, 2012). It may be possible to use the output maps to consider where these threats may have largest impacts to freshwater food provision. The model also does not explicitly consider the transport network that may be involved in supplying fisheries catches to extended markets. The majority of inland fisheries catch is artisanal or subsistence and therefore will not have a large distribution range, particularly in developing countries (Welcomme, 2011b). Given the high correlation between FFS and reported yields found here, it might be expected that larger waterbodies with higher yields are more likely to have increasingly large transportation networks, and thus increases are proportional to waterbody size, however this is not explored within the remit of this study .

The FFS model is designed to create a spatial representation of the importance of sub-catchments for fisheries. It is of note that although validation here has concentrated on fish catches, many other taxa including molluscs, crustaceans and even some reptiles and amphibians all have important contributions to make to FS (FAO, 2014b). The model may be a critical tool in management decisions; for instance, establishing areas where conservation measures may enhance both freshwater species preservation and FS, or where fisheries may in fact be having a negative impact on species. Similarly, a spatial understanding of the importance of fisheries allows an evaluation of the effect of spatially explicit threats, such as dams, pollution and water-extraction. By incorporating further factors such as a poverty index, the model can aim to capture the importance of inland waters to casual and subsistence fishers, for whom inland waters play a more pivotal role, and for which there is a lack of data available (excluding case studies). It is hoped that by introducing a simple tool it will now be easier for decision makers to spatially consider and protect freshwater systems, and the important food sources that are provided by them.

4.6 References

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5 Application and implications of the Freshwater Food Security model and freshwater biodiversity

5.1 Abstract

A lack of spatial data hampers efforts at many scales to answer questions at the interface between ecosystem service delivery and biodiversity. Here, it is shown how the output of the Freshwater Food Security model (FFS) from the previous chapter can be used to address current gaps in our knowledge. Used alongside freshwater species range maps it identifies where fisheries overlap with important areas for biodiversity, which may indicate potential synergies for water catchment protection to preserve food security and protect freshwater species. Equally, where there is a mismatch between fisheries harvests and threatened species can also be identified. Finally, it is demonstrated how FFS data can be incorporated into conservation planning. This information can be used by decision makers from the local to international level to inform management decisions that are beneficial to ecosystem service delivery and species conservation.

5.2 Introduction

It is recognised that ecosystems need to be managed for both native species and the ecosystem services they provide. However there is a paucity of macroecological data available with which to identify trade-offs and synergies of implementing an ecosystem approach, which hampers efforts of ecosystem managers and decision makers (Cowling *et al.*, 2008; Beck *et al.*, 2012; O'Riordan, 2014). Spatial data that is available is frequently out-dated and/or confined to national level statistics, particularly for less developed countries (Davies *et al.*, 2012). Yet policy makers must effect decisions from local scale to national or even international agreements such as the Aichi targets or Sustainable Development Goals, while it is recognised that vastly improved data and monitoring is required to do so (Lu *et al.*, 2015).

Inland water systems are among the most diverse habitats on earth, containing around a third of all known vertebrate species (Dudgeon *et al.*, 2006; Strayer & Dudgeon, 2010) but these systems and the species within them are the most threatened globally (Dudgeon *et al.*, 2006; WWF, 2014). Freshwater ecosystems are also a key resource for people, yet data sets for freshwater ecosystems and the services they provide are lagging behind terrestrial and marine data availability for much of the world (González Vilas *et al.*, 2015). Spatial fisheries data are all but non-existent. Simultaneously, fishery harvests are grossly under-reported with a particular data gap for subsistence fishers and consumers, many of whom are living in poverty, and who are most likely to be dependent on the food security and livelihoods the freshwater species provide (Mills *et al.*, 2011). Some inland fisheries are already seeing a decline in yields due to threats (Laë, 1994; Welcomme, 2001; Allan *et al.*, 2005; Richter *et al.*, 2010). The harvesting and management of fisheries has consequences for the people relying on the services provided by the freshwater species, but equally the freshwater species themselves are impacted (Beard *et al.*, 2011). Management is therefore essential but to date is based on limited knowledge.

In the previous chapter it was shown that the Freshwater Food Security (FFS) model output creates a good spatial proxy for hitherto non-existent data regarding inland fisheries, for the first time allowing for spatially explicit analyses of inland fisheries at any location and extent. This chapter examines how the validated FFS model can be used in combination with high quality species data from the IUCN to guide policy at the interface of inland water capture fisheries (a provisioning ecosystem service) and the conservation of freshwater species diversity. It considers the trade-offs and synergies between the delivery of fisheries as an ecosystem service and conservation – where management and protection of catchments would provide a win-win for protecting the fisheries and freshwater species, but also where it is the delivery of this ecosystem service which is threatening the species inhabiting the waterways. Finally, it is shown how this information can be used to inform conservation practises, by incorporating this information into protected area network planning.

5.3 Methods

The geographical focus for this chapter is Africa and Asia, as this is where the direct dependence on inland fisheries is greatest (Dugan *et al.*, 2010). The focus is on areas comprehensively assessed by IUCN, comprised of: (i) African mainland (accounting for 26% of global inland fisheries yield); and (ii) South Asia and Indo-Burma region (accounting for 40% of global inland fisheries yield) and southern parts of China, the highest yielding country in the world (FAO, 2014a). The Asian region used here includes and joins catchments that fall into the Western Ghats and Indo-Burma biodiversity hotspots (Myers *et al.*, 2000; Mittermeier *et al.*, 2005). Subsequently these focus regions will be referred to as Africa and sub-Asia.

The FFS model produces a predictive map of the relative importance of freshwater fisheries within the region of interest, mapped to the sub-catchment level. It incorporates both supply and demand metrics in tandem, by using the area of inland water to define buffered areas per sub-catchment, from which population data (e.g. Salvatorre et al. 2005) is extracted and summed to produce an overall FFS value per sub-catchment (see Chapter 4 for detailed methodology). These values are then mapped to portray the relative measure of importance of each sub-catchment for fisheries harvest within the study area.

Analyses and mapping are at the HydroBASINS Format 2 database level 8 (Lehner & Grill, 2013) unless otherwise stated, to be concordant with species data mapped by IUCN. In Africa there are 42,651 sub-catchments at level 8, with an area range of 0.15 km^2 to 69,842 km^2 and a mean and median area of 689 km^2 and 460 km^2 respectively. There are 9,091 sub-catchments in the sub-Asia region ranging from 0.19 km^2-11,389 km^2, with a mean of 623 km^2 and a median of 411 km^2. All calculations unless stated otherwise were carried out in R version 3.2.2 (R Core Team, 2015).

5.3.1 FFS for poverty

Harvest from subsistence fisheries catch is vastly underreported (FAO, 2014b). People living in poverty are more prone to rely on subsistence harvesting, and are therefore expected to be under-represented in official catch statistics, yet are particularly likely to be in critical need of fisheries as a resource (Béné *et al.*, 2007; Brooks *et al.*, 2014). By adapting the input data for the FFS model to incorporate poverty data rather than population data, maps were produced to predict where fisheries are of greatest important to these vulnerable groups. Most freely available spatial global datasets for poverty indicators rely on malnutrition rates. However given that this model is exploring where direct subsistence fishing can contribute to food security in spite of poverty, using malnutrition as a proxy would be inappropriate. The data was therefore based on infant mortality rates for the year 2000, mapped to a common quarter degree grid (CIESIN, 2005). Large holes in this spatial dataset in Egypt and southern Sudan were filled using country level values from The World Bank (available at http://data.worldbank.org/indicator/SP.DYN.IMRT.IN) in ArcMap 10.2, while leaving legitimate data holes caused by large waterbodies such as the African Great Lakes: a polygon layer that was the same size as the African region was created with World Bank country level infant mortality rates associated with Egypt and southern Sudan areas only. This polygon shapefile was then converted into a raster layer, and a conditional statement was used to replace any areas with no values in the original poverty layer with values from the new layer.

The FFS model was run with poverty data for Africa and sub-Asia. Spearman's rho correlations between FFS scores derived from population and poverty data per sub-catchment were calculated, due to non-normality of data. Spatial autocorrelation was accounted for by using a modified t-test in the SpatialPack package in R (Osorio & Vallejos, 2014), which adjusts the degrees of freedom based on an estimate of the effective sample size (Clifford *et al.*, 1989).

The spatial congruence of the highest scoring sub-catchments was tested by measuring the percentage overlap between the top 10%, 20% and 30% of FFS values from the population derived and poverty derived data. The spatial distribution was also compared for those same sub-catchments using Syrjala's test. Syrjala's test is a non-parametric test for the difference in distribution of two populations (Syrjala, 1996) and was carried out using the ecespa package in R (De la Cruz, 2008). This calculates an observed psi (Ψ) for the two distributions, defined as the sum of the square of the differences between the cumulative distribution functions of the two populations. This is then compared to the Ψ obtained from distributions from 999 randomised permutations of the data (Syrjala, 1996). The test is based on the Cramér-von Mises statistic and is very conservative; a significant difference between the spatial congruence of the tested observations is therefore difficult to achieve (Rey Benayas *et al.*, 2010).

5.3.2 Spatial overlap between high FFS areas and important biodiversity areas

Similarities in the FFS scores were compared to freshwater species richness (SR) calculated from IUCN species range maps at the sub-catchment level to examine overlap between areas important for food security and biodiversity. Bivariate maps were created (see Appendix D1) to visually assess areas of low to high importance for species richness and FFS scores simultaneously (Robertson & O'Callaghan, 1986). Data were transformed before plotting to normalise data to improve visual interpretation of the maps. Spearman's rho correlations between FFS scores and SR per sub-catchment corrected for spatial autocorrelations were calculated. The top 10%, 20% and 30% of sub-catchments for FFS scores and SR were compared by percentage overlap, and Syrjala's test, as above.

Several combinations of the data were tested. For FFS scores, variations included FFS scores derived from population data, and from the poverty data (see 5.3.1). For SR, three versions were used: freshwater species richness, all aquatic species richness and finally threatened freshwater species richness. Freshwater species richness includes all known fish, odonates (dragonflies and damselflies), molluscs and decapods. All aquatic species richness includes the freshwater species, plus amphibians, birds, mammals, reptiles and plants that require inland waters for at least part of their life cycle as defined by the IUCN guidelines (IUCN, 2013). Threatened freshwater species includes all fish, odonates, molluscs and decapods assessed as Critically Endangered, Endangered or Vulnerable by the IUCN (www.iucnredlist.org). Totals for each species category are given in Table 5.2. Correlations and spatial congruence tests were calculated for each permutation of FFS and SR datasets.

In order to consider the potential spatial conflict of fisheries with freshwater species, bivariate maps and correlations were derived as above between FFS scores and the richness of species assessed by IUCN as threatened by overfishing per sub-catchment.

5.3.3 Using FFS in conservation planning

Any spatial overlap between FFS and freshwater species richness implies that there are potential win-wins by managing freshwater catchments for multiple species, particularly

Table 5.1 Number of species for Africa and sub-Asia for each taxonomic group

SR dataset	Africa	Sub-Asia
Fish	2,966	1,781
Odonates	707	575
Molluscs	573	599
Decapods	183	481
Amphibians	606	469
Birds	675	436
Mammals	31	16
Reptiles	29	139
Plants	803	483
Threatened fish	558	258
Threatened odonates	38	18
Threatened molluscs	20	56
Threatened decapods	36	36
Threatened amphibians	134	77
Threatened birds	40	39
Threatened mammals	10	12
Threatened reptiles	1	43
Threatened plants	121	52

given the link between increased species richness and increased yield (Brooks *et al.*, 2016)). Equally, there is conflict between fishing and some threatened species. FFS was incorporated into the planning software Marxan with zones to test the prioritisation of a network to maximise benefit for species and people. Marxan with zones is a conservation planning tool which considers the spatial differences in cost of conservation networks with different levels of protection (Watts *et al.*, 2009). It is based on a heuristic optimisation algorithm which uses a simulated annealing process to optimise costs across a range of iterations from which either a 'best' case or summed solution can be garnered.

Marxan with zones was programmed to base a protection network on freshwater species richness by aiming to capture at least two sub-catchments per species based on Holland et al. (2012). The level of threat was included from the IUCN Red List categories by increasing the representation of Critically Endangered (CR) species to 100% of their range, with 75% of their range and 50% or their range represented for Endangered and Vulnerable species respectively. Basic cost was represented by the area of the sub-catchments to minimise the total area of protection. Protection was divided into two levels – low and high, with a third zone of 'available' included for catchments not included in the conservation network. Low level protection indicates areas that require management and policy to reduce external pressures on species within the inland waters, but would not exclude use by people outright. High level protection is species focused to the exclusion of human use. Area costs were weighted by 1 for the available zone, 2 for the low protection zone and 3 for the high protection zone. In order to keep the scenarios as simple as possible all other thresholds follow the Marxan with zones default user guidelines (Watts *et al.*, 2008).

There has been an increasing call to incorporate ecosystem services into conservation planning such as Marxan (Egoh *et al.*, 2007). To date this inclusion has focused on ecosystem services as an additional feature in Marxan to preserve alongside biodiversity (Chan *et al.*, 2011; Schröter *et al.*, 2014; Adame *et al.*, 2015). Here a novel approach is used to include the ecosystem service of FFS as a cost in the system, which can be weighted both positively and negatively to prioritise the catchments that are good for biodiversity and for FFS while simultaneously considering the increased human cost that a protected area may have on fishing harvests. To do this, FFS was weighted as 0 for the multiplier of the available zone (which means no cost or benefit is associated), -1 for the low protection zone (to reflect that well managed catchments are beneficial for fishery harvests), and 1 for the high protection zones (to reflect that a high level of protection will

Table 5.2 Scenario outlines for Marxan with zones. Area is sub-catchment area (km^2), FFS is output scores from Freshwater food security model.

Scenario	Cost	Targets for overharvested species
1	Area	No
2	Area and FFS	No
3	Area and FFS	Yes

come at the cost of fishery harvests). In order that area and FFS costs have comparable impact on the overall decision, area was included on a 0-1 scale, where 1 equals the area of the largest sub-catchment under consideration. Finally, species that were CR, EN or VU and have been assessed as threatened by overharvesting were targeted to include at least one sub catchment in the high protection zone.

To compare the effects of each of the parameters three scenarios outlined in Table 5.2 were run for both African and sub-Asian data. Marxan with zones does not aim to come up with a solution, but calculates which iterations offer the lowest cost within the parameters considered, which in this context is described as ‘best’ solution.

5.4 Results

5.4.1 FFS for poverty

Outputs from the FFS model based on population or poverty data correlate strongly (rho=0.80, df =1193.582, P <0.001 and rho=0.86, df = 1716.35, P <0.001 for Africa and sub-Asia respectively). While a substantial proportion of the top scoring catchments based on each data set overlap, the spatial distributions of these catchments are significantly different (Table 5.3). As an example in Africa, the Congo River is highlighted more strongly using poverty data than population data (Figure 5.1), and in Asia, using poverty data increases the importance of Tonle Sap in Cambodia and the Mekong river in respect to population data (Figure 5.2).

5.4.2 Spatial overlap between high FFS areas and important biodiversity areas

Larger waterbodies (both lakes and rivers) are generally associated with areas with high biodiversity and importance for fisheries (Figure 5.3 and Figure 5.4). In Africa, there are medium strength positive rank correlations between FFS scores and both freshwater species

Table 5.3 Spatial congruence of top sub-catchments for FFS scores derived from population data and poverty data. Psi (Ψ) values in bold highlight score pairs are significantly different (p<0.05).

% of sub-catchments	Region	Ψ	Overlap
Top 10%	Africa	**67.68**	40.01%
	Asia	**5.11**	48.94%
Top 20%	Africa	**130.17**	47.21%
	Asia	**7.05**	55.01%
Top 30%	Africa	**192.25**	53.55%
	Asia	**9.21**	58.67%

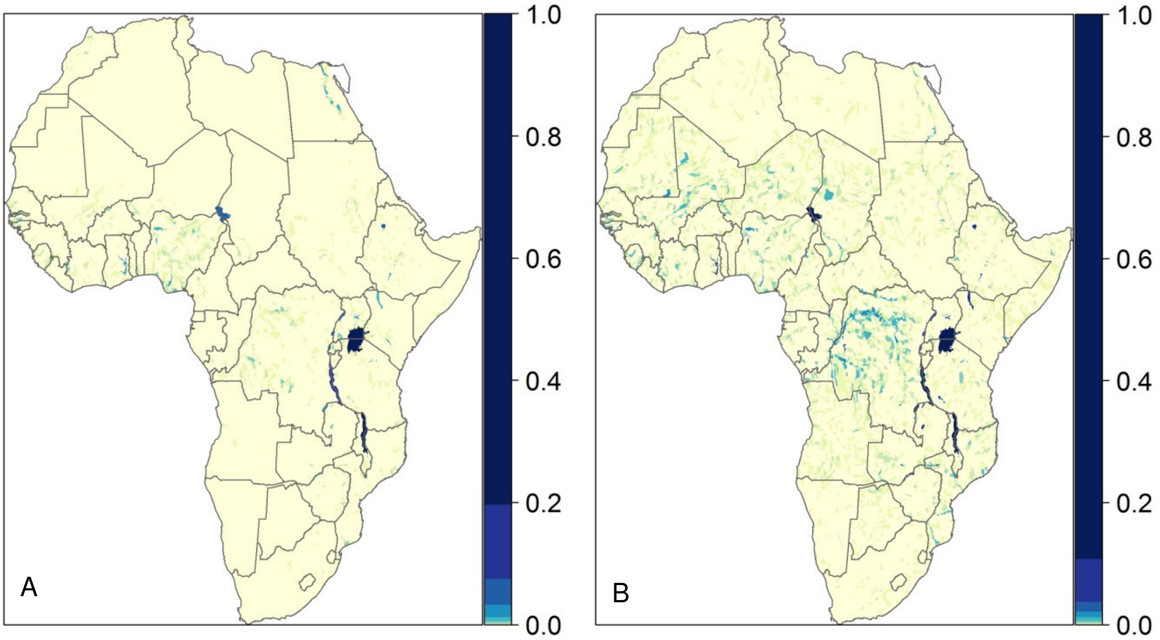

Figure 5.1 Relative importance of freshwater food security (FFS) in Africa. Based on A) population data, B) poverty data.

