Two Essays on Livelihood Susceptibility and the Economic Valuation of Inland Fisheries

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Author declaration

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This work has not previously been presented for an award at this, or any other, University. All sources are acknowledged as References. The two chapters in this thesis have been submitted as manuscripts for publication. On the date of submission Chapter 1, *Household susceptibility to hydrological change in the Lower Mekong Basin*, was under review for publication in *Regional Environmental Change* and Chapter 2, *Current status and future needs of economics research of inland fisheries,* had been accepted for publication in *Fisheries Management and Ecology*. In the case of both manuscripts I, Ruby Grantham, am the lead author and am responsible for the analysis and content. My supervisor, Murray Rudd is the co-author on both papers in recognition of the editing support he provided.

Introduction

Inland surface-waters (lakes, rivers, reservoirs and floodplains) often sustain highly biodiverse and productive ecosystems that provide various direct and indirect benefits that contribute to human well-being (Postel & Carpenter, 1997; MEA, 2005a). Freshwater may be used in-situ or extracted for agricultural, domestic and industrial uses and inland surface-waters can support various species of flora and fauna that provide food and fibre, and have a number of cultural and intrinsic values (MEA, 2005a). Fish are one of the most obvious examples, activities such as wildlife viewing have non-use values and, when extracted via inland fisheries, fish can also provide nutrition, income, and a source of recreation (e.g., Cooke & Murchie, 2013; Béné, Hersoug & Allison, 2010; Butler *et al.*, 2009; Coomes *et al.*, 2010; Henderson, Criddle & Lee, 1999; Holmlund & Hammer, 1999; Moreau & Coomes, 2008; Rudd, 2009; Welcomme *et al*., 2010; Stoll, Ditton & Stokes, 2009).

It is estimated that 96% of inland fisheries are small-scale (Mills *et al.*, 2011); 36% of people (more than 21 million) in the fishing sector are engaged in inland capture fisheries but these fisheries contribute only 13% of global catch (FAO, 2014). Subsistence fishing is a component of diversified livelihoods for millions of people in developing countries (FAO, 2014). In many African and Asian countries inland fish provide an important source of animal protein and micronutrients (Béné & Heck, 2005; FAO, 2014); in Cambodia, for instance, fish contributes 80% of animal protein consumed (Hortle, Lieng & Valbo-Jorgensen, 2004). In developed countries inland fish populations are mostly exploited by recreational anglers (Welcomme *et al.*, 2010). Recreational anglers attribute high value to certain fish species (Olaussen & Liu, 2011) and fishing experiences (Schuhmann & Schwabe, 2004), and their expenditures can contribute notably to economies (Munn *et al.*, 2010; McKean, Johnson & Taylor, 2011; Chen, Hunt & Ditton, 2003).

“The aspects of ecosystems utilized (actively or passively) to produce human well-being” (Fisher, Turner & Morling, 2009, p.645) have been termed ecosystem services (ESs). The concept grew from trends towards framing ecological concerns in an economic context in the 1970s and 80s (Gómez-Baggethun *et al.*, 2010) and since the mid 1990’s the number of studies that address ESs has increased exponentially (Fisher, Turner & Morling, 2009). Wide recognition that society is entirely dependent on ESs led to the United Nations Millennium Ecosystem Assessment (MA) - a collaboration of over 1,300 scientists with the objective to “assess the consequences of ecosystem change for human well-being” (MEA, 2005b). The assessment found that human activities have caused unprecedented environmental change in the past 50 years, and although there have been net gains associated with the changes, there has also been substantial degradation of ESs which may undermine our ability to achieve the Millennium Development Goals (MEA, 2005b).

There is evidence from across the globe that the degradation of inland surface-waters is having an adverse effect on the provision of ESs (Dudgeon *et al.*, 2006; Moreno-Mateos *et al.*, 2012). Inland water ecosystems are some of the most threatened ecosystems on the planet and many have already been degraded beyond recovery (Moreno-Mateos *et al.*, 2012). The numerous direct and indirect stressors faced by inland waters can all be attributed to anthropogenic activities: exploitation of the resources they provide and their waste assimilation capacities (Allan *et al.*, 2005); alteration of biological conditions through, for example the introduction of alien invasive species (Dextrase & Mandrak, 2006; Leuven *et al.*, 2009); hydrological and terrestrial development which alter the processes that occur as water moves through a landscape (Brauman *et al.*, 2007); and climate change, which is affecting the volume, timing, location and temperature of water flows (IPCC, 2014). Further, efforts aimed at increasing the benefits gained from a particular ES from inland surface-waters can undermine other services. For example the development of hydropower alters fluvial river habitats and so can compromise fish populations (Dudgeon, 2000).

Over the next few decades the supply of freshwater is projected to decrease and its demand to increase (Hejazi *et al.*, 2013). It is, therefore, becoming increasingly important to assess the most efficient allocation of freshwater between competing uses and users (FAO, 2012). However, our understanding of the status and functioning of inland waters is relatively limited (Turner *et al.*, 2000). Fisheries, along with many other ESs generated by inland surface-waters, are often overlooked or deemed unimportant in decision-making processes because their value is not fully recognized (FAO, 2012; Beard *et al.*, 2011; Cowx & Gerdeaux, 2004). Inland fisheries are difficult to monitor because they are often small-scale and informal (Welcomme *et al.*, 2010). Subsistence fisheries often do not contribute to taxes or GDP and so are not a political priority and internationally there is very little funding available for inland fisheries research (Arlinghaus, Mehner & Cowx, 2002; De Graaf *et al.*, 2015). Thus, at present catch estimates are unreliable (Welcomme *et al.*, 2010; FAO, 2014; Bartley *et al.*, 2015; Coates, 2002) and the other benefits derived from fisheries (e.g. nutritional contributions to human health) virtually unexamined. If the importance of maintaining healthy inland surface-water ecosystems and sustaining their service provision, in particular fisheries, are to be recognised in decision-making then they must be subject to greater research attention (De Groot *et al.*, 2006; Bartley *et al.*, 2015).

This dissertation comprises two parts, they are linked in their recognition that fisheries, as an ecosystem service of inland surface waters, are poorly understood and undervalued. These shortcomings will present challenges to their sustainable management. The first chapter explores how the distribution of livelihood strategies and assets may determine household sensitivity and adaptive capacity to hydrological change in the Lower Mekong Basin. It thus presents an assessment of relative *susceptibility*, which is an important step towards understanding the magnitude and distribution of vulnerability to changing conditions and ES provision across the basin.

The ways in which the impacts of environmental change manifest across socio-economic systems reflects differences in vulnerability. Vulnerability to environmental change is a function of exposure, sensitivity and adaptive capacity (McCarthy, 2001). That is, the vulnerability of individuals and households (and, at higher aggregation, communities or countries) is determined by the distribution of the external hazard and a combination of positive and negative factors internal to the system. The ecological and social impacts of the degradation of inland water-bodies are greatest in developing countries and disproportionately borne by the rural poor (MEA, 2005b). This is due to a combination of the physical nature of environmental change, direct livelihood dependency on sensitive ecosystems, and socio-economic conditions unfavorable for coping and adaptation (Gupta *et al.*, 2010; IPCC, 2014).

The assessment of climate change impacts has shifted away from focussing only on physical hazards and towards a vulnerability perspective (Adger *et al.*, 2004). There are examples across the literature of assessments of social vulnerability to changing environmental conditions, these range from local to national scales and focus on topics such as food security (Hughes *et al.*, 2012), agricultural practices (Li *et al.*, 2015; Antwi-Agyei *et al.*, 2012), health impacts (Confalonieri *et al.*, 2014; Oven *et al.*, 2012) and hazardous events (Emrich & Cutter, 2011; Koks *et al.*, 2015).

Vulnerability accounts for how changes in the flow of ESs are distributed physically and how the magnitude of the impacts vary as a result of social, economic and political factors. In the context of inland surface-water ecosystems, vulnerability assessments can: contribute to impact assessments for development planning, guide precautionary policies aimed at reducing vulnerability to possible future change in inland water ecosystems and inform efficient resource allocation to increase adaptive capacities or facilitate coping mechanisms. In light of the various threats to inland water ecosystems and the impacts this has on their ES provision, assessing vulnerability is pertinent.

The second chapter of this dissertation presents a structured review of inland fisheries economic research. The findings provide an overview of trends in study design and current understanding of inland fisheries economics and highlights knowledge gaps and methodological shortcomings. It thus identifies data needs and best practice recommendations for future fisheries economics research.

Economic valuation gives conservation political salience (Gómez-Baggethun *et al.*, 2010); it can, in theory, enable the various benefits derived from an ecosystem to be aggregated and accounted for in the economic assessments that underpin most decision-making processes, such as cost-benefit analysis (Beard *et al.*, 2011). For example, in a trade-off analysis for hydropower development, if fisheries have not been valued electricity generation will be automatically prioritised over food security (Brummett, Beveridge & Cowx, 2013; Ziv *et al.*, 2012). Economic valuation is also central to the Ecosystem Approach (EA) to environmental management (Boyd & Banzhaf, 2007). The EA focuses on sustaining ES provision and recognizes that to achieve this ecosystems must be considered as a whole. Traditional management approaches have often had a more limited focused on specific ecosystem components or single species, which in many cases degraded the functioning of ecosystems and therefore undermined their ability to generate human benefit (Pikitch *et al.*, 2004; Tallis & Polasky, 2009). Thus, for fisheries to be effectively and sustainably managed they must be economically valued and, inland surface water ecosystems must be managed holistically in accordance with the EA (Baron *et al.*, 2002).

Both studies provide important contributions to inland fisheries research. The first chapter provides geographically specific information that can be used to inform decision-making processes in the Lower Mekong Basin. The findings of the susceptibility assessment have potential policy implications that are in line with the precautionary principle and highlight the need to reduce uncertainty surrounding hydrological change and household exposure to that change. Further, potential short-comings of the policy implications are discussed, recognizing that poorly designed vulnerability policies may have unintended and adverse impacts. The second chapter is of wider relevance to inland fisheries research. By providing an overview of existing inland capture fishery studies from the novel perspective of economic credibility, the study presents valuable insights needed to enable fisheries to be managed according to the principles of EA and to give fisheries greater policy relevance.

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Household susceptibility to hydrological change in the Lower Mekong Basin

Abstract

The distinct hydrological cycles and conditions of the Lower Mekong Basin support a multitude of ecosystem services. Thus, climate change, hydropower development and other processes that influence the quantity, quality, timing and location of water flow in the Mekong River will have implications for the tens of millions of people whose livelihoods depend on these services. This study presents an assessment of livelihood susceptibility to hydrological change in the Lower Mekong Basin. Using an index-based approach based on household sensitivity to hydrological change and adaptive capacity, susceptibility scores were calculated for 2,703 households living within close proximity of the Mekong River. With those scores, we compared relative susceptibility across countries and ecozones. Due to their greater livelihood dependency on water-related activities, mean household susceptibility was higher in Vietnam than in Cambodia, Laos, or Thailand. Households in Northern Laos also had high susceptibility, which was attributed to their low adaptive capacity. The findings suggest that policies aimed at reducing vulnerability to hydrological change in the Lower Mekong Basin should account for geographic context. Further, it highlights how policies may be able to strategically target the most susceptible households but that poorly designed policies have the potential to exacerbate vulnerability. In the face of high uncertainty surrounding hydrological change in the Lower Mekong Basin, our assessment of susceptibility should help inform precautionary water management policies and provide baseline information needed for more comprehensive vulnerability assessments in the future.

1. Introduction

Inland water ecosystems, including fresh and brackish waterbodies such as rivers, lakes, reservoirs and floodplains, generate a wide range of valuable ecosystem services (Brauman *et al.*, 2007; Mitsch, Bernal & Hernandez, 2015; Postel, 2009) and sustain the livelihoods of tens of millions (Allison *et al.*, 2009; FAO, 2014). They are, however, among some of the most threatened ecosystems on the planet (Dudgeon *et al.*, 2006; Vörösmarty *et al.*, 2010; Welcomme *et al.*, 2010). The quantity, quality, timing, and location of water flow through a landscape influence how people use water and the ecosystem services water provides (Brauman *et al.*, 2007). Climate change, hydroelectric development, and land-use change can alter hydrological attributes, changing the benefits that people receive from ecosystem services (Postel, 2009).

Climate change, development, and population pressures are projected to be greatest in developing countries (IPCC, 2014). The adverse consequences of degraded ecosystems, particularly surrounding food and water security, are borne disproportionately by the poor (MEA, 2005). A lack of financial resources, weak governance, and lack of representation also often contribute to the vulnerability of individuals, households, and communities that depend on freshwater ecosystems (FAO, 2014). Vulnerability is a term used in many disciplines (Alwang, Siegel & Jorgensen, 2001) and its definition has evolved. In the context of socio-ecological systems (SESs) research, vulnerability is generally accepted to refer to the susceptibility of an SES to endure adverse consequences arising from exposure to an environmental hazard and the ability of that system to cope or recover (Adger, 2006; Füssel, 2007). The IPCC described vulnerability as a function of exposure, sensitivity, and adaptive capacity (McCarthy, 2001); more recent work also emphasizes the relationship of exposure to harm (Chapin III, Fole & Kofinas, 2009) and adaptation readiness (Ford & King, 2015). Exposure is the potential physical impact from a future external hazard, while sensitivity and adaptive capacity are negative and positive internal determinants, respectively. Sensitivity is "the degree to which a system will respond to a given change in climate including beneficial and harmful effects" and adaptive capacity is "the degree to which adjustments in practices, processes, or structures can moderate or offset the potential for damage to take advantage of opportunities created by a given change in climate" (McCarthy, 2001, p.89). These internal components determine a system's or household's susceptibility to stressors.

Vulnerability assessments consider why systems experiencing the same hazard are unequally affected. Understanding who will be most vulnerable to environmental change is essential for guiding interventions that facilitate effective, efficient, and sustainable freshwater ecosystem management (Vincent, 2007). The most appropriate method to assess vulnerability to environmental change remains contested (Cutter, Boruff & Shirley, 2003; Füssel, 2007; Hinkel, 2011; Polsky, Neff & Yarnal, 2007). Susceptibility, sensitivity and adaptive capacity are theoretical concepts so cannot be measured directly (Hinkel, 2011). To calculate a metric of *susceptibility*, a set of indicators based on measurable variables must be used to operationalize the concepts of sensitivity and adaptive capacity (Adger, 2006; Hinkel, 2011; Moss, Brenkert & Malone, 2001). Our focus on susceptibility in this paper aligns closely with "starting point" vulnerability. Starting point vulnerability assumes that internal and structural factors influence a system's ability to cope with future hazards, and that addressing those factors will therefore reduce vulnerability (Adger, Arnell & Tompkins, 2005; Kelly & Adger, 2000; O'Brien *et al.*, 2004). Thus, in being able to inform *ex-ante* action it has policy relevance in accordance with the precautionary approach (Füssel & Klein, 2006).

The Lower Mekong Basin (LMB) provides an important case study for assessing the vulnerability of household livelihoods to hydrological change. The capture of fish and other aquatic plants and animals is an important source of nutrition and income, and the river provides an essential source of water for domestic and agricultural use (MRC, 2010a). It was estimated that in 2003, of the 55 million people living in the LMB, 40 million were involved in the Mekong fisheries (MRC, 2003). The Mekong is home to at least 1,200 species of fish (Coates *et al.*, 2003); of the 165 species for which migration status is understood, 87% are migratory and most migrations are triggered by changing hydrological conditions (Baran, 2006). Distinct and extreme wet and dry seasons and a strong flood pulse characterize the LMB, and SESs have evolved according to those hydrological cycles (MRC, 2010a). Hence, fish populations, fisheries productivity, livelihoods, and human well-being in the LMB will all be impacted by changes in hydrology (Baran & Myschowoda, 2009; Barlow *et al.,* 2008).

Climate change is predicted to affect precipitation, temperature, and extreme weather events in the LMB (Kingston, Thompson & Kite, 2011), potentially changing water flow patterns and temperatures, and thereby placing increased physiological stress on fishes (Cooke, Paukert & Hogan, 2012). Land-use change throughout the LMB is affecting water run-off and water quality (MRC, 2015). Over 100 hydropower developments are planned or under construction on the Mekong and its tributaries (MRC, 2011), which are likely to impede fish passage (Bunn & Arthington, 2002; Cooke, Paukert & Hogan, 2012) and sediment transport (Kummu & Varis, 2007). Growing fishing pressure is also undermining Mekong fisheries production (Baran & Myschowoda, 2008).

Among LMB countries, Thailand is the most developed, whereas Laos PDR (hereafter Laos), Cambodia, and Vietnam suffer greater issues of poverty, malnutrition, inequality, and weak citizen-government relations (MRC, 2010a; Stuart-Fox, 2010). Those human and institutional conditions are typically indicative of vulnerability (Adger, 2006). Vulnerability assessments are becoming increasingly prominent in the management of the Mekong River; the Mekong River Commission (MRC) implemented their Social Impact Monitoring and Vulnerability Assessment (SIMVA) pilot study in 2008 which was followed up with a major household survey in 2011 and household and community surveys in 2014 (Hall & Bouapao, 2010; MRC, 2014). To date, other empirical research to assess livelihood vulnerability in the LMB has largely focused on terrestrial agriculture and farmer adaption in the Mekong Delta, Vietnam (Bastakoti *et al.,* 2014; Birkmann, 2011; Dang, Nuberg & Bruwer, 2014; Ling *et al.*, 2015). The high risks associated with climate change impacts, notably sea level rise, make vulnerability assessments and strategies an important research topic in this region (Dasgupta *et al.*, 2007; Yusuf & Francisco, 2009). However, hydrological regime changes will be felt across the LMB and may affect livelihood strategies and well-being for millions of people.

The aim of our research was to identify where within the LMB corridor and the major floodplains are those households that, as a result of internal socio-economic factors at the household level, are more likely to experience adverse impacts from hydrological change. That is, how do households across regions vary in their level of *susceptibility* to hydrological change? Our specific objectives were to: develop composite indices of *susceptibility* at basin, national, and ecozone scales, and analyze how relative *susceptibility* and its determinants differed across scales.

2. Methods

2.1 Design of *susceptibility* index

The most widely useful vulnerability metrics will be those that are transferrable and comparable (Luers *et al.*, 2003); a standardized framework, applicable in different contexts, is thus needed (Adger & Vincent, 2005; Eakin & Luers, 2006; Polsky, Neff & Yarnal, 2007). The use of even a relatively simple index can be justified as it provides directional advice that improves understanding of relative vulnerabilities. That provides policy-relevant information about potential intervention points that can help reduce sensitivity, increase resilience, and support proactive transformative policies that support environmental stewardship (Chapin III, Fole & Kofinas, 2009).

Our additive, unweighted index construct (below) is similar to that used in other studies to measure vulnerability associated with capture fisheries. For example, with an unweighted index Allison *et al.* (2009) used country level variables to compare the vulnerability of 132 national economies to climate change impact on capture fisheries. Cinner *et al.* (2012) assessed the vulnerability of 29 coastal communities to the impact of coral bleaching on fisheries using remote sensing and household survey data. They used a similar unweighted index and tested both additive and multiplicative forms of the index. Islam *et al.* (2014) compared livelihood vulnerability of fishers to climate variability and change in two Bangladeshi communities using meteorological data, household surveys and focus groups. They also tested additive and multiplicative relationships between exposure and sensitivity, and found a very high correlation between them. We used an additive function of dependency and adaptive capacity to develop an index of: $susceptibility = S + (1-AC)/2$, where S = sensitivity and AC = adaptive capacity. The expression $1-AC$ relates declining adaptive capacity to higher household *susceptibility*. Thus, a high *susceptibility* score is a function of high household sensitivity and/or low adaptive capacity.

Unlike conventional vulnerability indices, the *susceptibility* index used in this paper does not include a vector of exposure. While the MRC is developing models of changing hydrological conditions across the LMB (MRC, 2009b), these are not yet at a stage to incorporate in forward-looking community-scale exposure projections. Our index takes one important step in the process of developing the information needed for more comprehensive vulnerability assessments in the future.

2.2 Data

This study used household level data collected by the MRC SIMVA 2011 survey. The study location contains four countries (Cambodia, Laos, Thailand and Vietnam) and eight ecological subzones (ecozones, henceforth abbreviated as EZ) (Figure 1.1). The ecozones used in SIMVA 2011 were adapted from Integrated Basin Flow Management zones and that were based on hydrology, physiography, land cover and vegetation (MRC, 2009a). EZ1 and EZ2 straddle Laos and Thailand: EZ1 is located in the highlands and EZ2 covers the central plateau including Thailand’s Songkhram wetland and the southeastern highlands of Laos. EZ3 is the Mekong corridor running through the southeastern highlands and southern region of Cambodia. EZ4 is the Tonle Sap basin, and EZ5 and EZ6 are located in Vietnam and are the fresh and brackish water zones, respectively. The SIMVA 2011 survey targeted rural villages situated within a 15km corridor along the Mekong mainstream and 40km into its major tributary confluences and floodplains. A stratified random sampling strategy was used for villages and households within countries, while overall sampling effort was distributed evenly across countries (for equity reasons among the four LMB nations). A total of 2,720 households completed the SIMVA 2011 survey. Of those, 17 households answered "don't know" to questions central to this analysis and so were removed, leaving a sample of 2,703 households for our analysis. See Hall and Bouapao (2010) and MRC (2014) for details about the 2008-09 SIMVA pilot study and SIMVA 2011 design.

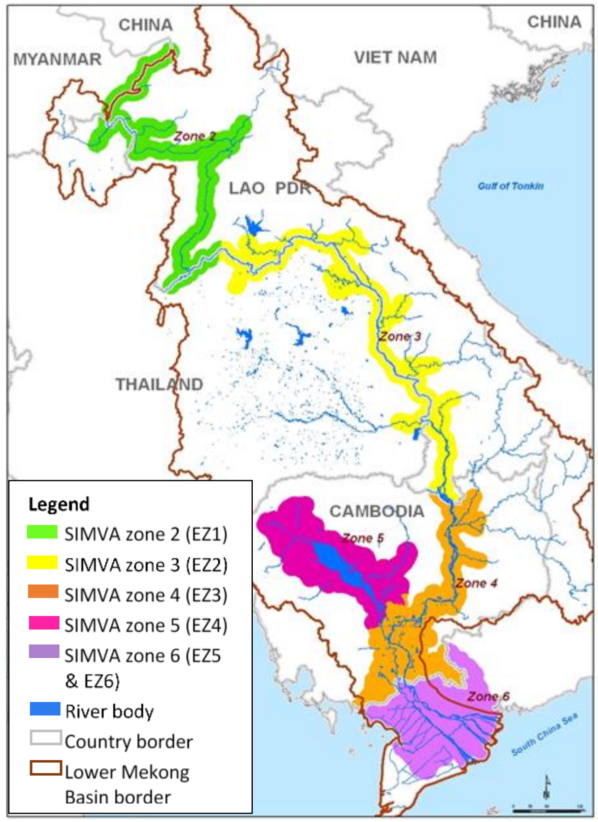

Figure 1.1. Map of SIMVA survey zones (and corresponding ecozones) in the Lower Mekong Basin. Source: Mekong River Commission.

