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ABSTRACT
The state of knowledge on the science and management of freshwater recreational fisheries is reviewed, with the objective of integrating insights from disparate fields such as fisheries science, environmental complexity theory, common-pool resource theory, and resilience theory. First, freshwater recreational fisheries are characterized as complex adaptive social-ecological systems (SESS). Subsequently, two interrelated frameworks, drawing on the Ostrom framework for the analysis of SESS and adaptive management as key foundations, are presented. These frameworks are useful to structure the complexity and apprehend the various feedbacks and links inherent in any particular recreational fisheries system. Moreover, the frameworks help to identify operational management strategies in the face of substantial social-ecological uncertainty. It is concluded that to understand and manage freshwater recreational fisheries as complex adaptive SESS, a sustained shift from disciplinary to inter- and sometimes transdisciplinary research as well as a focus on flexible, adaptive and generally enabling rather than command-and-control type governance and management are needed. Understanding and managing recreational fisheries as complex adaptive SESS will benefit from an increasing focus on (i) managing social-ecological feedbacks and processes, (ii) managing critical slow variables that either drive the system or maintain it in potentially undesirable states, and (iii) managing and maintaining social and ecological diversity. It is hoped that the frameworks presented in this article may guide future interdisciplinary inquiry to manage for sustainability by building resilience.

KEYWORDS
Angling, complex systems, heterogeneity, resilience, regime shift, scale, stocking, research frameworks, sustainability, uncertainty

1. Introduction
There is growing political recognition of the large economic, socio-cultural, and ecological importance of recreational fishing as a significant component of global capture fisheries (FAO, 2010, 2012; World Bank, 2012). Due to industrialization and an associated rise in personal wealth and leisure time, the main use of inland water bodies has shifted from subsistence and commercial fishing to recreational fisheries, particularly in the temperate world (Smith, 1986; Welcomme, 2001; Cowx et al., 2010; FAO, 2010). Accordingly, freshwater recreational fisheries now constitute the dominant or sole use of wild freshwater fish stocks in all industrialized countries (Arlinghaus et al., 2002; FAO, 2012). Recreational fisheries are also present in three quarters of the world’s exclusive economic zones in the marine environment (Mora et al., 2009), where many coastal stocks are now predominantly exploited by recreational anglers (Pawson et al., 2008; Ihde et al., 2011).
Recreational fisheries involve millions of people globally and generate billions of U.S. dollars in expenditures that support many full time jobs in a range of sectors (Arlinghaus et al., 2002; Ditton et al., 2002; Cisneros-Montemajor and Sumaila, 2010; FAO, 2012; World Bank, 2012; Arlinghaus et al., 2015a; Tufts et al., 2015). The average participation rate in recreational fishing is 10.5% across countries with reliable statistics (Arlinghaus et al., 2015a). Accordingly, there are about 118 million recreational fishers in North America, Europe and Oceania alone (Arlinghaus et al., 2015a). Globally, estimated numbers of recreational fishers range between 220 million (World Bank, 2012) and 700 million (Cooke and Cowx, 2004). The activity is also emerging as an important social and economic factor in many transitional economies (e.g., Argentina, Brazil, China, India; FAO, 2010; Freire et al., 2012; Bower et al., 2014) where it fuels an increasingly relevant angling tourism industry on top of increasing resident angling interest (Arlinghaus and Nahuelhual, 2007; FAO, 2010; Hargrove et al., 2015; Weyl et al., 2015). The popularity of recreational fisheries indicates that it generates substantial psychological, nutritional, and health benefits to the individual participant and aggregately social and cultural benefits (Hendee, 1974; Fedler and Ditton, 1994; Weithman, 1999; Parkkila et al., 2010). In fact, recreational fishing is a prime cultural ecosystem service produced by aquatic ecosystems and thereby contributes substantially to human well-being (Parkkila et al., 2010; Villamagna et al., 2014; Pope et al., in press). Recreational fishers target fish and invertebrates using a variety of gear following local custom (e.g., hook-and-line, gill nets, traps, spearguns, bows; Arlinghaus and Cooke, 2009). Despite this diversity, it has become commonplace to equate recreational fishing with angling because the most typical gear type used by recreational fishers is rod-and-reel with one or more hooks involving natural bait or artificial lures (Arlinghaus et al., 2007).

The sheer size of the recreational fishing industry increasingly demands dedicated management attention to safeguard biologically sustainable exploitation and manage stakeholder conflicts (Arlinghaus et al., 2002; Post et al., 2002; Pawson et al., 2008; Figueira and Coleman, 2010; Ihde et al., 2011; FAO, 2012). Recreational fishing can induce structural and functional changes in fish communities and aquatic ecosystems via actions such as excessive and selective harvest, hooking mortality in catch-and-release, accidental or deliberate release of organisms (including invasive or pathogenic species), litter, groundbaiting and associated nutrient input, and disturbance of the environment, wildlife and other users of aquatic ecosystems (Post et al., 2002; Arlinghaus, 2005; Cooke and Cowx, 2006; Lewin et al., 2006; Arlinghaus et al., 2007; Altieri et al., 2012; Post, 2013; Johnston et al., 2013, 2015). Reversing some of these ecological impacts may be slow or impossible, for instance when intensive and/or trait-selective recreational angling causes genetic changes in heavily exploited fish stocks or genetic drift through exploitation-induced population bottlenecks (Lewin et al., 2006; Biro and Post, 2008; Philipp et al., 2009; Saura et al., 2010; Matsumura et al., 2011; Sutter et al., 2012; Alós et al., 2014), or when intensive stocking with foreign genotypes contributes to the irreversible loss of locally adapted populations or to the establishment of an invasive species that cannot be eradicated (Johnson et al., 2009; Cowx et al., 2010; Laikre et al., 2010; van Poorten et al., 2011; Lorenzen et al., 2012). Responsible recreational fisheries management thus requires consideration of the broader impacts of fishing on the ecosystem as a whole, taking ecosystem traits, food webs and biodiversity across genetic, species and population levels into account (FAO, 2012; Pope et al., in press). Many declines in wild fish stocks are however only partly related to recreational fishing or its management practices (Arlinghaus et al., 2002). Other human activities affecting aquatic ecosystems, such as agriculture, damming, habitat homogenization due to channelization of rivers, deforestation, navigation, wetland reclamation, urbanization, water abstraction and transfer and waste disposal, have altered freshwater ecosystems profoundly (Welcomme, 2001; Cowx et al., 2010; Craig, 2016). Such anthropogenic change does not reduce the importance of sustainably managing recreational fishing, because anglers place additional pressures on already impoverished stocks and ecosystems (Cowx et al., 2010; Cooke et al., 2016). At the same time anglers play an important role in lobbying for and actively engaging in fish conservation and fisheries management, thereby contributing substantially to solve a range of local and regional sustainability issues in many countries of the world (Bate, 2001; Arlinghaus et al., 2002, 2007; Granek et al., 2008; Daedlow et al., 2011; Tufts et al., 2015). For example, in much of central Europe, anglers are not only stakeholders but leaseholders of fishing rights for most inland fisheries and in this position self-organize and self-manage a range of fisheries management and conservation actions such as conservation stocking to maintain threatened species or habitat restoration (Arlinghaus, 2006; Daedlow et al., 2011). Similarly, in the United States many restoration efforts are funded by excise taxes paid by the angler community (Tufts et al., 2015). At the same time, the popularity of recreational angling and their widespread presence at aquatic ecosystems, fostered by the intensity with which popular fisheries management actions such as stock enhancement are conducted, is also seen by some as a major potential disturbance to
nature (Arlinghaus, 2005). De Leeuw (2012), for example, argues that “harm caused to nature by both sport fisheries management and to fish by angling ‘outweigh’” angling related benefits to nature,” and, “consequently, the justification of angling on the basis of nature conservation is dubious at best.” In many places in central Europe, recreational fishing is indeed severely constrained or even banned in nature conservation areas, reflecting that the perspective just cited is indeed prevalent among some nature conservation agencies and environmental lobby groups, causing substantial intersectoral conflict in need of management (Arlinghaus, 2005).