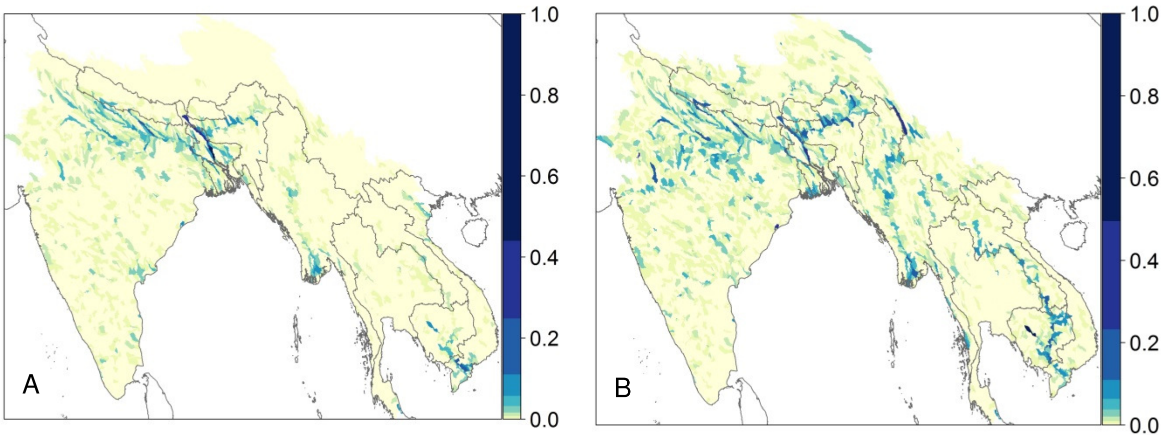

Figure 5.2 Relative importance of freshwater food security (FFS) in Asia. Based on A) population data, B) poverty data.

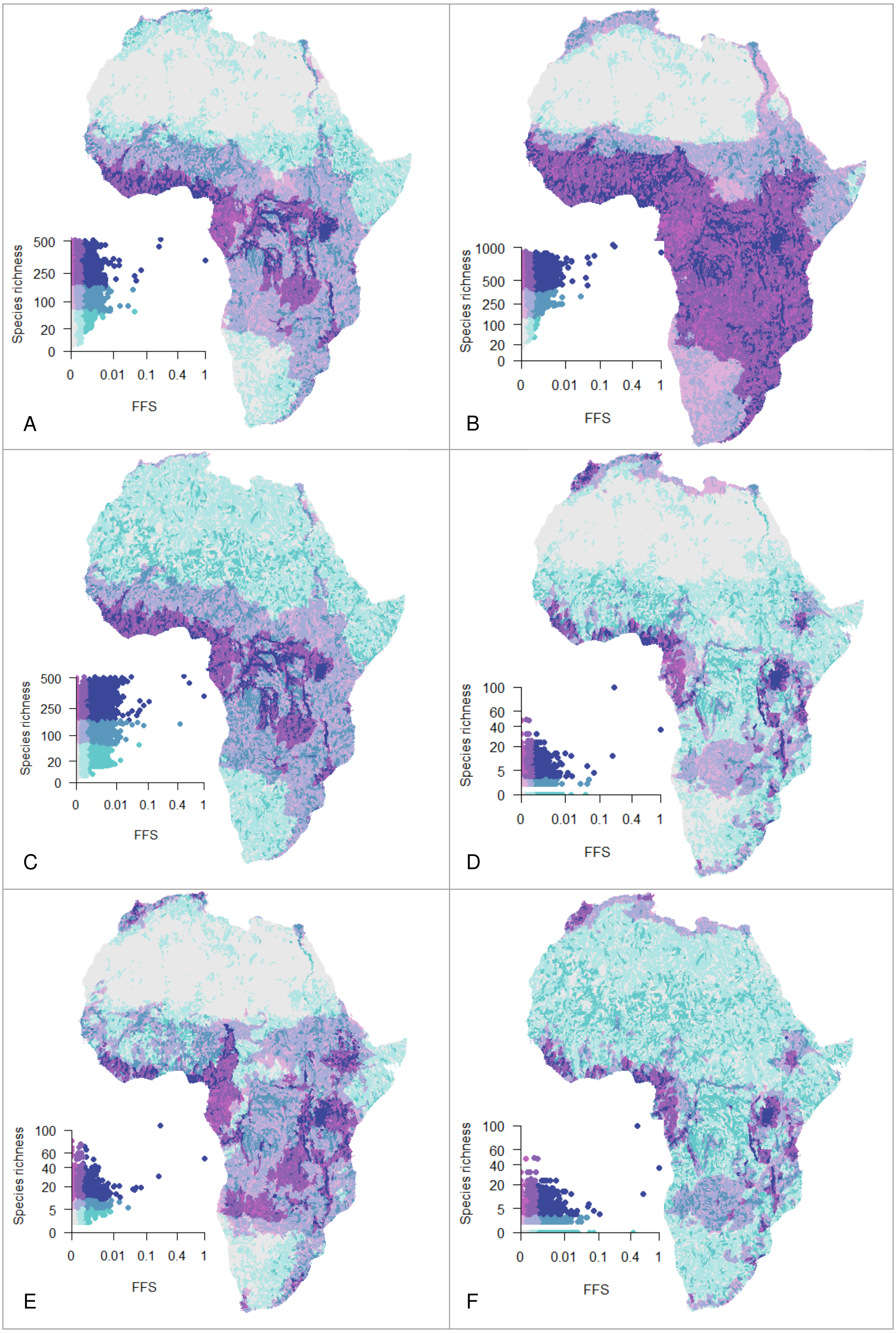

Figure 5.3 Bivariate choropleths of: A) Freshwater food security (FFS) vs freshwater taxa richness; B) FFS vs all aquatic species richness; C) Poverty FFS vs freshwater taxa richness; D) FFS vs threatened freshwater taxa richness; E) FFS vs threatened aquatic species richness; F) Poverty FFS vs threatened freshwater taxa richness. FFS axes are quarter root transformed and species richness axes are square root transformed. See Appendix D for details on colour legend.

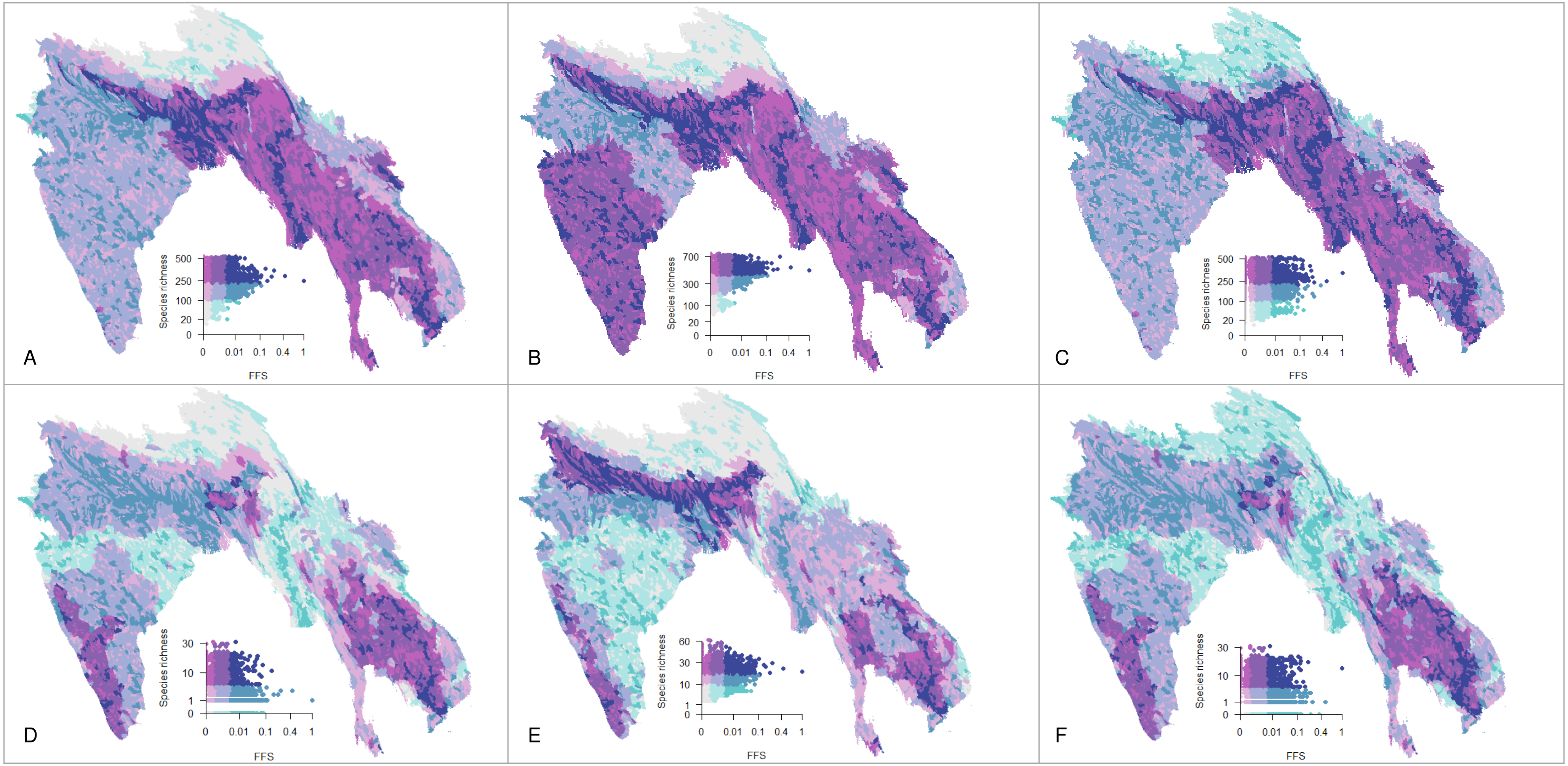

Figure 5.4 Bivariate choropleths of: A) Freshwater food security (FFS) vs freshwater taxa richness; B) FFS vs all aquatic species richness; C) Poverty FFS vs freshwater taxa richness; D) FFS vs threatened freshwater taxa richness; E) FFS vs threatened aquatic species richness; F) Poverty FFS vs threatened freshwater taxa richness. FFS axes are quarter root transformed and species richness axes are square root transformed. See Appendix D for details on colour legend.

richness and all aquatic species richness (Table 5.4). There was also a medium strength rank correlation between FFS scores and all threatened aquatic species, although the correlation was weaker against threatened freshwater species. FFS scores based on poverty data showed a weak positive rank correlation with freshwater species, and no correlation with threatened freshwater species. There were no significant correlations between FFS score layers and SR for sub-Asia, except between FFS scores based on population data and the richness of threatened freshwater species (Table 5.4).

The spatial distribution of top scoring catchments for FFS and species richness is significantly different in all data combinations explored for both Africa and sub-Asia (Table 5.5). The overlap between top FFS and species richness sub-catchments is greater in Africa than in sub-Asia, and for both regions the percentage overlap increases as the percentage of top catchments included is increased from the top 10% of catchments, to 20% and 30% (Table 5.5).

FFS scores correlate with the number of freshwater species threatened by overharvesting in both Africa (Figure 5.5) and Asia (Figure 5.6). However the distribution of top scoring catchments is significantly different for both regions, with a smaller percentage of overlap seen in top scoring sub-catchments (Table 5.6) than when FFS was compared to general species richness (Table 5.5).

Table 5.4 Spearman's rho of Freshwater food security (FFS) scores versus species richness (SR) per sub-catchment. Significant values (p<0.05) are in bold.

Region	FFS	SR	rho	Adj. df
Africa	Population	Freshwater sp.	**0.39**	52.29
	Population	All aquatic sp.	**0.42**	46.69
	Poverty	Freshwater sp.	**0.16**	216.13
	Population	Threatened freshwater sp.	**0.19**	173.67
	Population	Threatened aquatic sp.	**0.36**	60.42
	Poverty	Threatened freshwater sp.	0.06	711.77
Sub-Asia	Population	Freshwater sp.	0.06	110.84
	Population	All aquatic sp.	0.14	149.14
	Poverty	Freshwater sp.	-0.01	247.38
	Population	Threatened freshwater sp.	0.08	195.96
	Population	Threatened aquatic sp.	**0.19**	239.23
	Poverty	Threatened freshwater sp.	-0.03	490.72

Table 5.5 Spatial congruence of top scoring sub-catchments for Freshwater food security (FFS) scores and species richness. Psi (Ψ) values in bold highlight score pairs that are significantly different (p<0.05).

% of sub-catchments	FFS	SR	Africa		Asia	
			Ψ	Overlap	Ψ	Overlap
Top 10%	Population	Freshwater	**321.08**	12.62 %	**211.37**	4.58 %
	Population	All aquatic	**185.85**	13.24 %	**131.57**	5.78 %
	Poverty	Freshwater	**108.30**	12.08 %	**183.72**	5.42 %
	Population	Threatened freshwater	**221.86**	12.48 %	**164.91**	5.49 %
	Population	Threatened aquatic	**1440.87**	10.45 %	**41.09**	12.85 %
	Poverty	Threatened freshwater	**0.04**	12.08 %	**138.69**	6.27 %
Top 20%	Population	Freshwater	**453.81**	20.85 %	**253.34**	12.90 %
	Population	All aquatic	**295.02**	21.59 %	**177.78**	12.92 %
	Poverty	Freshwater	**131.58**	17.73 %	**213.21**	12.62 %
	Population	Threatened freshwater	**385.67**	21.26 %	**253.00**	10.89 %
	Population	Threatened aquatic	**2029.90**	19.57 %	**59.90**	20.57 %
	Poverty	Threatened freshwater	**131.58**	17.73 %	**217.14**	9.63 %
Top 30%	Population	Freshwater	**543.05**	30.60 %	**246.17**	17.98 %
	Population	All aquatic	**386.74**	31.63 %	**191.58**	21.97 %
	Poverty	Freshwater	**150.27**	24.76 %	**193.04**	18.71 %
	Population	Threatened freshwater	**3651.28**	29.98 %	**298.96**	18.73 %
	Population	Threatened aquatic	**1434.98**	32.80 %	**75.65**	27.06 %
	Poverty	Threatened freshwater	**150.27**	24.76 %	**253.20**	15.96 %

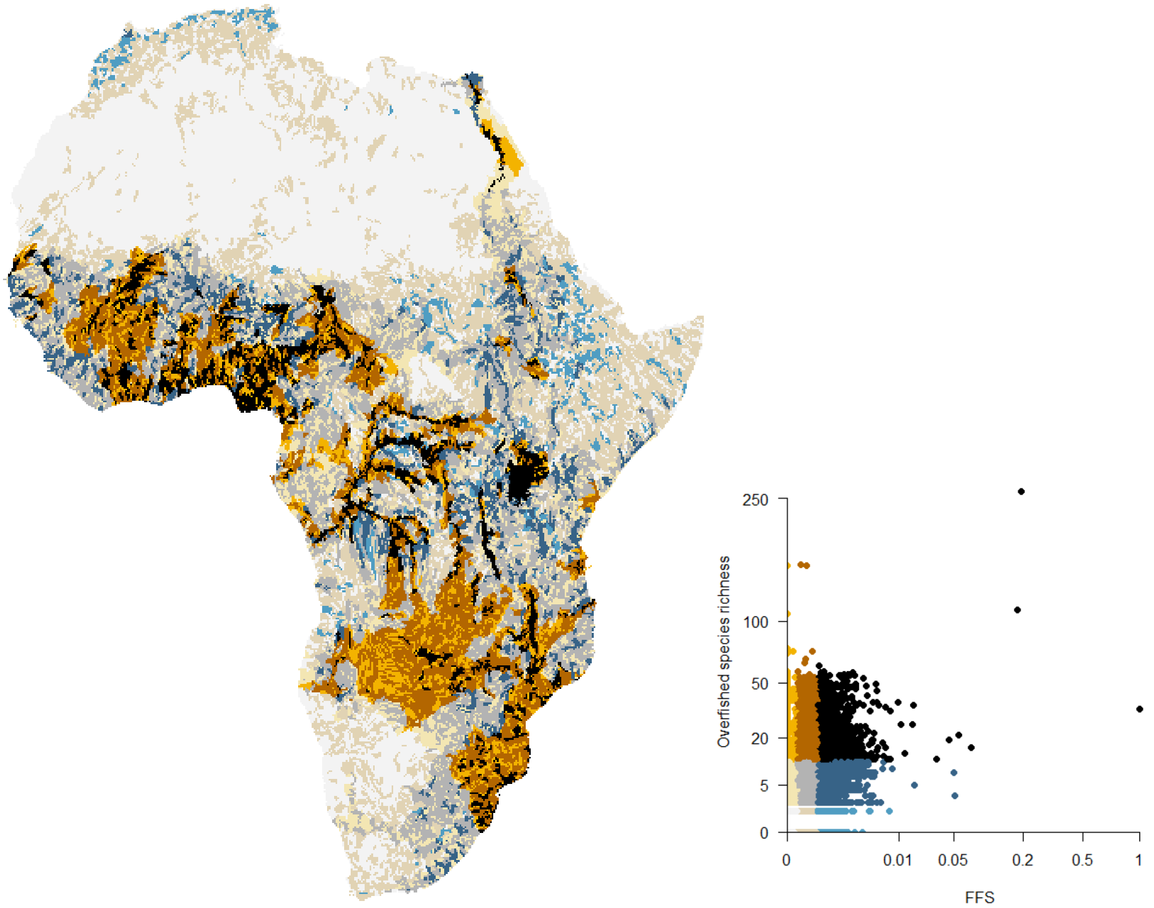

Figure 5.5 Bivariate choropleth of conflict in Africa between freshwater species assessed as threatened by overharvesting by IUCN, and freshwater food security (FFS) scores (rho=0.36, df=83.55, *P*<0.001). FFS axis is quarter root transformed and freshwater species richness axis is square root transformed. See Appendix D for details on colour legend.