2.3 Choice of indicators

From the array of data collected in the SIMVA 2011 survey, we selected variables as indicators of sensitivity and adaptive capacity in accordance with established theoretical concepts and past research findings, and within the limitations of the dataset. Sensitivity was used to describe whether a household's livelihood would be affected by changes in the flow of ecosystem services from natural capital (Reed *et al.*, 2013), specifically those services associated with or impacted by hydrological regime change. Livelihoods that are more dependent on a natural resource are more sensitive to changes in that resource (Allison *et al.*, 2009; Cinner *et al.*, 2012; Eakin & Bojorquez-Tapia, 2008; Smit & Wandel, 2006). Therefore, variables of household livelihood dependency on aquatic resources were used as sensitivity indicators: (1) income types – whether any household members were engaged in water-dependent income activities (scored as 0 if no income, 1 if mixed income sources including water-dependent, and 2 if entirely water-dependent); (2) income dependency – the percentage of total annual household income from water-dependent activities; (3) subsistence dependency – whether food from freshwater ecosystems eaten in the last 24 hours was caught by a household member or was home grown (count 1 for each of fish, other aquatic animals, and vegetables from a riverbank garden); (4) agricultural dependency – whether the main source of agricultural water was Mekong irrigation or other natural flooding (dummy); and (5) domestic dependency – whether river water was the main source of water used domestically (dummy).

Adaptive capacity describes the set of resources available to households to facilitate adaptation (Nelson, Adger & Brown, 2007). Past research in the LMB suggested that sustainable livelihoods are constrained by limited access to assets (Sok & Yu, 2015). Household adaptive capacity is therefore defined in terms of social, human, financial, physical and natural assets (Adger & Vincent, 2005; Yohe & Tol, 2002). Variables relating to each type of asset were used as adaptive capacity indicators (see supplementary information S1.1 for details).

For our analysis, the indicators *income types* and *education* were transformed into dummy variables to capture whether households engaged in any water-dependent income activities and whether any household member had education above primary school level. A composite variable, based on a count of the different types of physical assets a household owned and household income diversity, was used to capture a household's ability to spread risk (Berkes, 2007; Yohe & Tol, 2002). Per capita income was also included because, although it is a desired outcome of the SLF, it also feeds back into the process of wealth accumulation and reinforces sustainable livelihoods (DFID, 1999). Indicator values were constructed so that traits associated with high sensitivity and high adaptive capacity were reflected in high scores. Indicator scores were normalized from zero to one so that each was weighted equally within the relevant sub-index. The *proximity to road* indicator was normalized using an inverse function so that greater distance was reflected in lower scores. Relevant indicator scores were summed to calculate a value for each sub-index, which were then normalized so that sensitivity and adaptive capacity were equally weighted within the metric of *susceptibility*.

2.4 Data analysis

Each household in the SIMVA 2011 sample was assigned three *susceptibility* scores 1) relative to the whole sample, 2) relative to country sample, 3) relative to ecozone sample. The susceptibility scores calculated relative to the whole sample were used to compare susceptibility between countries, ecozones, and *susceptibility* classes. Scores calculated relative to individual country and ecozone samples were used to identify which factors determine the *susceptibility* metric by comparing *susceptibility* classes within each geographic region. *Susceptibilit*y classes were formed by splitting the sample into *susceptibility* score quartiles (following Allison *et al.*, 2009, Islam *et al.*, 2014).

Chi-squared tests were used to assess whether, for traits measured using dummy or ordinal variables, the proportion of households differed significantly between geographic or *susceptibility* class groups. For continuous variables, one-way ANOVA and Bonferroni adjusted post-hoc tests were used to compare the mean scores of different groups. Due to the large sample size only differences at 1% significance for main tests and 5% significance for post-hoc tests are discussed.

3. Results

The overall sample of 2,703 households consisted of 680 households from Cambodia, 679 households from Laos, 666 households from Thailand, and 678 households from Vietnam (Table 1.1). Descriptive statistics are summarized in Table 1.1: there were varying patterns of significant differences between countries for all demographic and other indicator variables.

3.1 Country level *susceptibility*

Mean household *susceptibility* scores were significantly different (F=132.7, 3 d.f., p<0.01) among countries (Table 1.1) and post-hoc tests indicated that all four countries differed from each other at the 5% significance level. Mean *susceptibility* was lowest in Thailand, which was characterized by low sensitivity and high adaptive capacity in the sub-component scores. However, Thailand was one of the countries containing the highest proportion of households who source income from water-dependent activities. In comparison to other countries, households in Thailand were smaller, contained the fewer working age members, and were more likely to have an elderly head of household.

Mean household *susceptibility* was second highest in Laos, where households were on average the second most sensitive and had the lowest adaptive capacity. Laos households had relatively high scores for *domestic dependency* and *subsistence dependency* but *agricultural dependency* was lower than in other countries. Households in Laos scored lowest for a number of adaptive capacity indicators including, among others, *reliable flood warning, income diversity,* and *distance to road*. However, households in Laos were significantly more likely to own agricultural land and livestock than households in any other country.

Table 1.1 Comparison of mean indicator, sensitivity, adaptive capacity, and composite *susceptibility* scores of countries.

	Cambodia	**Laos**	**Thailand**	**Vietnam**	**Statistic (df=3)**
Sample size	680	679	666	678	
Composite *susceptibility* score***	0.2^{a}	0.3^{b}	0.2^{c}	0.3^{d}	F=132.7
Sensitivity indicators					
^{D}Income types***	$23.4^{a,b}$	16.9^{c}	26.9^{b}	$20.2^{a,c}$	χ^2=33.6
% income water-dependent***	$6.3^{a,b}$	$4.6^{a,c}$	7.3^{b}	8.7^{b}	F=5.8
Subsistence dependency***	$0.3^{a,c}$	$0.4^{a,b}$	0.4^{b}	0.2^{c}	F=13.2
^{D}Agricultural dependency***	8.5^{a}	3.0^{b}	17.6^{c}	63.6^{d}	χ^2=871.5
^{D}Domestic dependency***	16.5^{a}	37.1^{b}	0.5^{c}	37.0^{b}	χ^2=360.7
Sensitivity score***	0.14^{a}	0.17^{b}	0.14^{a}	0.33^{c}	F=172.4
Adaptive capacity indicators					
HH size ***	5.0^{a}	5.7^{b}	4.0^{c}	4.3^{d}	F=108.8
No. working age members***	3.0^{a}	3.7^{b}	2.4^{c}	2.9^{a}	F=72.9
% working age members***	62.0^{a}	64.8^{a}	61.8^{a}	70.6^{b}	F=18.1
^{D}Non-elderly head ***	59.3^{a}	60.2^{a}	29.3^{b}	37.0^{c}	χ^2=199.9
^{D}Education***	27.9^{a}	43.9^{b}	61.1^{c}	84.1^{d}	χ^2=474.5
Asset value (US$)***	6598^{a}	7127^{a}	44003^{b}	35603^{c}	F=120.1
^{D}Credit, savings or remittances***	55.6^{a}	32.3^{b}	59.2^{a}	36.1^{b}	χ^2=149.9
^{D}Tech./trans. ownership***	58.5^{a}	65.4^{a}	94.1^{b}	99.1^{c}	χ^2=508.6
^{D}Water supply ownership***	36.5^{a}	39.9^{a}	42.6^{a}	90.6^{b}	χ^2=532.9
^{D}Fishing/farm equipment ownership***	32.9^{a}	56.1^{b}	61.0^{b}	34.4^{a}	χ^2=171.6
^{D}Livestock ownership***	76.3^{a}	85.7^{b}	47.6^{c}	48.7^{c}	χ^2=332.9
^{D}Stored rice/fish***	79.9^{a}	90.4^{b}	89.8^{b}	38.1^{c}	χ^2=656.5
^{D}Reliable flood warning***	71.8^{a}	10.2^{b}	59.6^{c}	69.6^{a}	χ^2=682.7
^{D}Agricultural land ownership***	77.1^{a}	94.4^{b}	81.5^{a}	76.8^{a}	χ^2=95.9
^{D}Non-local income***	58.7^{a}	29.2^{b}	28.8^{b}	34.4^{b}	χ^2=173.8
^{D}Male head***	77.2^{a}	90.9^{b}	73.7^{a}	84.7^{c}	χ^2=79.5
Distance to road***	0.1^{a}	10.4^{b}	0.8^{a}	0.8^{a}	F=268.5
^{D}Association membership***	25.6^{a}	53.0^{b}	88.3^{c}	58.1^{b}	χ^2=540.9
Asset diversity***	2.8^{a}	$3.4^{b,c}$	3.3^{b}	3.5^{c}	F=43.2
Income diversity***	3.1^{a}	2.2^{b}	3.2^{a}	2.5^{c}	F=74.7
Per capita income (US$)***	367^{a}	265^{a}	2510^{b}	1524^{c}	F=35.4
Adaptive capacity score***	0.7^{a}	0.6^{b}	0.2^{c}	$0.3^{a,c}$	F=55.9

^{D} = dummy variable, accompanying values are percentage of households who answered positively.
***significant at the 1% level, ** significant at the 5% level, * significant at the 10% level.

Cambodia had the second lowest mean household *susceptibility* score. Cambodia was one of the two lowest scoring countries for every sensitivity indicator except *income types*. The only adaptive capacity indicators in which Cambodia scored lowest were *education, association membership* and *asset diversity.* Only 27.9% of households surveyed in Cambodia contained a member who had education above primary school level.

Due to relatively high household sensitivity scores, mean household *susceptibility* was significantly higher in Vietnam than in any other country. Of households surveyed, 63.6% relied on natural flooding or irrigation for agriculture, a significantly greater proportion than other countries. *Domestic dependency* and *income dependency* were also high but mean *subsistence dependency* was significantly lower in Vietnam than in other LMB countries.

3.2 Ecozone vulnerability

Mean *susceptibility* scores varied significantly (F=65.1, 5 d.f., $p<0.01$) between ecozones (Table 1.2). Households' mean *susceptibility* across EZ2, EZ3 and EZ4 were the lowest and were statistically indistinguishable. The higher *susceptibility* score in EZ1 was attributed to significantly lower adaptive capacity among households there. Mean household sensitivity, adaptive capacity and *susceptibility* were similar across the two ecozones within Cambodia (EZ3 and EZ4). The only indicators for which mean scores differed between the two ecozones were *subsistence dependency, fishing/farm equipment ownership, stored rice/fish* and *asset diversity*. In Vietnam, households in EZ5 had higher mean sensitivity and, consequently, higher mean *susceptibility* compared to households in EZ6. The sensitivity of households in EZ5 and EZ6 was determined by a combination of different variables; households in both regions had high *agricultural dependency* but households in EZ5 also had much greater *domestic dependency* compared to households in EZ6. Households in EZ6 were more likely to engage in water-dependent activities and derive a greater proportion of their income from those activities. For most adaptive capacity indicators the mean score was similar across both Vietnamese ecozones.

Straddling Laos and Thailand, EZ1 and EZ2 were the only ecozones that crossed political borders. The ecozone and country indices were used to assess differences between households in the same ecozone but different countries and between households in the same country but different ecozones (supplementary information S1.2). In EZ1, households from Laos had significantly higher sensitivity, lower adaptive capacity, and higher *susceptibility* than their Thai counterparts sharing EZ1. Across EZ2 in Laos and Thailand, the mean scores for sensitivity, adaptive capacity, and *susceptibility* were not significantly distinguishable.

In Laos, households in EZ1 had significantly higher sensitivity than households in EZ2 because a higher proportion of households in EZ1 relied on river water for domestic uses; for most other dependency indicators, households in EZ1 scored significantly lower than households in EZ2. Households in EZ1 also had significantly lower adaptive capacity

Table 1.2 Comparison of mean indicator, sensitivity, adaptive capacity and composite *susceptibility* scores of ecozones.

	EZ1	EZ2	EZ3	EZ4	EZ5	EZ6	Statistic (df=5)
Sample size	676	669	340	340	340	338	
Composite *susceptibility* score***	0.3^{a}	0.2^{b}	0.2^{b}	0.2^{b}	0.3^{c}	0.3^{d}	F=65.1
Sensitivity indicators							
[D]Income types***	14.9^{a}	31.5^{b}	21.8^{a,c}	25.0^{b,c}	7.9^{d}	32.5^{b}	χ^2=115.7
% income water-dependent***	3.3^{a}	8.6^{b}	6.0^{a,b}	6.5^{a,b}	1.2^{c}	16.3^{d}	F=30.8
Subsistence dependency***	0.3^{a}	0.4^{b}	0.2^{c}	0.4^{a,b}	0.2^{c}	0.2^{a,c}	F=15.4
[D]Agricultural dependency***	8.6^{a}	11.8^{a}	9.4^{a}	7.6^{a}	73.2^{b}	53.8^{c}	χ^2=869.2
[D]Domestic dependency***	28.0^{a}	9.9^{b}	18.2^{c}	14.7^{b,c}	54.1^{d}	19.8^{a,c}	χ^2=281.1
Sensitivity score***	0.2^{a}	0.2^{a}	0.1^{a}	0.1^{a}	0.4^{b}	0.3^{b}	F=*105.8*
Adaptive capacity indicators							
HH size***	4.7^{a,b}	5.0^{a}	4.9^{a,b}	5.1^{a}	4.5^{b}	4.0^{c}	F=13.4
No. working age members**	3.0^{a}	3.1^{a}	3.0^{a}	3.1^{a}	3.1^{a}	2.8^{a}	F=2.7
% working age members***	64.4^{a}	62.2^{a}	61.2^{a}	62.9^{a}	70.4^{b}	70.8^{b}	F=10.6
[D]Non-elderly head***	48.4^{a,b,c}	41.4^{c,d}	57.4^{b,e}	61.2^{e}	32.9^{d}	41.1^{a,c,d}	χ^2=82.5
[D]Education***	49.6^{a}	55.3^{a}	30.3^{b}	25.6^{b}	85.9^{c}	82.2^{c}	χ^2=441.2
Asset value (US$)***	29088^{a}	21646^{a}	6215^{b}	6981^{b}	41379^{a}	29792^{a}	F=30.5
[D]Credit, savings or remittances***	38.5^{a}	52.8^{b}	52.1^{b}	59.1^{b}	36.5^{a}	35.8^{a}	χ^2=83.0
[D]Tech./trans. ownership***	74.4^{a}	84.9^{b}	55.3^{c}	61.8^{c}	99.1^{d}	99.1^{d}	χ^2=366.7
[D]Water supply ownership***	26.5^{a}	56.2^{b}	38.5^{c}	34.4^{a,c}	90.0^{d}	91.1^{d}	χ^2=652.3
[D]Fishing/farm equipment ownership***	53.6^{a}	63.5^{b}	23.8^{c}	42.1^{d}	30.6^{c,e}	38.2^{d,e}	χ^2=208.5
[D]Livestock ownership***	68.3^{a,b}	65.3^{b}	74.7^{a,c}	77.9^{c}	49.4^{d}	47.9^{d}	χ^2=121.4
[D]Stored rice/fish***	94.7^{a}	85.5^{b}	73.5^{c}	86.2^{b}	49.4^{d}	26.6^{e}	χ^2=731.9
[D]Reliable flood warning***	31.7^{a}	37.7^{a}	72.9^{b}	70.6^{b}	74.4^{b}	64.8^{b}	χ^2=364.4
[D]Agricultural land ownership***	89.6^{a}	86.4^{a,b}	73.2^{c}	80.9^{b,c}	76.8^{c}	76.9^{c}	χ^2=66.7
[D]Non-local income***	23.2^{a}	34.8^{b}	60.9^{c}	56.5^{c}	35.6^{b}	33.1^{b}	χ^2=194.9
[D]Male head***	85.1^{a}	79.7^{a,b}	74.4^{b}	80.0^{a,b}	86.8^{a}	82.5^{a,b}	χ^2=25.6
Distance to road***	8.7^{a}	2.6^{b}	0.1^{c}	0.2^{c}	0.8^{c}	0.7^{c}	F=91.1
[D]Association membership***	70.1^{a}	70.9^{a}	25.9^{b}	25.3^{b}	52.4^{c}	63.9^{a}	χ^2=380.4
Asset diversity***	3.1^{a}	3.6^{b}	2.7^{c}	3.0^{a}	3.5^{b}	3.5^{b}	F=36.9
Income diversity***	2.5^{a}	2.9^{b,c}	3.1^{b}	3.1^{b}	2.6^{a,c}	2.5^{a}	F=17.8
Per capita income (US$)***	1499^{a}	1253^{a,b}	428^{b,c}	306^{c}	1782^{a}	1264^{a,b,c}	F=6.1
Adaptive capacity score***	0.6^{a}	0.7^{b}	0.7^{c}	0.7^{b,c}	0.7^{b}	0.7^{b,c}	F=27.6

[D] = dummy variable, accompanying values are percentage of households who answered positively. ***significant at the 1% level, ** significant at the 5% level, * significant at the 10% level.

than EZ2 households. In Thailand, the difference across EZ1 and EZ2 was due to EZ1 households having higher sensitivity, characterized by a higher proportion of households engaging in water-dependent income activities. However, in the Thai portion of EZ2, a greater proportion of households derived income from water dependent activities and households had higher *subsistence dependency*.

3.3 Determinants of susceptibility

Comparing indicator and index scores between *susceptibility* quartiles can help identify which factors determine *susceptibility* scoring (Islam *et al.*, 2014; Smit & Wandel, 2006). Every indicator except *per capita income* differed significantly between at least two *susceptibility* classes across the whole LMB (Table 1.3). For those significant indicators, in all but *household size*, *water supply*, and *male head*, the direction they followed aligned with *a priori* assumptions. When the relationship between an indicator and *susceptibility* classes was not linear, or was opposite in effect to that the index construct presumed, it is reasonable to infer that it was not a determinant of *susceptibility*.

Differences between *susceptibility* classes were assessed across countries and ecozones to identify which factors influenced the *susceptibility* metric (Table 1.4). In almost all cases, every sensitivity indicator differed significantly between classes and followed *a priori* assumptions. That is, sensitivity indicator scores increased with *susceptibility* and therefore determined the *susceptibility* metric. Comparatively, in every index a number of *adaptive capacity* indicators were found not to differ significantly between *susceptibility* classes. In Vietnam and its ecozone sub-regions very few adaptive capacity indicators differed significantly between *susceptibility* classes and the adaptive capacity sub-index itself was not a determinant of the *susceptibility* metric. Which combination of adaptive capacity indicators determined the *susceptibility* metric varied somewhat between indices and was not always concurrent between a country and its component ecozones. In Cambodia, for instance, *reliable flood warning* was significantly and negatively related to *susceptibility* class but it was not significant in analyses for EZ3 or EZ4.

Table 1.3 Comparison of mean indicator scores of *susceptibility* classes to identify determinants of the *susceptibility* metric

	Low	**Moderate**	**High**	**Very high**	**Mean**	**Statistic (df=3)**
Sample size	676	675	676	676		
Composite *susceptibility* score***	0.1^{a}	0.2^{b}	0.3^{c}	0.4^{d}	0.3	F=6113.6
Sensitivity indicators						
^{D}Income types***	6.1^{a}	17.0^{b}	30.3^{c}	36.5^{c}	0.2	χ^2=216.5
% income water-dependent***	0.5^{a}	1.9^{a}	8.2^{b}	16.1^{c}	6.8	F=108.8
Subsistence dependency***	0.2^{a}	0.3^{b}	0.4^{b,c}	0.4^{c}	0.3	F=28.6
^{D}Agricultural dependency***	0.2^{a}	10.4^{b}	34.0^{c}	48.1^{d}	0.2	χ^2=543.9
^{D}Domestic dependency***	0.2^{a}	3.0^{b}	21.0^{c}	67.3^{d}	0.2	χ^2=1107.8
Sensitivity score***	0.0^{a}	0.1^{b}	0.2^{c}	0.4^{d}	0.2	F=1436.7
Adaptive capacity indicators						
HH size***	5.1^{a}	4.5^{b}	4.5^{b}	4.8^{c}	4.7	F=17.0
No. working age members***	3.4^{a}	2.8^{b}	2.8^{b}	3.0^{b}	3.0	F=22.3
% working age members***	67.9^{a}	63.7^{b}	62.9^{b}	63.2^{b}	64.8	F=5.8
^{D}Non-elderly head***	52.7^{a}	47.9^{a,b}	42.8^{b}	42.9^{b}	0.5	χ^2=18.2
^{D}Education***	63.3^{a}	51.0^{b}	52.4^{b}	50.1^{b}	1.8	χ^2=30.8
Asset value (US$)***	2897^{a}	21740^{a,b}	20911^{b}	20741^{b}	23222	F=4.1
^{D}Credit, savings or remittances***	62.7^{a}	47.4^{b}	40.8^{c}	32.0^{d}	0.5	χ^2=137.7
^{D}Tech./trans. ownership***	92.2^{a}	80.6^{b}	76.3^{b}	67.8^{c}	0.8	χ^2=126.9
^{D}Water supply ownership***	61.5^{a}	46.8^{b}	52.2^{c}	49.1^{b,c}	0.5	χ^2=34.0
^{D}Fishing/farm equipment ownership***	55.3^{a}	42.5^{b}	44.2^{b}	42.0^{b}	0.5	χ^2=32.1
^{D}Livestock ownership***	77.8^{a}	61.0^{b}	60.2^{b}	59.6^{b}	0.7	χ^2=68.4
^{D}Stored rice/fish***	89.6^{a}	76.9^{b}	65.4^{c}	66.0^{c}	0.7	χ^2=139.0
^{D}Reliable flood warning***	61.1^{a}	52.6^{b}	51.9^{b}	45.4^{c}	0.5	χ^2=33.7
^{D}Agricultural land ownership***	89.2^{a}	78.8^{b}	80.0^{b}	81.8^{b}	0.8	χ^2=30.4
^{D}Non-local income***	55.6^{a}	34.2^{b}	34.0^{b}	27.4^{c}	0.4	χ^2=130.4
^{D}Male head***	86.8^{a}	78.2^{b}	78.0^{b}	83.6^{a}	0.8	χ^2=25.2
Distance to road***	0.7^{a}	2.1^{b}	1.9^{a,b}	7.4^{c}	3.0	F=83.5
^{D}Association membership***	65.5^{a}	55.1^{b}	55.9^{b}	47.8^{c}	0.6	χ^2=43.7
Asset diversity***	3.5^{a}	3.1^{b}	3.1^{b}	3.0^{b}	3.3	F=56.1
Income diversity***	3.1^{a}	2.7^{b}	2.7^{b}	2.4^{c}	2.8	F=29.1
Per capita income (US$)*	1508^{a}	1237^{a}	898^{a}	967^{a}	1160	F=2.1
Adaptive capacity score***	0.8^{a}	0.7^{b}	0.7^{b}	0.6^{c}	0.7	F=201.5

^{D} = dummy variable, accompanying values are percentage of households who answered positively. ***significant at the 1% level, ** significant at the 5% level, * significant at the 10% level.

Table 1.4 Direction of relationship between indicators and *susceptibility* class to identify determinants of the *susceptibility* metric within each index.