Given the increasing awareness of the large social, economic, ecological and evolutionary impact of recreational fisheries world-wide and the social conflicts that recreational fishing might induce, it is not surprising that dedicated research efforts on recreational fisheries are on the rise. Research on recreational fishing began in the early 20th century and was established as its own subdiscipline in fisheries science by about 1950 in the United States (Arlinghaus, 2004). Since then, the number of scientific articles including the term “recreational fisheries” has steadily increased across the world (Rocklin et al., 2014). In this context, relatively little research effort has been devoted to studies of the human dimensions (psychology, economics, and governance) aspects of recreational fisheries compared to the larger body of fisheries biological work (Arlinghaus, 2006b; Ditton, 2004; Hunt et al., 2013). This contrast is noteworthy because many social and economic issues, including conflicts, trade-offs and contested political configurations are characteristic of managing recreational fisheries (Arlinghaus, 2005; Pope et al., in press), and solving these issues requires studying the interactions among managers, anglers, fish stocks and ecosystems from an interdisciplinary and integrative perspective (Wesley, 2002; Arlinghaus et al., 2013, 2014a,b; Ward et al., 2016; Pope et al., in press). Researchers have repeatedly argued that fisheries management is as much about people management as it is about fish stock management (Wilen et al., 2002; Arlinghaus, 2004; Hilborn, 2007)—a call to put the human dimensions of recreational fisheries on equal footing to the ecological dimensions (Aas and Ditton, 1998). But the demand goes further toward a call for integration of the applied social and ecological sciences with the realm of social-ecological research and management. This article is such a call toward change in how the science and management of recreational fisheries is pursued (see also Pope et al., 2014, in press for similar arguments).

Recreational fisheries are prime examples of strongly coupled social-ecological systems (SEs) (Carpenter and Brock, 2004; Arlinghaus et al., 2013; Hunt et al., 2013; Schlüter et al., 2014; Hinkel et al., 2014, 2015; Cooke et al., 2016). The tight linkage of people and nature in coupled SEs (Figure 1) is not new to fisheries scientists and managers (Larkin, 1978; Royce, 1983, Arlinghaus, 2004; Hilborn, 2007). Unfortunately maybe, despite the early recognition of the importance of the social aspects in fisheries science (Gordon, 1954; Larkin, 1978; Arlinghaus, 2014), the ecological and social sciences of recreational fisheries have largely developed independently (Arlinghaus et al., 2008; Fenichel et al., 2013a; Heck et al., 2015). Academic tendencies toward increasing disciplinarity have ultimately led to a separation rather than integration of the ecological and social sciences in resource management, as evidenced by the formation of niche journals, such as the Human Dimensions of Wildlife, or textbooks either focused on the human dimensions of fish and wildlife (Decker et al., 2012) or fisheries ecology (Sass and Allen, 2014; Craig, 2016) with only tangential treatment of the interdisciplinary aspects of fisheries. Even within the human dimensions, social-psychologists and economists have long expressed different world-views, theories and assumptions leading to little cross-fertilization and cross-referencing of each other’s work (Fenichel et al., 2013a). Such academic silos have prevented the effective integration of social and ecological sciences in recreational fisheries (Heck et al., 2015), thereby constraining our understanding of the dynamics and effects of feedbacks among the social and ecological subsystems of recreational fisheries (Arlinghaus et al., 2008, Arlinghaus, 2014; Ward et al., 2016). Better appreciating the various feedbacks among humans

![Figure 1. A schematic view of recreational fisheries as a social-ecological system at the highest level of aggregation.](image-url)
and nature in a structured way and learning how these feedbacks along with all involved structures (e.g., organizations) give rise to specific outcomes (such as conflicts or biological unsustainable exploitation) is however essential for building sustainability in fisheries and in other coupled SESs (Biggs et al., 2012; Larroza et al., in press).

The historic development of academic silos is very unfortunate because recreational fisheries science provides an exceptional opportunity ("model") for cutting-edge work in the emerging field of SES science. In contrast to many other SESs, recreational fisheries are characterized by very direct interactions between humans and nature through the consumptive use of renewable natural resources and its active management (Figure 1), while at the same time displaying great diversity in behaviours, values, ecosystems, organizations and policies at the landscape level. Importantly, as mentioned above, anglers in many areas of the world are not only consumers of wild fishes, but also engage directly in their management. This renders the SES of recreational fishing useful for understanding general drivers inherent in human-environment relationships (Post et al., 2002; Carpenter and Brock 2004; Brock and Carpenter, 2007; Biggs et al., 2009; Roth et al., 2010; Horan et al., 2011; Hunt et al., 2011). Put simply: by learning how recreational fisheries function, one can learn a great deal about how humans in general like to interact with nature and the consumable products of nature.

In recent years there has been an increasing emphasis on interdisciplinary studies linking social and ecological dynamics in recreational fisheries (Carpenter and Brock, 2004; Biggs et al., 2009; Johnston et al., 2010; Hunt et al., 2011; Horan et al., 2011; Johnston et al., 2013, 2015). Many of the key feedbacks, among anglers and managers on the one hand and among anglers and freshwater ecosystems on the other hand, involve behavioral dynamics, which is why studying interactions in recreational fisheries from the perspective of the behavioral sciences has become prominent both in the social but also in the ecological domains (reviewed by Ward et al., 2016). Studies on coupled SESs provide fundamental insights into how humans interact with nature and the outcomes that these interactions produce (Ostrom, 2007, 2009; Hinkel et al., 2014, 2015). Such insights are urgently needed to steer development toward sustainability while maximizing human well-being within the limits of the earth’s life support systems (Carpenter et al., 2009; Röckstrom et al., 2009). The central purpose of SES science is to identify the conditions that produce sustainable outcomes in the face of feedbacks, cross-scale interactions, change and uncertainty. To conclude, evaluating recreational fisheries as a coupled SES not only promises to help us better understand local and regional fisheries dynamics, but may also inform a much broader literature on general human–environment interactions.

The objectives of this article are to present an emerging systems-view of freshwater recreational fisheries and to integrate this view into two interacting interdisciplinary frameworks. The article begins with a review of theory on complex adaptive systems (CAS) (Mahon et al., 2008; Levin et al., 2013) and SESs (Ostrom, 2007, 2009) to illustrate whether freshwater recreational fisheries display characteristics of such systems. Subsequently, insights from environmental complexity theory, common-pool-resource theory, resilience thinking and fisheries science are integrated into two interrelated frameworks. Frameworks are important because they help the analyst (i.e., us) in organizing diagnostic, descriptive, and prescriptive inquiry (McGinnis and Ostrom, 2014). The first analytical framework is designed to stimulate and structure future interdisciplinary study of recreational fisheries as complex adaptive SESs by focusing on issues that foster understanding on how recreational fisheries systems work. Although it is difficult to be value-free, even as a scientist, the aim of the first framework thus is to foster basic scientific understanding (“positive science” in the wording of Fenichel et al., 2013b), rather than management application or the evaluation of fishery outcomes. The second related framework aims to help the analyst choose management actions that are robust to uncertainty using ideas of adaptive management (Walters, 1986; Hansen et al., 2015). The article ends with reflection on key principles that emerge when recreational fisheries are viewed as complex adaptive SESs.

In the narrative examples given throughout the rest of the article, a freshwater, rather than marine, focus on recreational fisheries is chosen for two reasons. First, anglers are the main user of freshwater fisheries and most research on recreational fisheries has thus focused on freshwater fisheries (Craig, 2016). Second, freshwater fisheries are characterized by a large degree of social-ecological complexity stemming from individual and spatial and temporal heterogeneity in both the social and ecological systems (Arlinghaus et al., 2002, 2013; Johnston et al., 2015). Hence, developing a view of freshwater recreational fisheries as complex adaptive SESs may be particularly relevant to the emerging field of studying human exploitation systems as CASs (e.g., commercial fisheries: Mahon et al., 2008; forestry; Messier et al., 2015) where individual and spatial heterogeneity are key sources of uncertainty that need to be understood and potentially modelled (Levin et al., 2013).