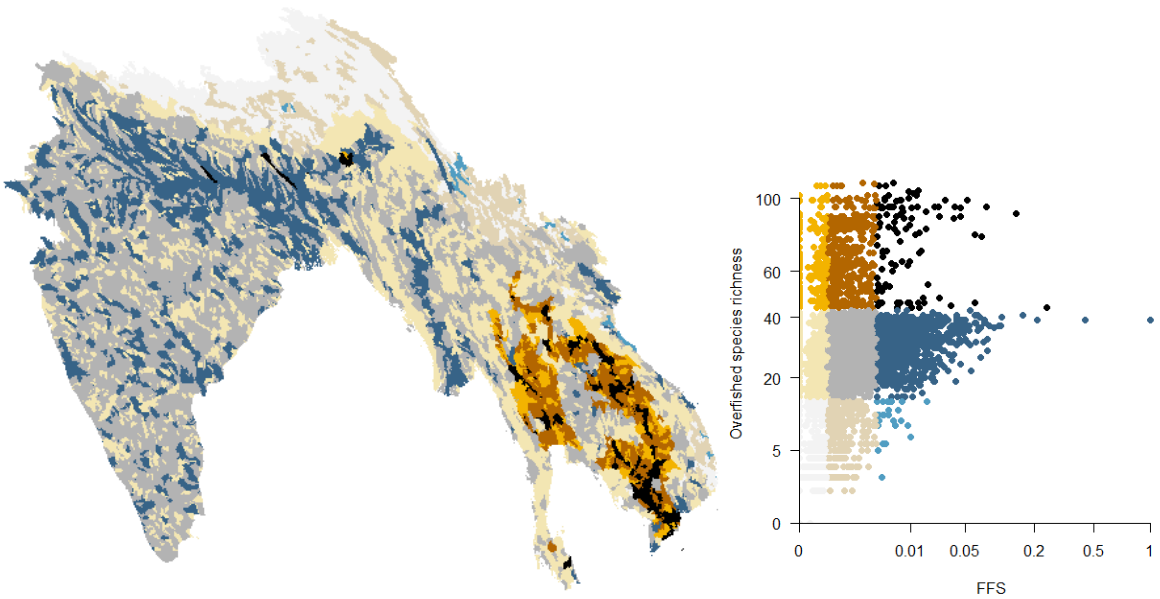

Figure 5.6 Bivariate choropleth of conflict in sub-Asia between freshwater species assessed as threatened by overharvesting by IUCN, and freshwater food security (FFS) scores (rho=0.25, df=329.37, *P*<0.001). FFS axis is quarter root transformed and freshwater species richness axis is square root transformed. See Appendix D for details on colour legend.

Table 5.6 Spatial congruence of top scoring sub-catchments for freshwater food security (FFS) scores and richness of freshwater species threatened by overharvesting. FFS and overfished species richness layers are described in main text. Psi (Ψ) values in bold highlight score pairs that are significantly different (*p*<0.05).

% of sub-catchments	FFS	SR	Africa		Asia	
			Ψ	Overlap	Ψ	Overlap
Top 10%	Population	Overfished freshwater	**270.96**	5.89 %	**199.99**	5.56 %
Top 20%	Population	Overfished freshwater	**451.07**	10.62 %	**135.35**	10.25 %
Top 30%	Population	Overfished freshwater	**552.87**	11.37 %	**123.79**	18.05%

5.4.3 Incorporating FFS data into protected area network planning

Each of the scenarios led to a different spatial configuration of zones deemed to be the best solution of the 1000 permutations tested in Marxan with zones (Figure 5.7). In Africa, costing the protected area network by area alone assigned 90% of the total area and 83% of the sub-catchments to the available zone, and 10% of the area (17% of the sub-catchments) to the low protection zone. Including FFS as a cost as well reduced the available zone to 39% of the area (41% of sub-catchments) and increased low protection zone area to 61% (totalling 59% of sub-catchments). Neither of these scenarios included any high protection designations. Including overfished species added only 0.5% of the area to high protection zone (29, or 0.07% of sub-catchments).

In sub-Asia, costing the protected area network by area alone assigned 2% of the total area and 0.7% of the sub-catchments to the available zone, and 98% of the area (99% of the sub-catchments) to the low protection zone. Including FFS as a cost as well increased the available zone to 40% of the area (30% of sub-catchments) and increased low protection zone area to 60% (totalling 70% of sub-catchments). Again, neither of these scenarios included any high protection designations. Including overfished species added only 0.4% of the area to high protection zone (46, or 0.5% of sub-catchments).

5.5 Discussion

Management of water catchments requires consideration of the needs of the local (as well as wider) population that depend on the resources available, and of the species within, but

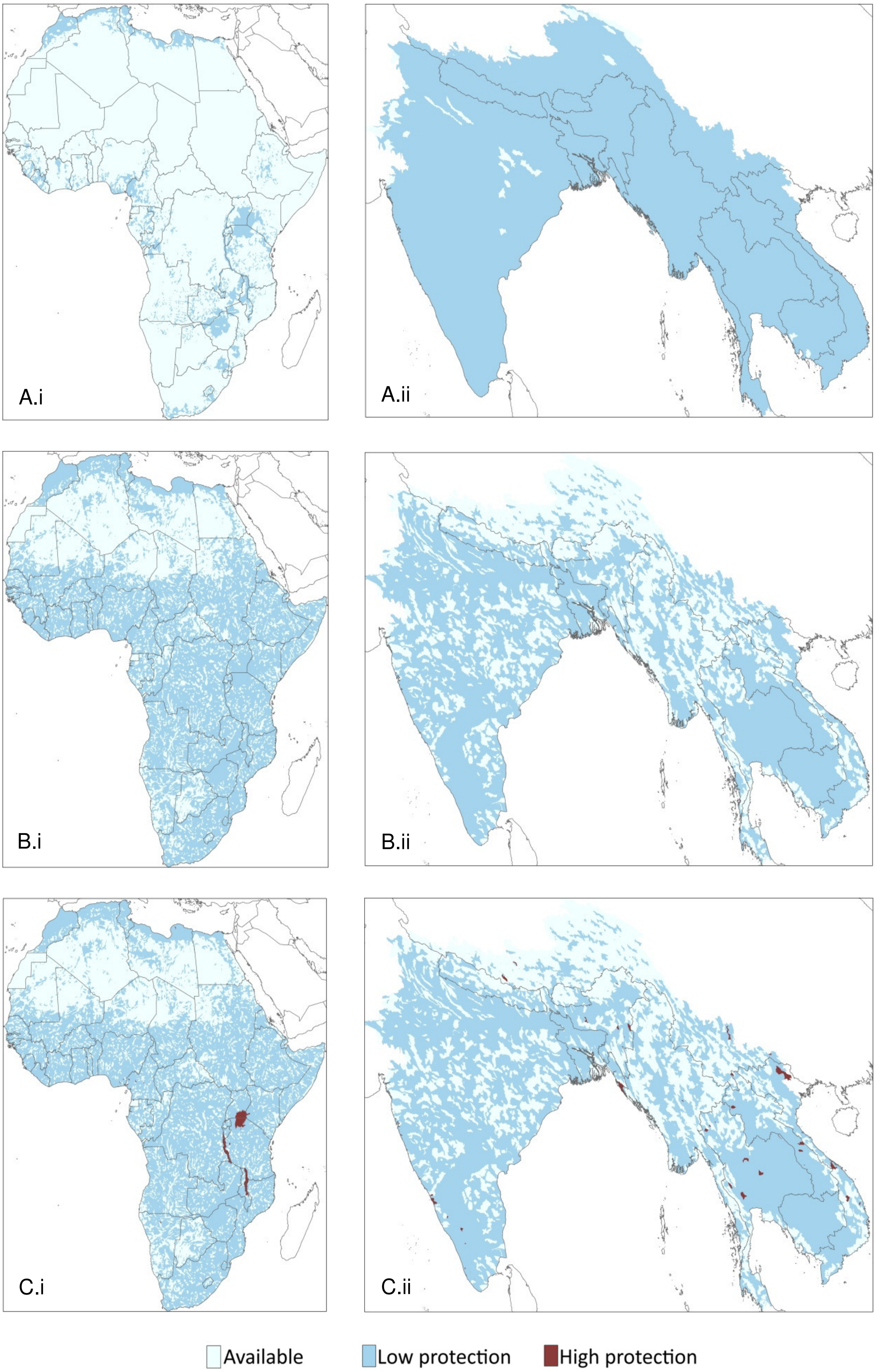

Figure 5.7 Spatial distribution of zones of different levels of protection. Best solution obtained from Marxan with zones for freshwater sub-catchments in i) Africa and ii) sub-Asia for: A) scenario 1 - costing by area; B) scenario 2 – costing by area and FFS; and C) scenario 3 – costing by area and FFS and including targets for overfished threatened species.

supporting data is often missing (Carpenter *et al.*, 2009). With freshwater systems under such threat, the provision of food and livelihoods of many fishers is in jeopardy, as are the fauna and flora inhabiting these systems (Dudgeon *et al.*, 2006; Dugan *et al.*, 2010). Rarely these days does (or should) conservation planning consider only the needs of species at threat, without considering the impact conservation initiatives would have on the local population (Ban *et al.*, 2013). With well thought out and designed practises there is a real chance of working towards protection of food security and of freshwater species simultaneously (Cowx & Portocarrero Aya, 2011). Results here demonstrate an approach to informing decisions where required management data is not available. It is shown how data can be used to inform where levels of protection can ensure the future both of fisheries and of species conservation, while illustrating the potential for a disconnect between the two. These issues can only be resolved through careful deliberation and planning in order not to endanger either the ecosystem services or the species providing them.

Being able to predict the fish catches of the most vulnerable groups could ensure that their needs and impacts are considered in fishery issues, a harvest which is currently massively under-reported (FAO, 2014b). It is unsurprising that ranked FFS scores using population data and poverty data correlate – where there are more people there are more likely to be more poor people, particularly in the study areas here of Africa and sub-Asia (CIESIN, 2005). The differences between spatial distribution of the population and poverty datasets do highlight however that the needs of the poorest people are not necessarily spatially analogous with the population of the study area as a whole, and should be expressly considered.

The most common factors identified for predicting trade-offs in an ecosystem service approach are a private interest in the natural resources, interest in a provisioning service and stakeholders acting at a local scale, all of which are applicable to most fisheries. Taking account of these trade-offs is more likely to result in synergies than prioritising a win-win approach (Howe *et al.*, 2014). Analysing the FFS output with overfished species distribution identifies where trade-offs in fisheries management and conservation might occur. Although their ranks correlate there is little overlap between key FFS sites and sub-catchments with the greatest number of overfished species (Figures 5.5-5.6 and Table 5.6) which suggests a good potential for balanced management, where prioritising the protection of the most threatened species may be achieved without maximum cost to ecosystem service delivery.

As noted in Chapter 4, Spearman's rho correlations were calculated due to non-normality of data, but there are limitations to this associated with a loss of information and alternative goodness-of-fit measures could be considered (see section 4.5). A range of tools is needed to assess and interpret the spatial relationship, which is why the spatial overlap and congruence were also calculated in the scenarios explored here. While there is some spatial correlation between human needs for fisheries and species richness this rank correlation is not necessarily strong and spatial congruence between the most important sub-catchments for each metric is not implicit (Table 5.4 and Table 5.5). The FFS output has been used alongside species ranges to indicate where catchment management is a potential win-win for species and fisheries harvest protection. It has been shown how this can be done simply and directly such as with the bivariate choropleths (Figure 5.3 and Figure 5.4), or in a more complex analysis such as exampled here using Marxan with zones.

Marxan with zones has proved to be an incredibly useful tool in conservation planning for terrestrial, marine and freshwater systems. It is an applied tool that is used to directly inform protected area networks (e.g. Agostini et al. 2010), but has been used here as a theoretical exercise and a proof of concept for incorporating FFS output into decision making. The results should be viewed with caution – to correctly apply the conservation planning software there are multiple parameters that should be adjusted to meet the requirements for the region and scenario in question (Watts *et al.*, 2008, 2009). In addition there may be further considerations to incorporate such as other socio-economic factors (Ban & Klein, 2009) and connectivity of the zones, a particular but surmountable challenge in freshwater systems (Hermoso *et al.*, 2011, 2015). It has been a purposeful decision to not adapt these thresholds as recommended and to keep all inputs as basic as possible, in order to realise the difference that incorporating FFS has upon the results in different scenarios and in different regions. For this reason we see extremes for both Africa and Asia when costing zones by area alone, albeit of an opposite nature – in Africa only 10% of the area is assigned to a protected zone, while 98% of sub-Asia is. This is a result of the large number of species for consideration without a target protected area size defined. But in both cases the addition of FFS as a cost refines the network to approximately 60% of each region. Adding FFS as a cost rather than a feature means that the network has been effectively prioritised by mutual benefits to both fisheries and species, and the two are incorporated in tandem rather than considered separately as with other studies integrating ecosystem services (Chan *et al.*, 2011; Schröter *et al.*, 2014; Adame *et al.*, 2015).

Without entire regions being managed as protected zones it would be difficult to meet the species area targets for either the low or high protection zones, and it would be necessary to ensure that results would at least meet an acceptable level of species protection. For instance, in this simple exercise the addition of overfished species has added high protection zones but very few, and these would be inadequate to meet the requirements of protecting all highly threatened species. These requirements could be better represented through the correct application of the threshold and variable options within the Marxan with zones software, for instance by making use of the feature penalty factors for not meeting defined targets (Watts *et al.*, 2008). All outcomes should be considered in a holistic context: it is unrealistic to expect the entirety of African Great Lakes to be designated high protection zones as suggested by the outputs presented here if that defined them as a no fishing zone, as they are of huge importance to millions of people for food and indeed for a suite of additional ecosystem services (Swallow *et al.*, 2009; Downing *et al.*, 2014). The technique used here produced a simple three level system of protection but for real-world application a more nuanced scale is likely to be appropriate.

The choice of data is of course critical. In examining the spatial overlap between high-scoring FFS areas and species richness, the bivariate choropleths differ depending on whether richness of freshwater species, all aquatic species or threatened freshwater species are used (Figure 5.3-5.4). Any datasets used to aid decision makers should therefore closely adhere to the goals of the management in question, while appropriately representing the social and ecological components affected. The examples used here also highlight the importance of the quality of the data, as well as understanding how underlying data effects model results; for large parts of Africa poverty data was only available at the national level, which artificially increases the FFS output in inappropriate levels such as Saharan Africa, where actual population levels (poor or not) are incredibly low (Salvatorre *et al.*, 2005). It is interesting however that the African population and poverty FFS scores correlate more strongly with each other, and that neither the datasets for Africa or sub-Asia are spatially congruent, despite sub-Asia poverty data having a finer resolution of sub-national data. Overall the difference in correlation, spatial congruence and Marxan with zones results between Africa and sub-Asia in this Chapter emphasise that there are regional differences, and data and tools used in research must reflect local needs, and at an appropriate scale.

As demonstrated here, the maps produced by the FFS model can be used in conjunction with biodiversity datasets to explore spatial questions that lack of data previously inhibited.

This data augments a recent drive to increase spatially explicit information to aid the understanding of freshwater habitats and the benefits they incur (Domisch *et al.*, 2015; González Vilas *et al.*, 2015). Critically, data of this sort allows for extending the relationship between this ecosystem service and biodiversity with models of spatially explicit threats, such as dams (Mulligan *et al.*, 2009; Winemiller *et al.*, 2016), pollution (Mekonnen & Hoekstra, 2015) and water-extraction (Vörösmarty *et al.*, 2010), and allows for better prediction of how inland water systems and their benefits are expected to change over time. Data such as these presented here are relevant not only for important international agreements such as Aichi targets 6, 12 and 14, but the interface between them. The development of this model and the application and analysis of the spatially explicit predictive data it produces can support policy makers and land managers to make informed decisions, which may not only maximise benefit for freshwater fisheries but simultaneously considers species protection and conservation.

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6 Discussion

There is growing emphasis towards ecosystem services (ES) in environmental management and policy as it is seen as an approach that considers the needs of people and the natural world in tandem. How well this approach integrates with conservation in its practical application is varied and often unclear (Fisher & Brown, 2014). Business and governments have identified pressures on freshwater resources as being the principal risk to society over the next 10 years, with increasing tension and competition over their use (World Economic Forum, 2016). Simultaneously freshwater species are the most threatened globally and in dire need of conservation actions (Collen *et al.*, 2014; WWF, 2014). There is therefore a pressing need to be able to incorporate human and species needs to a system which is expected to be receiving rapid modifications and intense interest. This thesis has examined the freshwater ES of food provision, with a focus on inland water capture fisheries, and examined the interaction between freshwater biodiversity and food security from these fisheries. This chapter aims to synthesise the findings of the thesis in the context of ecosystem approach and species conservation, and make suggestions for future directions for research and policy.

Throughout the thesis a wide range of analyses and statistical techniques have been used, including novel approaches to best answer the research questions. A range of scales have been considered, from localised case-studies in Chapter 2 and district level modelling and analyses in Chapter 4, to large scale cross-continental analysis in Chapters 3 and 5 and even global mapping in Chapter 4. This reflects the range of scales at which ES delivery is considered.

There has been a focus throughout on using open source tools and data. The application of environmental management and conservation is often restricted by funds and technical access. The methods used here are widely applicable, by using freely available datasets (Chapters 2, 3, 4 and 5) and open access software such as R (R Core Team, 2015) and Marxan with zones (Watts *et al.*, 2008, 2009). It is hoped that this project has provided theoretical and practical improvements to better inform ES focused water catchment management and policy.