Indicator		Sample	Lao	Thai	EZ1	EZ2	Cam	EZ3	EZ4	Viet	EZ5	EZ6
Sensitivity	Income source	↑	↑	↑	↑	↑	↑	↑	↑	↑	↑	↑
	Income dependency	↑	↑	↑	X	↑	↑	↑	↑	↑	↑	↑
	Subsistence dependency	↑	↑	↑	↑	↑	↑	↑	↑	↑	X	↑
	Agricultural dependency	↑	↑	↑	↑	↑	↑	↑	↑	↑	↑	↑
	Domestic dependency	↑	↑	X	→	↑	↑	↑	↑	↑	↑	↑
	Sensitivity score	↑	↑	↑	↑	↑	↑	↑	↑	↑	↑	↑
Adaptive capacity												
Human capital	Non-elderly head	↓	X	↓	X	X	X	X	X	↓	↓	X
	HH size	→	X	X	X	X	→	↓	X	X	X	X
	Number of working age members	↓	X	↓	X	↓	↓	↓	↓	X	X	X
	% working age members	↓	X	↓	↓	↓	↓	↓	X	X	X	X
	Education	↓	↓	↓	↓	↓	↓	↓	↓	X	X	X
Financial capital	Asset value	↓	↓	↓	↓	X	↓	↓	↓	X	X	X
	Credit, savings or remittances	↓	↓	X	↓	↓	X	X	X	↓	↓	X
Physical capital	Technology/transport ownership	↓	↓	↓	↓	↓	↓	↓	↓	X	X	X
	Water supply ownership	→	↓	↓	↓	↓	↓	↓	↓	X	X	X
	Fishing/farming equipment ownership	↓	→	→	↓	→	↑	X	↑	X	→	X
	Livestock ownership	↓	→	↓	X	↓	↓	↓	↓	X	X	X
	Stored rice/fish	↓	↑	↓	X	↓	↓	↓	X	↑	↑	X
	Reliable flood warning	↓	↓	↓	↓	X	↓	X	X	X	↓	X
Natural capital	Agricultural land ownership	↓	X	→	X	↓	↓	↓	↓	↑	↑	↑
Social capital	Non-local income	↓	↓	↓	↓	↓	→	↓	↓	↓	X	↓
	Male head	→	X	↓	X	X	↓	↓	X	X	X	X
	Proximity to road	↓	↓	X	↓	↓	X	X	X	↓	↓	↓
	Association membership	↓	→	↓	↓	X	↓	X	→	X	X	X
Other	Asset diversity	↓	↓	→	↓	↓	↓	↓	→	→	↑	↑
	Income diversity	↓	→	→	→	X	→	X	→	X	X	X
	Per capita income	X	↑	↓	X	X	↓	X	↓	↑	X	X
	Adaptive capacity score	↓	→	↓	↓	↓	↓	↓	→	X	→	X
	***Susceptibility* score**	↓	↓	↓	↓	↓	↓	↓	↓	↓	↓	↓

↑= variable score increases with *susceptibility*, ↓= score decreases with *susceptibility*, →= no linear relationship with *susceptibility*, X = no relationship with *susceptibility.* Paler coloured symbols used to indicate variables that are not determinants of the *susceptibility* metric. Based on statistical analysis between *Susceptibility* classes, data can be found in Table 8 and Supporting information (S1.3)

4. Discussion

Mean *susceptibility* to hydrological change varied between countries and ecozones in the LMB, implying that strategies to reduce vulnerability at the household level should be tailored to national and socio-ecological context. The factors to prioritize depends at which scale a policy or intervention can be implemented because the determinants of relative *susceptibility* are scale-specific. Policy design must take account of wider environmental and socio-economic factors to avoid irrelevance or even increasing vulnerability. While our index-based approach may be simple, in the face of uncertain hydrological change in the LMB and with a rich household-level dataset, it provides a metric that can help provide guidance for precautionary policies to reduce livelihood *susceptibility*.

4.1 Need for vulnerability assessments in the Lower Mekong Basin

4.1.1 Full cost accounting of hydrological and ecosystem services

Increasing energy demands and improved relations with China have caused a renewed interest in hydropower development in the LMB and in this context the demise of fisheries has been framed as a necessary and inevitable trade-off (Molle, Lebel & Foran, 2009). Without promoting adaptation to environmental change, development of the Mekong's hydropower resources may, in fact, result in the loss of livelihood for millions of people (Dugan *et al.*, 2010). The research community is thus calling for a new narrative in which it is recognized that the sustainable management of Mekong fisheries can actually contribute to economic development (Béné, 2009; Friend, Arthur & Keskinen, 2009; Friend & Blake 2009). This new narrative requires the value of fisheries and the wider ecosystem services of LMB aquatic ecosystems to be accounted for and reflect the value they provide to a full range of beneficiaries which, in the past, they have not (Molle, Lebel & Foran, 2009). As demonstrated by the controversial Pak Mun Dam in Thailand, failure to account for livelihood impacts of hydropower development can incur high financial compensation costs (Foran & Manorom, 2009). Quantifying relative *susceptibility* can help inform a no-regrets approach, enhancing adaptive capacity and helping to mitigate negative impacts in the near-term will reduce future vulnerability and have long-term benefits (IPCC, 2014; Sok & Yu, 2015). Further, assessing current *susceptibility* is pertinent, as addressing current conditions and risk is often an immediate policy priority (Lim & Burton, 2005).

4.1.2 Scale of analysis

Understanding the distribution of impacts across geographic regions and social scales is essential for assessing costs and developing effective adaptation strategies (Adger

& Kelly, 1999; Miller, 2014; Moss, Brenkert & Malone, 2001). Vulnerability manifests at household and community levels as a result of internal factors and wider political, economic and institutional conditions (Gupta *et al.*, 2010; Smit & Wandel, 2006). To develop practical and feasible mitigation strategies, vulnerability must be understood at the household level (Eriksen & Kelly, 2007; Smit & Wandel, 2006), but should be framed within a national context to be compatible with decision-making processes (Adger, Arnell & Tompkins, 2005; Bastakoti *et al.,* 2014; Kelly & Adger, 2000).

To account for the multiple levels of LMB governance and the non-localized and interactive nature of hydrological change, impact assessments should provide information at a range of scales (Keskinen, 2008). The MRC is an intergovernmental organization that acts as an advisory and facilitating platform for the management of the Mekong River. Ultimately, however, individual countries have decision-making sovereignty and the interests of riparian states often conflict (Ratner, 2003). Local decision-making and governance structures also affect resource management. Our analysis presents insights into relative household *susceptibility* at national and ecozone scales. This provides a first step in a continuum of assessments that could help identify key factors impacting household *susceptibility* and vulnerability from local- to national-scale.

4.2 Differences across geographic scale

The use of different geographic scales and areas in this study highlight how index outcomes depend on the context in which relative *susceptibility* is assessed. That is, the findings at different scales and in different regions reflect variations in underlying economic, social, political and environmental conditions. Households in Vietnam were more *susceptible* on average than households in the other countries. High agricultural dependency in both the freshwater and saline regions of Vietnam reflected extensive rice cultivation and aquaculture. The delta is the most important agricultural region of the country (MRC, 2010a); from a total of 3.0 m ha of agriculturally productive land in the delta, by 2010 a total of 1.9 m ha were dedicated to rice crops and 0.5 m ha used for aquaculture (MRC, 2012).

Households in the northern highlands of Laos were particularly *susceptible*. One notable trait in Laos in general, and in the highlands in particular, was that homes are located far from roads. Laos has a very low road density of 6.1 km per 1,000 people and only 13.6% of roads are paved (ADB, 2010). Provincial and rural roads are often impassable in the wet season (Menon & Warr, 2008).

Although households in Thailand had the lowest mean *susceptibility* they scored lowest for a number of human asset variables (*household size, number of working-age members, non-elderly head*). Fertility policies in Thailand since the 1960s caused rapid population decline and, consequently, smaller households and an aging population; due to fears of labor shortages policies now focus on population stabilization (Prachuabmoh & Mithranon, 2003). Households in the north of Thailand are more *susceptible* than households in the south as they are more likely to source income from water-dependent activities.

A particularly low proportion of households in both the Tonle Sap and Mekong mainstream regions of Cambodia contained any member educated beyond primary school level. Admission to lower secondary education is much lower in remote and rural areas than in urban areas (MoEYS, 2005). A weak education system and low levels of education in the generation who were teenagers in the mid-1970s is the long-term legacy of the Khmer Rouge (Ahrens & McNamara, 2013; De Walque, 2006).

4.3 Policy relevance

Adverse impacts of environmental change can be reduced through hazard mitigation and adaptation. Hazard mitigation reduces exposure; it has overarching effects on all exposed systems but sometimes requires international cooperation and benefits may not be felt for decades (Füssel & Klein, 2006). In contrast, adaptation focuses on enhancing factors internal to SESs that reduce vulnerability; it can have immediate effects but these are often localized (Füssel & Klein, 2006). By identifying how the internal factors that facilitate household adaptation are distributed across the LMB, the findings of the *susceptibility* assessment in this paper are directly relevant for informing adaptation policies

Vulnerability is a concept that incorporates value judgments regarding equity and social justice; some argue that those who have the capacity to reduce vulnerability should be considered accountable to do so (Eakin & Luers, 2006). However, policies aimed at reducing household livelihood *susceptibility* to hydrological change must be designed and implemented with caution to avoid causing net damage. The determinants of adaptive capacity are not independent and therefore policies that target a specific determinant in isolation may be ineffective (Smit & Wandel, 2006; Sok & Yu, 2015). Strategies may have negative, unintended consequences on other attributes of SESs (Settele *et al.*, 2014). Some forms of adaptation can act as trade-offs for other forms (Reed *et al.*, 2013) or lead to second-order adaptation (Birkmann, 2011), and the costs involved in implementing adaptation technologies may exclude the poor from the benefits (Anderson, 1995). Crucially, too, adaptation process may deplete a household's asset base and lead to a new and sometimes increased levels of vulnerability in the future (Adger, 2006; Chambers, 2006; Gaillard, 2010).

4.3.1 Policy options

At the basin scale in the LMB there are numerous ways in which policies could reduce HH *susceptibility*, examples include reducing income dependency on water resources and increasing the diversity of income sources. Strategically, policies that affect HH size, the gender of HH head, ownership of water supply or per capita income should not be prioritised because these factors do not determine which HHs are more vulnerable. At the national scale policies should address particular factors and target particular geographic regions to most effectively benefit the most vulnerable HHs. Within Laos and Thailand policies should seek to increase the adaptive capacity of HHs in the northern highlands, in Vietnam HHs in the saline region are the most vulnerable, whilst in Cambodia there is no difference in the relative susceptibility of HHs in different regions but the determinants of relative *susceptibility* differ. Policy options and limitations are discussed below.

We found that a high proportion of households derived income from water-dependent activities in the northern region of the Thailand section of the Mekong corridor. If there were suitable livelihood alternatives available, then, perhaps contrary to common belief, a shift away from traditional rural activities may not necessarily be a negative result for households (Bouahom, Douangsavanh & Rigg, 2004; Rigg, 2006). If policies in Thailand were to encourage households to reduce water-dependent income activities, complimentary strategies would be needed to ensure that the shift was economically and socially beneficial. New industries or economic development activities would need to be established or expanded to cope with the increase in labor availability and training may need to be provided so that unskilled workers were able to access these new livelihood strategies. Out-migration is a likely consequence of declining rural livelihood viability and may cause important changes in rural social structures (UN, 2002). Finally, water-dependent activities may have value beyond their income-generating potential, so shifting patterns of livelihood may also impact important subsistence and cultural values.

In northern Laos, the expansion of road networks to connect rural and remote villages may help increase household adaptive capacity as road access often helps decrease poverty incidence (Warr, 2010). Improved transportation may encourage labor mobility, enable rural households to access markets and other services, and create opportunities for households to seek financial support outside of their villages. Road networks may also improve communication and therefore increase access to associations and information such as flood warnings. However, past improvement of road networks in the region opened up international trade routes, causing a shift towards commercial farming of cash crops (which can increase deforestation and agricultural chemical use), an influx of seasonal Chinese laborers, increased land privatization, and inequitable distribution of profits among local people (Thongmanivong & Fujita, 2006). Reducing vulnerability in the Northern regions of Laos requires efforts beyond simply expanding road networks and may need to be accompanied with policies that fall well outside of the mandate of natural resource managers.

In Cambodia, enhancing human capital by increasing access to education above primary school level would have a strong effect on household-level *susceptibility* to hydrological regime change. Equitable access to, and improved quality of, education are existing policy priorities in Cambodia but poor attendance and high dropout rates remain challenges (Tan, 2007). The main reason that people aged 6-17 years in Cambodia do not attend school is that they must contribute to household income-generating activities (NIS, 2013). For education policies to be successful in Cambodia they must be accompanied by mechanisms that ensure the benefits of schooling outweigh the costs of attending; possible options include financial support to cover losses of labor input to family activities and improving the quality of education. Those mechanisms should help target the most marginalized households, who are typically the most vulnerable (Adger, 2006; Gaillard, 2010).

In Vietnam, reducing agricultural dependence on natural flooding and irrigation of Mekong water would, in theory, address the main determinant of relatively high levels of household *susceptibility*. However, water resources in the Mekong Delta are already heavily engineered and past experiences suggest further alteration of hydrological regimes may not be an effective solution (Käkönen, 2008). Starting in the 1980s, engineering projects focused on increasing and intensifying rice production by reducing dependence on natural conditions and reducing saline intrusion (Käkönen, 2008). One consequence has been major environmental degradation and a decline in aquatic biodiversity (Hoanh *et al.*, 2003; Reis, 2012). The focus on increasing rice production restricted livelihood flexibility and diversification, and undermined a boom in shrimp farming (Hoanh *et al.*, 2003). Water management policies in the delta now strive to accommodate both rice and aquaculture

(Käkönen, 2008). Arguably, the most environmentally and socially sustainable policy may be to reduce human interference in delta hydrology and encourage households to resume traditional adaptation strategies of diversified and flexible livelihoods (Käkönen, 2008).

4.4 Study limitations

4.4.1 Exposure

The use of different modeling approaches (Kingston, Thompson & Kite, 2011) and uncertainty of future scenarios (Kummu & Sarkkula, 2008; MRC, 2010b; Räsänen *et al.*, 2012) has led to a variety of predictions for hydrological change in the LMB; some common themes have arisen. Climate change is expected to cause an increase in average temperatures and precipitation (Eastham *et al.*, 2008) but the effects are likely to be non-linear (Kingston, Thompson & Kite, 2011). Rising sea levels will interact with changes in basin hydrology and have cumulative effects in the Mekong Delta (Vastila *et al.*, 2010). Damming of the Mekong mainstream and its tributaries will affect hydrology, biodiversity and productivity (Kummu & Sarkkula, 2008; Kummu & Varis, 2007; Ziv *et al.*, 2012). Hydropower is expected to have a greater influence on Mekong flow regimes than climate change by smoothing out natural flood pulses (Lauri *et al.*, 2012). However, the magnitude and distribution of impacts is dependent on the combination of locations and capacities of hydropower developments (Ziv *et al.*, 2012), and reservoir operations (Räsänen *et al.*, 2012). The complex interactions between different causes of hydrological change in the LMB are not well understood (Lauri *et al.*, 2012). In the context of this study, exposure scenarios would still be highly speculative and focusing on a single hazard in isolation would be unrepresentative of future hydrological conditions (Keskinen *et al.*, 2010).

Increasing our knowledge regarding system sensitivity is, however, one part of the solution for designing pro-active policies that ultimately reduce harm and facilitate transformations towards SES sustainability (Chapin III, Fole & Kofinas, 2009). As research continues to reduce the uncertainty surrounding hydrological change in the LMB, it should be possible to develop an exposure vector to compliment the *susceptibility* index. In the context of environmental change, however, efforts must focus on improving management capacity for dealing with the unexpected as well as reducing uncertainty (Berkes, 2007; Birkmann, 2011; Cutter, 2003). The index of *susceptibility* offers a widely useful and adaptable approach that can inform precautionary policies (Adger & Kelly, 1999).

4.4.2 Index construct

The widely recognized shortcomings of using index-based approaches to assess vulnerability are applicable to the assessment of *susceptibility* used in this study. Firstly, combining multiple indicators to form a single metric of vulnerability can mask important detail and variation (Adger, 2006; Alwang Siegel & Jorgensen, 2001). However, a common metric provides a way to assess relative vulnerability and helps science to be translated into policy. As in virtually all indicator-based systems designed to support real-world management decisions, there are inherent trade-offs between the capacity to develop scientifically credible models that describe complex SESs and to communicate information to decision-makers in a usable form.

Secondly, there is no widely agreed set of vulnerability indicators (Vincent, 2007). Which factors affect vulnerability and their relationships with vulnerability (i.e., direction and weighting) are context dependent, so a "one-size fits all" approach is unsuitable (Chambers, 2006; Cutter, 2003; Gaillard, 2010). Indicators may be selected according to deductive, inductive, normative, or "non-substantial" arguments (Hinkel, 2011). The subjectivity of indicator choice and function can introduce bias to the metric (Brooks, Adger & Kelly, 2005).

Thirdly, there is no agreed function for the index construct itself (Adger & Vincent, 2005). Index construct should arguably be determined by the context of the assessment (Adger *et al.*, 2004) and therefore informed by local knowledge. The widely adopted IPCC definition of vulnerability does not specify the how the vectors of sensitivity, adaptive capacity, and vulnerability should be combined. As Hinkel (2011) highlighted, this has led to the interaction and weightings of those concepts often being overlooked.

Fourthly, because there is no single proxy for vulnerability (Eakin & Bojorquez-Tapia, 2008), indices cannot be used to identify the determinants of vulnerability (Smit & Wandel, 2006); indices can only be used to identify which factors significantly affect relative scores within the constructed metric. Factors that determine vulnerability can be empirically identified through revealed vulnerability, for example using recall to assess a household's socio-economic status prior to exposure to a hazard, and to regress such variables against loss and recovery times (Birkmann & Fernando, 2008). However, recall is unreliable except over relatively short timeframes.

Finally, it is not possible to validate a vulnerability index. How vulnerable a household was to a past hazard cannot be used to validate an index constructed on present day data because present status is inclusive of adaptation or recovery implemented in response to the past hazard (Adger, 2006; Vincent, 2007). Following exposure to a hazard, it may be possible to assess whether index predictions were correct, but it is likely that the impacts of, and responses to, the hazard will cause changes in the system itself, perhaps making the index unreliable for predicting future responses.

4.4.3 Dataset

There were a number of shortcomings in the dataset. Most importantly, data on two key adaptive capacity indicators, housing quality and human health, were absent (Vincent, 2007). The survey also did not collect information on household perceptions of hazards (Anderson, 1995), knowledge of adaptation strategies (Füssel & Klein, 2006), or willingness to adapt. Therefore the index we develop was indicative only of potential ability to adapt (Adger & Vincent, 2005; Brooks, Adger & Kelly, 2005; Luers *et al.*, 2003). Due to a lack of community-level data collected in the SIMVA 2011 survey, it was not possible to include community-level variables that could also have a strong influence on the adaptive capacity of households within particular communities (e.g., community infrastructure, services, topography, social networks, etc…).

4.5 Future efforts

Ongoing research conducted by the MRC should in the near future provide data needed to develop more robust and comprehensive assessments of *susceptibility* to hydrological change. The SIMVA surveys are scheduled to be conducted periodically over the coming years; SIMVA 2014 was recently completed. The data collected in SIMVA surveys are cross-sectional and so provide a snapshot of household vulnerability at the time of data collection (Chaudhuri, 2003). Carrying out susceptibility assessments with data from each SIMVA cohort could thus provide a profile of changing relative *susceptibility* across the region. The data collected in SIMVA 2014 included village-level information regarding infrastructure, topography, and adaption activities. As a result, the next round of *susceptibility* assessments should be able to provide more in-depth cross-sectional information about household level adaptive capacity. Further, better community-level information may allow much more detailed assessments of how *susceptibility* varies across scales down to the provincial and district levels. In addition to the SIMVA surveys (conducted by MRC Environment Program), the MRC Fisheries Program is conducting survey research on the socio-economic impacts and social-implications of reduced capture fisheries in the LMB. That project is closely aligned with SIMVA, using a selection of the communities sampled in SIMVA 2014, but collecting detailed data on livelihood strategies, fish capture and consumption, and asset ownership, as well as household perceptions on the attractiveness and implementation of adaptation and adjustment strategies. The combination of the MRC datasets may ultimately allow the development of Ricardian models (Mendelsohn, Nordhaus & Shaw, 1994) to assess the economic impacts of hydrological regime change in the LMB.

5. Conclusion

The assessment of *susceptibility* can contribute to the design of policies aimed at reducing the socio-economic impacts of hydrological change and developing more sustainable SESs. It is also an important step towards more comprehensive vulnerability assessments. Understanding household livelihood vulnerability to hydrological change is crucial in the LMB; millions of people are dependent on the hydrologic and ecosystem services associated with the Mekong River, which are threatened by various anthropogenic activities. Despite limitations inherent in the use of simple indices, the rich MRC dataset provides a way to identify which type of households are more *susceptible* to changes in the hydrological systems of the Mekong River. We found that the determinants of relative *susceptibility* vary according to the spatial scale of analysis and between countries and ecozones. Hence, strategies to reduce vulnerability should be tailored according to geographic context.

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Current status and future needs of economics research of inland fisheries

Abstract

Inland capture fisheries (ICFs) provide ecosystem services – fish for food, livelihoods, and recreation – to people and therefore have an economic value. Economic valuation can inform the sustainable management of ICFs and ensure they are recognised in trade-off analysis and decision-making. This study assesses existing ICFs economic research to identify knowledge gaps. Bibliographic databases were searched for suitable peer-reviewed articles. The selected studies (n=75) were analysed for coverage, valuation methodologies, and value metrics. A majority of existing studies value recreational ICFs in developed countries. Studies have employed a wide range of valuation methodologies and therefore provide a variety of economic values measured at different units and scales. This study highlights the need for a greater quantity of ICFs economic research that covers a representative sample of ecosystems and fishery types globally. Best practice recommendations are made. These aim to ensure future ICFs research generates economically credible and comparable values.

1. Introduction

Inland capture fisheries (ICFs) are the activity through which fish are extracted from inland natural or constructed waterbodies containing fresh or brackish water (Welcomme *et al.,* 2010). ICFs provide income, nutrition, and leisure opportunities, and are an important scientific resource (Holmlund & Hammer, 1999). ICFs are classified as commercial (where the main objective is to generate profit through trading fish catch in formal markets), recreational (where catch may be released or used for personal consumption but is not traded) or subsistence (small-scale exploitation where fish are mainly consumed by the fishing household or given away, bartered, or sold in local markets).