2. Recreational fisheries as complex adaptive SESs

There has been a tendency among freshwater recreational fisheries scholars and managers to take a
“piscicentric” (Cowx et al., 2010), “reductionistic” (Aas, 2002) and overall “narrow” (Pope et al., in press) view that tends to focus on local biological and social issues and particular target species. For example, many articles in recreational fisheries science deal with issues of how much recreational harvesting a local exploited fish population can withstand or descriptively describe the human dimensions (demographics, motivations, etc.) of a regional or state-level population of anglers. Although such “case-study” based research is certainly valuable, the perspective omits the complex social and economic dimensions that often drive local- or regional-scale features and patterns such as cumulative fishing effort or the size of a local angling subpopulation (Arlinghaus et al., 2008; Hunt et al., 2013; Fenichel et al., 2013a; Pope et al., in press). In reality, all local recreational fisheries are embedded in other higher-order SESs, such as land use systems and society at large (Hunt et al., 2013; Pope et al., in press). Changes and developments in the ecological and social fabrics of this broader SES will almost always exert effects, directly or indirectly, on a particular fishery and create many trade-offs related to a range of ecosystem services provided by ecosystems (Pope et al., in press). Relatively, a local recreational fishery will be sensitive to changes in regional ecological factors, technology, communication channels, altered norms and expectations of society about what fisheries management and anglers ought to do (e.g., fish welfare debate in central Europe; Arlinghaus et al., 2012), or altered fishing participation in response to urbanization and demographic change (Arlinghaus et al., 2015a). Therefore, a “reductionistic” perspective risks missing important cross-scale effects. Similarly, a predominantly ecological view of recreational fisheries is bound to lead to substantial implementation uncertainty because human behavioural responses to new management tools or to changes in society are often poorly considered or ignored when setting management targets and associated regulations (Johnston et al., 2010; Fulton et al., 2011; Fenichel et al., 2013a).

When taking a systems perspective, all regional-level (i.e., macro-scale) outcomes (e.g., overfishing across a landscape of spatially structured lakes and river sections; Hunt et al., 2011) are an emergent property of local fishery-level (i.e., micro-scale) interactions of anglers with fish stocks and local ecosystems that in turn span their effects across spatio-temporal scales. Hence, outcomes observed at the macro-scale will always arise from multiple local-scale interactions among anglers and their interplay with fish stocks and ecosystems (Post et al., 2008). To understand macro-scale patterns, science must thus address the processes and mechanisms that operate locally and across local to regional and even international scales. A holistic understanding of recreational fisheries can in consequence only be achieved through integration of the social and biological sciences across scales and sectors (Arlinghaus et al., 2008; Post et al., 2008; Hunt et al., 2011, 2013; Hansen et al., 2015), which is defined here and elsewhere (Arlinghaus et al., 2014a) as interdisciplinary science. The goal of such research is to arrive at insights into what systematically drives recreational fisheries and how recreational fishing-related activities (e.g., harvesting, stocking) feedback to influence the food web, ecosystems and other stakeholders traditionally not considered part of the recreational fisheries system (e.g., hydropower plant operators, wildlife viewers, nature conservation agencies, Pope et al., in press). To give credit to the complex set of interactions on different spatial scales that characterize most contemporary recreational fisheries (e.g., in fishing tourism; Ditton et al., 2002, or in relation to harvesting pressure, Chizinski et al., 2014), it is thus useful to take a systems view and conceptualize recreational fisheries as complex adaptive SESs characterized by many interactions, feedbacks and processes within and among social and ecological components and over multiple scales (Levin, 1998; Levin et al., 2006; Ostrom, 2009; Schütter et al., 2012; Hunt et al., 2013; Hinkel et al., 2014, 2015). In addition to the obvious coupling of social and ecological dimensions, any coupled SES is characterized by three key features, viz.: (i) complex adaptive dynamics, (ii) cross-scale relationships and hierarchies, and (iii) non-linearities (Berkes et al., 2003; Levin et al., 2013). In the following subsections it is shown and argued that recreational fisheries are defined by all of these qualities.

### 2.1. Complex adaptive systems

All SESs are also CASs (Levin et al., 2013). Because features of a CAS do not necessarily demand an interaction with nature to emerge (Levin, 1998, 1999), first those properties of recreational fisheries are described that fall within the traditional realm of CAS, before adding those features that are specific to them being coupled SESs.

An adaptive system is characterized by a set of interacting entities that produce a coherent whole, which responds to environmental change by altering its interacting parts (Levin, 1999). Change and adaptation is hence an inherent feature of adaptive systems. Examples of adaptive systems include the liver or other tissues, individual organisms, natural ecosystems, and human-constructed organizations (such as a fisheries management agency).

CASs, as introduced by Levin (1998), are characterized by an additional set of three key attributes: (i) diversity and individuality of components, (ii) localized...
micro-scale interactions among components that lead to emergent macro-scale patterns, and (iii) an autonomous, self-organized process that uses the outcomes of local interactions as feedback for adaptation through selection and evolution. Note it has become commonplace to talk about “evolution” in the literature on CAS, where it either can mean biological evolution (i.e., genetic change) or simply development, change and adaptation in a non-biological sense as shown by human agents (Levin, 1999). For example, in adaptive management, environmental feedback could lead to the sorting of competing management tools by the outcomes emerging from their implementation, in turn leading to decisions about which concrete management direction to follow in the future and which ones to omit (Walters, 1986).

All three key descriptors of CAS mentioned above apply to recreational fisheries. Examples include: (i) angler populations exhibiting individual variability in demographic characteristics and values, beliefs, attitudes, norms and preferences, leading to substantial diversity among anglers in expectations, desires, opinions and expressed behavioural patterns (Bryan, 1977; Fisher, 1997; Aas et al., 2000; Arlinghaus, 2004; Dorow et al., 2010; Beardmore et al., 2011), (ii) angler behaviour at the micro-scale level affecting macro-scale patterns such as the spatial configuration of effort and relatedly regional overharvesting patterns (Carpenter and Brock, 2004; Post et al., 2008; Hunt et al., 2011), and (iii) local patterns of fishing quality and fisheries management affecting the decisions of anglers to renew licenses (e.g., to drop in or out of the pool of anglers) through patterns of self-selection and self-organization (Loomis and Fix, 1998; Hunt et al., 2007; Post et al., 2008; Abbott and Fenichel, 2013; Dabrowska et al., 2014).

The three qualities of CAS foster the potential for (largely unexpected and difficult-to-predict) emergent properties caused by individual and spatio-temporal heterogeneity of the system’s components, and its self-organizing potential. As in biological evolution, diversity provides the raw material for selection and change, which can lead to system outcomes that are different from ones where all actors, species or genes would be identical. For example, Ward et al. (2013a, b) reported that anglers’ ability to catch fish (catch-ability) and decisions about where to fish were both influenced by the angler types present in the regional fishery. Where certain angler types will be fishing is a result of the self-organized effort distribution pattern on the landscape, which has both biological and social consequences (Johnston et al., 2010), such as localized overfishing near urban areas and underutilization of other fisheries (Hunt et al., 2011; Post and Parkinson, 2012). Similarly, it is likely that angler types differ in the technologies they use (e.g., boats), which through a self-regulatory process of site choice can lead to angler-type-dependent spread of non-native organisms attached to boats or released from bait buckets (Johnson et al., 2009; Drake and Mandrak, 2014). As these two examples demonstrate, the self-organization properties of a CAS will not necessarily lead to socially optimal outcomes (Levin et al., 2013; for examples from recreational fisheries see Johnston et al., 2010; Fenichel and Abbott, 2014). A focus on heterogeneity is thus very important in studies of human-environment systems (Levin et al., 2013).