6.1 Advances in our understanding emerging from this study

6.1.1 Inland capture fisheries are an undervalued resource

Chapter 2 used small-scale case studies in Asia to examine the difference in perceived value of a collection of freshwater ES against a monetary valuation of the same services. In the eight services tested fisheries were economically the least valuable, yet the non-monetary approach found fisheries to be assessed as one of the highest value services at the local scale, in fact being scored as the most important service for two of the four sites (Brooks *et al.*, 2014). Disaggregating the stakeholders showed that people who are likely to be influential in land management decision-making (government officials and business owners) most closely reflected the values of an economic assessment and not those of poorer people whose livelihoods depended most directly on the land (fishers and to a lesser extent, farmers). This suggests that the true value of fisheries is poorly represented and the value to the poor and most in need may be considerably neglected. This is of particular concern when 80% of the total reported harvest from inland capture fisheries come from low-income food-deficit countries (Kapetsky, 2003). Coupled with this is the inadequate reporting of yields, particularly at the artisanal and subsistence level. Case studies documenting household consumption or direct yield monitoring in Asia and Africa frequently report harvest two to four times greater than official catch data reports (e.g. Welcomme 1976, Coates 2002, Béné et al. 2007, Hortle et al. 2008, Lymer et al. 2008).

There is therefore a critical need to be able to account for representative levels of catch in order to consider the benefits of inland fisheries (Youn *et al.*, 2014). The lack of accurate data undervalues the global importance of inland water capture fisheries, and makes analysis and decision-making problematic (Bartley *et al.*, 2015). Chapter 4 addressed this data gap by creating a validated model which can spatially predict the importance of inland capture fisheries. Chapter 5 advanced the model to illustrate that the importance of under-reported catches of subsistence fishing for those in poverty in particular can be represented, thus allowing specific consideration of the benefits garnered to the most vulnerable and potentially neglected groups in the absence of robust data.

6.1.2 High species richness is beneficial for fishery yields

While ES delivery relies on species, a positive relationship between biodiversity and ES is not implicit (Hooper *et al.*, 2005). Unlike marine fisheries, inland fisheries tend to harvest a range of species with almost no by-catch, making the most of the fishery resources available to them and spreading the benefits widely (Welcomme *et al.*, 2010). On one hand a greater diversity in species may mean a greater number of species harvested. Yet a large proportion of fisheries harvest targets single high income species and for many countries official catch data shows that five or fewer species make up more than half of the total country yield (Table B6 in Appendix B).

Chapter 3 demonstrated that countries with higher species richness reported higher total yields even when accounting for other macro-ecological and human drivers. In addition, the variation in annual yield was lower in African countries with higher species richness (Brooks *et al.*, 2016). This implies that protecting inland waters for the benefit of multiple species is not only valuable for the overall yield but for the resilience of that yield over time, an integral pillar of food security and critical for peoples livelihoods. There is therefore potential synergy between ecosystem service delivery and fish species conservation in inland waters. No doubt ecosystem management and conservation to protect fish species will have knock-on benefits to a range of freshwater taxa, although it should be noted that fish conservation cannot be assumed to be a panacea for freshwater biodiversity conservation as a whole, due to a lack of spatial congruence between different taxonomic groups (Collen *et al.*, 2014). Chapter 5 went on to study how data can be used to guide policy at the interface between fisheries and species conservation, and consider both the spatial trade-offs and win-wins that occur.

6.2 From knowledge to practice

The link between species richness and fishery yield totals and stability shown here is an important relationship and is a valuable consideration for management practitioners worldwide. There is a global commitment to the conservation and sustainable use of biological diversity agreed upon by the Convention on Biological Diversity (CBD) which is currently governed by the Aichi Biodiversity Targets adopted at the 10th Conference of the Parties (COP). These targets incorporate an ecosystem services approach by recognising that biodiversity underpins ecosystem functioning, and conservation and restoration of biodiversity is essential for sustaining and delivering benefits for human well-being (see www.cbd.int). Not only are the results reported here directly relevant to many of the targets such as targets 6, 12 and 14 (Box 6.1), but examine the interface between them. Understanding this relationship allows for measures to be adopted that will make progress towards reaching multiple targets simultaneously, for the benefit of species and human well-being.

Box 6.1 Aichi Biodiversity targets. from www.cbd.int/sp/targets/

Target 6: By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe ecological limits.

Target 12: By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained.

Target 14: By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable.

6.2.1 Benefits beyond fisheries

While fish production is one of the most important ecosystem services provided by inland water ecosystems there are significant trade-offs between capture fisheries and other ES such as hydropower generation and land-use change (e.g. draining for agriculture), which are often high value and high impact (Brugere *et al.*, 2015). It is these trade-offs which are most likely to need balancing at a local scale, and by the value of fishery yields alone there is limited incentive to prioritise water catchment management for the benefit of fisheries and species protection, particularly if there is a disconnect between decision-makers and the benefactors of ES such as fisheries (Brooks *et al.*, 2014). However inland water fisheries provide more than just food, including recreational and cultural services, and empowerment to individuals (Lynch *et al.*, 2016). It has been recognised that river basin management plans designed to improve ecosystem status do not necessarily improve human well-being without deliberate consideration of benefits (Terrado *et al.*, 2016). The benefits of managing water catchments for healthy fisheries are multiple and far-reaching, with the positive contributions to human well-being out-weighing the negative impacts that fisheries cause (Table 6.1) and contributing a far higher value to local communities than an assessment of harvest value alone would conclude. In comparison to the delivery of other ecosystem services that inland waters deliver such as those considered in Chapter 2, fisheries generate fewer trade-offs and negative externalities, and investment in sustainable fisheries can be considered a replenishment of natural capital which will increase the flow of ecosystem services from inland water ecosystems (Brugere *et al.*, 2015).

Table 6.1 Provisioning, regulating, supporting and cultural services provided by freshwater fish populations and inland capture fisheries. Green, positive contribution; Red, negative contribution or self-inflicted impact (from Brugere *et al.*, 2015).

Provisioning	Regulating	Supporting	Cultural
Proteins and other nutrients	Nutrient cycling	Biodiversity	Recreation and tourism
Medicinal products	Biological regulation	Food webs and trophic structures	Education
Income/revenue	Sedimentation regulation	Ecological balance	Research
Aquafeeds	Water quality	Aquaculture	Cultural and spiritual identity and heritage
Livelihood options			
Health, food security			

Despite their demonstrably large contribution, inland fisheries generally receive little consideration in water allocation and land management decisions due to a lack of support and political will (Cooke *et al.*, 2013). One of the greatest challenges is to marry the knowledge exchange between scientists and environmental decision makers which is a surmountable yet poorly implemented practice (Cvitanovic *et al.*, 2015). In the marine realm an ecosystem approach is increasingly recognised in fisheries policy, yet in a review of 1,200 marine fish stocks only 24 considered any sort of ecosystem drivers to the tactical management of the stocks and a single-species focus was still the norm (Skern-Mauritzen *et al.*, 2016). It is likely to be a realistic reflection of the situation for inland water catch fisheries also. Aronson *et al.* (2010) argued that there are three "great divides" causing the disconnect between knowledge and application: an economic development divide between the rich and the poor; an information divide which obstructs communications between scientists, public opinion, and policy makers; and an ideological divide between economists and ecologists. Chapter 2 examined how an economic and social divide caused a disconnect in valuations of ES, however by recognising and accounting for a breadth of stakeholders, differences in values can in incorporated into an ES approach. More recent research also acknowledges this and advances that recognition of the difference in stakeholder values needs to be taken one step further to understand the factors that contribute to differences among stakeholders, and how these can be compared to inform a more holistic approach to ES policy (Tadaki *et al.*, 2015; Horcea-Milcu *et al.*, 2016). To broach both the information gap and the ideological divide a greater understanding of the processes and consequences is needed which can be translated to a range of audiences. The results in this thesis can help bridge the gap by increasing the understanding of the value of inland fisheries in both literal and ecological terms. It also provides tangible tools such as recognising the importance of non-monetary valuations in ecosystem accounting. The creation of a model in Chapter 4 provides practical data output on inland fisheries which can be used to aid spatial analyses and decision-making, as demonstrated in Chapter 5.

6.3 Further research to advance our understanding

6.3.1 Species richness versus functional diversity

Species richness is the simplest and most commonly used metric for biodiversity (Magurran, 2004), and has been used throughout this thesis due to the availability of comprehensive data. While there is mounting evidence that species richness and diversity can enhance ecosystem functioning, functional traits are a better indicator of ecosystem functioning than a taxonomic approach (Gagic *et al.*, 2015). As such measures of functional diversity are increasingly used in studies of ecosystem functioning (e.g. Vandewalle *et al.*, 2010; Lefcheck & Duffy, 2015; Martins *et al.*, 2015). The vast majority of this evidence comes from plant communities, however the relevance for capturing functional diversity is supported in freshwater studies also: Increased functional diversity improves fish growth above and beyond species richness (Leduc *et al.*, 2015), and the loss of even a few freshwater fish species can result in a substantial loss of functional diversity (Matsuzaki *et al.*, 2013).

The importance of functional diversity in ecosystem functioning not only relies on the traits and unique species contributions, but the interactive effects within species assemblages (Dalerum *et al.*, 2010). No one functional diversity metric is sufficient for capturing all key criteria and a combination of indices is recommended (Villéger *et al.*, 2008; Gagic *et al.*, 2015). The influence of these factors on overall functional diversity of a community are scale dependent (Dalerum *et al.*, 2010) meaning a detailed consideration of the functional attributes of species must be identified for the ecosystem of interest, a difficult task for freshwater communities which are comparatively poorly understood. Being able to identify these traits and interactions and test them with the relationships explored here would be a valuable addition to our knowledge on the explicit contribution of freshwater species to food security and other ecosystem services and functions.

6.3.2 Threats to inland waters and fisheries

One of the main motivations for understanding the processes that underpin ES delivery is to forecast how the benefits may change over time, predominantly as a result of direct or indirect anthropogenic impacts upon the ecosystem. The three predominant threats to freshwater species are habitat loss and degradation, water pollution and over-exploitation (Collen *et al.*, 2014). This also highlights that fisheries must be managed sustainably in order to minimise the threats they themselves cause, however the most pressing threats will vary with the region and scale considered. A valuable undertaking would therefore be to predict how results from this study may be affected in the future. As discussed in Chapter 5, it is now possible to do this for instance by testing the spatial overlap between FFS delivery as derived from Chapter 4, and spatially explicit threats such as water pollution (Mekonnen & Hoekstra, 2015), over-abstraction of water for other services (Vörösmarty *et al.*, 2010) and dams (Mulligan *et al.*, 2009; Winemiller *et al.*, 2016). In this way it would be possible to model areas of potential trade-offs of different ES prioritisation, and where it is most likely to affect fisheries. This might be particularly important to consider for high production fisheries, but also for key fisheries, such as those providing food security to communities with few stable alternatives.

One of the most prominent global threats is climate change, and like all ecosystems inland waters will be affected. Freshwater species are particularly vulnerable as they are restricted within their water bodies and river networks, and are expected to suffer declines, extirpations and extinction (Xenopoulos *et al.*, 2005). The effects of climate change on fish will in turn ultimately lead to a decline in food security for local communities (Weatherdon *et al.*, 2016). There is already a recognition that proactive management strategies are needed, with an emphasis on the removal of other stressors from inland water systems to sustain fisheries (Ficke *et al.*, 2007), and the reduction in water consumption a major factor for prevention of fish extinctions (Xenopoulos *et al.*, 2005). As discussed above, within this thesis biodiversity was defined by species richness but in many cases functional diversity may be more informative for understanding ecosystem processes, particularly at a local scale. By increasing our understanding of the functional traits of freshwater fish (and other taxa) and coupling those traits to species and their biological tolerances, it would be possible not only to predict which species would be affected by climate change (as in Xenopoulos *et al.*, 2005) but also other threats and even multiple stressors, and determine the impact subsequent declines and species losses would have upon fisheries.

6.4 Conclusion

This thesis has focused on the benefits of inland water capture fisheries as a provider of sustenance and nutrition. Wild freshwater species provide all four pillars of food security (FAO, 2006) and livelihoods to many millions worldwide (Richter *et al.*, 2010). This thesis has added to the evidence that these benefits are particularly valuable for those living in poverty. Those most dependent on inland waters for their livelihoods and well-being suffer the most when aquatic ecosystems are modified (Brugere et al., 2015). However it is possible to consider fisheries as a service to these groups specifically. Managing water catchments for the benefit of fisheries need not conflict with the protection of species, indeed it has been made evident here that there are potentially synergistic advantages for freshwater communities and ecosystem service delivery. The gains for people and conservation are a strong argument to raise the agenda for sustainable capture fisheries.

6.5 References

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Appendices

Appendix A Supporting information for Chapter 2

A.1 Site descriptions of the HighARCS projects

A.1.1 Shaoguan, China

The site incorporates three fishing communities along the Beijiang (Pearl River catchment) river, upstream and downstream of the city of Shoaguan in Guangdong Province, southern China. The livelihoods of fishing communities are declining due to the decrease of fishing resources, and the marginalization by policies neglecting the fishing community. Aquatic resources and other ecosystem services are declining mainly due to the impacts of dams, industrial scale sand dredging of the river bed, and water pollution from industrial sources.

A.1.2 Buxa, West Bengal, India

The project site is in the forested hills of the Jalpaiguri District in West Bengal, India and incorporates three villages, all of which are within the Buxa Tiger Reserve (BTR) the core of which is a National Park. Most of the people living within the BTR are poor and rely upon agriculture which is supplemented by animal husbandry, manual labour and the use and selling of non-timber forest products. The site is rich in biodiversity including many globally threatened species and has more than ten rivers, which together supply important ecosystem services to the local communities. As Buxa is within a Forest Reserve (BTR) governance regarding resource use and management is strongly influenced by the Department of Forestry, which often leads to conflicts with local communities use of natural resources.

A.1.3 Dakrong Highland Commune, Quang Tri, Vietnam

The site incorporates three communities along the Dakrong River, in the hills of central Vietnam in Quang Tri Province. Within the three communities it is the poorest households that are more dependent upon aquatic resources as they have more limited access to good agricultural land, and the river provides water and power through micro-generators to all in the villages. However aquatic resources at the site are declining due to the impacts of hydropower dams, deforestation, gold mining and overfishing which is impacting the livelihoods of the communities. The legal framework governing aquatic resources and biodiversity is also complex with a range of overlaps between legislation, policies and institutions, and suffers from a lack of guidance and poor capacity for implementation on the ground.

A.1.4 Phu Yen District, Son La, Vietnam

The site includes a number of communes along the Song Da Reservoir (dammed in 1979) in the mountains of eastern Son La province, northern Viet Nam. Many of these communities are poor whose livelihoods are highly dependent upon fishing and harvesting aquatic resources. However, the aquatic resources and other ecosystem services in this area are declining due to policies driving economic development (including historic and future dam development), intensification of agriculture in the upper catchment, illegal and destructive fishing practices, aquaculture and fisheries development in the reservoir, the introduction of non-native invasive species, and the operation of the Hoa Binh dam being principally for power generation (i.e. with little regard to its knock on affects).

A.2 Site specific ecosystem service selection

Table A1 Aquatic ecosystem services used at each site, and included in site-specific stated preference technique.

Shaoguan	Buxa	Quang Tri	Son La
Agricultural water supply†	Daily water use (humans) †	Commercial fishing/shrimping§	Commercial fishing/shrimping§
Domestic water supply†	Daily water use (animals) †	Subsistence fishing/shrimping§	Subsistence fishing/shrimping§
Industrial water supply†	Rainfall	Daily water use (humans) †	Daily water use (humans) †
Aquatic products	Cleaning pollution	Daily water use (livestock) †	Daily water use (livestock) †
Sand	Flood control	Water for gold mining	Agricultural water supply †
Transportation	Subsistence fishing	Gold mining	Transportation
Hydropower	Commercial fishing	Transportation	Hydropower
Fishing	Sand and stone	Hydropower	Wetland water storage during dry season
Air moisture regulation	Medicinal plants	Wetland water storage during dry season	Flood control
Air temperature regulation	Habitat provision for animals	Flood control	Water purification
Flood control	Tourism	Water purification	Habitat provision for economic species (fish/shrimp)
Disease reduction	Transportation	Habitat provision for animals	Climate regulation
Clean environment	Basic health services	Biodiversity	Biodiversity protection
Cleaning pollution	Education	Tourism	Maintain genetic resources of valuable fish
Biodiversity	Skill based training for livelihoods	Recreation	Tourism
Sailing‡	Research on aquatic resources	Aesthetic value	Recreation
Tourists	Research of renewable energy	Spiritual value	Aesthetic value
Swimming‡	Biodiversity	Research	Spiritual value
Living environment for boat people	Stable environment	Education	Research
Aesthetic value			Education
Spiritual value			
Education			
Research			

†Averaged to produce water supply score, ‡Averaged to produce recreation score' § Averaged to produce fisheries score

Appendix B Supporting information for Chapter 3

B.1 Controlling for effort in Yield

Yield is a factor not only of fisheries productivity but also of human effort and demand. This is often controlled for by either reporting yield per unit area or yield per unit effort (Kantoussan *et al*. 2014). No per unit effort or number of fishers data is comprehensively available at the country level for inland fisheries. As a proxy measure for fisheries effort we used the sum of the population living within 10 km of inland waterbodies for each country. The size of the population surrounding the waterbody would be expected to reflect the reliance upon the waterbody, and thus act as a proxy for fishing effort. In an area with a limited population, yield would be expected to be lower regardless of the amount of fish potentially available.