Global production from ICFs reached 11.6 million tonnes in 2012 (FAO, 2014) but it is widely accepted that this estimate is based on unreliable data and likely to be a significant underestimate (Allan *et al.,* 2005; Welcomme *et al.,* 2010; Mills *et al.,* 2011; FAO, 2014; Bartley *et al.,* 2015; de Graaf *et al.,* 2015). ICFs are often poorly monitored and their social, economic and cultural importance is undervalued (Welcomme *et al.,* 2010; Arthur & Friend, 2011; Mills *et al.,* 2011; FAO, 2012; Bartley *et al.,* 2015). A number of practical complexities explain why understanding of ICFs is limited: ICFs are located in many different water bodies with non-distinct landing sites, they are exploited by a large and diverse population of fishers; there is high variation in seasonal catches and in the species composition of catch; ICF stocks are heavily influenced by external factors; and a large proportion of catch is not traded on formal markets (Arlinghaus, Mehner & Cowx, 2002; Welcomme *et al.,* 2010; FAO, 2012). The common assumption that fishing is a last-resort livelihood activity of the poor means that the sustainable management of ICFs is rarely a policy focus; investment focuses on projects that are deemed to contribute more directly to economic development (Welcomme *et al*., 2010; Arthur & Friend, 2011; Béné & Friend, 2011; FAO, 2014).

The need to recognize the full suite of benefits provided by ICFs, so that they are accounted for in decision-making processes, is being voiced with growing urgency (Welcomme *et al.,* 2010; UNEP, 2010; Beard *et al.,* 2011; Cowx & Portocarrero Aya, 2011; Brummett, Beveridge & Cowx, 2013). ICFs are directly dependent on inland aquatic ecosystems that are threatened by wide-ranging anthropogenic activities and are often already highly degraded (Vörösmarty *et al.,* 2000; Johnson, Olden & Vander Zanden, 2008; Kundzewicz *et al.,* 2008; Cooke, Paukert & Hogan, 2012; Auerbach *et al.,* 2014). Fishing itself comprises part of the problem (Allan *et al.,* 2005; Raby *et al.,* 2011; Cooke & Murchie, 2013) but external stressors also greatly disrupt hydrological patterns and cause environmental degradation and habitat loss (Allan *et al.,* 2005; Welcomme *et al.,* 2010; Beechie *et al.,* 2013;

van Vliet, Ludwig & Kabat, 2013), a pattern that will only intensify (Ficke, Myrick & Hansen, 2007; FAO, 2012).

The income, nutritional, and cultural benefits gained from fish landed by ICFs are final ecosystem services of aquatic ecosystems. Ecosystem services are "the aspects of ecosystems utilized (actively or passively) to produce human well-being" (Fisher, Turner & Morling, 2009: 645); final ecosystem services – commonly classified as provisioning, regulating and maintenance, and cultural – are those that provide direct benefits for which people (and firms) would be willing to pay for an increase in (Johnston & Russell 2011). Considering whole ecosystems and the services they provide (the 'ecosystems approach' to environmental management) has, over the past decade, been widely embraced by the international science community (Carpenter *et al.,* 2009; Braat & de Groot, 2012; Mooney, Duraiappah & Larigauderie, 2013) and for government and international assessments and policy-making (e.g., MEA, 2003; CBD, 2004; UK National Ecosystem Assessment, 2011).

A key feature of the ecosystem approach is the economic valuation of final ecosystem services (Heal *et al.,* 2004; Fisher *et al.,* 2008; Johnston & Russell, 2011; Bartley *et al.,* 2015). Valuation provides a common monetary metric for the multitude of factors that influence household's (and firm's) quality of life (and profitability). Willingness to pay (WTP) can be used to assess the trade-offs that individuals and households are willing to make for improvements in environmental quality. WTP, as a theoretically-based measure of economic well-being known as consumer surplus (CS), has been widely adopted by decision-makers and governments to support cost-benefit analysis (CBA) weighing the relative merits of conservation policies and interventions (Howarth & Farber, 2002; Heal *et al.,* 2004). CS has proven powerful in its predictive capacity and can be thought of as an indicator of humans' willingness to trade money for other attributes describing, in the ICF case, the quality and productivity of the fishing experience or event (see Rudd, Folmer & van Kooten, 2002). For firms, their measure of benefit, producer surplus (PS), is functionally the profit they make by catching and selling fish. CBA sums changes in both CS and PS over time to calculate whether the economic benefits of a change in ICF quality or productivity arising from an investment or intervention outweigh their costs or, alternatively, to calculate the costs of inaction.

The benefits derived from ICFs must be valued to enable the fish populations and aquatic ecosystems that support them to be incorporated into CBA and managed according to the ecosystem services approach. This paper reviews existing inland fisheries economics literature to summarize the state of knowledge and identify knowledge gaps that may limit using an ecosystem approach in fisheries management. The objectives are to: identify existing coverage of ICF valuation research across geographic regions, ecosystem types and fishery classifications; to quantify the range of valuation methodologies employed; and to assess the consistency with, and credibility of, the economic values generated. This paper then makes best practice recommendations for future ICFs economic research based on the need for that research to provide certain basic economic information necessary to support an ecosystem approach in inland fishery management decisions.

2. Methods

2.1 Study selection

The literature used in this review was constructed drawing on general procedures developed for systematic reviews (Haddaway & Pullin, 2014). However, as an exploratory study, it did not follow the entire process that involves an analysis of study biases and treatment effects; it is therefore best considered a structured review. Structured and systematic reviews are increasingly popular in environmental science (Liquete *et al.,* 2013; Bilotta, Milner & Boyd, 2014; Hejnowicz *et al.,* 2014) and help highlight the state of current research practices, knowledge gaps, and future research needs.

Bibliographic databases *Econlit*, *Greenfile*, *Scopus*, *Science Direct*, and *Web of Science* were searched for studies that valued ICFs. *Google Scholar* was searched (using *Publish or Perish*) to supplement the academic database searches; only articles cited at least twice were retained. Search terms for the title, abstract and keywords of literature included [*freshwater* or *inland*], [*fisher** or *fishing*] and [*economic** or *socioeconomic* or *socio-economic*] (supporting information S2.1). Duplicates, studies written in a language other than English, editorials, conference proceedings, technical and consultancy reports, and books and book chapters were removed. Books and book chapters were not included as the information contained in them, if relevant, were usually also reported in the primary literature. While there is a substantial body of relevant economic research in the grey literature, including only peer-reviewed literature provided a cut-off that in theory should help ensure a minimum quality standard regarding the economic approaches used.

The initial sample of articles from all searches was collated in *Endnote*. Articles were then progressively screened at the title, abstract, and full text levels, ensuring their relevance to the valuation of ICFs. To be included studies must have used primary data to calculate the economic value of ICFs and the value for ICFs had to be distinguishable from the value of other activities or ecosystem services. Only studies using primary data were included to avoid the risk of double counting and because studies that used secondary data often did not provide sufficient detail to describe the original methodology used. To ensure full coverage of the literature, a snowballing strategy was used to further identify candidate publications from the citations in articles that reached the full text screening stage and to publications that had been excluded during title and abstract screening, but that were heavily referenced within other ICF studies (e.g., in other review papers).

2.2 Study coding

Information coded from each article included year of publication, central research focus, country and waterbody of study location, fishery type, data collection method, type of economic value calculated and unit of measure. For studies with a broad focus, only information directly related to the valuation of ICFs was coded.

3. Results

3.1 Study characteristics

Of 3,939 documents identified in the bibliographic database searches, 44 articles were retained for analysis after three stages of screening and a further 31 articles were identified during the snowballing process (Figure 2.1) (see supporting information S2.2 for full list of articles). The research focus of the articles fell into three distinct categories: 44% specifically valued ICFs; 32% focused on ICFs but not specifically on economic valuation; and 24% included ICFs as one component in wider research. Valuation was most commonly used to quantify the impact on fisheries from a change in environmental or management factors (32%) or the contribution of fishing to livelihoods (27%). The earliest study retained was published in 1987 (Marchand, 1987); there was a sharp increase in the number of ICF valuation studies from the year 2000 onwards (Figure 2.2).

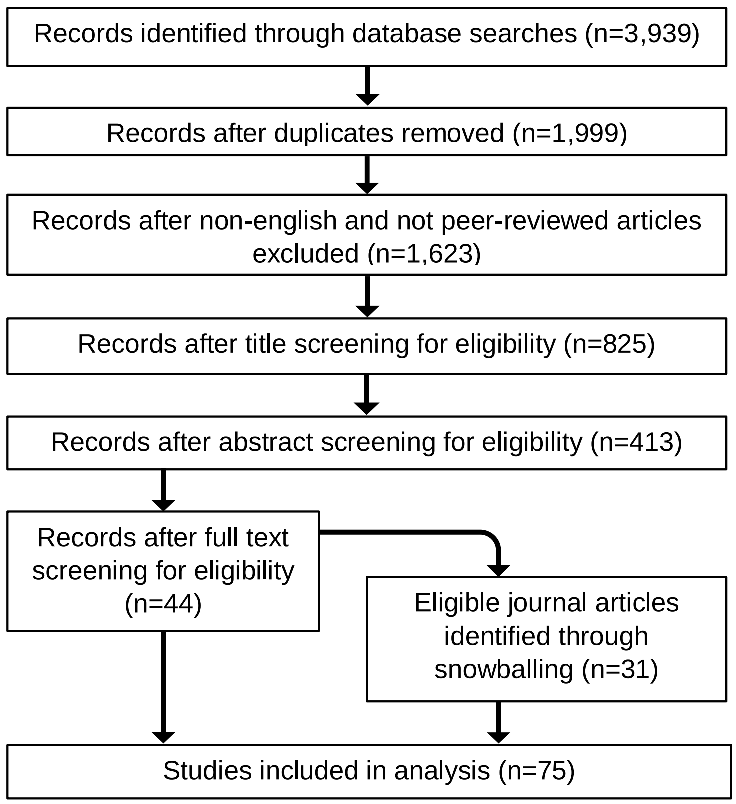

Figure 2.1 Flow diagram of study selection and screening process.

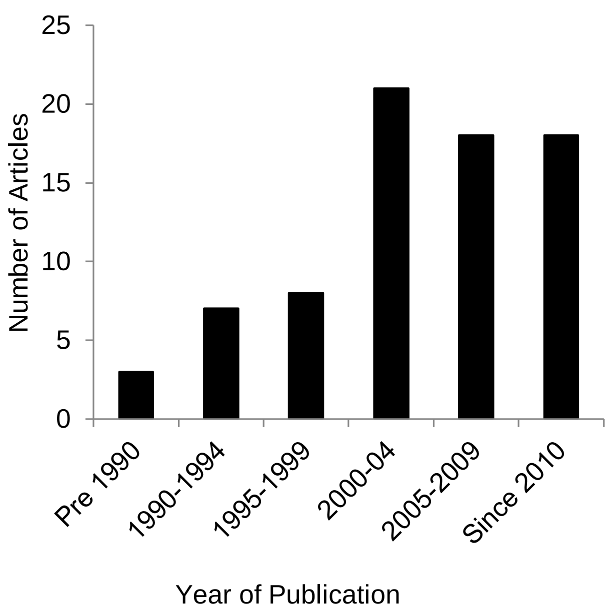

Figure 2.2 Number of studies by year of publication

3.2 Study Distribution

3.2.1 Geographic region

ICFs were valued in 31 countries across 6 continents, primarily (63%) in developed countries. North America studies comprised 33% of the total and, of those, 92% were from the USA. ICFs were valued in 12 different European countries, accounting for 25% of the retained articles. South American ICFs were valued across 14 studies (19%) in four different countries. Fisheries in Brazil and Peru were each valued in six studies, accounting for 86% of the South American research. ICFs in Asia were valued in 12% of studies across seven different countries but these did not include any articles from China or India. Five African studies each valued ICFs in different countries. Finally, two studies valued ICFs in Australia and one in New Zealand.

At the time of publication, the lead author for 77% of the retained studies was based in a developed country (based on World Bank classifications). Some 39% of studies in developing countries were authored by researchers based in developed countries. Of the 47 studies located in developed countries, for all but one (Shrestha, Seidl & Moraes, 2002) the lead author was based in the country of study. Comparatively, in only 11 (De Camargo & Petrere, 2001; Walter & Petrere Jr, 2007; Weyl *et al.,* 2007; Navy & Bhattarai, 2009; Rana *et al.,* 2009; Akwetaireho & Getzner, 2010; Freire, Machado & Crepaldi, 2012; Smederevac-Lalic *et al.,* 2012; Hallwass *et al.,* 2013; Chesoh & Lim, 2014; Le Xuan *et al.,* 2014) of the 28 studies carried out in developing countries was the lead author based in the country of study.

3.2.2 Fishery types

Many more studies valued recreational fisheries (61%) than subsistence (32%) or commercial (13%) fisheries (n.b., five studies valued multiple fishery types). Most recreational fishery studies were located in North America (50%) and Europe (37%). In contrast, studies on subsistence fisheries were mostly from Africa (38%), South America (21%) and Asia (33%) (Figure 2.3). These findings reflect how as economies develop ICFs switch from being subsistence to recreational (Arlinghaus, Mehner & Cowx, 2002). Three of the commercial ICFs valued (Kvist *et al.,* 2001; Coomes, Barham & Takasaki, 2004; Moreau & Coomes, 2008) focused on Peruvian aquarium and ornamental fisheries.

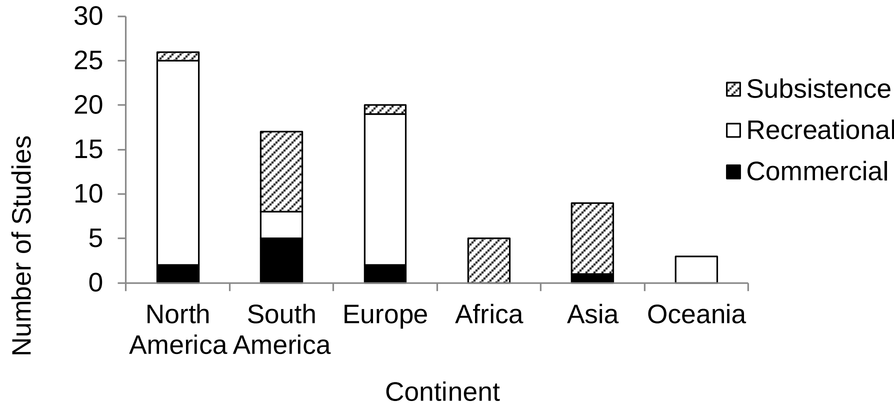

Figure 2.3 Number of studies valuing commercial, recreational or subsistence fisheries according to region.

3.2.3 Waterbodies

The number of studies that valued ICFs in each waterbody type is summarized in Figure 2.4. The majority of articles (51%) valued river fisheries; other studies focused on a variety of waterbodies, including lakes, reservoirs, floodplains/wetlands, and canals. In 15% of articles the type of waterbody studied was not stated. Some specific waterbodies were valued in multiple studies. In Peru, for example, the coverage of the six articles was geographically concentrated: three studies valued ICFs of the Maranon and Ucayali Rivers (Kvist *et al.,* 2001; Takasaki, Barham & Coomes, 2001; Coomes *et al.,* 2010), one valued the fishery in only the Maranon River (Takasaki, Barham & Coomes, 2010), and another

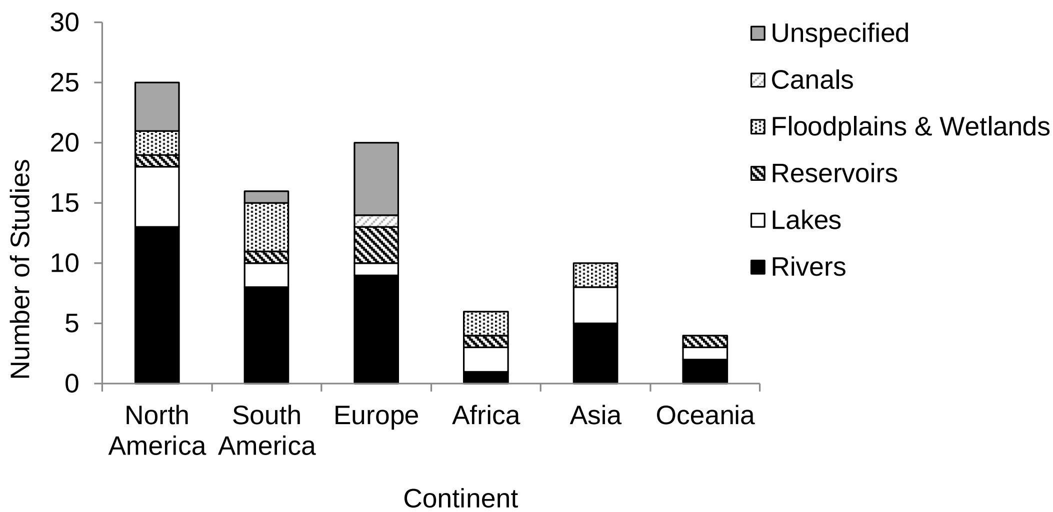

Figure 2.4 Number of fishery studies in each waterbody type according to region.

valued the fisheries in the Ucayali and Tapiche rivers (Moreau & Coomes, 2008). In Asia, all three studies in Cambodia valued ICFs of the Mekong River (Ringler & Cai, 2006; Israel *et al.,* 2007; Navy & Bhattarai, 2009) and both studies in West Sumatra were of the fisheries in Lake Singkarak (Yuerlita & Perret, 2010; Yuerlita, Perret & Shivakoti, 2013).

3.3 Valuation method

Studies were categorized according to the economic valuation method they employed (Table 2.1). The Travel Cost Method (TCM) and "reporting values as given by respondents" were the most commonly used. Valuation of recreational fisheries used the most diverse range of methods including TCM, contingent valuation method (CVM), choice experiments (CE), permit prices, reported values stated by respondents, and economic impact calculated from input-output models. Only three of the papers that used CVM estimated non-use values (Baker & Pierce, 1997; Peirson *et al.,* 2001; Toivonen *et al.,* 2004). Four studies (Johnson & Adams, 1988; Train, 1998; Henderson, Criddle & Lee, 1999; Ringler & Cai, 2006) used the economic value of ICFs in scenario modelling to estimate how changing conditions would impact fishery values.

Table 2.1 Total number of papers that employ each valuation methodology, number of articles using each valuation methodology according to fishery type, and number of articles that use scenario modelling

Valuation Method	Studies	Fishery Type: Commercial	Fishery Type: Recreational	Fishery Type: Subsistence	Scenario Modelling
Market price of fish[a]	14	6	0	10	0
Given by respondents[b]	18	4	6	9	0
Permit price[c]	2	0	2	0	0
Gains – losses[d]	6	2	0	5	1
Travel cost [e]	20	0	20	1	1
Contingent valuation[f]	13	0	13	0	1
Choice experiment[g]	3	0	3	0	0
Input-output modelling[h]	6	0	6	0	1

[a] fish valued according sales price; [b] survey respondents asked to state income from fish or value of fish consumed; [c] price paid for angler permits used to value fishery; [d] fish valued as a combination of gains (i.e. income) minus a combination of losses (i.e. expenditures); [e] fishery valued as the price anglers are willing to pay to travel to the fishery; [f] respondents asked to state their willingness to pay or accept compensation for a change in the fishery; [g] fishery valued according to respondent's choice preference from a set of hypothetical scenarios that include different associated costs; [h] assessment of the impacts of angler expenditures on national or regional economies.

3.4 Value types

Seven different types of value were used in ICF valuation studies (Table 2.2). Credible valuation for CBA purposes requires net or surplus values. CS is the difference between consumers' WTP and the market price paid for a recreational fishing trip (note that suppliers' revenues from goods and services sold to recreational anglers captures only a portion of overall WTP - see Rudd, Folmer, H. & van Kooten, 2002). Producer surplus (PS) for suppliers is gross income less costs of production and allowances for returns to management and risk; net income is usually an acceptable proxy for PS. A variety of other ad hoc measures and proxies of economic value (market value, gross income, economic impact and others) were coded as they emerged in the selected articles.

Table 2.2 Type of economic measure to value inland capture fisheries and examples of the types of values given by the studies (provided to illustrate the range of measures, terminology, and values).

Value Type	Studies	Example of values
Consumer Surplus	33	Average anglers WTP (in addition to permit price) per year for increase in wild fish €11.4-18.0 depending on target fish species (Changeux *et al.* 2001)
Producer Surplus	2	Net income per trip of the Santarem fleet ranges from US$53-520 according to boat size (Almeida *et al*. 2001)
Market Value	10	Total estimated annual value of fish yield from all 10 dams studied in the North West Province of South Africa 5,051,000R (Weyl *et al*. 2007)
Net Income	6	Net household income from fishing for rural households in Siem Reap province from motorised fishing 4,380,680/6,651,880 riels during wet/dry season (Israel *et al*. 2007)
Gross Income	12	Annual earnings from aquarium fish US$283 for independent fishers and US$6,890 for expedition fishers on Ucayali and Tapiche rivers (Moreau & Coomes 2008)
Economic Impact[1]	10	Rod fisheries in the Spey catchment generate a gross annual output of £12.6 million, salmon and sea trout fisheries contribute £11.6 million of this (Butler *et al.,* 2009)
Other	2	In 2001 annual state expenditure on inland fishery programs US$432,000-39,276,052 and income from programs US$32,000 to $30,000,000 (Gabelhouse 2005)

[1] Includes expenditure values.

Thirty-three articles valued CS; 11 of those (Johnson & Adams, 1988; Parsons & Kealy, 1992; Pendleton & Mendelsohn, 1998; Willis & Garrod, 1999; Changeux, Bonnieux & Armand, 2001; Lupi *et al.,* 2003; Paulrud & Laitila, 2004; Schuhmann & Schwabe, 2004; Dorow *et al.,* 2010; Olaussen & Liu, 2011; Beville, Kerr & Hughey, 2012) did not provide baseline values but stated only marginal changes in CS associated with a change in the fishery. All studies that calculated CS valued recreational fisheries; one study (Henderson *et al.,* 2003) valued CS for both subsistence and recreational fisheries. One study used PS to value commercial fisheries (Almeida, McGrath & Ruffino, 2001). Another two studies, categorised as "other," collected income and profit values associated with organizations that managed recreational fisheries (Williams & Moss, 2001; Gabelhouse, 2005). Fishers' income was measured in studies of commercial and subsistence fisheries; 12 studies measured gross income and six measured net income. The market value of catch (i.e., sales revenue) was the only economic value given in 10 studies, one of which (Acuña *et al.,* 2013) provided marginal values associated with a change in fish population but no baseline values. Another 10 studies (Kircheis, 1998; Marta *et al.,* 2001; Wedekind, Hilge & Steffens, 2001; Chen, Hunt & Ditton, 2003; Henderson *et al.,* 2003; Butler *et al.,* 2009; Munn *et al.,* 2010; McKean, Johnson & Taylor, 2011; Freire, Machado & Crepaldi, 2012; Perez-Bote & Roso, 2014) provided only economic impact values such as angler expenditures and fishery contribution to local or regional economies.

3.5 Units of measure

Across the literature the value of ICFs are given for a variety of different units of measure and time dimensions. The most commonly used units were human-based (69%), including values of fish or fishing per person, household, group, or for a particular population. Eight studies (11%) used spatial units of measure in their valuation (e.g., per hectare or kilometre of stream). Three studies (4%) calculated ICF values for a region or waterbody but did not calculate those values with a spatial unit of measure. The time dimension used ranged from values per day to social benefit gained over a period of 50 years (van Vuuren & Roy 1993); annual values were most common (33%).