2.2. Cross-scale relationships and hierarchical organization

Complex adaptive SESs are hierarchically organized into structural arrangements that determine and are reinforced by the links, flows, feedbacks and interactions among the system components (Levin, 1999). Such hierarchies are also a key feature of many freshwater recreational fisheries (Figure 2). For example, management organizations are usually established either at national-levels or state-levels. Their policies and regulations affect local and regional angler behaviour, which affect patterns of site choices and angling effort; changes in local or regional patterns of effort and exploitation might in turn induce policy and operational management responses (Daedlow et al., 2011). Hierarchical structures and cross-scale interactions are self-evident in a landscape of freshwater fisheries where anglers interact with localized fish stocks, each with its own internal dynamic, which are then linked through anglers’ mobility (Figure 2). The outcomes of the interactions are managed by local, regional or sometimes even national management organizations that add spatial heterogeneity to the system (Figure 2). Declines in perceived fishing quality often lead to calls by anglers for compensatory fisheries management actions, such as stocking, which may develop into a rigid panacea reinforced by negative (stabilizing) feedback (Box 1; van Poorten et al., 2011). Such cross-scale feedbacks are the primary reason to study complex adaptive SESs simultaneously at different scales, leading to important challenges for modelling (Schlüter et al., 2012; Levin et al., 2013). To make matters more complex, recreational fisheries are embedded and affected by other SESs, such as water management or agricultural systems (Hunt et al., 2013). Phenomena at each scale have their own emergent properties, which feedback onto other levels and systems and vice versa. Gunderson and Holling (2002) coined the term “Panarchy” to refer to the adaptive nature of interactions and feedbacks of various subsystems in SESs that are hierarchically organized at several spatial, temporal and organizational scales (e.g., individual fish, nested in stocks, nested in lakes, nested...
in a landscape of lakes) all governed by an overarching management agency or a set of independent decision-makers such as anglers in angling organizations (Figure 2). Each subsystem follows a different temporal and spatial dynamic and is influenced by subsystems above and below (represented by the “figure-eight” graphic, Figure 2). Each subsystem constantly changes and adapts, creating micro- and macro-level outcomes (e.g., ecosystem services) that are valued by and elicit responses from both anglers and managers (Figure 2).

### 2.3. Non-linearity

The complex adaptive nature of SESs fosters the emergence of non-linearity, which fundamentally is about uncertainty and unpredictability and involves the potential for regime shifts and alternative stable states (Box 2) (Berkes et al., 2003; Mahon et al., 2008; Levin et al., 2013). Non-linearities exist both at the system and at the variable level, and it is important to grasp the conceptual differences among the two.

CASs change primarily through events that alter local-level interactions and rules (Levin, 1998). Many of these events are chance events, such as responses to environmental stochasticity, random establishment of a new species, technological innovation altering exploitation pattern or ecological impacts or societal-level value changes. These events and associated interactions “set the stage” (i.e., the path) locally at the micro-level, on which the CAS unfolds specific features and qualities at the macro-level through selection, development and cross-scale interactions. Because there is a large number of possible chance events and resulting trajectories governed by local interactions, it follows that the potential for alternative developmental pathways is large in any CAS (Levin, 1998), hence the term non-linearity. For example, the unexpected establishment of European catfish (Silurus glanis) in the Ebro delta in Spain (which goes back to illegal release by anglers) has completely altered all sorts of interactions, processes and outcomes, ecologically and socially, in the local recreational fishery. Today, there is significant foreign tourism and associated service sectors surrounding the recreational fishery for non-native giant catfish that was not present before the fish was released and the fishery was small scale, resident-only and dependent on native species only. The very same (eco)system could today also have existed in a completely different state and configuration.

**Figure 2.** A schematic view of cross-scale interactions (solid arrows in black and grey) among and across subsystems in a freshwater recreational fisheries landscape, each of which is constantly changing and adapting (visualized by the “figure-eight” model from Gunderson and Holling 2002). The size of the subsystem is largest and the speed of change slowest to the upper right and smallest and fastest to the lower left. “Figure-eight” symbols are $K = \text{conservation}$, $\Omega = \text{release}$, $\alpha = \text{reorganization}$, $r = \text{exploitation}$. $K$ indicates a mature system state at equilibrium; $\Omega$ indicates the sudden release of the system from its equilibrium after a disturbance, which opens a window of opportunity for change and adaptation; $\alpha$ indicates the phase of reorganization and change; and $r$ is the phase where the reorganized system accumulates structure and capital toward a new steady-state. Changes at lower levels of organization can induce a release phase above (“revolt”) while higher order levels may provide capital and resources (“remember”) such that a lower level systems may be geared toward maintaining structure and function following a disturbance (e.g., compensatory reserve in fish stocks to dampen exploitation).
It is important to avoid confusion with the more traditional use of the term non-linearity in ecology or social sciences. Traditionally, non-linearity refers to non-linear relations of one or two variables (e.g., stock size and recruitment rate). By contrast, in the literature on CAS, non-linearity refers to the fact “that the local rules of interaction change as the system evolves and develops” (Levin, 1998). Taking this perspective, path dependency and the potential for multiple possible outcomes of dynamics are consequences of non-linearity (Levin, 1998).

Box 1. Cross-scale interactions maintaining fish stocking as panacea in German recreational fisheries

Over a five-year period, the Besatzfisch project (www.besatz-fisch.de) studied key ecological and social determinants of fish stocking decisions in Germany. Fish stocking is the dominant management tool, and stocking is considered a management panacea in Germany as in many other central European countries where local fisheries are managed via fishing rights commonly held by about 10,000 independent clubs and angler associations (Daedlow et al., 2011). The lack of adaptation of the current stocking regime is caused by a range of stabilizing (negative) feedbacks across the entire SES as illustrated in Figure Box 1 and outlined in detail in Arlinghaus et al. (2015b). The brief stylized summary given here cannot give credit to the full complexity of local level decision-making. In brief, the locally self-organized stocking system is largely driven by three actors operating at local and regional scales—fisheries managers in local angling clubs, mobile anglers and hatchery operators. Although fisheries managers in agencies operate at the highest organizational level in the German states, there is only a tangential role for these actors in stocking decision-making. The lack of involvement of agency personnel including fisheries scientists, coupled with the weak regulatory feedback associated with private fishing rights has strongly contributed to the emergence of stocking as a local management panacea. In the absence of any fishery-independent monitoring of stocking outcomes, most feedback about current system status reaches local fisheries managers in the club through catch reports and individual complaints. As a result, the local fishery is not able to assess objective signals of stocking success or failure, preventing learning by both local managers and anglers about whether and when stocking is needed. Angling clubs work under the objective of sustainable exploitation in line with Federal Nature Conservation and State-Level Inland Fisheries Legislation, with a secondary objective of maximizing club-level membership satisfaction. The general perception among both managers and anglers in the clubs is that most lakes and rivers are impacted by anthropogenic habitat loss and hence inland water bodies are in “need” of supplemental stocking to maintain both species from a nature conservation perspective and fisheries. A long history of stocking in freshwater has created path dependencies and a habitual reliance on stocking relative to other tools (Klein, 1996). Psychologically, the biggest drivers of local managers’ decisions to stock are strong personal norms that stocking is necessary to maintain fish populations and angling quality. These personal norms are reinforced by the lack of alternatives that could be as easily implemented as stocking (e.g., habitat management) and that do not risk social conflicts (e.g., effort controls). Moreover, the decision makers in the club are affected by strong pro-stocking social norms from the club’s membership whose satisfaction is largely catch-dependent (Arlinghaus et al., 2014b). Angler satisfaction, however, often widely fluctuates from year to year due to natural stochasticity in stock recruitment (van Poorten et al., 2011), which creates an incentive for anglers to call for stocking to increase stability. The limited ability of anglers to learn whether stocking works or not in turn fosters a belief that stocking may be needed to maintain catches and that stocking is an immediate and desired investment of their local angling license fees (Arlinghaus et al., 2015b). In summary, high pro-stocking norms fostered by the lack of easily implementable alternative management tools and a lack of involvement of government agencies and fishery scientists to assist in local stocking decisions lead to a largely self-organized management system that almost exclusively relies on stocking to provide quality fisheries. Loss aversion (Kahneman et al., 1991), i.e., the reluctance to provisionally stop stocking because aversion to a potential loss in catch outweighs desire for potential gains, prevents objective assessment of stocking outcomes and results in maintenance of current practices. In addition, the third key player—commercial hatchery operators—has an economic interest in selling fish for stocking. There are strong social and informational networks among club managers and regional hatchery operators. Given the vested economic interest to sell stocking material, there is a tendency for hatchery operators to advise club managers to stock regularly. With limited supply of local-origin stocking material, stocking often happens with foreign stocks and genotypes, fostering large-scale transfers of non-native genotypes across catchments (Eschbach et al., 2014; Arlinghaus et al. 2015b). As there is little preference among German club anglers for wild fishes (Arlinghaus et al., 2014b), these large-scale fish transfers across catchments are not a source of concern for anglers.
A consequence of path dependency, in turn, is the existence of multiple basins of attraction in ecosystem development and the potential for threshold behavior, regime shifts (Box 2) and qualitative shifts in system dynamics under changing environmental influences (Levin, 1998). Thus, the type of non-linearity characteristic for CAS at the systems level can render precise modelling or forecasting untenable. As one reviewer of this article elegantly observed, this brings us to the decisive difference among system and variable level non-linearity: system-level non-linearity is what makes CAS “complex” and (largely) unpredictable, whereas the many non-linear processes known to the fisheries scientist or ecologist (e.g., the non-linear relationships between stock and recruitment, Ricker, 1954; Beverton and Holt, 1957, the non-linear relationship of catch-per-unit effort and fish abundance, Hilborn and Walters, 1992; Ward et al., 2013b, or the non-linear relationship of angler satisfaction and catch rates, Johnson and Carpenter, 1994; Arlinghaus et al., 2014b; Beardmore et al., 2015) makes system understanding merely “difficult.”