To test the efficacy of waterside population as a proxy for fishing effort, population within a 5, 10 and 20 km buffer (all log transformed) of 28 African lakes for which the number of fisher men and women could be collated (Henderson & Welcomme, 1974; Vanden Bossche & Bernacsek, 1990; Mölsä et al., 1999; van Zwieten & Njaya, 2003; Weyl, 2003; FAO, 2007; Weyl et al., 2010; Marshall & Mkumbo, 2011) was extracted. Population was extracted from a raster layer of the 2000 global rural population reported by FAO at the 5 arc-minute resolution (Salvatorre et al. 2005) using a buffer around each of the sample lakes. The 10 km buffer was found to have the strongest correlation with the number of fishers (Pearson's $r = 0.75$, $P<0.0001$), compared to the population within a 5km buffer correlation (Pearson's $r = 0.74$, $P<0.0001$) or a 20 km buffer (Pearson's $r = 0.70$, $P<0.0001$). The analysis focused on Africa as: (a) collating fisher data on all lakes globally was not tractable and; (b) Africa has almost half (49) of the countries examined in this study and is a region where inland fisheries are known to be particularly important for rural livelihoods (Dugan *et al*. 2010).

Yield per unit area was not used in these analyses as the surface area of a country's inland waters does not reflect the yield relationship at the country-level; for instance Denmark and Bangladesh contain a similar area of water, yet the yield for Bangladesh far exceeds that of Denmark (Figure B1.A). Figure B1.B shows that population far better correlates with the differences in yield between these two countries. Furthermore, yield per unit area is not regarded as a robust way of controlling for area in this type of ecological analysis (García-Berthou 2001).

Neither surface area nor waterside population are independent as they correlate with each other (Figure B1.C). Similarly, fish species richness also correlates with area (Figure B1.D). There are therefore multiple confounding factors, which are controlled for by including all of them as predictor variables within the multiple regression models. The covariation between variables is likely to reduce the expected size of independent effects, and therefore it is of particular interest to compare the relative importance and effect sizes between variables.

B.2 PCA of climatic variables

It is necessary to control for productivity in the system; however no spatial data of global freshwater productivity currently exists. A surrogate for ambient and productive energy in the system – the two are highly correlated - can be achieved using climatic variables, including water metrics (i.e. precipitation) and seasonality (Hawkins et al., 2003). Climatic variables were derived by performing a principal components analysis (PCA) on a range of climatic data layers. Use of a PCA reduces multidimensionality and eliminates collinearity between variables. Nineteen spatial climatic data layers were accessed from www.worldclim.org, including mean and seasonality for temperature and precipitation variables (Hijmans et al., 2005). Mean values for each of the 19 data layers available were extracted at the country level and used within the PCA (see Table B1). A broken stick model was used as a stopping rule in order to avoid under- or over-estimating the influence of data by including the correct number of non-trivial components (Jackson 1993). These newly derived components were then included as climatic variables within the models. The broken stick stopping model resulted in the retention of the first two principal components for inclusion in further analysis. The eigenvalue of component 1 equalled 10.6, explaining 55.7% of the variance in the data, and component 2 had an eigenvalue of 4.6, explaining 24.1% of the variance. The correlations of the components with each climatic variable are shown in Table B1.

B.3 Variance in yield over time

When examining the link between biodiversity and variation in fisheries yield, CV would not differentiate a difference between a yield which is steadily increasing or decreasing and one which is unstable but fluctuating in similar increments around the mean (see Figure B3). Therefore a variation metric (V) has been adapted which uses differences of year on year yield to calculate the variability of yield over time within the system, where x_i is fisheries yield for a given year and n is the number of years within the study period:

$$V = \frac{\sqrt{\sum(x_{i+1} - x_i)^2}}{n-1} \div \overline{x}$$

Total fishing yield was extracted per country for each of the years 1981-2010. Records prior to 1981 were excluded due to the higher chance of inaccuracies and extrapolated figures with older data. The variability of yield (t) was calculated for each country for decadal increments from the years 1981 – 2010, and the mean was compared with linear regression to fish SR per country. As before, Box-Cox methodology was used to determine the most appropriate transformation. Analysis was repeated for country data subset by continent for comprehensive datasets; Africa and Europe. A comparison of CV and adjusted variability (V) results is shown in Table B5.

B.4 Relationship with species richness considering multiple groups

A subset of the main analyses detailed in the main text (Figure 3.2, Table 3.1 and Table B2) were repeated to incorporate the species richness of other freshwater taxonomic groups. Correlations between spatial patterns of fish SR and SR of the other freshwater faunal groups at the country level were examined using Spearman's rho due to non-normality of the data, with corrected degrees of freedom calculated using Dutilleul's modified test to account for spatial autocorrelation.

When SR was expanded to include odonates, molluscs and decapods, results were largely concordant with the fish only results, with a 1.35-1.45 increase in AICc (Figure B4 and Tables B3-B4). The strength of relationship between CV and multiple freshwater taxa species richness is equal to that with fish species richness (Figure B5). However, in both of these analyses it is not possible to disentangle the effect of overall SR from that of fish richness as there is a strong correlation between fish SR and SR of the other freshwater faunal groups at the country level ($r_{17.15} = 0.88$, $P < 0.0001$).

B.5 Supplementary Figures

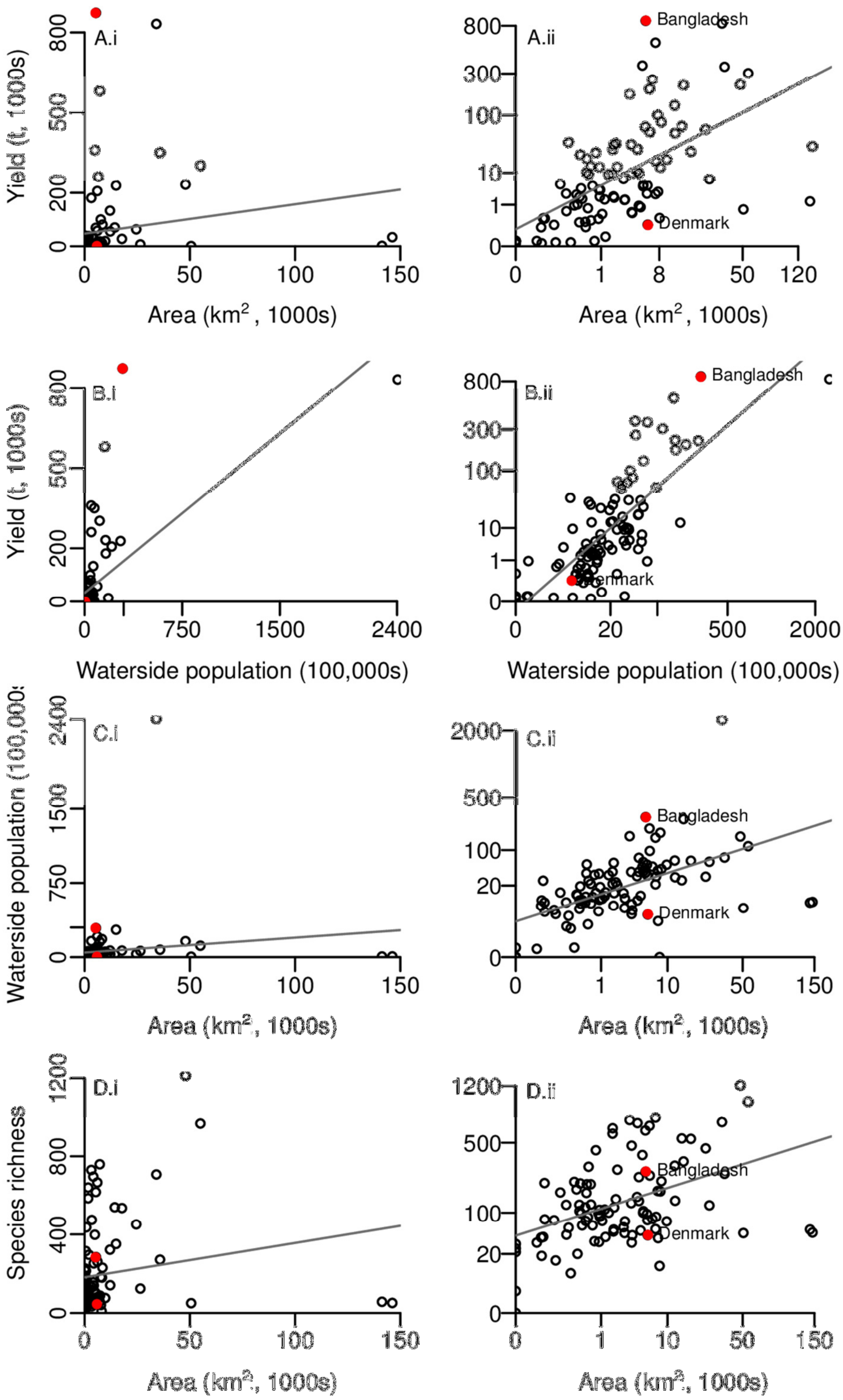

Figure B1 Relationship between confounding model variables at the country level (N=100), (i) raw data and (ii) transformed data: A) Yield ~ Inland water surface area (cor = 0.59, *P*<0.0001); B) Yield ~ Mean population (cor = 0.70, *P*<0.0001); C) Inland water surface area ~ Mean population (cor = 0.53, *P*<0.0001), and; D) Area ~ Fish species richness (cor = 0.44, *P*<0.0001). Yield, area and population are quarter root transformed, fish species richness is cubic root transformed. Red dots indicate data from Bangladesh and Denmark for comparison.

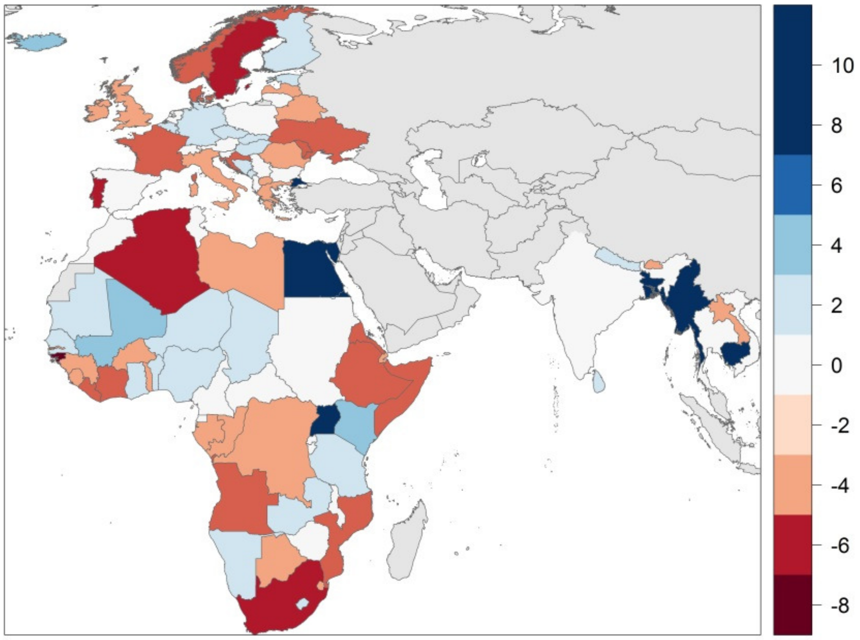

Figure B2 Residual difference between the observed and expected values from the full spatial simultaneous autoregressive model predicting inland water capture fishery yields per country. Blue countries indicate where FAO reported yield is higher than expected from the model, red countries are where yield is lower than expected. Full model as calculated in SAR_{err}: $\sqrt[4]{Yield} = \sqrt[3]{SR} + \sqrt[4]{P} + C1 + C2 + \sqrt[4]{A} + \sqrt[3]{E}$. SR = species richness of fishes, P = human population living within 10km of inland waterbodies, C1 = first principal component of climatic variables, C2 = second principal component of climatic variables, A = inland water area in km^2, E = mean elevation (m).

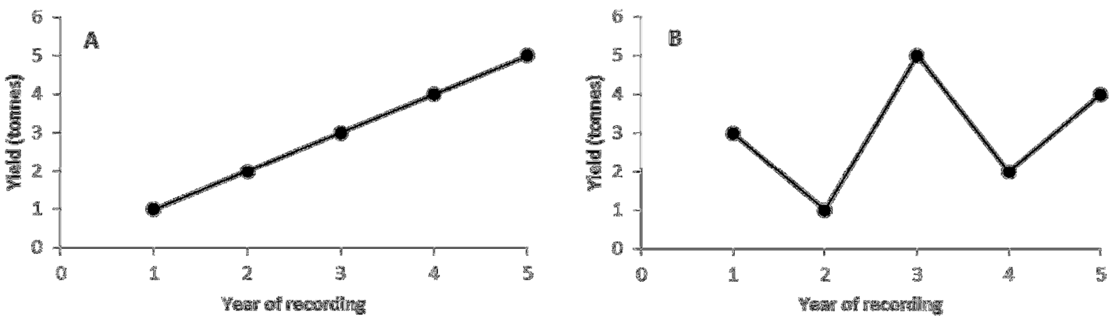

Figure B3 Hypothetical fluctuations in yield showing that a steady increase in yield over time has the same coefficient of variation as a yield with more fluctuations, but a lower variability in year-to-year differences (V). The mean and CV for A and B are the same at 3 and 0.83 respectively. The adapted variability measure (V) is greater in B (V = 0.48) than in A (V = 0.17).

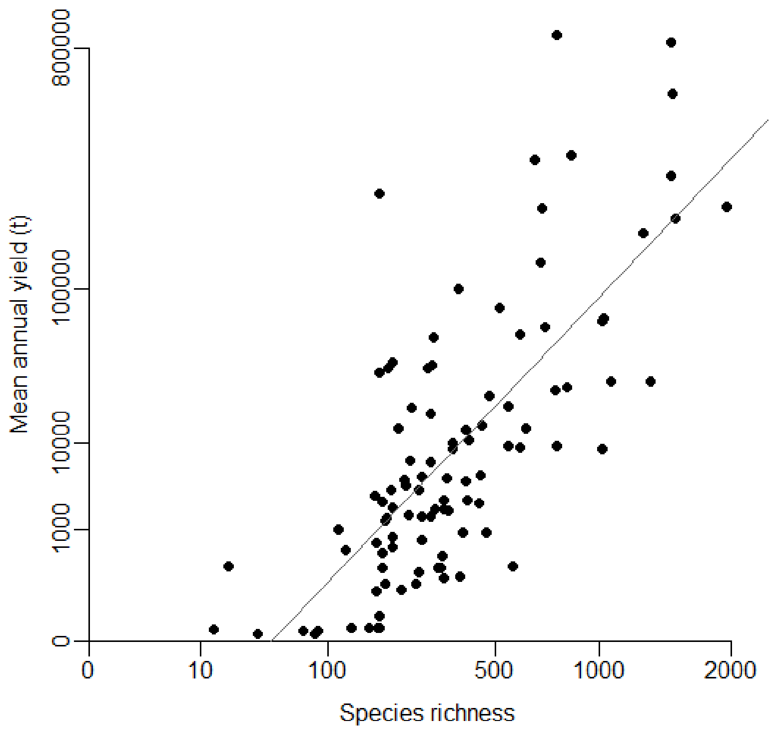

Figure B4 Relationship between inland water capture fisheries yield (t) (axis quarter root transformed) and freshwater species richness (axis cubic root transformed), R^2=0.54, F= 116.8, df=98, P<0.001.

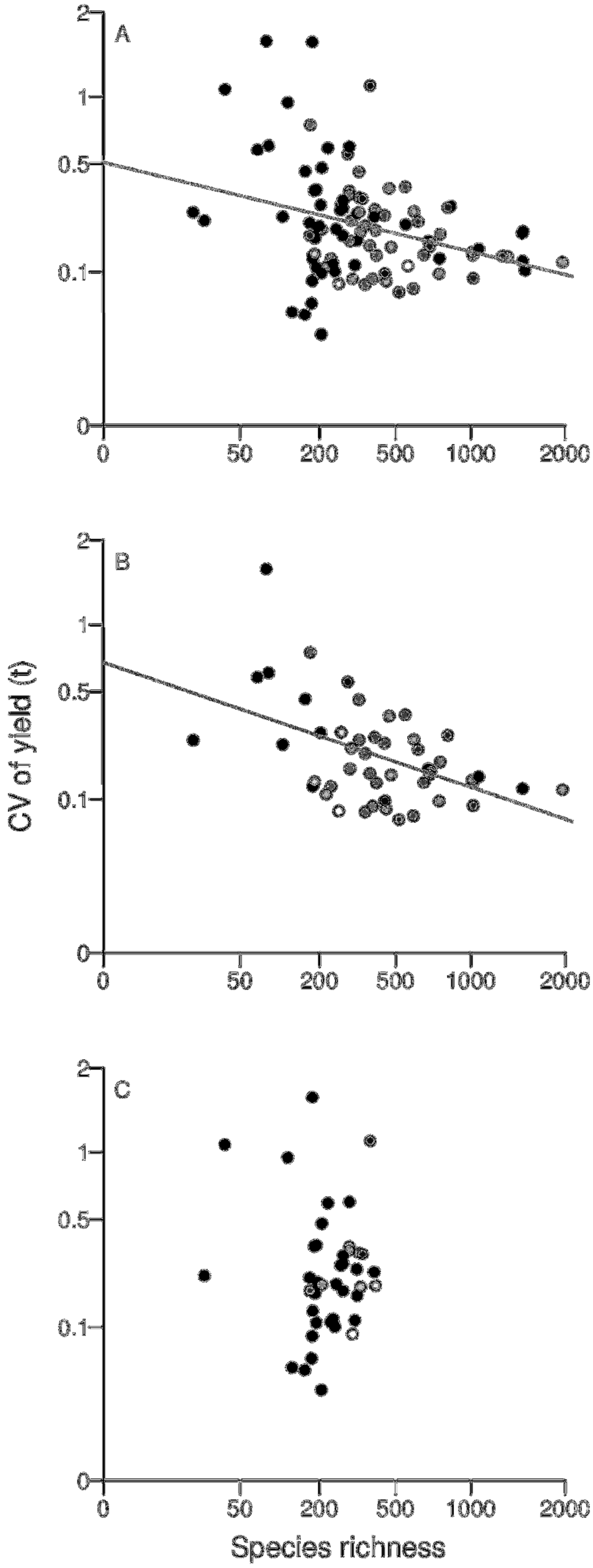

Figure B5 Relationship between freshwater species richness and mean coefficient of variation of yield (kg) per capita (both cubic root transformed). A) All countries within boundaries of this study, R^2=0.03, *F*=3.66, *df*=98, *P*=0.06. B) African countries, R^2=0.16, *F*=9.68, *df*=46, *P*=0.003. C) European countries, R^2=-0.03, *F*=0.01, *df*=39, *P*=0.92. Proportion of FAO country data that has been estimated or extrapolated by FAO is graded from white (all years estimated) to black (all actual data).