4. Discussion

4.1 Synthesis of findings

This structured review found that, to date, a majority of inland fisheries economics research has been carried out on river and lake recreational fisheries in developed regions, mostly in the USA. ICFs have been valued using a wide variety of methodologies; the most common methodology was the TCM used to value recreational fisheries. Across the literature, the value of ICFs are given according to a variety of different units of measure. The majority of studies valued fisheries according to a human measure, for example per person per day, which reflects the value of a fishery according to its usage. The shortcomings of the existing body of literature and how to address them are discussed below.

4.2 Challenges arising

The sustainable management of fisheries is limited by a lack of research funding (Arlinghaus, Mehner & Cowx, 2002). Primary economic valuation research is expensive, so there has been much interest in the use of ‘benefits transfer’ (Brouwer, 2000; Bergstrom & Taylor, 2006; Navrud & Ready, 2007; Plummer, 2009). Benefits transfer entails using estimates of ecosystem service CS derived in one place or situation in other contexts. While benefits transfer estimates are subject to a variety of biases (Johnston, 2007; Rosenberger & Johnston, 2009) and can exhibit notoriously high transfer errors (±100% is common and much higher is possible), the numbers may still be useful for policy purposes (Navrud & Ready, 2007), especially with sensitivity analyses across key parameters (Akter & Grafton, 2010). Estimates help counter the tendency in CBA to assign a value of zero for all ecosystem services where valuation estimates have not yet been quantified (Liu *et al.,* 2011; Navrud & Ready, 2007). Recall that CBA sums changes in both CS and PS, so anything with a zero value is inconsequential in CBA. While it is widely recognised that some ecosystem services are difficult to value monetarily (Heal *et al.,* 2004; UK National Ecosystem Assessment, 2011), the qualitative text that often accompanies CBA calculations can be ignored if decision-makers focus simply ‘on the numbers.’ Benefits transfer thus requires there to be a body of available data that provides economically credible values for a variety of fisheries.

4.2.1 Knowledge gaps

In the existing literature on the economic value of inland fisheries there are large gaps in the valuation of certain ICF types and regions. Relative to their size and productivity, ICFs in developing regions are vastly under-represented in the valuation literature. This geographic under-representation is synonymous with an under-representation of subsistence fisheries. Data from developed regions are unsuitable as secondary data for valuing ICFs in developing regions because of differences in the fisheries and the wider social, environmental and economic context (Plummer, 2009). Thus the existing distribution of research poses challenges for employing the ecosystem services approach to fisheries management.

The Ramsar Convention classifies inland wetlands into 30 different types; of these, four do not support fish populations (wastewater treatment areas, salt exploitation sites) or support only stocked/farmed fish populations (aquaculture, human-made ponds). The Ramsar classification of wetland types (Ramsar, 2009) can be used to identify gaps in the coverage of ecosystems by existing ICF research (Table 2.3). There is poor coverage of all wetland types except permanent rivers/streams/creeks and permanent freshwater lakes. Although many studies provide economically credible values from their research, these are most often not reported according to a spatial unit of measure.

Table 2.3 Number of studies located in each wetland type according to the classification system for Ramsar sites (Ramsar, 2012) and the credibility and metric of the values

Wetland type	Economically credible value per spatial unit	Economically credible value, non-spatial unit	Spatial unit, not economically credible value	Neither economically credible nor spatial measure	Total studies for wetland type
Permanent inland deltas	x	x	1	x	1
Permanent rivers/streams/ creeks	3	19	1	13	36
Seasonal/ intermittent /irregular rivers/streams/ creeks	x	x	x	x	0
Permanent freshwater lakes	1	9	x	4	14
Seasonal/ intermittent freshwater lakes	x	x	x	x	0
Permanent saline/brackish/ alkaline lakes	x	x	x	x	0
Seasonal/ intermittent saline/brackish/ alkaline lakes and flats	x	x	x	x	0
Permanent saline/brackish/ alkaline marshes/pool	x	x	x	1	1

Table 2.3 continued

Seasonal/ intermittent saline/brackish/ alkaline marshes/pool	x	x	x	x	0
Permanent freshwater marshes/pools	x	x	x	2	2
Seasonal/ intermittent freshwater marshes/pools on inorganic soils	x	3	x	x	3
Non-forested peatlands	x	x	x	x	0
Alpine wetlands	x	x	x	x	0
Tundra pools	x	x	1	x	1
Shrub-dominated wetlands	x	x	x	x	0
Freshwater, tree dominated wetlands	1	x	x	1	2
Forested peatlands	x	x	x	x	0
Freshwater springs, oases	x	x	x	x	0
Geothermal wetlands	x	x	x	x	0
Karst and other subterranean hydrological systems	x	x	x	x	0
Irrigated land	x	1	x	x	1
Seasonally flooded agricultural land	x	x	x	x	0
Water storage areas	x	3	1	5	9
Excavations	x	x	x	x	4
Canals and drainage channels, ditches	x	1	x	x	1
Karst and other subterranean hydrological systems (human-made)	x	x	x	x	0
Unspecified/ general	x	5	x	5	10

4.2.2 Methodological diversity

The plethora of often *ad hoc* valuation methodologies and economic values used in inland fisheries economics research creates challenges relating to comparability and defensibility of valuation estimates. Some of the methodologies do not actually capture the net value of the fishery. Values must be economically credible to increase the likelihood of fisheries being incorporated into assessments and enhance the leverage that fisheries research has in CBA. The most common method employed to value subsistence and commercial fisheries was according to the market price of fish, which is not a measure of net value or benefit and does not, therefore, reflect the true value of fisheries (Rudd, Folmer & van Kooten, 2002). Input-output models, as used in some articles in this review, capture economic impact, which is the short-term effect of an activity on business revenues and income (Crutchfield, 1962). Economic impact is not a credible measure of the economic value – the sum of CS and PS – of a fishery and is therefore unsuitable for use in analysis such as CBA and benefits transfer. However, because job creation and business spin-off impacts are often important politically, input-output models can be useful for management agencies (Seung & Waters, 2006) where regional economic development is a political objective and for justifying their own budget allocations from scarce government financial resources.

4.2.3 Non-comparability of values

Fisheries are just one of many ecosystem services arising from wetlands and inland fish populations and so in decision-making processes the value of fisheries will not be considered in isolation. There is therefore a need for comparability and compatibility of ICF valuation estimates with values from other ecosystem services. As the ecosystem approach to environmental management has matured, ecosystem service valuation studies based on benefits transfer have increasingly sought to use spatially explicit values to assess changes in ecosystem service provision (Wilson *et al.,* 2005; Costanza *et al.,* 2006; Liu *et al.,* 2011; Brander *et al.,* 2012; Raheem *et al.,* 2012). Spatially measured economic values are transferrable and comparable, they can aid the mapping of ecosystem services (Fisher *et al.,* 2008; Liquete *et al.,* 2013), and can be used in spatial modelling to provide global estimates of the value of inland fisheries (de Graaf *et al.,* 2015). Primary valuation research on ICFs has, for a number of reasons, not entirely kept in step with this trend; very few studies used spatial units for fisheries valuation. Further, across all unit types, even within the broad categories used in this study, there were few compatible studies.

4.3 Recommendations for future research

4.3.1 Methodological approaches

Agreement on a study design that generates credible and widely useful values is needed for our understanding of ICFs to effectively inform their sustainable management. There is a lack of consensus regarding the most appropriate study design for ecosystem service valuation and applied economics journals often favour research that demonstrates theoretical or methodological advances, which may tend to discourage standardisation.

Valuation studies of recreational ICFs should employ theoretically-based stated preference models (CVM and CE) or a combination of stated and revealed preference (TCM) models. Although both stated and revealed preference models capture surplus values, revealed preference models should not be used alone if non-use values are likely play an important role in CBA; non-use values comprise an important component of total economic value because they reflect the wider value of a resource to society (Rudd, Folmer, H. & van Kooten, 2002). For ICFs, non-use values can be equal in magnitude to use values (Wilson *et al.,* 2005). For capturing the use-value of ICFs, revealed preference methods may, however, be preferable to stated preference methods because they reduce the risk of bias and are less resource intensive (Neill *et al.,* 1994; Brouwer, 2000). Of the stated preference methods, CEs may be considered preferable to CVM for ICFs research. CVM asks respondents to state their willingness to pay for a particular ecosystem service, whereas in CE respondents are asked to state their WTP for a number of hypothetical scenarios. CEs are less subject to bias, more efficient for collecting a wider breadth of information, and can reflect environmental and socio-demographic differences so generate values more useful for benefits transfer (Morrison *et al.,* 2002; Jiang, Swallow & McGonagle, 2005; Colombo, Calatrava-Requena & Hanley, 2007; Johnston, 2007).

Subsistence fisheries should ideally be valued using household level surveys. The net economic value of subsistence fisheries may be primarily comprised of either PS or CS, depending on context. Where households derive value from fish via sales revenue, a household production function approach can be used with net income acting as a close proxy for PS. Net income is sales revenue less the costs of the inputs to fishing, including labour and depreciation but excluding capital costs. Where households consume the fish directly, savings on food purchases, according to market price (i.e. replacement costs), can be used as an imperfect proxy for CS. Although a number studies included in this review used household level surveys to collect income data, many did not collect the information needed to calculate net income or CS. Cultural or intrinsic (non-use) values associated with a subsistence fishery, as with recreational fisheries, should be captured through stated preference methods whenever possible because peoples' opinions, rather than behaviour, are needed to value hypothetical changes in those types of non-use values. These can be incorporated into household surveys.

The net economic value of commercial fisheries is net profit, which is usually a good proxy for PS (and also accounts for returns to risk and management). Net profit is the revenue taken from the sale of fish less the fishing costs. The best way to collect reliable primary data is through cost and earning surveys of commercial fishers. However, these surveys can be very expensive and response rates can be low (e.g., DFO, 2007) making it challenging to collect reliable and thorough data for calculating PS. Although gross income is not a credible measure of economic value, it may, in the context of commercial ICFs, be the best valuation option at the present. Gross income data are relatively straightforward to collect and can be used as a proxy for net economic value on the assumption that profit or net income are roughly equal to 10% to 30% of gross income (e.g., DFO, 2006), depending on prevailing interest rates and 'normal' returns in risky resource harvesting activities. If estimating gross income based on official national or regional landing statistics, additional caution must be exercised. National fishery statistics are often considered to be unreliable (Allan *et al.,* 2005; Welcomme *et al.,* 2010; WorldBank, FAO & WorldFish Center, 2010; Welcomme, 2011; Pauly & Zeller, 2014; de Graaf *et al.,* 2015) due to issues of trust between regulators (governments) and the regulated (fishers) and due to monitoring difficulties. Whenever examining the economics of commercial fisheries, it is important to consider the possible effects of subsidization on profitability of firms that would otherwise be operating at an economic loss (Sumaila *et al.,* 2010).

4.3.2 Primary research prioritization

Research should strategically target fisheries that provide data to fill the crucial information gaps identified by this review, be widely useful in benefits transfer, and generate the most useful information possible (i.e., economic research on ICFs itself needs to be subject to a cost-effectiveness analysis – see Allen & Loomis, 2008). The valuation of fisheries in inland wetland ecosystems other than rivers and lakes should be prioritised. In many cases it is not appropriate to transfer values from one type of wetland to another type because of differences in hydrologic and geomorphic conditions, species composition, and biodiversity. Hence the valuation of fisheries in all types of ecosystems is needed to provide values suitable for benefits transfer between similar areas and to compare the value of the services provided by different types of ecosystems.

Geographically the greatest knowledge gaps are in the ICFs of developing countries, where ICFs are predominantly exploited for subsistence purposes. If it is assumed that funding for research usually originates from the country that the lead author was affiliated with at the time of publication (Liquete *et al.,* 2013), then the large proportion of studies carried out in developed regions reflects the availability of resources and the prioritisation of domestic research. Investment in primary research into developing country ICFs is needed because the sustainable management of subsistence ICFs can contribute to poverty alleviation, food security and environmental protection and so help achieve Millennium Development Goals (UNEP, 2010; Brummett, Beveridge & Cowx, 2013).

In recognition of the importance of ICFs, a number of existing efforts to encourage foreign investment and research into the status and value of ICFs are underway. Examples include the UK-based ESPA (http://www.espa.ac.uk/), an interdisciplinary programme that focuses on promoting sustainable environmental resource management (including fisheries) in developing countries, and the Canadian-funded Too Big to Ignore (http://toobigtoignore.net/) partnership that focuses on global small scale fisheries in both the marine and freshwater environment. In January 2015, the FAO hosted the Global Conference on Inland Fisheries (http://inlandfisheries.org/), which was the first global cross-sectoral conference dedicated to inland fisheries; economic and social assessment was a key theme. Despite these and other efforts, however, it is clear that global research investments in ICFs are very limited relative to their importance for food security, income generation, and human development and poverty alleviation.

4.3.3 Spatially-explicit valuation efforts

Economic research into inland fisheries should collect basic information on where fishers fish and/or the size of the fishery so that ICF values can be given according to a spatial unit. Unlike some ecosystem services (Raheem *et al.,* 2012), it is appropriate and relatively straightforward to spatially value fisheries, for example “per ha” or “per stream km”. However, it is important to also consider the spatial scale from which data are collected as ecosystem services require a minimum area to function (Fisher *et al.,* 2008). The spatial scale of ICF research should take account of the nature of the specific fishery being valued. It should be noted that adjusting the scale of values according to different geographical areas or to be compatible with values from other studies can lead to a loss in important detail and contextual variation (Bergstrom & Taylor, 2006).

4.3.4 Clarity of methodology

Across the ICFs literature there are inconsistencies in the level of detail given of study design and in the breakdown of values and calculations. In some cases, the most basic demographic and geographic information was absent from journal publications. For ICF values to be credible and suitable for inclusion in analyses based on the ecosystem services approach (or in standard economic CBA or benefits transfer), the details of study design must be transparent and clearly presented to include:

- Description of the geographic location, spatial scale, key landscape or freshwater features and ecosystems, type and target species of the fishery;
- Details of data collection methodology, study design, and rationale;
- Summary of the sample including target population, sample size, and demographic and socio-economic profile (information on any sample bias relative to the whole population would also be useful);
- Any data used as inputs for final values (for example, if a study calculates the total value of a wetland the individual value attributed to the associated fishery should be accessible);
- Studies that calculate the marginal value associated with a change in a fishery should also state baseline values whenever possible;
- Explanation and formula for any calculations and analysis;
- Key statistical outputs;
- Economic values accompanied by a clear unit of measure, preferably spatially explicit (e.g., per ha, km^2, or km of watercourse or shoreline); and
- For economic welfare estimates calculated with TCM, CVM, or CE methodologies, confidence intervals for WTP should be provided.

There are examples in the literature of each of the best practice recommendations made by this study but no individual study can be taken as an exemplar that considers all factors related to economic theory, methodology, and presentation. With forethought about study design and knowledge of what economic information is needed to inform fisheries management in the future, it should be possible for research teams, even those composed primarily of natural scientists, to still gather information that can be strategically used to further economic understanding of ICFs. Several of the past studies in this review illustrate how the future utility of ICF research could be extended into the economic realm with relatively minor modifications to research strategy and implementation. To fill knowledge gaps and maximise the utility of valuation efforts in the future, primary research should be prioritised according to their: (1) potential importance of the ecosystem type for various types of ICFs; (2) ability to fill important gaps in poorly or under-studied wetland types; and (3) potential for collection of economically credible values at relevant spatial resolution.

5. Conclusion

This study provided an overview of existing economics research for ICFs and found significant shortcomings in quantity, quality, and geographic coverage. Economic values are needed to ensure that fisheries are recognised as providing economically valuable ecosystem services in decision-making and development trade-offs. ICFs are threatened by multiple stressors but the field of study is limited by scarce resources. Therefore, it is essential to ensure primary valuation studies generate widely useful values and as much information as possible, for example, by providing data useful for benefits transfer. Shortcomings in the existing body of research have been used to inform best practice recommendations. Future studies should endeavour to collect the data needed to provide economically credible values (surplus or net values) measured in spatial units. To fill knowledge gaps primary research should prioritise the valuation of ICFs in wetland ecosystems other than rivers and lakes and there is a need for more extensive coverage of subsistence fisheries in developing regions, particularly in Africa and Asia.

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Conclusion

The two parts of this dissertation explored different but complimentary topics, each related to recognizing the importance of fisheries as an ecosystem service of inland surface-water ecosystems. The research presented in the first chapter assessed household susceptibility to hydrological change in the Lower Mekong Basin. By comparing the livelihood sensitivity and adaptive capacities of 2,703 households, the study provided an insight into relative *susceptibility* across the Mekong corridor and within individual geographic subsections. The second chapter was a structured review of economics research of inland fisheries. A systematic approach was used to select relevant literature and to code information on the nature of each study. A total of 76 peer-reviewed articles were selected for analysis. The study provided an extensive overview of the current status of inland fisheries economics knowledge and trends in study design.

Both studies provide valuable contributions to inland fisheries research. The extensive data coverage of the first study is unprecedented for the assessment of household vulnerability to hydrological change in the Mekong Basin; it highlighted the importance of understanding *susceptibility* to inform precautionary principles and identified how geographic scale affects the findings and therefore the policy implications of vulnerability assessments. As far as I am aware, the second study is the first review of inland fisheries literature that explicitly focused on an overview of economics research. Thus, it provides a novel insight into what is needed from future studies to help align fisheries management with the ecosystem approach.

The management of an ecosystem should account for all associated ecosystem services in concert and all beneficiaries of those services. Services that are not traded on markets, for example cultural values, and services that are exploited for subsistence are often overlooked or undervalued. This can hamper sustainable environmental management and lead to certain groups of people disproportionately bearing the costs of environmental change.

The Lower Mekong Basin (LMB) provides a good case study for assessing the importance of wetland ecosystem services in concert. The LMB is a cross boundary river basin and is home to the world's most productive inland fishery (Baran & Myschowoda, 2009). Total fishery production is estimated to be over 2.6 million tonnes (van Zalinge *et al.*, 2004). However, fisheries data from the LMB are considered to be underestimates due to the failure to fully account for all individuals involved in the fishery (MRC, 2010). Including part-time and seasonal fishers, two-thirds of the basin's population are involved in the fishery, this proportion varies regionally (MRC, 2003). Attempts to economically value the fishery have generally focused on market value and are inaccurate due to unreliable data and the exclusion of consumption and in-direct use values (Baran, Jantunen & Chong, 2007; Hortle & Bush, 2003). Various other aspects of livelihoods in the LMB are also dependent on services generated by aquatic ecosystems. For example flooding and irrigation are used in agriculture and rice paddies, the Mekong is an important transportation corridor, and the river is of cultural significance. Thus, the impacts on ecosystem service provision from impending changes to hydrological regimes in the LMB will have important implications on livelihoods and well-being (Dugan *et al.*, 2010; Orr *et al.*, 2012).

When considering the basin as a whole, household *susceptibility* to hydrological change varied between countries and ecozones. Within each region it was different factors that indicated which households would be most *susceptible*. The importance of geographic scale and location in identifying which factors determine relative *susceptibility* is relevant to the various levels at which the Mekong River is managed. The significance of different sensitivity and adaptive capacity variables in each region highlights that vulnerability mitigation policies should be tailored according to the specific target population and that the management of the Mekong must account for various ecosystem services. The findings of the first chapter can therefore inform policy prioritization and resource allocation sensitive to the region that particular decision-making processes will affect. If sustainability is to become the dominant narrative in the management of the Mekong, its fisheries and other ecosystem services must be comprehensively valued (van Zalinge *et al.*, 2004) so that the distribution of costs can be accounted for.

The exploitation of inland fish populations by fisheries provides an important source of food and income and offers highly valued opportunities for recreation. However inland fisheries economics have been subject to only limited research attention and so are poorly understood (Welcomme *et al.*, 2010; Bartley *et al.*, 2015; Mills *et al.*, 2011; Beard *et al.*, 2011). As an ecosystem service of inland waterbodies, inland fisheries need to be accounted for in decision-making processes that affect the wider functioning of wetlands and hydrological cycles (Beard *et al.*, 2011), for example climate change and hydropower development. The ecosystems approach is being encouraged in fisheries management (Beard *et al.*, 2011; Cowx & Gerdeaux, 2004; FAO, 2003; Suuronen & Bartley, 2014), which requires economic valuation. In light of this, chapter two assesses the current status of inland fisheries economics research.

To date, peer-reviewed studies have mostly focused on use-values associated with recreational fisheries, mainly in North America. Few studies accounted for the non-use values associated with fisheries and, relative to their productivity and livelihood importance, there has been little research into the value of subsistence fisheries in developing regions. As well as being unequally distributed geographically, studies were limited to only a narrow range of wetland types. Inland fisheries research has employed a variety of valuation methodologies, a number of which do not calculate an economically credible measure of the value of a fishery.

Both chapters identified shortcomings in understanding and methodology; these shortcoming may pose serious challenges for the sustainable management of inland fisheries. Although the index approach used in chapter one is a popular method of assessing vulnerability there are some fundamental shortcomings in its design that mean its validity is contested. The vulnerability index approach simplifies a theoretical concept into a single metric, which is both a strength and a weakness. By reducing the complexity of livelihood *susceptibility* the findings are compatible with policy-making processes. However, this simplification can mask important detail. The outcome of an index assessment is heavily influenced by its construction, which is subjective because of the researcher's choice of variables. Thus researcher bias is likely to be entrenched in the findings of index assessments, including in this study. Sensitivity analysis could provide a way to illustrate how index specifications influence outcomes and to refine indicator choice. As an exploratory study into the influence of geographic scale on *susceptibility* assessments this research used a broad range of indicators selected according to practical and theoretical guidelines laid down by others working in the same field. These guidelines may change as knowledge in the subject continues to develop. The identification of factors that contribute to vulnerability would also benefit from stakeholder consultation. A further short-coming found in the first chapter was that the immediately obvious solutions for addressing susceptibility may lead to perverse recommendations. For example, widespread construction of roads in northern Lao may increase adaptive capacity but the net affects may not be beneficial for well-being due to negative consequences arising from increased road access. Hence, the assessment identifies the distribution of *susceptibility*, but management must take into account wider environmental, social and economic factors. Failure to do so may lead to ineffective policies or potentially policies that increase vulnerability.

In the second chapter, I found economics research of inland fisheries to be lacking in quality and quantity. As highlighted in the original paper, the literature demonstrates that there is poor understanding of economic methodologies among ecologically-oriented researchers. Methodological limitations are a key cause of economic data shortcomings in ICFs research. In most cases economic research design demands are relatively simple. Survey methods are appropriate for most subsistence and commercial fishery valuations so longs as the information collected is sufficient to calculate net values. Similarly, methodologies to value recreational fisheries, including revealed and stated preference models, must capture sufficient information to reflect the true value of the fishery including non-use values. The agreement on and implementation of a standardised framework would help address the lack of consistent data quality. Without credible valuation the monetary values attributed to fisheries are unreliable and cannot be used in the ecosystem approach. The poor research coverage of subsistence fisheries in developing regions indicates that the largest knowledge gaps are exactly where there is the greatest need to understand the importance of inland fisheries and other water-based ecosystem services. Further, the distribution of fisheries research across geographic locations and ecosystem types limits the ability to use benefits transfer that, given limited resource availability, is crucial.