Understanding the systematic effect of non-linear social-ecological interactions at the variable level is nevertheless important and particularly appealing to both academics and managers, because these processes alone can already lead to unexpected and potentially undesirable outcomes at the macro-level (Box 1). But, merely “difficult” non-linearities at the variable level could potentially be fully understood with better data and models, and accounting for key non-linear relationships in models could then be used to analyze the outcome of possibly very complicated policy scenarios. As an example, consider a spatially structured freshwater recreational fishery that is composed of several lakes and rivers that are linked through angler movement and are governed by regional management authorities and organizations (Figure 2). Lake-specific productivity and abundance and size-structure of vulnerable fishes will dictate local catch rates and size of fish captured (Ward et al., 2013a,b), and in turn drive local angler satisfaction (Beardmore et al., 2015). Catch outcomes experienced by an angler feed into the angler’s non-linear utility function (i.e., a relational description of factors producing benefits; Fenichel...
et al., 2013a; Arlinghaus et al., 2014b), but the utility and site choice also depends on other factors that have nothing to do with expected catches (e.g., distance, accessibility, scenic beauty, expected crowding; Aas et al., 2000; Hunt, 2005; Post et al., 2008; Dorow et al., 2010; Beardmore et al., 2013; Arlinghaus et al., 2014b). Changes in a local fishery of any of these catch or non-catch attributes will attract or repel certain angler types contingent on their preferences (Post et al., 2008; Johnston et al., 2010; Beardmore et al., 2011; Hunt et al., 2011). The resulting local fishing pressure not only reduces abundance and alters size structure, but surviving fishes might also respond by reducing their intrinsic vulnerability, e.g., learning after catch-and-release (Beukema, 1970; Raat, 1985; van Poorten and Post, 2005; Askey et al., 2006; Klefoth et al., 2013), potentially adding another layer of non-linearity (Ward et al., 2013b). Any ecological and behavioural changes will affect future catch rates as anglers learn from past experience, leading to emergent and non-linear patterns of effort (e.g., some anglers with weak skills and high expectations may abandon a local fishery earlier than others; Ward et al., 2013a, but see Seekell et al., 2013). These self-organized processes can in turn lead to other unexpected emergent properties, such as alterations in density-dependent population-level catchability (Ward et al., 2013b), which can keep average catch rates high despite declining fish abundance. The emergence of non-linear relationships of abundance and catch rate (e.g., hyperstability in catches, Post et al., 2002; Erismann et al., 2011) may potentially result in further non-linear, depensatory processes that increase the probability of stock collapse (Post, 2013). Alternatively, reduced vulnerability of surviving fishes might decouple fishery dependent stock assessments from actual stocks due to hyperdepletion effects (Alós et al., 2015a,b), which might foster ill-fitting management responses, such as intensive stocking to satisfy the expectations of (often dissatisfied) anglers (van Poorten et al., 2011). As this narrative example has shown, the interplay of localized interactions of lake-specific fish populations and diverse mobile anglers interacting with spatially and temporally heterogeneous lakes not only leads to non-linear self-organized patterns of angling pressure at the macroscopic scale, but could also create misleading signals about localized stock abundance. These signals could lead to system-level collapses that are difficult to foresee (Post, 2013). Only a social-ecological perspective on recreational fishing can address such complex interactions, and the system may even be considered more complex when one accounts for mobility of fish stocks through interconnected systems of lakes and rivers. But the good news is—all of the above is, in theory at least, quantifiable and could be integrated in (certainly complex) social-ecological models.

Non-linearity at the systems-level will, however, always be very difficult or impossible to be fully addressed, but is nevertheless important to be alert about, for instance because it can foster difficult to foresee regime shifts in systems that exhibit alternative stable states maintained by internal feedback processes (so-called negative feedback; see Scheffer and Carpenter, 2003; Scheffer et al., 2001; Boxes 1 and 2). Identifying indicators for such regime shifts is of ongoing academic interest (Cline et al., 2014). Regime shifts in freshwater systems (Angeler et al., 2014) are often driven by changes in critical slow variables, which are differentiated from fast variables by the speed of turnover from year to year (Carpenter et al., 2011). Slow variables are critically important for driving the functioning of the system and affect the type of regime a given system may be “locked in” (Carpenter et al., 2011; Angeler et al., 2014). Prototypical examples of critical slow variables of relevance for recreational fisheries include the presence of spawning and refuge habitat, genetic diversity at the fish population level or diversity in social values, organizations and institutions (Roth et al., 2007; Biggs et al., 2009; Roth et al., 2010; Horan et al., 2011). Slow, but steady, changes in these critical slow variables will be without an obvious effect until a threshold is surpassed, after which the system can flip into a different “basin of attraction” (i.e., the alternative stable state) from which reversal is difficult (Scheffer et al., 2000; Persson et al., 2007; Carpenter et al., 2011; Box 2). The exact transition configuration for the critical slow variable is usually not known and hence it is very difficult to predict and manage. Moreover, patch dependencies and chance events can lead to “lock ins” in particularly undesirable states (Box 2). Generally, managers and recreational fishers tend to be unresponsive to changes in (and may not even aware of) critical slow variables, until it may be too late and the system has flipped into an alternate state (Pope et al., 2014; Hansen et al., 2015). Examples of abrupt changes were reported in several recent articles focusing on recreational fisheries and include stable loss of top predators and erosion of size structure, the sudden establishment of stocked genotypes and replacement of wild fish, the spread of non-native organisms pervasively changing ecosystem configurations, and rapid and stable shifts in governance and regulatory environments (Box 2; Persson et al., 2003, 2007; Arlinghaus, 2007; Brock and Carpenter, 2007; Biggs et al., 2009; Roth et al., 2010; Carpenter et al., 2011; Arlinghaus et al., 2012; Daedlow et al., 2013). Unfortunately perhaps, data that can be used to identify “leading indicators” of regime shifts (i.e., indicators that warn that a system shift is about to happen; Carpenter and Brock, 2010; Cline et al., 2014) is often absent or inadequate in many recreational fisheries (Hansen et al., 2015). One should, however, note that most of the examples given above are from models and
Regime shifts may lead to “traps” or “lock ins” from which escape is unlikely without massive intervention (Carpenter and Brock, 2008). For example, recreational fisheries exist in various stable domains of attraction (“states”) with and without controls on angling practices in terms of ethics-based fish welfare. The slowly changing variables in this case are pro-environmental and pro-animal welfare cultural values and the degree to which animal welfare legislation is enforced in relation to angling (Manfredo, 2008; Arlinghaus et al., 2012; Riepe and Arlinghaus, 2014a,b). Up until the 1970s there was little debate about fish welfare in recreational fisheries in Germany. Anglers engaged in a range of practices such as competitive fishing with catch-and-release or the use of live bait. Change in environment-related cultural values during and after World War II led to the establishment of general animal welfare legislation. In the wake of the social reform movement in the 1960s until the 1980s, social pressure and public attention toward animal welfare accumulated and fostered increased scrutiny of popular angling practices (Arlinghaus, 2007). After a series of court cases against angling practices such as competitive fishing with catch-and-release, a rapid change in the moral acceptability of certain angling practices occurred in Germany in the mid 1980s and Switzerland in the 2000s. Now it is illegal to voluntarily catch-and-release fish of harvestable size and to engage in fishing competitions that involve catch-and-release, based on the assumption that catching and releasing food for fun is not morally defensible and is traumatizing to fish (Arlinghaus et al., 2009; Riepe and Arlinghaus, 2014a,b). In addition, anglers now have to pass examinations including training in animal welfare to be entitled to fish recreationally (Arlinghaus, 2007; Arlinghaus et al., 2012). A range of legal, social, political and psychological forces keep the German governance system with respect to fish welfare in a “locked” state, with little opportunity for debating novel management regulations, such as the value of protecting large highly fecund fishes from harvest through harvest slot regulations (Arlinghaus et al., 2010; Gwinn et al., 2015). Despite accumulation of biological support for the need to protect both immature and very large fecund fishes (Hixon et al., 2013), German fisheries professionals are reluctant to consider these novel regulations because they are presumed (at odds with reality; Riepe and Arlinghaus, 2014a,b) to be perceived by the public as unethical. Moreover, for similar reasons there are strong forces to constrain stocking of catchable fishes as well as put-and-take fisheries despite the mounting evidence that larger, rather than smaller, stocking sizes often produce greater fisheries benefits when released in naturally reproducing stocks (Lorenzen, 2005). The concern is that animal welfare critics might argue that releasing fish that had already been raised to market-size into a put-and-take fishery or a local angling pond is unnecessarily cruel to fish (unnecessary recycling of harvestable fishes). Recent highly visible media reports in Germany have indeed portrayed anglers operating in put-and-take fisheries as sadists, contributing to the maintenance of status quo policy within fisheries agencies (Arlinghaus, 2014). As a result, fisheries professionals commonly pressure fishing rights holders in Germany to maintain status quo regulations (e.g., one size fits all minimum-length limits or stocking of only small juveniles or fry), despite evidence of suboptimal outcomes. This is an example of a social regime shift driven by feedback processes related to social and political change at higher levels of societal organization trickling down to affect local fishing practices and management approaches.