B.6 Supplementary Tables

Table B1 Principal Component Analysis of country climatic variables. The first two PCA axes explained 56% and 24% of the total variability in climate conditions, respectively, and were retained as predictors for inland water fisheries yield models**.**

Variable	Comp 1	Comp 2	Comp 3	Comp 4	Comp 5
Annual mean temp.	0.98	-0.09	-0.09	0.03	0.13
Mean diurnal range*	0.65	-0.56	0.08	0.04	-0.08
Isothermality†	0.87	0.20	-0.20	-0.27	-0.09
Temp. seasonality‡	-0.85	-0.37	0.20	0.29	0.11
Max temp. of warmest month	0.86	-0.40	-0.06	0.23	0.19
Min temp. of coldest month	0.95	0.14	-0.23	-0.08	0.09
Temp. annual range§	-0.56	-0.66	0.29	0.38	0.07
Mean temp. of wettest quarter	0.87	-0.13	0.20	-0.02	0.24
Mean temp. of driest quarter	0.89	-0.08	-0.30	0.09	0.01
Mean temp. of warmest quarter	0.89	-0.32	-0.05	0.20	0.23
Mean temp. of coldest quarter	0.98	0.05	-0.14	-0.06	0.05
Annual precipitation	0.29	0.93	0.13	0.15	0.02
Precip. of wettest month	0.56	0.72	0.32	0.20	-0.06
Precip. of driest month	-0.77	0.43	-0.27	-0.01	0.28
Precip. seasonality‖	0.84	-0.25	0.28	0.03	-0.11
Precip. of wettest quarter	0.53	0.73	0.34	0.22	-0.07
Precip. of driest quarter	-0.69	0.55	-0.30	-0.01	0.30
Precip. of warmest quarter	0.09	0.72	0.58	-0.16	0.16
Precip. of coldest quarter	0.06	0.59	-0.54	0.50	-0.19
Variability explained	55.73%	24.07%	7.81%	4.26%	2.40%

* Mean diurnal range = (Mean of monthly (max temp-min temp))
† Isothermality = Mean diurnal range/Temp annual range*100
‡ Temp seasonality = Standard deviation*100
§Temp annual range = Max temp of warmest month-min temp of coldest month
|| Precip. seasonality = coefficient of variation

Table B2 Multimodel averaged parameter estimates of top model set of SAR models of country-level inland water fisheries yield (t). Abbreviations as in Table 3.1.

Parameter	Estimate	SE	z	95% Lower CI	95% Upper CI	Relative importance
(Intercept)	-1.05	1.61	0.65	-4.21	2.11	NA
SR	1.07	0.32	3.36	0.45	1.69	1.00
P	0.13	0.03	4.12	0.07	0.18	1.00
C1	0.40	0.18	2.26	0.05	0.75	1.00
C2	0.15	0.19	0.78	-0.23	0.53	0.28
A	0.49	0.13	3.76	0.24	0.75	1.00
E	-0.37	0.17	2.12	-0.71	-0.03	1.00

Table B3 SAR models of country-level inland water fisheries yield (t) (quarter root transformed) of the two best fitting models. Shaded cells indicate which of the biodiversity, climatic and geographic variables were included in the model. SR = species richness of freshwater taxa (cubic root transformed), P = human population living within 10km of inland waterbodies (quarter root transformed), C1 = first principal component of climatic variables, C2 = second principal component of climatic variables, A = inland water area in km^2 (quarter root transformed), E = mean elevation (m) (cubic root transformed). ΔAICc = difference between the AICc of each model and that of the best model, Wi = Akaike weights. Full model as calculated in SAR_{err}: $\sqrt[4]{Yield} = \sqrt[3]{SR} + \sqrt[4]{P} + C1 + C2 + \sqrt[4]{A} + \sqrt[3]{E}$

SR	P	C1	C2	A	E	Pseudo R^2	AIC_c	ΔAIC_c	W_i
■	■	■		■	■	0.76	545.42	0	0.44
■	■	■	■	■	■	0.76	547.24	1.82	0.18

Table B4 Multimodel averaged parameter estimates of top model set of SAR models of country-level inland water fisheries yield (t). Abbreviations as in Table B3.

Parameter	Estimate	SE	z	95% Lower CI	95% Upper CI	Relative importance
(Intercept)	-3.45	1.85	1.87	-7.07	0.16	NA
SR	1.13	0.31	3.60	0.52	1.75	1.00
P	0.12	0.03	3.73	0.06	0.18	1.00
C1	0.45	0.16	2.82	0.14	0.76	1.00
C2	0.15	0.19	0.80	-0.22	0.53	0.29
A	0.52	0.12	4.24	0.28	0.77	1.00
E	-0.37	0.17	2.18	-0.71	-0.04	1.00

Table B5 Comparison of variation in yield against species richness metrics. SR = Fish species richness (cube root transformed), CV = Coefficient of variation (cube root transformed), Y = Yield (t), V = Adapted variability measure (see above for details, cube root transformed). Significant relationships in bold.

Model	All countries				Africa				Europe			
	R^2	F	df	P	R^2	F	df	P	R^2	F	df	P
CV of Y ~ SR	0.02	3.14	98	0.08	**0.16**	**9.81**	**46**	**0.003**	-0.02	0.21	39	0.65
V of Y ~ SR	**0.04**	**5.66**	**98**	**0.02**	**0.14**	**8.71**	**46**	**0.005**	-0.02	0.07	39	0.79

Table B6 Species breakdown of FAO recorded freshwater fish catch harvest per country in 2010 for the countries where this data exists. Percentage of yield (t) produced by number of species, e.g. 90.24% of Denmark's total yield comes from just five fish species, and the total amount harvested from 10 species or fewer. Where yield has not been identified down to species level it is either categorised as grouped (e.g. identified as torpedo-shaped catfishes), or unidentified (identified as freshwater fishes not elsewhere included). Countries are ordered by yield (t), lowest to highest. Data from FAO (2011). Due to the numerical and categorical nature of the data they cannot be statistically analysed, however it is apparent that where catch has been identified down to species level, five species or fewer account for a large proportion of the yield for the majority of countries, and that instances of six or more species identified to contribute significantly to total country yield do not increase as yield (t) increases.

Country	1-5 sp.	6-10 sp.	11+ sp.	Grouped yield	Unidentified
Denmark	90.24%	9.76%	0.00%	0.00%	0.00%
Lesotho	44.44%	0.00%	0.00%	0.00%	55.56%
Botswana	0.00%	0.00%	0.00%	98.33%	1.67%
Ireland	100.00%	0.00%	0.00%	0.00%	0.00%
Slovenia	73.37%	15.98%	7.10%	3.55%	0.00%
Macedonia	53.39%	3.81%	0.00%	40.25%	2.54%
Iceland	100.00%	0.00%	0.00%	0.00%	0.00%
Latvia	79.33%	17.02%	1.22%	0.00%	2.43%
Croatia	87.72%	10.96%	1.32%	0.00%	0.00%
Belgium	73.39%	10.76%	0.00%	15.85%	0.00%
Montenegro	44.01%	0.00%	0.00%	40.45%	15.54%
Norway	100.00%	0.00%	0.00%	0.00%	0.00%
Belarus	62.21%	12.37%	0.22%	25.08%	0.11%
Greece	68.12%	4.13%	0.22%	8.27%	19.26%
Bulgaria	89.20%	6.46%	4.34%	0.00%	0.00%
Sweden	84.21%	15.79%	0.00%	0.00%	0.00%
Lithuania	79.92%	12.37%	4.35%	2.90%	0.46%
Slovakia	85.01%	6.90%	2.18%	5.91%	0.00%

Country	1-5 sp.	6-10 sp.	11+ sp.	Grouped yield	Unidentified
Switzerland	31.40%	1.33%	0.00%	67.15%	0.12%
Netherlands	89.76%	0.00%	0.00%	3.41%	6.83%
Romania	75.95%	12.86%	3.58%	7.45%	0.16%
United Kingdom	93.93%	0.00%	0.00%	6.07%	0.00%
France	53.60%	0.00%	0.00%	4.40%	42.00%
Estonia	94.70%	4.60%	0.66%	0.00%	0.03%
Albania	63.70%	8.60%	0.00%	11.40%	16.30%
Italy	1.14%	0.00%	0.00%	27.41%	71.44%
Czech Republic	91.53%	5.19%	2.03%	0.00%	1.25%
Gambia	0.00%	0.00%	0.00%	25.91%	74.09%
Ukraine	91.55%	5.87%	1.65%	0.94%	0.00%
Spain	41.11%	0.00%	0.00%	0.00%	58.89%
Serbia	47.78%	13.19%	4.41%	0.00%	34.62%
Togo	0.00%	0.00%	0.00%	80.00%	20.00%
Hungary	70.01%	6.11%	0.00%	17.91%	5.97%
Morocco	0.36%	0.00%	0.00%	85.80%	13.84%
Gabon	0.00%	0.00%	0.00%	49.95%	50.05%
Zimbabwe	75.24%	0.00%	0.00%	9.52%	15.24%
Rwanda	100.00%	0.00%	0.00%	0.00%	0.00%
Burkina Faso	0.00%	0.00%	0.00%	65.43%	34.57%
Germany	5.16%	1.48%	0.24%	13.15%	79.96%
Burundi	96.25%	1.37%	0.12%	0.00%	2.27%
Ethiopia	18.55%	0.00%	0.00%	78.29%	3.16%
Poland	9.28%	2.99%	1.16%	0.00%	86.57%
Benin	3.82%	0.00%	0.00%	80.38%	15.80%
Finland	82.36%	12.96%	2.20%	0.00%	2.48%
Laos	0.00%	0.00%	0.00%	15.86%	84.14%
Senegal	25.29%	1.55%	0.12%	13.83%	59.21%
Turkey	74.76%	4.50%	0.25%	19.88%	0.61%
Niger	19.25%	0.00%	0.00%	65.75%	15.00%
Mozambique	28.84%	0.00%	0.00%	0.00%	71.16%
Sri Lanka	0.00%	0.00%	0.00%	53.90%	46.10%
Sudan	62.00%	0.00%	0.00%	0.00%	38.00%
Zambia	10.24%	0.00%	0.00%	0.00%	89.76%
Malawi	0.00%	0.00%	0.00%	95.29%	4.71%
Mali	62.15%	0.00%	0.00%	19.00%	18.85%

Country	1-5 sp.	6-10 sp.	11+ sp.	Grouped yield	Unidentified
Kenya	85.26%	0.00%	0.00%	12.10%	2.64%
Thailand	55.56%	0.30%	0.00%	9.96%	34.19%
Egypt	77.85%	1.16%	0.00%	15.14%	5.84%
Tanzania	70.91%	0.00%	0.00%	23.61%	5.48%
Nigeria	27.82%	1.17%	0.00%	67.38%	3.63%
Uganda	44.47%	0.00%	0.00%	54.83%	0.70%
Bangladesh	11.70%	0.00%	0.00%	0.00%	88.30%
India	0.62%	0.00%	0.00%	62.68%	36.71%

B.7 Data sources and supporting references

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Appendix C Supporting information for Chapter 4

C.1 Supplementary figures

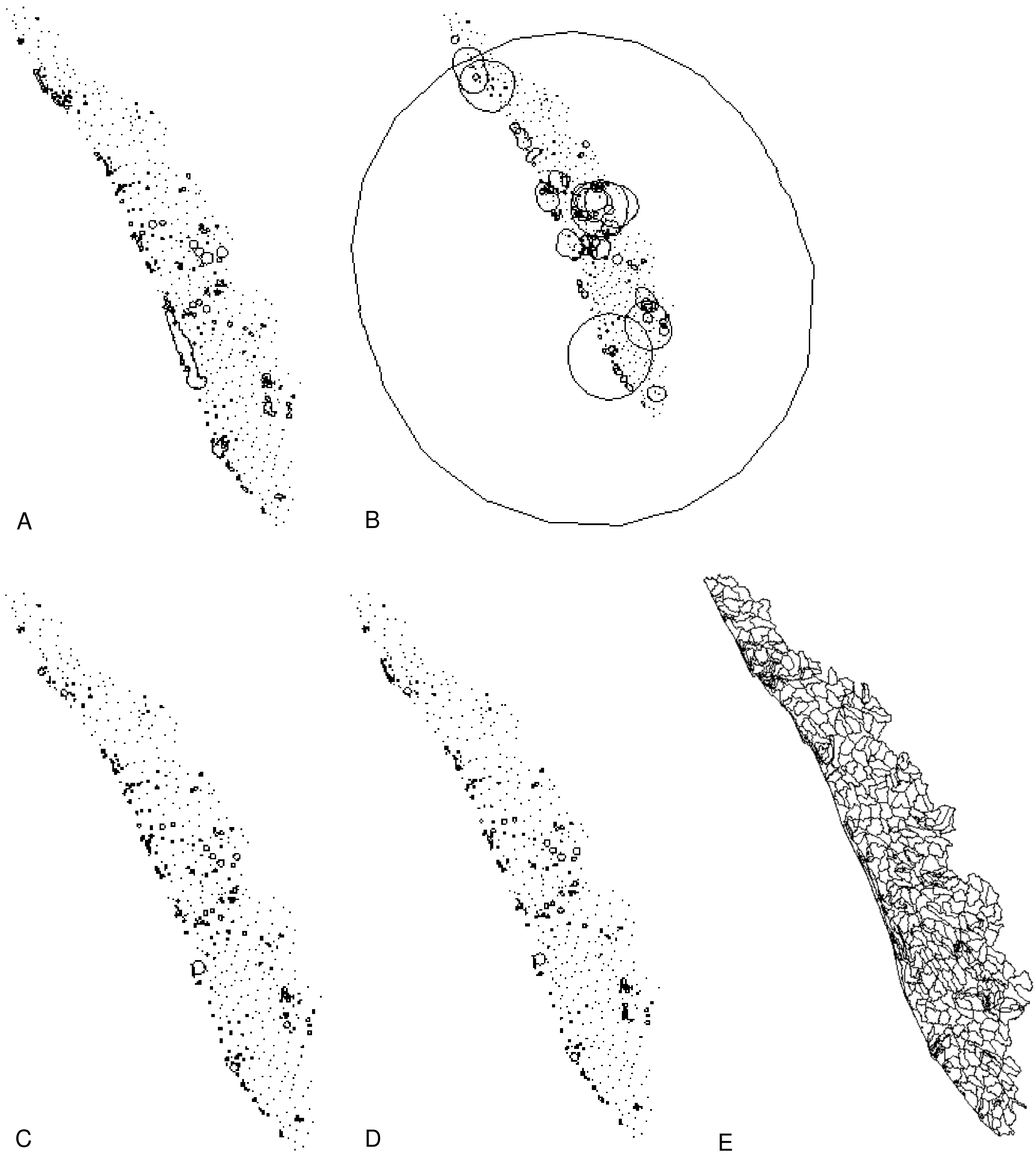
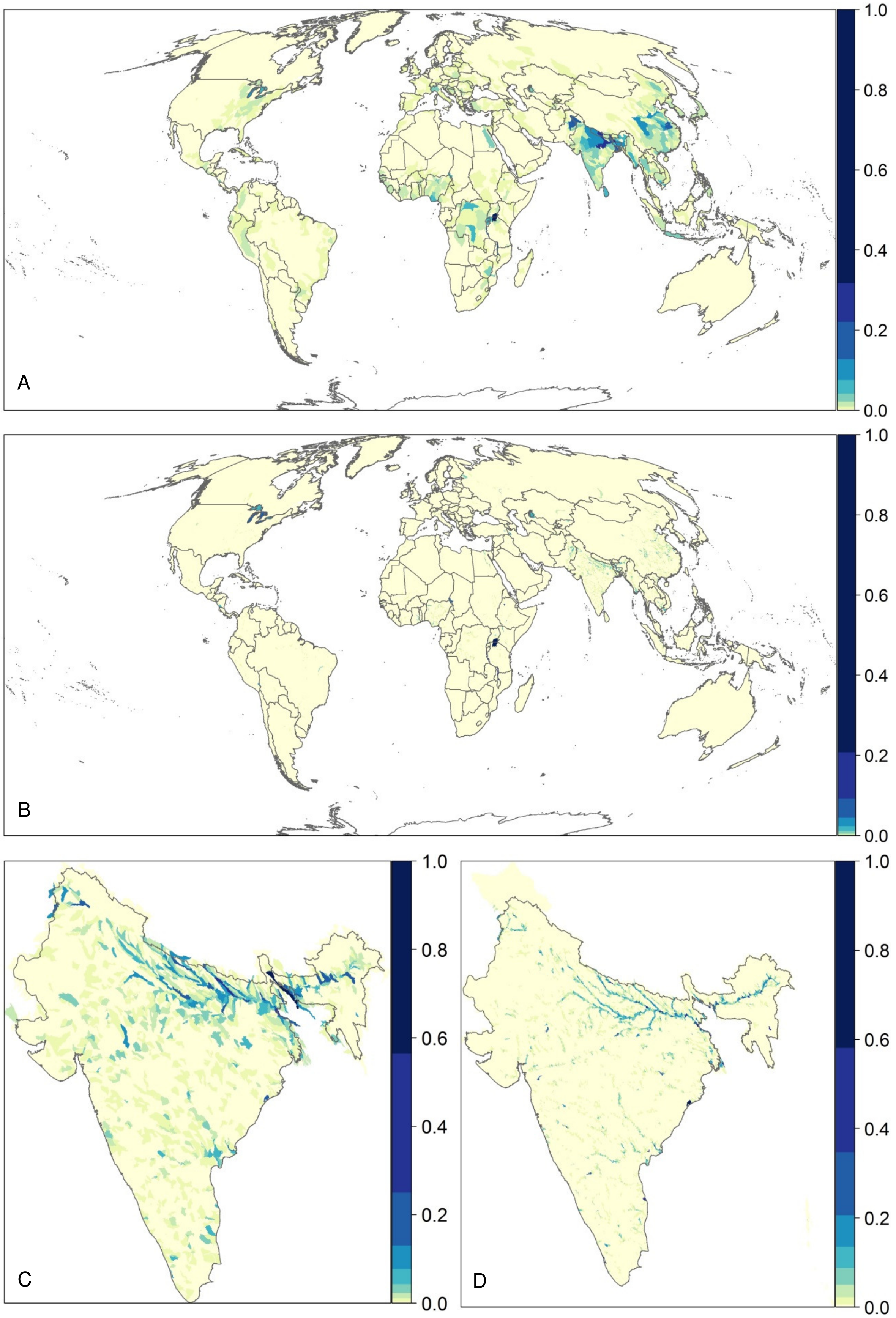


Figure C1 Rescaled level 12 HydroBASINS for Kerala State, India, based on different buffer algorithms. A) square root of inland water surface area x 1000, B) area × 1000, c) cube root area x 1000, D) log(area+1) x 1000, and E) original level 12 HydroBASINS.