This dissertation has highlighted that the management of inland fisheries has environmental, economic and social implications, and that for these to be accounted for in decision-making processes fisheries must be economically valued to capture the value of the fishery as fully as possible, i.e. to include use and non-use values of all beneficiaries. This presents a vast challenge. Ultimately, unless future research strategically addresses the economic research shortcomings identified in this dissertation, inland fisheries will continue to be overlooked in decision-making and their sustainable management will be crippled by our lack of understanding (Beard *et al.*, 2011). The development of an adequate body of fisheries economic data will require greater international investment and coordination and cooperation between researchers so that future studies are widely useful and compatible.

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Supplementary information S1.1 Adaptive capacity indicators

Asset type	Asset description (according to DFID)	Indicators	Variable type
Human	Factors that enable people to pursue livelihood strategies and utilise other forms of asset including health, skills, knowledge and ability of labour.	HH size Number of working age members (14-59yrs) % working age members Non-elderly head (<60 years) Education	Count Count Percentage Dummy Scale (0=none, 1=primary, 2=lower secondary, 3=higher secondary, 4=tertiary)
Financial	Stocks and flows of financial resources that enable people to pursuit livelihood strategies.	Asset value Credit, savings or remittances	$US Dummy
Physical	Infrastructure and goods that support sustainable livelihoods including transport, shelter, water and sanitation, energy and information.	Technology/transport ownership Water supply ownership Fishing/farming equipment ownership Livestock ownership Stored rice/fish Reliable source of flood warning	Dummy Dummy Dummy Dummy Dummy Dummy
Natural	Stock of a natural resource from which goods and services flow	Agricultural land ownership	Dummy
Social	The social resources that facilitate and support people's livelihood pursuits, includes networks and connectedness, membership to formal groups and informal social safety nets.	Non-local income Male head Proximity to road Association membership	Dummy Dummy Km Dummy
Other	Diversity spreads risk and offers flexibility to adjust to fluctuations or shocks	Asset diversity Income diversity	Count (1 for each type) Count (1 for each type)
	Income determines a household's ability to employ preventative, coping and recovery strategies	Per capita income	$US

Supplementary information S1.2 *Susceptibility* in Laos and Thailand according to ecozone.

Table S1.2.1 *Susceptibility* scores for Laos and Thailand

		Laos	**Thailand**	***F* (df=1)**
EZ1	Mean sensitivity score***	0.3	0.1	72.4
	Mean adaptive capacity score***	0.5	0.6	60.8
	Mean composite *susceptibility* score***	0.5	0.3	125.9
EZ2	Mean sensitivity score**	0.1	0.2	0.0
	Mean adaptive capacity score	0.7	0.7	0.0
	Mean composite *susceptibility* score**	0.3	0.3	5.7

***significant at the 1% level, **significant at the 5% level, * significant at the 10% level

Table S1.2.2 *Susceptibility* scores for Ecozones 1 and 2 within Laos

Indicator	EZ1	EZ2	Statistic (df=1)
Sample size	339	340	
Composite *susceptibility* score***	0.5	0.3	F=52.4
Sensitivity indicators			
[D]Income types***	10.6	23.2	X^2=19.2
% income water-dependent***	2.5	6.6	F=10.4
Subsistence dependency***	0.3	0.4	F=8.0
[D]Agricultural dependency	2.9	2.9	X^2=0.0
[D]Domestic dependency***	55.8	18.5	X^2=100.7
Sensitivity score***	0.3	0.1	F=19.9
Adaptive capacity indicators			
HH size	3.8	5.8	F=0.4
Number of working age members	3.6	3.7	F=1.0
% working age members	64.7	64.8	F=0.0
[D]Non-elderly head ***	66.7	53.8	X^2=11.7
[D]Education***	35.4	52.4	X^2=19.8
Asset value (US$)***	5224.2	9023.3	X =17.6
[D]Credit, savings or remittances***	17.7	46.8	X^2=65.6
[D]Technology/transport ownership***	52.5	78.2	X^2=49.6
[D]Water supply ownership***	10.6	69.1	X^2=242.2
[D]Fishing/farming equipment ownership***	46.9	65.3	X^2=23.3
[D]Livestock ownership***	92.3	79.1	X^2=24.2
[D]Stored rice/fish***	98.8	82.1	X^2=55.1
[D]Reliable flood warning***	4.1	16.2	X^2=27.0
[D]Agricultural land ownership***	99.1	89.7	X^2=28.4
[D]Non-local income***	33.3	66.7	X^2=30.8
[D]Male head***	96.8	85.0	X^2=28.3
Distance to road***	16.7	4.1	F=132.0
[D]Association membership	52.5	53.5	X^2=0.1
Asset diversity***	3.0	3.8	F=17.6
Income diversity***	1.9	2.5	F=41.7
Per capita income (US$)***	194.0	336.3	F=16.3
Adaptive capacity score***	0.5	0.7	F=76.8

[D] = dummy variable, accompanying values are percentage of households who answered positively.
***significant at the 1% level, ** significant at the 5% level, * significant at the 10% level.

Table S1.2.3 *Susceptibility* scores for Ecozones 1 and 2 within Thailand

Indicator	**EZ1**	**EZ2**	**Statistic (df=1)**
Sample size	329	337	
Composite *Susceptibility* score***	0.4	0.3	F=36.2
Sensitivity indicators			
[D]Income types***	40.1	19.3	χ^2=34.7
% income water-dependent***	4.1	10.6	*F*=36.5
Subsistence dependency***	0.3	0.4	*F*=7.6
[D]Agricultural dependency**	21.0	14.2	χ^2=5.2
[D]Domestic dependency*	0.9	0.0	χ^2=3.1
Sensitivity score***	0.2	0.1	F=35.1
Adaptive capacity indicators			
HH size***	5.7	4.2	*F*=8.2
Number of working age members	2.4	2.4	*F*=0.0
% working age members**	64.0	59.5	*F*=4.3
[D]Non-elderly head	28.6	30.0	χ^2=0.2
[D]Education	58.4	63.8	χ^2=2.1
Asset value (US$)***	53094.3	34691.3	*F*=10.8
[D]Credit, savings or remittances	59.0	59.4	χ^2=0.0
[D]Technology/transport ownership**	91.8	96.4	χ^2=6.5
[D]Water supply ownership	42.9	42.4	χ^2=0.0
[D]Fishing/farming equipment ownership	61.7	60.2	χ^2=0.2
[D]Livestock ownership*	51.1	44.2	χ^2=3.1
[D]Stored rice/fish	89.1	90.5	χ^2=0.4
[D]Reliable flood warning	59.9	59.4	χ^2=0.0
[D]Agricultural land ownership	83.0	80.1	χ^2=0.9
[D]Non-local income	30.7	27.0	χ^2=1.1
[D]Male head	74.2	73.3	χ^2=0.1
Distance to road***	0.7	0.9	*F*=10.8
[D]Association membership	88.8	87.8	χ^2=0.1
Asset diversity	3.2	3.3	*F*=0.5
Income diversity	3.1	3.2	*F*=0.5
Per capita income (US$)	2812.5	2200.0	*F*=0.8
Adaptive capacity score	0.6	0.6	F=0.0

[D] = dummy variable, accompanying values are percentage of households who answered positively.
***significant at the 1% level, ** significant at the 5% level, * significant at the 10% level.

Supplementary information S1.3 Determinants of *susceptibility* classes according to country and ecozone

Table S1.3.1 Cambodia

Indicator	Low	Moderate	High	Very high	Statistic (df=3)
Sample size	170	170	170	170	
Composite *susceptibility* score***	0.2[a]	0.3[b]	0.4[c]	0.7[d]	F=1158.5
Sensitivity indicators					
Income types***	0.0[a]	8.2[b]	27.1[c]	58.2[d]	X^2=190.2
% income water-dependent***	0.0[a]	0.6[a]	2.5[a]	21.9[b]	F=81.1
Subsistence dependency***	0.0[a]	0.1[a]	0.4[b]	0.6[c]	F=55.9
Agricultural dependency***	0.0[a]	0.0[a]	8.2[b]	25.9[c]	X^2=97.3
Domestic dependency***	0.0[a]	0.0[a]	13.5[b]	52.4[c]	X^2=227.0
Sensitivity score***	0.0[a]	0.0[a]	0.2[b]	0.5[c]	F=604.0
Adaptive capacity indicators					
HH size***	5.5[a]	4.5[b]	4.6[b,c]	5.2[a,c]	F=10.0
Number of working age members***	3.6[a]	2.7[b]	2.7[b]	3.1[b]	F=13.6
% working age members***	67.6[a]	61.8[a,b]	57.7[b]	61.0[a,b]	F=5.3
Non-elderly head	61.8[a]	57.1[a]	57.6[a]	60.6[a]	X^2=54.1
Education***	48.8[a]	27.1[b]	19.4[b]	16.5[b]	X^2=54.1
Asset value (US$)***	10762.4[a]	7243.2[a,b]	4251.0[b]	4133.4[b]	F=10.7
Credit, savings or remittances	62.4[a]	50.6[a]	56.5[a]	52.9[a]	X^2=5.4
Technology/transport ownership***	87.1[a]	55.9[b]	45.9[b]	45.3[b]	X^2=81.0
Water supply ownership***	55.9[a]	32.4[b]	34.1[b]	23.5[b]	X^2=41.6
Fishing/farming equipment ownership***	27.6[a]	22.9[a]	34.1[a,b]	47.1[b]	X^2=25.3
Livestock ownership***	95.9[a]	72.4[b]	69.4[b]	67.6[b]	X^2=49.1
Stored rice/fish***	95.9[a]	81.8[b]	77.6[b]	64.1[c]	X^2=54.2
Reliable flood warning***	81.8[a]	65.3[b]	70.0[a,b]	70.0[a,b]	X^2=12.5
Agricultural land ownership***	92.9[a]	71.8[b]	74.7[b]	68.8[b]	X^2=34.0
Non-local income***	74.7[a]	59.4[b]	42.4[c]	58.2[b]	X^2=36.7
Male head***	89.4[a]	68.2[b]	75.3[b]	75.9[b]	X^2=22.7
Distance to road*	0.1[a]	0.1[a]	0.2[a]	0.1[a]	F=2.6
Association membership**	31.8[a]	17.1[b]	24.7[a,b]	28.8[a,b]	X^2=10.9
Asset diversity***	3.6[a]	2.6[b]	2.6[b]	2.5[b]	F=28.8
Income diversity***	3.3[a]	2.8[b]	3.1[a,b]	3.3[a]	F=5.1
Per capita income (US$)***	462.4[a]	459.9[a,b]	257.1[b]	289.8[a,b]	F=4.0
Adaptive capacity score***	0.7[a]	0.5[b]	0.5[b]	0.5[b]	F=50.9

***significant at the 1% level, **significant at the 5% level, * significant at the 10% level; matching superscripts a, b, c, d, indicate non-significant differences between categories in post-hoc tests.

Table S1.3.2 Laos

Indicator	Low	Moderate	High	Very high	Statistic (df=3)
Sample size	170	169	170	170	
Composite *susceptibility* score***	0.2^{a}	0.3^{b}	0.5^{c}	0.7^{d}	F=1778.8
Sensitivity indicators					
Income types***	0.6^{a}	9.5^{b}	29.4^{c}	28.2^{c}	X^2=73.2
% income water-dependent***	0.0^{a}	0.6^{a}	5.8^{b}	11.8^{c}	F=20.1
Subsistence dependency***	0.1^{a}	$0.4^{b,c,d}$	0.4^{c}	0.5^{d}	F=21.0
Agricultural dependency***	0.0^{a}	0.0^{a}	$2.9^{a,b}$	8.8^{b}	X^2=30.8
Domestic dependency***	0.0^{a}	0.6^{a}	60.6^{b}	87.1^{c}	X^2=418.7
Sensitivity score***	0.0^{a}	0.1^{b}	0.3^{c}	0.5^{d}	F=768.8
Adaptive capacity indicators					
HH size	5.9^{a}	5.7^{a}	5.9^{a}	5.5^{a}	F=1.1
Number of working age members	4.0^{a}	3.5^{a}	3.7^{a}	3.4^{a}	F=3.2
% working age members	68.1^{a}	64.1^{a}	64.1^{a}	62.7^{a}	F=1.9
Non-elderly head	60.0^{a}	59.2^{a}	59.4^{a}	62.4^{a}	X^2=0.5
Education***	61.2^{a}	43.8^{b}	45.3^{b}	14.4^{c}	X^2=72.8
Asset value (US$)***	11502.5^{a}	6737.3^{b}	6072.1^{b}	4192.0^{b}	F=12.1
Credit, savings or remittances***	51.8^{a}	27.8^{b}	30.0^{b}	19.4^{b}	X^2=44.4
Technology/transport ownership***	92.4^{a}	62.1^{b}	70.0^{b}	37.1^{c}	X^2=117.3
Water supply ownership***	64.7^{a}	47.3^{b}	30.6^{c}	17.1^{d}	X^2=90.6
Fishing/farming equipment ownership***	75.9^{a}	38.5^{b}	65.3^{a}	44.7^{b}	X^2=63.2
Livestock ownership***	90.6^{a}	78.1^{b}	90.6^{a}	$83.5^{a,b}$	X^2=15.2
Stored rice/fish***	$88.8^{a,b}$	82.8^{a}	96.5^{c}	$93.5^{c,b}$	X^2=20.8
Reliable flood warning***	15.9^{a}	$8.3^{a,b}$	$11.8^{a,b}$	4.7^{b}	X^2=12.8
Agricultural land ownership	94.1^{a}	92.3^{a}	95.3^{a}	95.9^{a}	X^2=2.4
Non-local income***	41.8^{a}	25.4^{b}	$28.2^{a,b}$	21.2^{b}	X^2=19.5
Male head	91.2^{a}	87.0^{a}	91.8^{a}	93.5^{a}	X^2=4.7
Distance to road***	5.1^{a}	$7.8^{a,b}$	9.9^{b}	18.9^{c}	F=28.2
Association membership***	$61.2^{a,b}$	$47.9^{b,c}$	63.5^{a}	39.4^{c}	X^2=26.5
Asset diversity***	4.2^{a}	3.2^{b}	3.5^{c}	2.8^{d}	F=51.9
Income diversity***	2.5^{a}	2.0^{b}	2.5^{a}	1.8^{b}	F=10.6
Per capita income (US$)***	377.1^{a}	$249.4^{a,b}$	$256.3^{a,b}$	178.0^{b}	F=5.5
Adaptive capacity score***	0.7^{a}	0.6^{b}	0.6^{c}	0.5^{d}	F=61.8

***significant at the 1% level, **significant at the 5% level, * significant at the 10% level; matching superscripts a, b, c, d, indicate non-significant differences between categories in post-hoc tests.

Table S1.3.35 Thailand

Indicator	Low	Moderate	High	Very high	Statistic (df=3)
Sample size	167	166	166	167	
Composite *susceptibility* score***	0.2[a]	0.3[b]	0.4[c]	0.6[d]	F=1221.1
Sensitivity indicators					
Income types***	0.5[a]	8.6[b]	33.5[c]	57.4[d]	X^2=221.7
% income water-dependent***	0.0[a]	0.5[a]	5.6[b]	23.1[c]	F=90.1
Subsistence dependency***	0.1[a]	0.3[a,b]	0.3[b]	0.8[c]	F=39.7
Agricultural dependency***	0.0[a]	0.0[a]	14.5[b]	55.7[c]	X^2=239.7
Domestic dependency**	0.0[a]	0.0[a]	0.0[a]	1.8[a]	X^2=9.0
Sensitivity score***	0.0[a]	0.0[a]	0.1[b]	0.4[c]	F=466.1
Adaptive capacity indicators					
HH size	4.2[a]	3.8[a]	3.8[a]	4.1[a]	F=2.1
Number of working age members***	2.8[a]	2.3[b]	2.2[b]	2.4[b]	F=7.4
% working age members***	70.5[a]	60.5[b]	56.4[b]	59.7[b]	F=8.0
Non-elderly head***	38.9[a]	33.1[b]	22.9[b]	22.2[b]	X^2=16.1
Education***	79.0[a]	56.0[b]	55.4[b]	53.9[b]	X^2=30.3
Asset value (US$)***	65131.7[a]	44771.7[a,b]	32620.4[b]	33426.0[b]	F=7.2
Credit, savings or remittances**	66.5[a]	55.4[a]	62.0[a]	52.7[a]	X^2=8.1
Technology/transport ownership***	99.4[a]	95.2[a,b]	88.0[b]	94.0[b]	X^2=20.2
Water supply ownership***	60.5[a]	30.7[b]	36.7[b]	42.5[b]	X^2=33.7
Fishing/farming equipment ownership***	70.1[a]	46.4[b]	52.4[b]	74.9[a]	X^2=39.3
Livestock ownership***	56.9[a]	39.2[b]	43.4[a,b]	50.9[a,b]	X^2=12.4
Stored rice/fish***	96.4[a]	90.4[a,b]	83.7[b]	88.6[b]	X^2=14.9
Reliable flood warning***	69.5[a]	53.0[b]	60.8[a,b]	55.1[b]	X^2=11.3
Agricultural land ownership***	93.4[a]	81.9[b]	67.5[c]	83.2[b]	X^2=37.8
Non-local income***	44.3[a]	25.9[b]	21.7[b]	23.4[b]	X^2=26.8
Male head***	85.0[a]	71.7[b]	70.5[b]	67.7[b]	X^2=15.4
Distance to road	0.8[a]	0.8	0.9	0.8	F=0.3
Association membership***	95.2[a]	90.4[a,b]	83.1[b]	84.4[b]	X^2=15.1
Asset diversity***	3.8[a]	2.9[b]	2.9[b]	3.5[c]	F=23.2
Income diversity***	3.4[a]	2.7[b]	3.2[a]	3.3[a]	F=6.9
Per capita income (US$)***	4163.6[a]	3134.7[a,b]	1558.9[a,b]	1180.5[b]	F=4.1
Adaptive capacity score***	0.7[a]	0.6[b]	0.5[b]	0.6[b]	F=37.1

***significant at the 1% level, **significant at the 5% level, * significant at the 10% level; matching superscripts a, b, c, d, indicate non-significant differences between categories in post-hoc tests.

Table S1.3.4 Vietnam

Indicator	Low	Moderate	High	Very high	Statistic (df=3)
Sample size	170	169	169	170	
Composite *susceptibility* score***	0.3[a]	0.5[b]	0.6[c]	0.8[d]	F=1542.8
Sensitivity indicators					
Income types***	6.5[a]	14.2[a,b]	25.4[c,b]	34.7[c]	X^2=48.7
% income water-dependent***	1.4[a]	5.4[a,b]	10.9[b,c]	17.1[c]	F=16.4
Subsistence dependency***	0.1[a]	0.1[a,b]	0.3[b,c]	0.3[c]	F=11.7
Agricultural dependency***	19.4[a]	74.6[b]	66.9[b]	93.5[c]	X^2=218.6
Domestic dependency***	1.2[a]	10.7[b]	54.4[c]	81.8[d]	X^2=312.0
Sensitivity score***	0.1[a]	0.3[b]	0.4[c]	0.6[d]	F=807.6
Adaptive capacity indicators					
HH size	4.3[a]	4.0[a]	4.4[a]	4.4[a]	F=1.2
Number of working age members	2.9[a]	3.0[a]	2.9[a]	2.9[a]	F=0.1
% working age members*	68.8[a]	75.2[a]	68.2[a]	70.2[a]	F=2,5
Non-elderly head***	47.1[a]	39.1[a,b]	33.1[a,b]	28.8[b]	X^2=13.6
Education	82.9[a]	85.8[a]	80.5[a]	87.1[a]	X^2=3.3
Asset value (US$)	29389.9[a]	39867.4[a]	35755.3[a]	37424.4[a]	F=1.2
Credit, savings or remittances***	46.5[a]	32.5[a,b]	39.6[b]	25.9[b]	X^2=17.5
Technology/transport ownership*	100.0[a]	100.0[a]	97.6[a]	98.8[a]	X^2=7.4
Water supply ownership*	86.5[a]	93.5[a]	89.3[a]	92.9[a]	X^2=6.4
Fishing/farming equipment ownership**	30.6[a]	27.8[a]	41.4[a]	37.6[a]	X^2=8.8
Livestock ownership	43.5[a]	52.1[a]	46.7[a]	52.4[a]	X^2=3.8
Stored rice/fish***	24.1[a]	33.1[a,b]	45.0[b,c]	50.0[c]	X^2=29.5
Reliable flood warning	68.8[a]	70.6[a]	74.1[a]	64.7[a]	X^2=4.6
Agricultural land ownership***	49.4[a]	88.8[b,c,d]	78.7[c]	90.6[d]	X^2=103.7
Non-local income***	41.8[a]	37.3[a]	35.5[b]	22.9[c]	X^2=14.7
Male head	80.6[a]	85.2[a]	86.4[a]	86.5[a]	X^2=3.0
Distance to road***	0.5[a]	0.6[a,b]	0.8[c,b]	1.1[c]	F=9.8
Association membership	68.8[a]	71.0[a]	74.6[a]	64.1[a]	X^2=4.5
Asset diversity***	3.1[a]	3.6[b]	3.5[c]	3.7[d]	F=12.0
Income diversity	2.4[a]	2.6[a]	2.6[a]	2.6[a]	F=1.8
Per capita income (US$)***	939.0[a]	1467.6[a,b]	2132.0[b]	1560.8[a,b]	F=4.2
Adaptive capacity score*	0.5[a]	0.6[a]	0.6[a]	0.6[a]	F=2.3

***significant at the 1% level, **significant at the 5% level, * significant at the 10% level; matching superscripts a, b, c, d, indicate non-significant differences between categories in post-hoc tests.