**Box 2. An example of a social regime shift related to fish welfare in Germany**

Regime shifts may lead to “traps” or “lock ins” from which escape is unlikely without massive intervention (Carpenter and Brock, 2008). For example, recreational fisheries exist in various stable domains of attraction (“states”) with and without controls on angling practices in terms of ethics-based fish welfare. The slowly changing variables in this case are pro-environmental and pro-animal welfare cultural values and the degree to which animal welfare legislation is enforced in relation to angling (Manfredo, 2008; Arlinghaus et al., 2012; Riepe and Arlinghaus, 2014a,b). Up until the 1970s there was little debate about fish welfare in recreational fisheries in Germany. Anglers engaged in a range of practices such as competitive fishing with catch-and-release or the use of live bait. Change in environment-related cultural values during and after World War II led to the establishment of general animal welfare legislation. In the wake of the social reform movement in the 1960s until the 1980s, social pressure and public attention toward animal welfare accumulated and fostered increased scrutiny of popular angling practices (Arlinghaus, 2007). After a series of court cases against angling practices such as competitive fishing with catch-and-release, a rapid change in the moral acceptability of certain angling practices occurred in Germany in the mid 1980s and Switzerland in the 2000s. Now it is illegal to voluntarily catch-and-release fish of harvestable size and to engage in fishing competitions that involve catch-and-release, based on the assumption that catching and releasing food for fun is not morally defensible and is traumatizing to fish (Arlinghaus et al., 2009; Riepe and Arlinghaus, 2014a,b). In addition, anglers now have to pass examinations including training in animal welfare to be entitled to fish recreationally (Arlinghaus, 2007; Arlinghaus et al., 2012). A range of legal, social, political and psychological forces keep the German governance system with respect to fish welfare in a “locked” state, with little opportunity for debating novel management regulations, such as the value of protecting large highly fecund fishes from harvest through harvest slot regulations (Arlinghaus et al., 2010; Gwinn et al., 2015). Despite accumulation of biological support for the need to protect both immature and very large fecund fishes (Hixon et al., 2013), German fisheries professionals are reluctant to consider these novel regulations because they are presumed (at odds with reality; Riepe and Arlinghaus, 2014a,b) to be perceived by the public as unethical. Moreover, for similar reasons there are strong forces to constrain stocking of catchable fishes as well as put-and-take fisheries despite the mounting evidence that larger, rather than smaller, stocking sizes often produce greater fisheries benefits when released in naturally reproducing stocks (Lorenzen, 2005). The concern is that animal welfare critics might argue that releasing fish that had already been raised to market-size into a put-and-take fishery or a local angling pond is unnecessarily cruel to fish (unnecessary recycling of harvestable fishes). Recent highly visible media reports in Germany have indeed portrayed anglers operating in put-and-take fisheries as sadists, contributing to the maintenance of status quo policy within fisheries agencies (Arlinghaus, 2014). As a result, fisheries professionals commonly pressure fishing rights holders in Germany to maintain status quo regulations (e.g., one size fits all minimum-length limits or stocking of only small juveniles or fry), despite evidence of suboptimal outcomes. This is an example of a social regime shift driven by feedback processes related to social and political change at higher levels of societal organization trickling down to affect local fishing practices and management approaches.

2.4. Synthesis and key research needs

From the above, five key features can be synthesized that characterize the complex adaptive SES of freshwater recreational fisheries and that in turn demand increased attention in future studies. First, recreational anglers are a particularly heterogeneous group of natural resource users. Anglers differ strongly in their expectations, preferences, and behaviours, which makes reliable prediction of angler behaviour a constant challenge (Johnston et al., 2010; Ward et al., 2013a,b). Second, the utility functions commonly used to model commercial fisheries are dominated by profit maximization given costs. By contrast, utility functions of anglers and the related concept of angler satisfaction are multi-dimensional involving both...
catch and non-catch components (Fedler and Ditton, 1994; Manfredo et al., 1996; Hunt, 2005). As a result, a priori prediction of angler behaviour is particularly challenging (Hunt, 2005; Beardmore et al., 2011, 2013; Arlinghaus et al., 2014b), and many aspects beyond catch rate or yield must be accounted for in setting management objectives for recreational fisheries (Johnston et al., 2010, 2013, 2015). Moreover, anglers value non-catch aspects of the fishing experience, and environmental degradation, difficult access, and crowding are important contributors to (dis)utility (Hunt, 2005; Beardmore et al., 2013; Arlinghaus et al., 2014b). Consequently, recreational fisheries are not necessarily self-regulating in a biological sense (i.e., anglers do not necessarily leave a fishery when catches decline, but see Askey et al., 2013), and in fact anglers often continue fishing even on dwindling stocks because of other factors that are attractive to anglers (Post et al., 2002; Hunt et al., 2011; Post, 2013). Hence, the overexploitation potential can be strong in highly populated areas with high latent angling effort (Hunt et al., 2011). Third, freshwater angling systems are usually spatially structured, with individual anglers linking multiple sites within a fishery and also interacting across multiple fisheries (lakes, river segments) in a region (Post et al., 2008; Hunt et al., 2011). This structure adds fundamental spatial and temporal variation as productivities and impacts vary strongly among lakes, even those nearby (Johnston et al., 2016). The situation differs from many marine systems where multiple stocks form one large metapopulation that tends to be driven by similar internal dynamics on wider spatial scales. Although fishers’ mobility is also key in marine systems, the situation is more complex in freshwater fisheries when anglers link fish stocks from multiple ecologically independent resource patches (Hunt et al., 2011). Important complexity thus far poorly tackled by scientists also includes spatial variation associated with variable governance structures. For example, in North America, the dominant situation is that anglers are managed by some state-level authority and access to all state waters is in principle open to license holders, while in central Europe fishing rights are private and anglers are organized in numerous angling clubs with fishery access confined to club waters only (Daedlow et al., 2011). Such patterns create a mosaic of many independent “mobility patches” within clubs with relatively minor spill-over of effort across club boundaries. Notably, anglers may join several clubs or fish beyond state boundaries on commercially operated fisheries or in coastal zones where no club structures are present. Different clubs or management units are governed, potentially, by different rules, regulations and norms, including variation in state-specific fisheries legislation – all of which adds additional layers of spatial heterogeneity that has thus far not been addressed. Fourth, management approaches to angling have often been ad-hoc with little scientific underpinning, which may have contributed to the development of one-size-fits-all policies (e.g., one uniform harvest regulation applied to entire landscapes in all lakes and rivers; Carpenter and Brock, 2004) in many areas of the world. Because of the spatial heterogeneity of regional recreational fisheries, it is naïve to assume that one will ever have enough resources and manpower to develop management strategies based on stock-assessments as are typically employed for high value industrialized fisheries (Cooke et al., 2014). Thus, research on and management of recreational fisheries has to account for the data-poor nature of the activity and come up with practical solutions for monitoring, assessment and management guidance (Lester et al., 2003, 2014) against some abstract principles of sustainable management, such as resilience (Biggs et al., 2012, see for details further below). Fifth and finally, a particular feature of many recreational fisheries is the nested structure of the SES that is strongly affected by larger scale systems, such as agriculture or water management or societal values (Box 2) that are often beyond the control of fisheries managers. Hence, many solutions to local or regional management problems may be completely inaccessible to fisheries managers, demanding networks and partnerships with agencies responsible for other sectors such as agriculture or water management. Recreational angling also sometimes has low socio-political priority, operating in a data poor setting over large spatial scales involving thousands of people (Shuter et al., 1998; Lester et al., 2003, 2014; de Kerkhove et al., 2015).