Appendix C



1.0
0.8
0.6
0.4
0.2
0.0
A
1.0
0.8
0.6
0.4
0.2
0.0
B
1.0
0.8
0.6
0.4
0.2
0.0
C
1.0
0.8
0.6
0.4
0.2
0.0
D

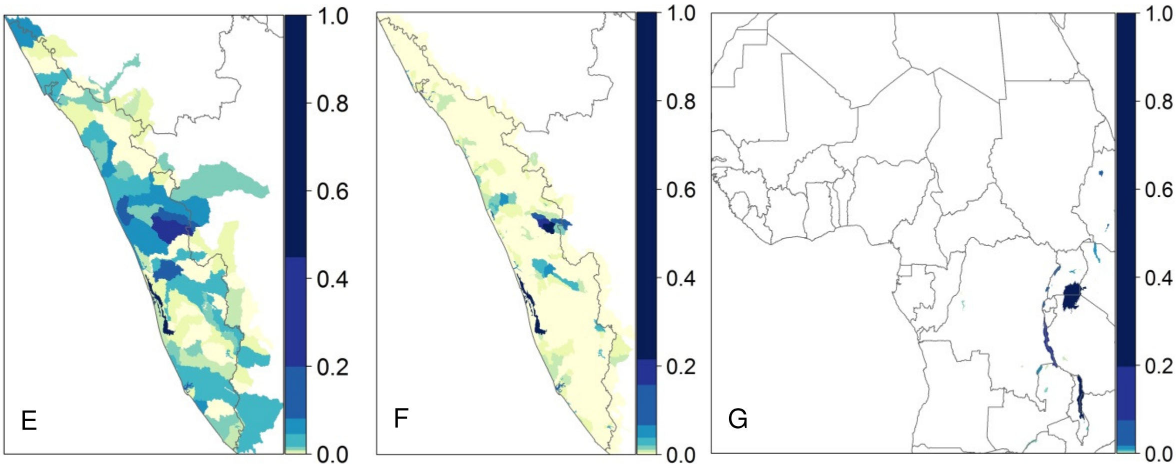


Figure C2 Importance of freshwater food security (FFS) based on population demand data only. A) Global data, using level 5 HydroBASINS sub-catchments. B) Global, level 8 HydroBASINS. c) India, level 8 HydroBASINS. D) India, level 12 HydroBASINS. E) Kerala, level 8 HydroBASINS. F) Kerala, level 12 HydroBASINS. G) African lakes from GLWD. Note that values are relative and therefore maps are not directly comparable.

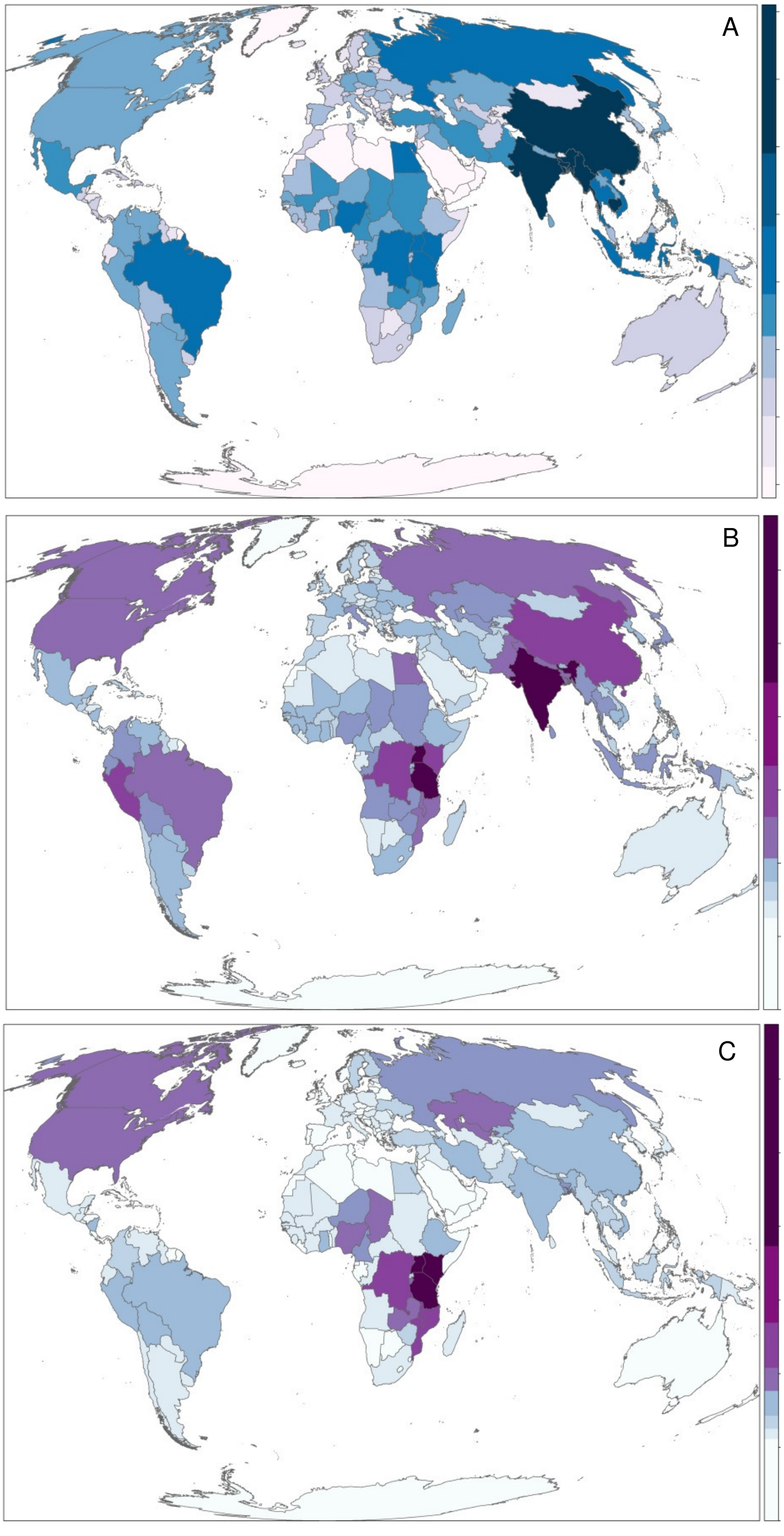


Figure C3 A) Global reported inland waters fishery Yield (from FAO 2014a), B) Level 5 HyrdoBASINS FFS scores aggregated to country level

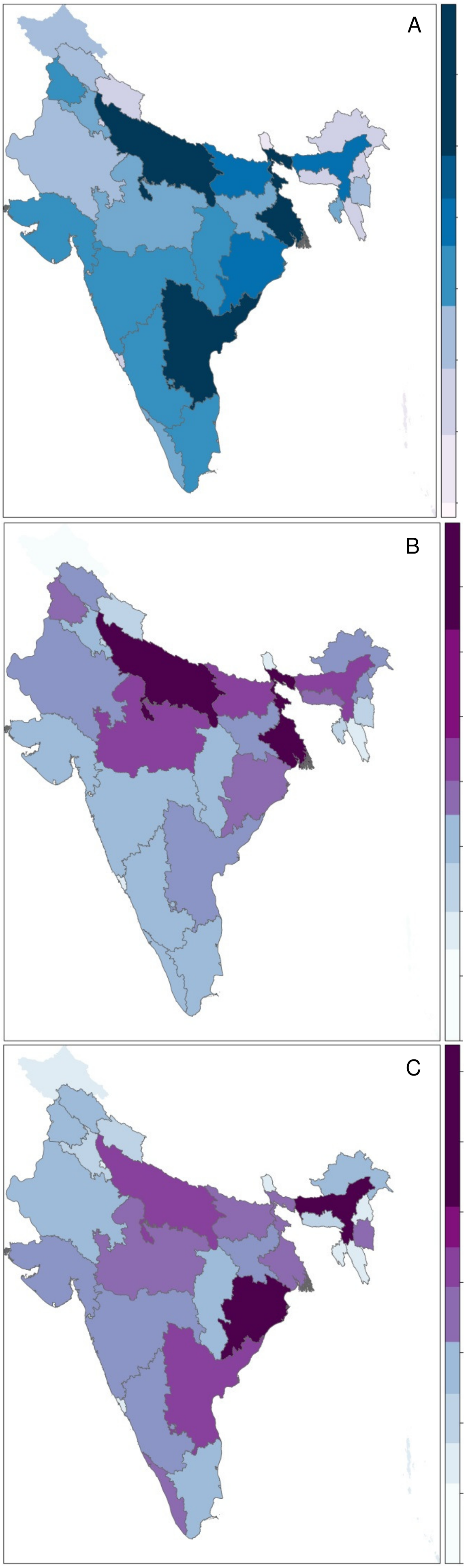


Figure C4 A) reported inland waters fishery Yield per Indian state (Latha, 2010), B) Level 8 HyrdoBASINS FFS scores aggregated to state level c) Level 12 HyrdoBASINS FFS scores aggregated to state level. Low (pale) to high (dark).

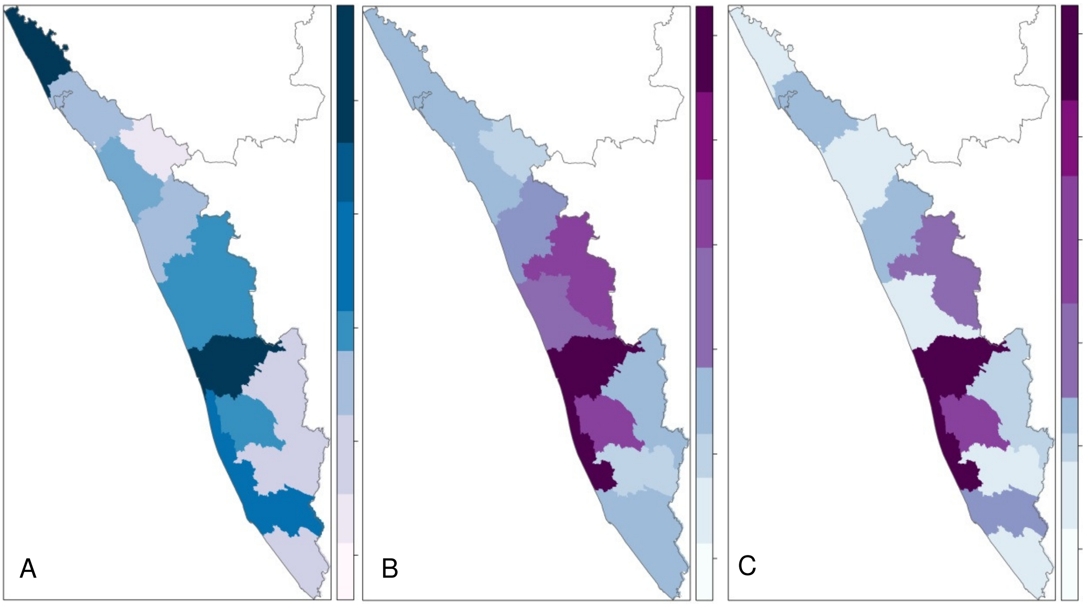


Figure C5 A) Reported inland waters fishery yield per Keralan district (Latha, 2010), B) Level 8 HyrdoBASINS FFS scores aggregated to district level c) Level 12 HyrdoBASINS FFS scores aggregated to district level. Low (pale) to high (dark).

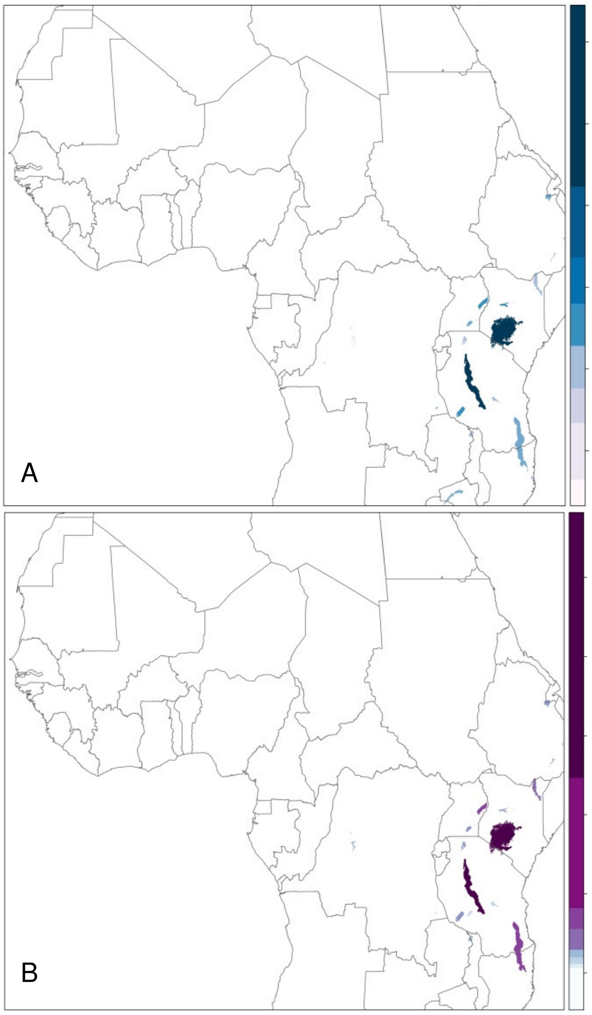


Figure C6 A) Reported inland waters fishery yield for 33 African lakes (see Table C1), B) FFS scores per lake. Low (pale) to high (dark).

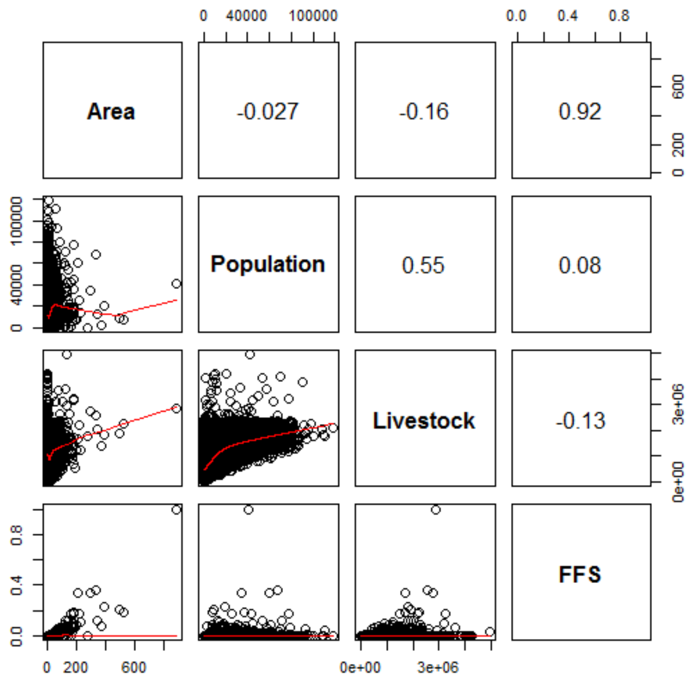


Figure C7 Correlation matrix for FFS model inputs and outputs, calculated at the HydroBASINS level 12 for India. Red line depicts a non-parametric smoother of the data. All correlations *p*<0.0001.

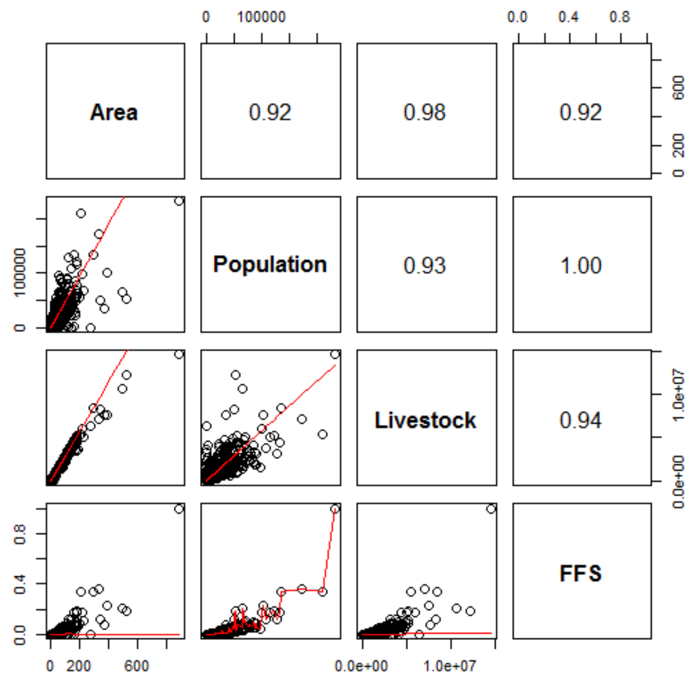


Figure C8 Correlation of FFS model inputs and outputs, calculated at the rescaled sub-catchment level for India. Red line depicts a non-parametric smoother of the data. All correlations *p*<0.0001.