Table S1.3.5 Ecozone 1

Indicator	Low	Moderate	High	Very high	Statistic (df=3)
Sample size	*99*	*289*	*145*	*143*	
Composite *susceptibility* score***	0.1[a]	0.5[b]	0.5[b]	0.5[b]	F=79.0
Sensitivity indicators					
Income types***	0.0[a]	17.3[b]	19.3[b]	16,1[b]	X^2=21.0
% income water-dependent**	0.0[a]	4.3[a]	4.1[a]	2.8[a]	F=3.2
Subsistence dependency***	0.1[a]	0.3[b]	0.4[b]	0.3[b]	F=9.4
Agricultural dependency***	0.0[a]	9.7[b]	11.7[b]	9.1[b]	X^2=11.6
Domestic dependency***	0.0[a]	37.7[b]	24.8[c]	30.8[b,c]	X^2=53.3
Sensitivity score***	0.0[a]	0.3[b]	0.3[b]	0.2[b]	F=39.6
Adaptive capacity indicators					
HH size	4.4[a]	4.9[a]	4.6[a]	4.7[a]	F=1.8
Number of working age members	3.1[a]	3.1[a]	2.8[a]	2.9[a]	F=1.4
% working age members***	73.8[a]	63.6[b]	61.4[b]	62.3[b]	F=5.8
Non-elderly head	44.4[a]	53.3[a]	42.1[a]	47.6[a]	X^2=5.8
Education***	80.8[a]	46.0[b]	44.8[b]	39.9[b]	X^2=46.8
Asset value (US$)***	56130.5[a]	21384.3[b]	25635.8[b]	29437.8[b]	F=6.9
Credit, savings or remittances***	63.6[a]	30.1[b]	44.8[b]	31.5[b]	X^2=40.5
Technology/transport ownership***	99.0[a]	70.9[b]	71.7[b]	67.1[b]	X^2=37.8
Water supply ownership***	61.6[a]	18.7[b]	23.4[b]	21.0[b]	X^2=74.7
Fishing/farming equipment ownership***	75.8[a]	50.5[b]	51.7[b]	46.2[b]	X^2=24.0
Livestock ownership**	68.7[a]	73.7[a]	59.3[a]	66.4[a]	X^2=9.6
Stored rice/fish	97.0[a]	93.8[a]	95.2[a]	94.4[a]	X^2=1.6
Reliable flood warning***	58.6[a]	23.9[b]	31.0[b]	29.4[b]	X^2=41.6
Agricultural land ownership	92.9[a]	90.3[a]	86.9[a]	88.8[a]	X^2=2.6
Non-local income***	39.4[a]	21.5[b]	18.6[b]	20.3[b]	X^2=17.4
Male head	90.9[a]	85.8[a]	82.8[a]	81.8[a]	X^2=4.6
Distance to road***	2.3[a]	8.9[b]	7.7[b]	13.6[c]	F=10.6
Association membership***	91.9[a]	63.0[b]	69.7[b]	69.9[b]	X^2=29.5
Asset diversity***	4.0[a]	3.0[b]	2.9[b]	2.9[b]	F=23.6
Income diversity***	3.4[a]	2.2[b]	2.6[c]	2.3[b,c]	F=16.6
Per capita income (US$)	2553.9[a]	1486.7[a]	1236.9[a]	1061.1[a]	F=1.6
Adaptive capacity score***	0.7[a]	0.5[b]	0.5[b]	0.5[b]	F=47.4

***significant at the 1% level, **significant at the 5% level, * significant at the 10% level; matching superscripts a, b, c, d, indicate non-significant differences between categories in post-hoc tests.

Table S1.3.6 Ecozone 2

Indicator	Low	Moderate	High	Very high	Statistic (df=3)
Sample size	*167*	*167*	*167*	*168*	
Composite *susceptibility* score***	0.1^{a}	0.2^{b}	0.4^{c}	0.6^{d}	F=1320.6
Sensitivity indicators					
Income types***	0.6^{a}	14.4^{b}	44.9	66.1	X^2=203.4
% income water-dependent***	0.0^{a}	1.0^{a}	8.0^{b}	25.2^{c}	F=73.4
Subsistence dependency***	0.1^{a}	0.4^{b}	0.4^{b}	0.8^{c}	F=38.9
Agricultural dependency***	0.0^{a}	0.0^{a}	15.0^{b}	32.1^{c}	X^2=113.02
Domestic dependency***	0.0^{a}	0.0^{a}	6.6^{b}	32.7^{c}	X^2=137.4
Sensitivity score***	0.0^{a}	0.1^{b}	0.2^{c}	0.4^{d}	F=535.2
Adaptive capacity indicators					
HH size	5.5^{a}	4.9^{a}	4.8^{a}	4.8^{a}	F=3.6
Number of working age members***	3.6^{a}	2.9^{b}	2.8^{b}	3.0^{b}	F=7.7
% working age members***	67.0^{a}	61.2^{a,b}	57.9^{b}	62.5^{a,b}	F=3.8
Non-elderly head**	48.5^{a}	46.1^{a}	32.9^{a}	38.1^{a}	X^2=10.7
Education***	70.7^{a}	53.3^{b}	53.9^{b}	43.5^{b}	X^2=25.9
Asset value (US$)	27085.7^{a}	18884.2^{a}	20418.7^{a}	20205.3^{a}	F=1.4
Credit, savings or remittances***	65.3^{a}	49.7^{b}	50.9^{b}	45.2^{b}	X^2=15.2
Technology/transport ownership***	95.8^{a}	88.6^{a,b}	77.2^{c}	78.0^{b,c}	X^2=31.2
Water supply ownership***	72.5^{a}	59.3^{a,b}	48.5^{b,c}	44.6^{c}	X^2=31.7
Fishing/farming equipment ownership***	75.4^{a}	47.9^{b}	57.5^{b}	73.2^{a}	X^2=37.3
Livestock ownership***	77.2^{a}	60.5^{b}	60.5^{b}	63.1^{b}	X^2=14.3
Stored rice/fish***	94.6^{a}	79.6^{b}	82.6^{b}	85.1^{b}	X^2=16.9
Reliable flood warning	35.3^{a}	34.7^{a}	42.5^{a}	38.1^{a}	X^2=2.7
Agricultural land ownership***	94.6^{a}	86.2^{a,b}	80.8^{b}	83.9^{b}	X^2=14.9
Non-local income***	50.9^{a}	32.3^{b}	28.1^{b}	28.0^{b}	X^2=26.2
Male head	86.2^{a}	79.0^{a}	77.8^{a}	75.6^{a}	X^2=137.4
Distance to road***	1.9^{a}	2.4^{a,b}	2.3^{a,b}	3.6^{b}	F=4.2
Association membership	74.9^{a}	68.9^{a}	71.9^{a}	67.9^{a}	X^2=2.4
Asset diversity***	4.2^{a}	3.4^{b}	3.2^{b}	3.4^{b}	F=19.8
Income diversity**	2.9^{a}	2.6^{a}	3.0^{a}	2.9^{a}	F=3.1
Per capita income (US$)*	2583.3^{a}	793.9^{a}	905.1^{a}	732.1^{a}	F=2.5
Adaptive capacity score***	0.8^{a}	0.6^{b}	0.6^{b}	0.6^{b}	F=9.4

***significant at the 1% level, **significant at the 5% level, * significant at the 10% level; matching superscripts a, b, c, d, indicate non-significant differences between categories in post-hoc tests.

Table S1.3.7 Ecozone 3

Indicator	Low	Moderate	High	Very high	Statistic (df=3)
Sample size	85	85	85	85	
Composite *susceptibility* score***	0.1[a]	0.2[b]	0.4[c]	0.7[d]	F=521.0
Sensitivity indicators					
Income types***	0.0[a]	9.4[b]	25.9[c]	51.8[d]	X^2=77.0
% income water-dependent***	0.0[a]	0.6[a,b]	2.4[a,b]	21.1[b]	F=34.6
Subsistence dependency***	0.0[a]	0.1[a]	0.3[b]	0.4[b]	F=15.0
Agricultural dependency***	0.0[a]	0.0[a]	7.1[a]	30.6[b]	X^2=62.9
Domestic dependency***	0.0[a]	1.2[a,b]	9.4[b]	62.4[c]	X^2=150.9
Sensitivity score***	0.0[a]	0.0[a]	0.1[b]	0.5[c]	F=240.8
Adaptive capacity indicators					
HH size***	5.5[a]	4.5[b]	4.4[b]	5.1[a,b]	F=6.8
Number of working age members**	3.6[a]	2.7[b]	2.5[b]	2.9[b]	F=8.6
% working age members***	67.7[a]	62.3[a,b]	55.1[b]	59.8[a,b]	F=3.9
Non-elderly head	63.5[a]	57.6[a]	48.2[a]	60.0[a]	X^2=5.8
Education***	54.1[a]	23.5[b]	20.0[b]	23.5[b]	X^2=30.8
Asset value (US$)***	10799.4[a]	5314.4[b]	4182.6[b]	4561.6[b]	F=9.1
Credit, savings or remittances	58.8[a]	50.6[a]	51.8[a]	47.1[a]	X^2=2.5
Technology/transport ownership***	83.5[a]	55.3[b]	36.5[b]	45.9[b]	X^2=40.5
Water supply ownership***	60.0[a]	37.6[b]	36.5[b]	20.0[b]	X^2=31.0
Fishing/farming equipment ownership	20.0[a]	18.8[a]	23.5[a]	32.9[a]	X^2=5.8
Livestock ownership***	96.5[a]	81.2[b]	62.4[c]	58.8[c]	X^2=39.9
Stored rice/fish***	96.5[a]	84.7[a,b]	68.2[b]	44.7[c]	X^2=65.9
Reliable flood warning	82.4[a]	69.4[a]	72.9[a]	67.1[a]	X^2=4.6
Agricultural land ownership***	91.8[a]	78.8[b]	67.1[b,c]	55.3[c]	X^2=36.9
Non-local income***	78.8[a]	58.8[a,b]	44.7[b]	61.2[a,b]	X^2=19.4
Male head***	89.4[a]	67.1[b]	67.1[b]	74.1[a,b]	X^2=12.7
Distance to road	0.1[a]	0.1[a]	0.1[a]	0.1[a]	F=1.2
Association membership	31.8[a]	18.8[a]	24.7[a]	28.2[a]	X^2=4.0
Asset diversity***	3.5[a]	2.7[b]	2.3[b,c]	2.1[c]	F=24.7
Income diversity	3.3[a]	3.1[a]	3.0[a]	3.1[a]	F=1.0
Per capita income (US$)	498.7[a]	588.4[a]	290.8[a]	334.9[a]	F=1.9
Adaptive capacity score***	0.8[a]	0.6[b]	0.5[c]	0.5[b,c]	F=41.4

***significant at the 1% level, **significant at the 5% level, * significant at the 10% level; matching superscripts a, b, c, d, indicate non-significant differences between categories in post-hoc tests.

Table S1.3.8 Ecozone 4

Indicator	Low	Moderate	High	Very high	Statistic (df=3)
Sample size	85	85	85	85	
Composite *susceptibility* score***	0.2[a]	0.3[b]	0.4[c]	0.7[d]	F=762.7
Sensitivity indicators					
Income types***	0.0[a]	2.4[a]	21.2[b]	76.5[c]	X^2=172.3
% income water-dependent***	0.0[a]	0.2[a,b]	1.8[a,b]	24.1[b]	F=59.1
Subsistence dependency***	0.0[a]	0.2[a]	0.5[b]	0.8[c]	F=38.6
Agricultural dependency***	0.0[a]	0.0[a]	12.9[b]	17.6[b]	X^2=29.5
Domestic dependency***	0.0[a]	1.2[a]	22.4[b]	35.3[b]	X^2=59.7
Sensitivity score***	0.0[a]	0.0[a]	0.2[b]	0.5[c]	F=376.0
Adaptive capacity indicators					
HH size**	5.6[a]	4.6[a]	4.9[a]	5.1[a]	F=3.4
Number of working age members***	3.7[a]	2.8[b]	2.9[b]	3.1[a,b]	F=6.6
% working age members**	69.1[a]	61.9[a]	59.4[a]	61.1[a]	F=3.0
Non-elderly head	60.0[a]	58.8[a]	61.2[a]	64.7[a]	X^2=0.9
Education***	47.1[a]	29.4[a,c]	16.5[b,c]	9.4[b]	X^2=36.6
Asset value (US$)***	13347.1[a]	6470.2[b]	4683.2[b]	3421.6[b]	F=6.8
Credit, savings or remittances	64.7[a]	50.6[a]	62.4[a]	58.8[a]	X^2=3.0
Technology/transport ownership***	91.8[a]	55.3[b,c,]	62.4[b]	37.6[c]	X^2=54.9
Water supply ownership***	54.1[a]	23.5[b]	30.6[b]	29.4[b]	X^2=18.0
Fishing/farming equipment ownership***	37.6[a]	23.5[a]	44.7[a,b]	62.4[c]	X^2=27.3
Livestock ownership***	94.1[a]	63.5[b]	78.8[b]	75.3[b]	X^2=27.3
Stored rice/fish**	95.3[a]	80.0[a]	88.2[a]	81.2[a]	X^2=10.7
Reliable flood warning**	81.2[a]	60.0[a]	69.4[a]	71.8[a]	X^2=7.4
Agricultural land ownership***	88.2[a]	70.6[b]	83.5[a,b]	81.2[a,b]	X^2=11.6
Non-local income***	72.9[a]	58.8[a,b]	41.2[b]	52.9[b]	X^2=18.1
Male head**	90.6[a]	71.8[a]	78.8[a]	78.8[a]	X^2=12.7
Distance to road	0.0[a]	0.2[a]	0.3[a]	0.1[a]	F=1.5
Association membership***	32.9[a]	11.8[b]	22.4[a,b]	34.1[a]	X^2=17.1
Asset diversity***	3.7[a]	2.4[b]	3.0[c]	2.9[b,c]	F=15.2
Income diversity***	3.3[a]	2.4[b]	3.1[a]	3.7[a]	F=11.8
Per capita income (US$)***	446.9[a]	300.3[a,b]	236.1[b]	242.1[b]	F=6.0
Adaptive capacity score***	0.7[a]	0.5[b]	0.6[c]	0.6[c]	F=26.3

***significant at the 1% level, **significant at the 5% level, * significant at the 10% level; matching superscripts a, b, c, d, indicate non-significant differences between categories in post-hoc tests.

Table S1.3.9 Ecozone 5

Indicator	Low	Moderate	High	Very high	Statistic (df=3)
Sample size	85	85	85	85	
Composite *susceptibility* score***	0.3 [a]	0.5 [b]	0.7 [c]	0.8 [d]	F=580.9
Sensitivity indicators					
Income types***	0.0 [a]	0.0 [a]	7.1 [a]	24.7 [b]	X^2=47.4
% income water-dependent***	0.0 [a]	0.0 [a]	0.7 [a]	3.9 [b]	F=8.9
Subsistence dependency	0.1 [a]	0.2 [a]	0.2 [a]	0.3 [a]	F=2.1
Agricultural dependency***	40.0 [a]	71.8 [b]	85.9 [b,c]	95.3 [c]	X^2=76.0
Domestic dependency***	5.9 [a]	31.8 [b]	87.1 [c]	91.8 [c]	X^2=182.4
Sensitivity score***	0.1 [a]	0.3 [b]	0.5 [c]	0.6 [d]	F=297.2
Adaptive capacity indicators					
HH size	4.6 [a]	4.4 [a]	4.4 [a]	4.6 [a]	F=0.4
Number of working age members	3.3 [a]	2.9 [a]	3.2 [a]	3.1 [a]	F=1.1
% working age members	73.1 [a]	65.3 [a]	73.7 [a]	69.3 [a]	F=1.8
Non-elderly head ***	43.5 [a]	24.7 [a,b]	41.2 [a,b]	22.4 [b]	X^2=13.8
Education	89.4 [a]	82.4 [a]	88.2 [a]	83.5 [a]	X^2=2.5
Asset value (US$)	36360.4 [a]	40662.8 [a]	49607.2 [a]	38887.0 [a]	F=0.8
Credit, savings or remittances***	51.8 [a]	28.2 [b]	40.0 [a,b]	25.9 [b]	X^2=15.6
Technology/transport ownership	100.0 [a]	100.0 [a]	98.8 [a]	97.6 [a]	X^2=3.7
Water supply ownership	84.7 [a]	92.9 [a]	92.9 [a]	89.4 [a]	X^2=4.3
Fishing/farming equipment ownership***	22.4 [a]	20.0 [a]	50.6 [b]	29.4 [a]	X^2=23.3
Livestock ownership	49.4 [a]	49.4 [a]	55.3 [a]	43.5 [a]	X^2=2.4
Stored rice/fish***	35.3 [a]	32.9 [a]	70.6 [b]	58.8 [b]	X^2=34.3
Reliable flood warning***	78.8 [a,b]	74.1 [a,b]	83.5 [a]	61.2 [b]	X^2=12.4
Agricultural land ownership***	55.3 [a]	75.3 [b]	85.9 [b,c]	90.6 [c]	X^2=35.1
Non-local income**	44.7 [a]	41.2 [a]	28.2 [a]	28.2 [a]	X^2=8.3
Male head*	81.2 [a]	88.2 [a]	94.1 [a]	83.5 [a]	X^2=7.2
Distance to road***	0.5 [a]	0.7 [a,b]	0.7 [a,b]	1.2 [b]	F=4.8
Association membership	56.5 [a]	44.7 [a]	58.8 [a]	49.4 [a]	X^2=4.3
Asset diversity***	3.1 [a]	3.4 [a]	3.8 [b]	3.5 [a,b]	F=6.9
Income diversity**	2.7 [a]	2.3 [a]	2.8 [a]	2.7 [a]	F=3.1
Per capita income (US$)**	1422.8 [a]	1351.8 [a]	2966.1 [a]	1387.3 [a]	F=3.6
Adaptive capacity score***	0.6 [a]	0.5 [a]	0.6 [b]	0.5 [a]	F=8.5

***significant at the 1% level, **significant at the 5% level, * significant at the 10% level; matching superscripts a, b, c, d, indicate non-significant differences between categories in post-hoc tests.

Table S1.3.10 Ecozone 6

Indicator	Low	Moderate	High	Very high	Statistic (df=3)
Sample size	85	84	84	85	
Composite *susceptibility* score***	0.3 ^{a}	0.5 ^{b}	0.6 ^{c}	0.9 ^{d}	F=657.7
Sensitivity indicators					
Income types***	9.4 ^{a}	17.9 ^{a}	42.9 ^{b}	60.0 ^{b}	X^2=62.2
% income water-dependent***	2.0 ^{a}	6.9 ^{a}	21.2 ^{b}	35.0 ^{c}	F=26.1
Subsistence dependency***	0.0 ^{a}	0.1 ^{a}	0.3 ^{b}	0.5 ^{b}	F=15.1
Agricultural dependency***	9.4 ^{a}	61.9 ^{b}	59.5 ^{b}	84.7 ^{c}	X^2=103.4
Domestic dependency***	0.0 ^{a}	7.1 ^{a,b}	15.5 ^{b}	56.5 ^{c}	X^2=102.3
Sensitivity score***	0.0 ^{a}	0.2 ^{b}	0.3 ^{c}	0.6 ^{d}	F=378.6
Adaptive capacity indicators					
HH size*	4.4 ^{a}	3.9 ^{a}	3.8 ^{a}	3.9 ^{a}	F=2.3
Number of working age members	2.9 ^{a}	2.9 ^{a}	2.6 ^{a}	2.6 ^{a}	F=1.2
% working age members	66.7 ^{a}	75.8 ^{a}	70.9 ^{a}	70.0 ^{a}	F=1.8
Non-elderly head**	45.9 ^{a,b}	52.4 ^{a}	35.7 ^{a,b}	30.6 ^{b}	X^2=10.1
Education	87.1 ^{a}	79.8 ^{a}	77.4 ^{a}	84.7 ^{a}	X^2=3.4
Asset value (US$)	29730.9 ^{a}	30142.2 ^{a}	28306.9 ^{a}	30974. ^{a} 7	F=0.1
Credit, savings or remittances	43.5 ^{a}	34.5 ^{a}	32.1 ^{a}	32.9 ^{a}	X^2=3.1
Technology/transport ownership	100.0 ^{a}	98.8 ^{a}	97.6 ^{a}	100.0 ^{a}	X^2=3.7
Water supply ownership	91.8 ^{a}	88.1 ^{a}	90.5 ^{a}	94.1 ^{a}	X^2=2.0
Fishing/farming equipment ownership*	29.4 ^{a}	39.3 ^{a}	35.7 ^{a}	48.2 ^{a}	X^2=6.7
Livestock ownership*	40.0 ^{a}	44.0 ^{a}	48.8 ^{a}	58.8 ^{a}	X^2=6.7
Stored rice/fish	17.6 ^{a}	33.3 ^{a}	26.2 ^{a}	29.4 ^{a}	X^2=5.8
Reliable flood warning	64.7 ^{a}	61.9 ^{a}	71.4 ^{a}	61.2 ^{a}	X^2=2.4
Agricultural land ownership***	49.4 ^{a}	85.7 ^{b}	86.9 ^{b}	85.9 ^{b}	X^2=48.5
Non-local income***	44.7 ^{a}	38.1 ^{a,b}	27.4 ^{a,b}	22.4 ^{b}	X^2=11.8
Male head	84.7 ^{a}	83.3 ^{a}	75.0 ^{a}	87.1 ^{a}	X^2=4.8
Distance to road***	0.4 ^{a}	0.6 ^{a}	0.7 ^{a}	1.2 ^{b}	F=10.7
Association membership**	70.6 ^{a}	64.3 ^{a}	51.2 ^{a}	69.4 ^{a}	X^2=8.7
Asset diversity***	3.1 ^{a}	3.6 ^{b}	3.6 ^{b}	3.9 ^{b}	F=8.5
Income diversity	2.3 ^{a}	2.5 ^{a}	2.5 ^{a}	2.6 ^{a}	F=1.1
Per capita income (US$)	876.8 ^{a}	1313.4 ^{a}	1375.2 ^{a}	1494.7 ^{a}	F=1.4
Adaptive capacity score	0.4 ^{a}	0.4	0.4	0.4	F=0.8

***significant at the 1% level, **significant at the 5% level, * significant at the 10% level; matching superscripts a, b, c, d, indicate non-significant differences between categories in post-hoc tests.

Supplementary information S2.1 Bibliographic database search terms

Econ Lit
((inland or freshwater) and (fisher* or fishing) and (economic* or socioeconomic or socio-economic)).ab. or ((inland or freshwater) and (fisher* or fishing) and (economic* or socioeconomic or socio-economic)).ti. or ((inland or freshwater) and (fisher* or fishing) and (economic* or socioeconomic or socio-economic)).kw.

Green File
AB ((inland OR freshwater) AND (fisher* OR fishing) AND (economic* OR socioeconomic OR socio-economic)) OR TI ((inland OR freshwater) AND (fisher* OR fishing) AND (economic* OR socioeconomic OR socio-economic)) OR KW ((inland OR freshwater) AND (fisher* OR fishing) AND (economic* OR socioeconomic OR socio-economic))

Science Direct
TITLE-ABSTR-KEY ((inland OR freshwater) AND (fisher* OR fishing) AND (economic* OR socioeconomic OR socio-economic))

Scopus
TITLE-ABS- KEY ((inland OR freshwater) AND (fisher* OR fishing) AND (economic* OR socioeconomic OR socio-economic)) AND (EXCLUDE (DOCTYPE, “ed”)) AND (LIMIT-TO (LANGUAGE, “English”))

Web of Science
TOPIC: ((inland OR freshwater) AND (fisher* OR fishing) AND (economic* OR socioeconomic OR socio-economic)) Refined by: RESEARCH AREAS: (WATER RESOURCES OR SOCIAL SCIENCES OTHER TOPICS OR FISHERIES OR AGRICULTURE OR AREA STUDIES OR BUSINESS ECONOMICS OR ENGINEERING OR CULTURAL STUDIES) AND [excluding] DOCUMENT TYPES: (LETTER OR MEETING OR NEWS OR EDITORIAL) AND LANGUAGES: (ENGLISH)

Publish or Perish
CONTAINS ALL: economic* fisher* CONTAINS ANY: inland freshwater. Only cites 2 or more per year

Supporting information (S2.2). Inland capture fishery studies retained in final selection organized by publication year.