Given these characteristics a number of key research areas ranging from ecological to institutional for understanding recreational fisheries as complex adaptive SESs are summarized in Table 1. The reader is also directed to Ward et al. (2016) for a summary of other key research priorities related to understanding bi-directional behavioural feedback processes. The institutional dimension is the one that probably will vary most across regions or nations, and property rights systems (Daedlow et al., 2011), and is also the most understudied in recreational fisheries science. Variation in governance systems across the world leads to either polycentric systems with many independent decision-makers (as in central European angling clubs), or to top-down systems where single agencies govern entire states such as the state-wide fisheries management agencies typical of most North American freshwater fisheries (Daedlow et al., 2011). Clearly, the particular configuration of property and management rights will not only strongly affect institutional dynamics and lead to path dependencies, but also
Table 1. Summary of key management-oriented research needs when viewing freshwater recreational fisheries as complex adaptive social–ecological systems.

| Effort dynamics and angler diversity: Dynamic angler responses in recreational fisheries (Larkin, 1978) are poorly known and rarely represented in the development of operational management policies (Askey et al., 2013; Mee et al., 2016). Monitoring programs tailored at observing how anglers react to social-ecological change are needed to properly understand the dynamics and provide responsive management. Ecologically, there is a need to understand “giving-up” densities and the exact form of the fish abundance–effort relationship for various angler types represented by varying catch-dependent or non-catch-dependent utilities and reward systems (Post, 2013). Angler effort dynamics may render populations prone to collapse (Post et al., 2002) or lead to self-regulatory outcomes that conserve fish stocks due to entry-exit dynamics related to intrinsic preferences of anglers for catching fish (Askey and Johnston, 2013) or aversion to strict harvest regulations (Johnston et al., 2011). Moreover, as angler types desire different rewards, diversified management is likely to generate more landscape-level benefits than one-size-fits-all policies. More research is needed on strategic management recommendations to better address the needs and desires of diverse angler populations. |
| Vulnerability and catchability: Individual fishes vary in their susceptibility to capture (Klefoth et al., 2013; Alós et al., 2015a,b). Similarly, anglers vary in their skill in capturing fish, and there is a systematic effect of climate and geography on vulnerability (Ward et al., 2013a; Mogensen et al., 2014). These processes are poorly understood, but drive effort dynamics and affect the ability of monitoring to foresee collapse. Inverse-density dependent catchability, e.g., due to effort sorting, may lead to hyperstable catch rates, increasing the likelihood of biological collapse (Post, 2013). Alternatively, changes in vulnerability with overexploitation may lead to hyperdepletion and thereby reduce the probability of collapse (Alós et al., 2015a,b). Evaluation of these competing hypotheses is needed, particularly to understand when angler catch rates indicate population declines. |
| Compliance with regulations: Replicated experiments are needed to directly measure regulatory compliance and to understand how to foster rule compliance in spatially-structured, diffuse freshwater fisheries, because lack of rule compliance undermines all regulatory efforts (Sullivan, 2002; Johnston et al., 2015). |
| Allee effects: A range of depensatory and food web mechanisms described in Post et al. (2002), Post (2013) and Walters and Kelch (2001) can lead to ecological Allee effects. The prevalence of this effect in freshwater systems remains an unanswered empirical question. |
| Demographic portfolio effect: Research is accumulating that the structure of the spawning stock is equally important to stability as overall spawning biomass. In general, there is an emerging hypothesis that diversity of age classes, genes and populations is important for safeguarding stability (Schindler et al., 2015; Gwinn et al., 2015), but whether this is a widespread phenomenon of importance in freshwater ecosystems is not currently known. The presence of an age-class diversity-based effect on recruitment would, however, fundamentally alter harvest regulations to move from minimum-length limits to harvest slots (Gwinn et al., 2015). |
| Fisheries-induced evolution: Fisheries can lead to changes in phenotypes, but also genotypes and affect behaviour and hence catchability. The degree and importance of fisheries-induced evolution is controversial. While fisheries-induced evolution might lead to increases in the compensatory ability of exploited stocks (Uusi-Helkkilä et al., 2015), it might also erode catchability and hence the quality of local fisheries (Philipp et al., 2009). |
| Food web effects: Emerging theory on size-structure in freshwater communities is challenging the traditional paradigm for modelling and understanding fisheries (Persson et al., 2014). The underlying size-structured theory has been mainly developed in small freshwater ecosystems where interaction strength is high, potentially leading to alternate stable states of stunted and unstunted predator populations (Persson et al., 2007; de Roos and Persson, 2013). By monitoring key variables, it is possible to study the empirical prevalence of this theory. |
| Landscape fisheries: Spatial heterogeneity among lakes is relevant to a range of issues, and adaptive management is necessary to learn how localized interactions lead to emergent properties at the landscape level. Combining adaptive management of landscapes with simulation models can help outline which policies to choose and test (Box 4). |
| Portfolio effect of policies: Freshwater fisheries offer ample opportunities to test different configurations of spatially heterogeneous management regulations, stocking and habitat enhancement, but the potential is untapped. There is the concern that engaging in such management would lead to overly complex outcomes (Pope et al., 2014). The reality is that managers already engage in spatially diverse management regimes without knowing whether landscape level outcomes are “optimal.” Adaptive management paired with modelling can help in answering this hugely important question. |
| Institutional dynamics: There is a paucity of knowledge on the drivers of institutional change and how decision-processes and incentives work in agencies and other organizations engaged in management (Hunt et al., 2013). By observing these issues in a systematic fashion insights can be gathered about how to foster desired institutional change and how to account for stakeholder interactions when identifying optimal decisions. Research could focus on how people redesign institutions when resource and social outcomes change and how and if these changes feedback through institutions and other organizations engaged in management (Hunt et al., 2013). Such studies could examine not only agency managers’ responses to changing outcomes but also how groups of fishers respond to these changes (e.g., develop and revise voluntary norms; Cooke et al., 2013). |
| Learning and decision-processes in social groups: Much research on the human dimensions of anglers has been directed at the individual, entirely omitting social dynamics. It is clear, however, that social dynamics and other contextual factors fundamentally drive decisions of individuals. It is also known that anglers in groups behave differently from the same anglers in experimental laboratory settings (Sooth et al., 2012). Hence, novel monitoring and experimentation is needed that considers collective decisions and social networks. |
| Effects and potentials of social media: Communication channels are rapidly changing (Papenfuss et al., 2015). Social media and apps offer huge potential to serve managers as tools for monitoring and for purposely directing effort. At the same time, social media revolutionizes the way anglers learn. Research in this area and on social networks in general (Little and McDonald, 2007; Martin et al., 2014) is of fundamental importance if we are to understand the future of recreational fisheries (Hunt et al., 2013). |