C.2 Supplementary tables

Table C1 Reported fishery yields from 33 African lakes, and their source. Listed in size (surface area) order.

Lake	Fish yield (t)	Source
Victoria	955,000	(Marshall & Mkumbo, 2011)
Tanganyika	200,000	(Mölsä *et al.*, 1999)
Malawi	33,000	(Thompson & Allison, 1997; Weyl *et al.*, 2010)
Turkana	9,069	(FAO, 2007)
Albert	38,240	(von Sarnowski, 2004; FAO, 2009b)
Mweru	55,000	(van Zwieten & Njaya, 2003)
Tana	24,900	(Jul-Larsen *et al.*, 2003; FAO, 2009b)
Kivu	9,500	(Vanden Bossche & Bernacsek, 1990; FAO, 2009b)
Edward	16,900	(FAO, 2004, 2009b)
Bangweulu	15,000	(Kolding, 2011)
Rukwa	5,990	(Vanden Bossche & Bernacsek, 1990)
Mai-Ndombe	1,000	(FAO, 2009b)
Kyoga	80,000	(Allison, 2003)
Chisi	47	(Vanden Bossche & Bernacsek, 1990)
Abaya	2,000	(Lewis, 1988)
Tumba	443	(Henderson & Welcomme, 1974)
Chilwa	15,500	(Vanden Bossche & Bernacsek, 1990)
Upemba	12,000	(Henderson & Welcomme, 1974)
Ziway	3,180	(Reyntjens & Wudneh, 1998)
Pool Malebo	3,500	(FAO, 2009b)
Ch'Amo	3,464	(Reyntjens & Wudneh, 1998)
Malombe	10,300	(FAO, 1993)
George	4,350	(Vanden Bossche & Bernacsek, 1990)
Nzilo	2,800	(Fernando & Holčík, 1982)
Guiers	2,250	(Fernando & Holčík, 1982)
Bisina	984	(Vanden Bossche & Bernacsek, 1990)
Langano	1,000	(Vanden Bossche & Bernacsek, 1991)
Baringo	63	(FAO, 2007)
Nokoue	1,331	(FAO, 2008)
Naivasha	108	(FAO, 2007)
Chiuta	1,400	(Vanden Bossche & Bernacsek, 1990)

Table C2 Comparison of inland water surface area metrics. Degrees of freedom (df) corrected for spatial autocorrelation.

Region	HydroBASINS level	$lakes\ 1{:}3\ rivers$		$lakes\ 1{:}1\ rivers$	
		rho	df	rho	df
Global	5	0.80	85.24	0.79	93.80
Global	8	0.78	88.07	0.74	101.67
India	12	0.76	19.94	0.82	20.66
India	8	0.86	21.55	0.80	23.32
Kerala	12	0.84	4.90	0.76	5.34
Kerala	8	0.71	5.41	0.77	5.39
African Lakes	GLWD	0.67	29.91	0.67	29.91

Table C3 Comparison of Spearman's rho for different buffer algorithms within FFS model at different regional scales and sub-catchment resolutions. Degrees of freedom (df) corrected for spatial autocorrelation.

Region	HydroBASINS level	$\sqrt{area} \times 1000$		$area \times 1000$		$\sqrt[3]{area} \times 1000$		$\log(area + 1) \times 1000$		No buffer	
		rho	df	rho	df	rho	df	rho	df	rho	df
Global	5	0.80	85.24	0.72	92.76	0.78	95.54	0.78	96.30	0.69	90.28
Global	8	0.78	88.07	0.63	73.80	0.81	87.26	0.80	93.06	0.59	96.16
India	12	0.76	19.94	0.72	21.14	0.80	20.98	0.85	20.27	0.84	20.37
India	8	0.86	21.55	0.83	22.48	0.87	21.04	0.89	20.57	0.87	20.22
Kerala	12	0.84	4.90	0.87	4.57	0.79	4.81	0.74	4.81	0.27	10.60
Kerala	8	0.71	5.41	0.74	5.33	0.67	6.13	0.71	5.67	0.29	11.32
African Lakes	GLWD	0.67	29.91	0.46	31.02	0.12	31.14	-0.26	31.65	0.33	35.63

Table C4 Comparison of Spearman's rho for different input layer options within FFS model at different regional scales and sub-catchment resolutions. Degrees of freedom (df) corrected for spatial autocorrelation.

Region	HydroBASINS level	$population \times livestock$		$population$		$livestock$		$\frac{population + livestock}{2}$	
		rho	df	rho	df	rho	df	rho	df
Global	5	0.80	85.24	0.82	91.87	0.53	26.18	0.68	84.76
Global	8	0.78	88.07	0.8	85.95	0.64	82.27	0.67	78.46
India	12	0.76	19.94	0.86	20.45	0.82	21.12	0.82	20.62
India	8	0.86	21.55	0.91	20.33	0.85	21.51	0.89	20.81
Kerala	12	0.84	4.90	0.77	6.07	0.77	6.07	0.71	5.10
Kerala	8	0.71	5.41	0.67	6.63	0.67	5.93	0.68	6.31
African Lakes	GLWD	0.67	29.91	0.7	29.91	0.67	26.46	0.69	27.07

Table C5 Normalised variance of differences in rank between fisheries catch data and summed FFS scores per administrative boundary.

Region	HydroBASINS level	Variance
Global	5	0.19
Global	8	0.30
India	12	0.26
India	8	0.25
Kerala	12	0.21
Kerala	8	0.20
African Lakes	GLWD	0.28

C.3 R code for Freshwater Food Security (FFS) model

```
##Import data
library(rgdal)
basinspoly<-      #Basin polygon shapefile in projected coordinate system
row.names(basinspoly)<-as.character(basinspoly$basin_ID) #Define row.names by
unique ID per basin
FWpoly<-      #Inland freshwaters shapefile in projected coordinate system,
overlaid by basins layer and including column of area

library(raster)
pop<-      #Population raster layer in same coordinate system as basin
shapefile
livestock<-     #Livestock raster layer in same coordinate system as basin
shapefile

###1 SUPPLY - Determine extent of influence of each basin by creating buffers

##Calculate water area per basin
#Sum area per basin from FWpoly:
bybasin<-data.frame(basinspoly)
FWtmp<-data.frame(FWpoly)
#Rivers are 3x more productive
FWtmp$H2O_Area[FWtmp$TYPE == "River"]<-FWtmp$H2O_Area[FWtmp$TYPE == "River"]*3
temp<-aggregate(FWtmp$H2O_Area,by=list(FWtmp$basin_ID),"sum") #Temporary file
with summed surface areas of water (from FWpoly) per basin
colnames(temp)<-c("basin_ID","FWarea") #Renames columns
bybasin<-merge(bybasin,temp,all.x=T)
bybasin$FWarea[is.na(bybasin$FWarea)] <- 0 #Changes NAs to 0s

#Turn back to spatial
row.names(bybasin)<-as.character(bybasin$basin_ID) #Change rownames to match
bybasinDF<-data.frame(bybasin)
bybasin<-SpatialPolygonsDataFrame(as(basinspoly,"SpatialPolygons"),data=bybasi
n) #Coerce dataframe back to shapefile

#Calculate initial buffwidth
bybasin$buffwid<-(sqrt(bybasin$FWarea)*1000) #Buffers based on sqrt of area of
GLWD
bybasin$buffwid[(bybasin$buffwid)==0] <- 0.001 #Value must be greater than 0
bybasin$buffwid[is.nan(bybasin$buffwid)]<-0.001 #Remove NaNs
bybasinDF<-data.frame(bybasin)

#Create buffer area parameters
#centroid buffer:
library(rgeos)
HBcent<-gCentroid(bybasin,byid=T,id=basinspoly$basin_ID) #Calculates centoid
Cbuff<-gBuffer(HBcent, byid=T,width=bybasin$buffwid)
Cbuffkm<-gArea(Cbuff,byid=T) #Calculate area (minusing holes) per buffer
Cbuffkm<-data.frame(Cbuffkm) #Convert to dataframe
Cbuffkm<-Cbuffkm/1000000 #Convert from m to km
bybasinDF<-merge(bybasinDF,Cbuffkm,all.x=T,by=0) #Attach output to bybasin data
frame
row.names(bybasinDF)<-bybasinDF$basin_ID
bybasin<-SpatialPolygonsDataFrame(as(basinspoly,"SpatialPolygons"),data=bybasin
DF) #Coerce dataframe back to shapefile

##Calculate new buffers so that they are the area of Cbuffs but in
correct polygon shape of the basin
bufferby <- function(spdf,ID){
```

```
  library(rgeos)
  precision <- 1e-6
  maxsteps <- 1000
  #Create empty buffer list based on spdf
  buffs<-vector("list",length(spdf))
  #For every item in SpatialPolygonDataFrame (spdf),
  for(i in seq_len(length(spdf))){
    #Loop buffer each polygon individually based on widths, and add to 'list'
    if(spdf[i,]$Cbuffkm>spdf[i,]$HB_Area){
      spgeom<-spdf[i,]
      spgeom.area <- spgeom$HB_Area*1000000 #Area of basin
      achieved.precision <- 1
      steps <- 0
      ratio<-spdf[i,]$Cbuffkm/spdf[i,]$HB_Area #Difference between Cbuff area
and basin area
      buffered.spgeom <- spgeom
      buffer.width<-sqrt((spdf[i,]$Cbuffkm*1000000)/pi) #Starting width =
radius of Cbuff

      while(abs(achieved.precision) > precision & steps < maxsteps) { #While
the achieved precision (regardless of sign) is greater than allowed precision,
& the number of steps is fewer than the allowed number of steps,
        buffered.perimeter<-gLength(buffered.spgeom) #Calculate the length of
the polygon
        buffer.width<-buffer.width-(achieved.precision*spgeom.area/buffered.per
imeter) #New buffer width = Previous buffer width minus (current precision X
current area / current length)
        buffered.spgeom<-gBuffer(spgeom, width=buffer.width) #Buffer by new
buffer width
        buffered.spgeom.area<-gArea(buffered.spgeom) #Calculate new area
        achieved.precision<-buffered.spgeom.area/spgeom.area-ratio
        steps<-steps+1 #Add one to number of steps
      }

      buffs[[i]]<-buffered.spgeom
      #Name each object in list by ID
      names(buffs)[i]<-as.character(ID[i]) #Needs to be as.character so
necessary zeros aren't accidentally removed
    }
    else
    {buffs[[i]]<-Cbuff[i,]
     names(buffs)[i]<-as.character(ID[i])
    }
  }
  #Filter results by those that worked (i.e. exclude any negative buffers if
necessary)
  buffs=Filter(function(x){!is.null(bbox(x))},buffs)
  #Convert the filtered results to spatial polygons
  buffsSP<-SpatialPolygons(lapply(1:length(buffs), function(i) {
    Pol <- slot(buffs[[i]], "polygons")[[1]]
    #Add names to be able to link to other datasets
    slot(Pol, "ID") <- names(buffs)[i]
    Pol
  }))
  #Project output (SP) based on input (spdf)
  proj4string(buffsSP)<-CRS(projection(spdf))
  buffsSP
}

newbuffs<-bufferby(bybasin,bybasin$basin_ID)
#Convert to SpatialPolygonsDataFrame
newbuffs<-SpatialPolygonsDataFrame(newbuffs,data=bybasinDF)
```

```
###2 DEMAND - Extract model data using Supply buffers

#POPULATION
#Extract population per buffered basin
newbuffcw<-extract(pop,newbuffs,small=T,weights=T,progress='text') #extract
'weights' of small polygons - fraction of cell covered by the polygon rounded
to 1/100
weights<-newbuffcw
library(gdata)
weights<-NAToUnknown(weights,0,force=T)
popbuff<-sapply(weights,function(x){ #Function to find weighted sums
  sum(x[,ncol(x)-1]*x[,ncol(x)]) #Multiply value of cell {x[,1]} by weight of
cell within polygon {x[,2]}
})
popbuffDF<-data.frame(popbuff)
popbuffDF$basin_ID<-newbuffs$basin_ID
popbuff<-merge(bybasinDF,popbuffDF,all.x=T)
popbuff$popbuff[is.na(popbuff$popbuff)]<-0
row.names(popbuff)<-popbuff$basin_ID
popbuff<-SpatialPolygonsDataFrame(as(basinspoly,"SpatialPolygons"),data=popbuf
f)
#Normalise scores
popbuffDF$popnorm<-popbuffDF$popbuff/max(popbuffDF$popbuff,na.rm=TRUE)

#LIVESTOCK
##Create raster of livestock per capita
#Resample population layer to match resolution and origin of livestock layer
percap<-resample(pop,livestock)
#Remove 0 values
percap[percap<1]<-1
livestock[livestock<1]<-1
livpercap<-livestock/percap
#Inverse pasture per capita
#Reclassify Inf to zero so don't affect cellStats
livpercap<-reclassify(livpercap, cbind(Inf,0))
##Inverse: Max value minus value
invliv<-calc(livpercap,fun=function(x){
  M<-cellStats(livpercap,max,na.rm=T)
  M-x})
#Get rid of NaNs
invliv[is.nan(invliv)]<-0
##Extract from livestock layer
invlivW<-extract(invliv,newbuffs,small=T,weights=T,progress='text') #extract
'weights' of small polygons - fraction of cell covered by the polygon rounded
to 1/100
weights<-invlivW
weights<-NAToUnknown(weights,0,force=T)
Ilivbuff<-sapply(weights,function(x){ #Function to find weighted sums
  sum(x[,ncol(x)-1]*x[,ncol(x)]) #Multiply value of cell {x[,1]} by weight of
cell within polygon {x[,2]}
})
IlivbuffDF<-data.frame(Ilivbuff)
IlivbuffDF$Ilivbuff[is.na(IlivbuffDF$Ilivbuff)]<-0
IlivbuffDF$basin_ID<-newbuffs$basin_ID
Ilivbuff<-merge(bybasinDF,IlivbuffDF,all.x=T)
row.names(Ilivbuff)<-Ilivbuff$basin_ID
Ilivbuff<-SpatialPolygonsDataFrame(as(basinspoly,"SpatialPolygons"),data=Ilivbu
ff)
#Normalise scores
```

```
IlivbuffDF$Ilivnorm<-IlivbuffDF$Ilivbuff/max(IlivbuffDF$Ilivbuff,na.rm=TRUE)

#Add population data
FFS<-merge(popbuffDF,IlivbuffDF)
#Get overall model score
FFS$FFS<-FFS$popnorm*FFS$Ilivnorm
bybasin<-data.frame(basinspoly)
FFS<-(merge(bybasin,FFS,all.x=T))
#Change NAs to zero
FFS$FFS[is.na(FFS$FFS)]<-0
row.names(FFS)<-FFS$basin_ID
#Coerce dataframe back to shapefile
FFS<-SpatialPolygonsDataFrame(as(basinspoly,"SpatialPolygons"),data=FFS)
```

C.4 Supplementary references

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Appendix D Supporting information for Chapter 5

D.1 Bivariate choropleth legends

Bivariate choropleth maps were used to map the relationship between FFS scores and species richness. Sub-catchments that scored highly for both FFS value and species richness are coloured with high saturation hues, while sub-catchments with low scores for both variables have a low saturation grey. Data was categorised into three classes for each variable in R using Jenks' natural breaks. The class for each variable was combined to define an overall colour category based on the colour legends shown in Figure D1. A scatterplot displaying the colour-coded data is also provided for each figure, to show the distribution of the data.

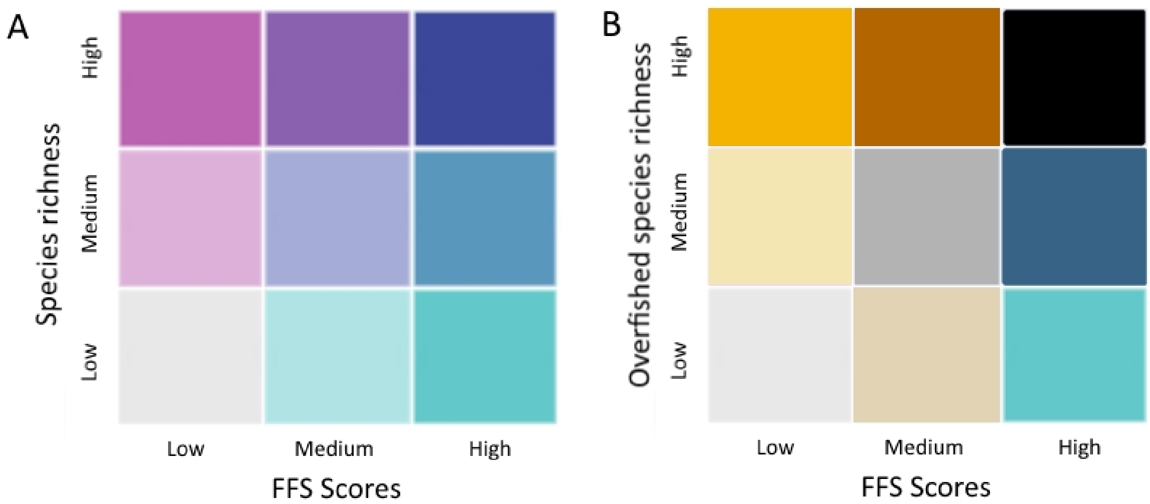

Figure D1 Colour legends to accompany bivariate choropleth maps in Chapter 5. A) Colour categories for Figures 5.3 and 5.4. B) Colour categories for Figures 5.5 and 5.6.