Year	Country of Fishery	Waterbody Type	Fishery Classification	Summary	Reference
2014	Thailand	Rivers	Subsistence	Impacts of thermal power plant on aquatic environment and social factors.	Chesoh, S. and Lim, A. (2014) Investigation of aquatic environment and social aspects of thermal power plant operation in southern of Thailand. *Asian Social Science,* 10(16), 168-175.
2014	Cambodia	Rivers	Commercial	The value chains of snakehead.	Le Xuan, S., Navy, H. and Pomeroy, R.S. (2014) Value chain of snakehead fish in the lower Mekong basin of Cambodia and Vietnam *Aquaculture Economics and Management,* 18(1), 76-96.
2014	Spain	Rivers	Recreational	Information on social and economic aspects of recreational fishing in Southern Europe.	Perez-Bote, J.L. and Roso, R. (2014) Recreational fisheries in rural regions of the south-western Iberian peninsula: A case study. *Turkish Journal of Fisheries and Aquatic Sciences,* 14(1), 135-143.
2013	Spain	Reservoirs	Commercial and recreational	Impact on ecosystem functioning and cost effectiveness of adding dead wood to streams.	Acuña, V., Díez, J.R., Flores, L., Meleason, M. and Elosegi, A. (2013) Does it make economic sense to restore rivers for their ecosystem services? *Journal of Applied Ecology* 50 (4), 988-997.
2013	Brazil	Rivers, lakes and floodplains/ wetlands	Subsistence	Effect of fisher's behaviour and environmental variables on fisher's catch and income.	Hallwass, G., Lopes, P.F.M., Juras, A.A. and Silvano, R.A.M. (2013) Behavioral and environmental influences on fishing rewards and the outcomes of alternative management scenarios for large tropical rivers. *Journal of Environmental Management* 128, 274-282.

2013	West Sumatra	Lakes	Subsistence	Relationship between socioeconomic characteristics and livelihood diversification.	Yuerlita, Perret, S.R. and Shivakoti, G.P. (2013) Fishing farmers or farming fishers? Fishing typology of inland small-scale fishing households and fisheries management in Singkarak Lake, West Sumatra, Indonesia. *Environmental Management,* 52(1), 85-98.
2012	New Zealand	Rivers & lakes	Recreational	The impact of Didymo on nonmarket values of recreational fishing.	Beville, S.T., Kerr, G.N. and Hughey, K.F.D. (2012) Valuing impacts of the invasive alga Didymosphenia geminata on recreational angling. *Ecological Economics* 82, 1-10.
2012	Brazil	Unspecified	Recreational	Overview of recreational fisheries.	Freire, K.M.F., Machado, M.L. and Crepaldi, D. (2012) Overview of inland recreational fisheries in Brazil. *Fisheries,* 37(11), 484-494.
2012	Serbia	Rivers	Subsistence	The characterisation of Serbian commercial fishers and fish catch.	Smederevac-Lalic, M., Pesic, R., Cvejic, S. and Simonovic, P. (2012) Socio-economic features of commercial fishery in the bordering upper Danube River area of Serbia. *Environmental Monitoring and Assessment,* 184(5), 2633-2646.
2011	USA	Rivers	Recreational	The economic impacts of recreational fishing in Idaho and how this will change with increased stocks of steelhead and salmon.	McKean, J.R., Johnson, D.M. and Taylor, R.G. (2011) Regional economic impacts of the Snake River steelhead and salmon recovery. *Society and Natural Resources,* 24(6), 569-583.
2011	Norway	Rivers	Recreational	The impact of proportion of wild salmon on angler WTP.	Olaussen, J.O. and Liu, Y. (2011) On the willingness to pay for recreational fishing - escaped versus wild atlantic salmon. *Aquaculture Economics and Management,* 15(4), 245-261.
2010	Uganda	Lakes and floodplains/ wetlands	Subsistence	Importance of wetland ecosystem goods and services for community.	Akwetaireho, S. and Getzner, M. (2010) Livelihood dependence on ecosystem services of local residents: a case study from Mabamba Bay wetlands (Lake Victoria, Uganda). *International Journal of Biodiversity Science Ecosystem Services and Management,* 6 (1-2), 75-87.

2010	Peru	Rivers	Subsistence	The role of artisanal fishing in household livelihoods.	Coomes, O.T., Takasaki, Y., Abizaid, C. and Barham, B.L. (2010) Floodplain fisheries as natural insurance for the rural poor in tropical forest environments: evidence from Amazonia. *Fisheries Management and Ecology,* 17(6), 513-521.
2010	Germany	Unspecified	Recreational	Preferences of eel anglers regarding management options for conservation and the associated welfare loss.	Dorow, M., Beardmore, B., Haider, W. and Arlinghaus, R. (2010) Winners and losers of conservation policies for European eel, Anguilla anguilla: an economic welfare analysis for differently specialised eel anglers. *Fisheries Management and Ecology*, 17(2), 106-125.
2010	USA	Unspecified	Recreational	Economic impact of expenditures from fishing, hunting and wildlife-associated recreational activities	Munn, I.A., Hussain, A., Spurlock, S. and Henderson, J.E. (2010) Economic impact of fishing, hunting, and wildlife-associated recreation expenditures on the southeast US regional economy: an input–output analysis. *Human Dimensions of Wildlife,* 15(6), 433-449.
2010	Peru	Rivers	Subsistence	Adjustments made by households to manage crop income losses from floods.	Takasaki, Y., Barham, B.L. and Coomes, O.T. (2010) Smoothing income against crop flood losses in Amazonia: rain forest or rivers as a safety net? *Review of Development Economics,* 14(1), 48-63.
2010	Finland	Lakes	Recreational	Impacts of water clarity on recreational swimming, fishing and boating.	Vesterinen, J., Pouta, E., Huhtala, A. and Neuvonen, M. (2010) Impacts of changes in water quality on recreation behavior and benefits in Finland. *Journal of Environmental Management,* 91(4), 984-994.
2010	West Sumatra	Lakes	Subsistence	Socioeconomics, fishing behaviour and problems faced by communities.	Yuerlita, and Perret, S.R. (2010) Livelihood features of small-scale fishing communities: a case from Singkarak Lake, West Sumatra, Indonesia. *International Journal of Environment and Rural Development*, 1-2, 94-101.

2009	Ghana	Lakes	Subsistence	The social and economic impacts of intensifying fishing using *acadja*.	Béné, C. and Obirih-Opareh, N. (2009) Social and economic impacts of agricultural productivity intensification: The case of brush park fisheries in Lake Volta. *Agricultural Systems,* 102(1-3), 1-10.
2009	Congo	Rivers	Subsistence	Fisheries as a source of income, and the relationship between poverty and fishing.	Béné, C., Steel, E., Luadia, B.K. and Gordon, A. (2009) Fish as the “bank in the water”–Evidence from chronic-poor communities in Congo. *Food Policy,* 34(1), 108-118.
2009	Scotland	Rivers	Recreational	The economic impact of recreational rod fisheries for specific species.	Butler, J.R.A., Radford, A., Riddington, G. and Laughton, R. (2009) Evaluating an ecosystem service provided by Atlantic salmon, sea trout and other fish species in the River Spey, Scotland: the economic impact of recreational rod fisheries. *Fisheries Research,* 96(2-3), 259-266.
2009	Cambodia	Rivers	Subsistence	Economic profitability and viability of small-scale capture fisheries.	Navy, H. and Bhattarai, M. (2009) Economics and livelihoods of small-scale inland fisheries in the Lower Mekong Basin: a survey of three communities in Cambodia. *Water Policy,* 11(supplement 1), 31-51.
2009	Bangladesh	Floodplains/ wetlands	Subsistence	Community socioeconomic status and dependency on *haors*.	Rana, M.P., Chowdhury, M.S.H., Sohel, M.S.I., Akhter, S. and Koike, M. (2009) Status and socio-economic significance of wetland in the tropics: a study from Bangladesh. *IForest* 2(5), 172-177.
2009	Myanmar	Rivers	Subsistence	The effect of dolphin cooperation on size and composition of fisher's catch.	Smith, B.D., Tun, M.T., Chi, A.M., Win, H. and Moe, T. (2009) Catch composition and conservation management of a human-dolphin cooperative cast-net fishery in the Ayeyarwady River, Myanmar. *Biological Conservation,* 142(5), 1042-1049.
2008	Peru	Rivers	Commercial	Differences in structure and importance of two aquarium fisheries.	Moreau, M.A. and Coomes, O.T. (2008) Structure and organisation of small-scale freshwater fisheries: aquarium fish collection in western Amazonia. *Human Ecology,* 36(3), 309-323.

2007	Chile	Lakes	Recreational	Economic benefits of a recreational fishery and the implications for management.	Arismendi, I. and Nahuelhual, L. (2007) Non-native salmon and trout recreational fishing in Lake Llanquihue, Southern Chile: Economic benefits and management implications. *Reviews in Fisheries Science,* 15(4), 311-325.
2007	Cambodia	Lakes	Subsistence	Valuation of aquatic resources.	Israel, D.C., Ahmed, M., Petersen, E., Hong, Y.B. and Chee, H.M. (2007) Economic valuation of aquatic resources in Siem Reap province, Cambodia. *Journal of Sustainable Agriculture,* 31(1), 111-135.
2007	Australia	Reservoirs	Recreational	The economic value of improving recreational fish catch.	Rolfe, J. and Prayaga, P. (2007) Estimating values for recreational fishing at freshwater dams in Queensland. *Australian Journal of Agricultural and Resource Economics,* 51(2), 157-174.
2007	Brazil	Reservoirs	Subsistence	The socioeconomic importance of small-scale fisheries in an urban waterbody.	Walter, T. and Petrere, Jr M. (2007) The small-scale urban reservoir fisheries of Lago Paranoá, Brasília, DF, Brazil. *Brazilian Journal of Biology,* 67(1), 9-21.
2007	South Africa	Reservoirs	Subsistence	The economic and subsistence fishery potential of dams.	Weyl, O.L.F., Potts, W., Rouhani, Q. and Britz, P. (2007) The need for an inland fisheries policy in South Africa: a case study of the North West Province. *Water SA,* 33(4), 497-504.
2006	USA	Rivers	Recreational	The economic impacts on the local economy from increased catches and fish size	Loomis, J.B. (2006) Use of survey data to estimate economic value and regional economic effects of fishery improvements. *North American Journal of Fisheries Management,* 26(2), 301-307.
2006	Laos, Thailand, Cambodia and Vietnam	Rivers and floodplains/ wetlands	Subsistence	The impacts of alternative water use on the value of fisheries	Ringler, C. and Cai, X. (2006) Valuing fisheries and wetlands using integrated economic-hydrologic modeling - Mekong River Basin. *Journal of Water Resources Planning and Management,* 132(6), 480-487.

2005	USA	Lakes	Recreational	Value of recreational fishery in reservoir with low visitation rates.	Chizinski, C.J., Pope, K.L., Willis, D.B., Wilde, G.R. and Rossman, E.J. (2005) Economic value of angling at a reservoir with low visitation. *North American Journal of Fisheries Management,* 25(1), 98-104.
2005	USA	Unspecified	Commercial and recreational	Relationship between income of fisheries programs and expenditures, funding and staffing.	Gabelhouse, D.W. (2005) Staffing, spending, and funding of State inland fisheries programs. *Fisheries,* 30(2), 10-17.
2005	USA	Reservoirs	Recreational	Non-market valuation of fishing and environmental concern of different angler groups.	Oh, C.-O., Ditton, R.B., Anderson, D.K., Scott, D. and Stoll, J.R. (2005) Understanding differences in nonmarket valuation by angler specialization level. *Leisure Sciences,* 27(3), 263-277.
2005	Trinidad	Floodplains/ wetlands	Commercial and subsistence	Distribution of fish resources and revenue from fish sales.	Ramsundar, H. (2005) The distribution and abundance of wetland ichthyofauna, and exploitation of the fisheries in the Godineau Swamp, Trinidad - Case study. *Revista De Biologia Tropical,* 53(1), 11-23.
2004	Peru	Floodplains/ wetlands	Commercial and subsistence	Factors affecting income reliance on fishing, hunting and resource extraction.	Coomes O.T., Barham B.L. and Takasaki Y. (2004) Targeting conservation–development initiatives in tropical forests: insights from analyses of rain forest use and economic reliance among Amazonian peasants. *Ecological Economics,* 51(1), 47-64.
2004	Sweden	Rivers	Recreational	Economic benefit from increased trip frequency and utility resulting from improved fish management.	Paulrud, A. and Laitila, T. (2004) Valuation of management policies for sport-fishing on Sweden's Kaitum river. *Journal of Environmental Planning and Management,* 47(6), 863-879.

2004	Finland	Reservoirs	Commercial	Changes in commercial catch.	Salonen, E. and Mutenia, A. (2004) The commercial coregonid fishery in northernmost Finland - A review. *Annales Zoologici Fennici,* 41(1), 351-355.
2004	USA	Rivers	Recreational	Impact of congestion on fisher's utility and how factors influence angler's WTP.	Schuhmann, P.W. and Schwabe, K.A. (2004) An analysis of congestion measures and heterogeneous angler preferences in a random utility model of recreational fishing. *Environmental and Resource Economics,* 27(4), 429-450.
2004	Denmark, Finland, Iceland, Norway and Sweden	Unspecified	Recreational	Factors that determine WTP for recreational angling.	Toivonen, A.-L., Roth, E., Navrud, S., Gudbergsson, G., Appelblad, H., Bengtsson, B. and Tuunainen, P. (2004) The economic value of recreational fisheries in Nordic countries. *Fisheries Management and Ecology,* 11(1), 1-14.
2003	Germany	Unspecified	Recreational	The potential of specialised carp anglers to reduce carp populations vs. their contribution to phosphorous eutrophication.	Arlinghaus, R. and Mehner, T. (2003) Socio-economic characterisation of specialised common carp (Cyprinus carpio L.) anglers in Germany, and implications for inland fisheries management and eutrophication control. *Fisheries Research,* 61(1-3), 19-33.
2003	USA	Lakes	Recreational	The economic impacts, at local and state scales, associated with trophy largemouth bass fishery.	Chen, R., Hunt, K. and Ditton, R. (2003) Estimating the economic impacts of a trophy largemouth bass fishery: issues and applications. *North American Journal of Fisheries Management,* 23(3), 835-844.
2003	USA	Lakes	Recreational	Anglers perceptions and the economic impacts of aquatic plant coverage	Henderson, J.E., Kirk, J.P., Lamprecht, S.D. and Hayes, W.E. (2003) Economic impacts of aquatic vegetation to angling in two South Carolina reservoirs. *Journal of Aquatic Plant Management* 41, 53-56.

2003	USA	Rivers	Recreational	Economic benefits of suppressing Sea Lamprey population.	Lupi, F., Hoehn, J.P. and Christie, G.C. (2003) Using an economic model of recreational fishing to evaluate the benefits of sea lamprey (*Petromyzon marinus*) control on the St. Marys River. *Journal of Great Lakes Research,* 29(1), 742-754.
2002	USA	Rivers	Recreational	The value of the blue-ribbon trout fishery and values that would be lost if fishing was prohibited.	Kerkvleit, J., Nowell, C. and Lowe, S. (2002) The economic value of the Greater Yellowstone's blue-ribbon fishery. *North American Journal of Fisheries Management,* 22(2), 418-424.
2002	USA	Lakes	Recreational	Modelling the trip-taking behaviour of anglers.	Provencher, B., Baerenklau, K.A. and Bishop, R.C. (2002) A finite mixture logit model of recreational angling with serially correlated random utility. *American Journal of Agricultural Economics,* 84(4), 1066-1075.
2002	Brazil	Floodplains/ wetlands	Recreational	Value of recreational fishery according to different models.	Shrestha, R.K., Seidl, A.F. and Moraes, A.S. (2002) Value of recreational fishing in the Brazilian Pantanal: a travel cost analysis using count data models. *Ecological Economics,* 42(1), 289-299.
2001	Brazil	Rivers	Commercial	Characterisation of the commercial fishing fleet and fishermen.	Almeida, O.T., McGrath, D.G. and Ruffino, M.L. (2001) The commercial fisheries of the lower Amazon: an economic analysis. *Fisheries Management and Ecology,* 8(3), 253-269.
2001	Brazil	Rivers	Subsistence	Factors which affect fishers' income.	De Camargo, S.A. and Petrere, J.R. (2001) Social and financial aspects of the artisanal fisheries of Middle São Francisco River, Minas Gerais, Brazil. *Fisheries Management and Ecology,* 8(2), 163-171.

2001	France	Unspecified	Recreational	Description of national management plans and whether angler's WTP for restoration of wild fish covers restorations costs.	Changeux, T., Bonnieux, F. and Armand, C. (2001) Cost benefit analysis of fisheries management plans. *Fisheries Management and Ecology,* 8(4-5), 425-434.
2001	Peru	Rivers	Commercial and subsistence	The socioeconomic importance of different livelihood activities.	Kvist, L.P., Gram, S., Cácares, C.A. and Ore, B.I. (2001) Socio-economy of flood plain households in the Peruvian Amazon. *Forest Ecology and Management,* 150(1), 175-186.
2001	Portugal	Rivers and reservoirs	Recreational	Characterisation of recreational fishers and their fishing activities.	Marta, P., Bochechas, J. and Collares-Pereira, M.J. (2001) Importance of recreational fisheries in the Guadiana River Basin in Portugal. *Fisheries Management and Ecology,* 8(4-5), 345-354.
2001	UK	Rivers	Recreational	Use and non-use values, social benefits and community impacts of fisheries.	Peirson, G., Tingley, D., Spurgeon, J. and Radford, A. (2001) Economic evaluation of inland fisheries in England and Wales. *Fisheries Management and Ecology,* 8(4-5), 415-424.
2001	Peru	Rivers	Subsistence	The role of wealth and geographical factors in shaping livelihood strategies.	Takasaki, Y., Barham, B.L. and Coomes, O.T. (2001) Amazonian peasants, rain forest use, and income generation: the role of wealth and geographical factors. *Society and Natural Resources,* 14(4), 291-308.
2001	Germany	Unspecified	Recreational	National status and value of inland fisheries.	Wedekind, H., Hilge, V. and Steffens, W. (2001) Present status, and social and economic significance of inland fisheries in Germany. *Fisheries Management and Ecology,* 8(4-5), 405-414.
2001	UK	Unspecified	Recreational	The attitude of recreational clubs located in shallow SSSI sites towards conservation.	Williams, A.E. and Moss, B. (2001) Angling and conservation at Sites of Special Scientific Interest in England: economics, attitudes and impacts. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 11(5), 357-372.

1999	USA	Rivers	Recreational and subsistence	Impact of aquatic vegetation on economic value of fishery.	Henderson, M.M., Criddle, K.R. and Lee, S.T. (1999) The economic value of Alaska's Copper River personal use and subsistence fisheries. *Alaska Fishery Research Bulletin,* 6(2), 63-69.
1999	UK	Rivers	Recreational	The benefits to anglers and other users, of increasing the flow of low-flow rivers.	Willis, K.G. and Garrod, G.D. (1999) Angling and recreation values of low-flow alleviation in rivers. *Journal of Environmental Management,* 57(2), 71-83.
1998	USA	Unspecified	Commercial	Species composition and economic value of baitfish industry.	Kircheis, F.W. (1998) Species composition and economic value of Maine's winter baitfish industry. *North American Journal of Fisheries Management,* 18(1), 175-180.
1998	USA	Unspecified	Recreational	The impact of a doubling in CO^2 on fish catch and recreational angler's welfare.	Pendleton, L.H. and Mendelsohn, R. (1998) Estimating the economic impact of climate change on the freshwater sportsfisheries of the Northeastern US. *Land Economics,* 74(4), 483-496.
1998	USA	Rivers	Recreational	Difference in WTP calculated using random-parameter and standard logit models.	Train, K.E. (1998) Recreation demand models with taste differences over people. *Land Economics,* 74(2), 230-239.
1997	Australia	Rivers	Recreational	Importance of considering non-use values in fishery management strategies.	Baker, D.L. and Pierce, B.E. (1997) Does fisheries management reflect societal values? Contingent Valuation evidence for the River Murray. *Fisheries Management and Ecology,* 4(1), 1-15.
1997	Northern Ireland	Rivers	Recreational	The consequential value of wild smolts.	Kennedy, G.J.A. and Crozier, W.W. (1997) What is the value of a wild salmon smolt, Salmo salar L.? *Fisheries Management and Ecology,* 4(2), 103-110.

1996	USA	Rivers	Recreational	The impact of policy initiatives on angler consumer surplus.	Layman, R.C., Boyce, J.R. and Criddle, K.R (1996) Economic valuation of the Chinook salmon sport fishery of the Gulkana River, Alaska, under current and alternate management plans. *Land Economics,* 72(1), 113-128.
1993	Canada	Rivers	Recreational	Comparison of travel cost models to demonstrate importance of using a multi-level nested-logit model.	Morey, E.R., Rowe, R.D. and Watson, M. (1993) A repeated nested-logit model of Atlantic salmon fishing. *American Journal of Agricultural Economics,* 75(3), 578-592.
1993	USA	Rivers	Recreational	Value of wildlife associated recreation.	Shafer, E.L., Carline, R., Guldin, R.W. and Cordell, H.K. (1993) Economic amenity values of wildlife: Six case studies in Pennsylvania. *Environmental Management,* 17(5), 669-682.
1993	Canada	Floodplains/ wetlands	Recreational	Differences in the social and private net benefits from preserving wetlands vs. converting them to agriculture.	Van Vuuren, W. and Roy, P. (1993) Private and social returns from wetland preservation versus those from wetland conversion to agriculture. *Ecological Economics,* 8(3), 289-305.
1992	USA	Rivers	Recreational	The impacts of water-use allocation on wildlife and fisheries habitats and recreational activities.	Creel, M. and Loomis, J. (1992) Recreation value of water to wetlands in the San Joaquin Valley: linked multinomial logit and count data trip frequency models. *Water Resources Research,* 28(10), 2597-2606.
1992	USA	Lakes	Recreational	Demonstrate use of randomly drawn opportunity sets to estimate benefits of water quality improvements.	Parsons, G.R. and Kealy, M.J. (1992) Randomly drawn opportunity sets in a random utility model of lake recreation. *Land Economics,* 68(1), 93-106.

1991	UK	Canals	Recreational	The value of open access recreation in inland waterways.	Willis, K.G. and Garrod, G. (1991) Valuing open access recreation on inland waterways: On-site recreation surveys and selection effects. *Regional Studies,* 25(6), 511-524.
1990	USA	Floodplains/ wetlands	Commercial	Economic value of baitfish fishery.	Carlson, B.N. and Berry, C.R. (1990) Population size and economic value of aquatic bait species in palustrine wetlands of eastern South Dakota. *Prairie Naturalist,* 22(2), 119-128.
1988	USA	Rivers	Recreational	The benefits of improving stream flow on recreational steelhead angling.	Johnson, N.S. and Adams, R.M. (1988) Benefits of increased streamflow: the case of the John Day River Steelhead Fishery. *Water Resources Research,* 24(11), 1839-1846.
1988	USA	Rivers	Recreational	The economic effects of changes in timber harvest on recreational and commercial salmon and steelhead fisheries.	Loomis, J.B. (1988) The bioeconomic effects of timber harvesting on recreational and commercial salmon and steelhead fishing: a case study of the Siuslaw National Forest. *Marine Resource Economics***,** 5(1), 43-60.
1987	Mali	Floodplains/ wetlands	Subsistence	The value of floodplain productivity.	Marchand, M. (1987) The productivity of African floodplains. *International Journal of Environmental Studies,* 29(2-3), 201-211.