3. A framework to analyze the complex adaptive SES of recreational fisheries

Studying recreational fisheries using a complex adaptive SES perspective can be greatly facilitated by the use of a coherent framework (Hunt et al., 2013). Frameworks are descriptors of the basic structural elements and causal relationships to be expected in a given system (McGinnis and Ostrom, 2014). Recreational fishing has featured prominently in the development of the increasingly popular “Ostrom framework for the study of SESs” (Ostrom, 2007, 2009; Schüller et al., 2014; Hinkel et al., 2014, 2015). This framework constitutes the most widely used framework to structure relationships among social and ecological systems (Basurto et al., 2013; Binder et al., 2015).
2013; Thiel et al., 2015). The benefits of following a general SES framework, such as Ostrom’s, are manifold, not least to overcome inconsistent use of variables and concepts across applications in SES studies (Thiel et al., 2015). The framework aims to be general enough to guide future empirical interdisciplinary analysis of many recreational fisheries, but to be applied it requires the analyst to draw on theory to identify and select variables that are important to understand specific processes, dynamics and outcomes (Schlüter et al., 2014). In an SES context, diverse ecological and evolutionary theories and hypotheses (e.g., predator-prey theory, population dynamics theory, community ecology, food web theory, eco-evolutionary feedback theory) and social theories (game theory, transaction cost theory, choice theory, welfare economics, social identity theory, social-psychological theory, theories of common-pool-resources) apply from which the analyst has to choose those that are most relevant in a given context (McGinnis and Ostrom, 2014).

### 3.1. The analytical social-ecological framework

#### 3.1.1. Background

Elinor Ostrom’s framework for the analysis of the sustainability of SESs (Ostrom, 2007, 2009; McGinnis and Ostrom, 2014) has become popular to analyze human-environment systems in a wide variety of settings, including those specific to small-scale commercial fisheries (Basurto et al., 2013; Leslie et al., 2015; Partelow and Boda, 2015) and recreational fisheries (van Poorten et al., 2011; Schlüter et al., 2014; Hinkel et al., 2014, 2015). It is thus natural to draw heavily on the Ostrom framework to represent the SES of recreational fisheries, but a number of issues in the original framework are further developed and clarified to increase its appeal to the recreational fisheries case.

The framework developed here was built on an early framework for the SES analysis of recreational fishing developed by part of the authors (Daedlow et al., 2007, for latest version see Daedlow, 2015), which by coincidence was presented at about the same time when the first version of Ostrom’s framework was published (Ostrom, 2007). All relevant elements and the specific vocabulary inherent in Ostrom’s latest version of her framework (McGinnis and Ostrom, 2014) were integrated and further refinements related to recreational fisheries were considered (Hinkel et al., 2014, 2015). In this context, a new graphical representation of the SES framework was developed that accounts for the nested structure of some of Ostrom’s “first-tier” categories (for definition, see below) in both the ecological and social domains. Not only are the hierarchical structures inherent in both the social and ecological systems presented, but in addition the explicit interactions and ecological processes (horizontal lines) and outcomes (vertical lines) not present in Ostrom’s original visual representation of her framework are considered (Figure 3). Most importantly, however, the ideas of the authors were combined with Ostrom’s and adaptive control theory applied to fisheries and natural resource management (Walters and Hilborn, 1978; Walters, 1986, Schindler and Hilborn, 2015) by adding an explicit dynamic component of monitoring and learning to the SES framework (the outer loop in Figure 3). As will be elaborated in the next section and highlighted by Hansen et al. (2015) in detail,
adaptive management, i.e., purposely testing management interventions to learn about critical uncertainties at the system level (Walters, 1986), and its further development into adaptive co-management (Pope et al., in press) explicitly deals with uncertainties in how exploited SESs react to change and manipulation. Adaptive management and its analogue in multi-stakeholder environments, adaptive co-management, is thus strongly recommended for sustainable recreational fisheries to learn how the complex processes in SESs lead to observable outcomes that are valued by stakeholders (FAO, 2012; Hansen et al., 2015; Schindler and Hilborn, 2015; Pope et al., in press). Echoing others, adaptive management is identified as a key component of studies of recreational fisheries as complex adaptive SESs and is therefore explicitly incorporated into the present framework in line with previous recommendations for recreational fisheries (FAO, 2012; Pope et al., 2014; Hansen et al., 2015; Pope et al., in press).

The ultimate goal of the analysis framework is to foster an integrative view of recreational fisheries as complex adaptive SESs and guide interdisciplinary analysis. To cover regional variability inherent in recreational fisheries, the scientific analysis framework is by design as broad as possible. Therefore, when the framework is applied to understand and study a specific recreational fishery, the analyst must “zoom in” to the specific details in either the social or the ecological environments to identify key players, actors, variables and processes relevant for their study (Hunt et al., 2013; Schlüter et al., 2014). On a conceptual level, this has already been done for the social system of recreational fisheries in the article by Hunt et al. (2013), and it is hoped that this work will stimulate future empirical applications specifically directed at the feedbacks among the social and ecological components of the SES. The aim for now is to provide a broad categorization of the major elements that are present in any complex adaptive recreational fishery conceptualized as an SES. Note that the framework shown in Figures 3 and 4 is meant to guide scientific inquiry without normative emphasis (evaluation) of what follows from the science for management or governance.

### 3.1.2. Analytical framework for the social-ecological analysis of recreational fisheries

The basic framework consists of the social and the ecological environment (Figure 3). The social environment is composed of two levels. First, there are the institutions (defined as the existing formal and informal rules, i.e., institutions are not organizations), which includes formal fisheries regulations as well as social norms of proper behaviour for managers, anglers, and society at large (Cooke et al., 2013). The second social component includes the actors (i.e., anglers, decision-makers, policy makers, other stakeholders) and the organizational governance structures developed to steer the system (i.e., governmental management agencies, user organizations and non-governmental organizations, NGOs). The ecological environment involves macro-scale boundary conditions, the set of water bodies in a given landscape and the associated biophysical conditions and fish populations. Within the ecological environment there is a conceptual integration of the individual resource units (individual fish and fishing sites) as common-pool resources (CPR) that anglers desire to appropriate (i.e., possess and possibly harvest) and for which there is rivalry in consumption among fishers.

The two main components in the framework, the social and ecological system, are not meant to be studied in isolation as in traditional fisheries science. Rather, both components in Figure 3 are interlinked within the broader SES. The focus of SES study is particularly directed at the feedbacks between the compartments in the central area of Figure 3, which are often behaviour-based (Ward et al., 2016). Perturbations in any part of the system will propagate throughout the entire system, through direct or indirect pathways and generate expected or unexpected outcomes. The two-dimensional framework of Figure 3 can also be thought of as a hierarchically organized system as depicted in Figure 2, where small and fast resource dynamics affect overarching angling dynamics, which in turn feedback to large and slow institutional dynamics.

**First-tier variables.** At its broadest level, the SES framework adopts the vocabulary of Ostrom’s SES by specifying several broad structural categories, or so-called first-tier variables (McGinnis and Ostrom 2014), which are abbreviated by capital letters in Figures 3 and 4.

Actors (A) are characterized by individual traits and behaviours. They are situated within and outside government and can form actor groups (e.g., an angler organization or groups of like-minded anglers). Actors, and by the same token the subcategory of actor groups/organizations, act within a specific Governance System (GS) at various geographical scales (Leslie et al., 2015).

The GS defines the overarching “rules of the game” (Ostrom, 1990) and conceptually include the governance structures (agencies, clubs, regional management bodies, NGOs) engaged in management or lobbying. Hinkel et al. (2014) are followed in nesting governance structures as components of the GS and as a subcomponent of Actors because they are a special form of actor groups (Figure 3). Institutions are composed of constitutional rules (e.g., rules that define who has the right or privilege to use and manage fisheries resources), collective choice
Figure 4. The Ostrom social-ecological framework in a formalized presentation modified from Hinkel et al. (2014) and Schlüter et al. (2014). Boxes denote concepts, black arrows pointing down denote attribution relationships, black arrowheads pointing up denote subsumption relationships, brown links denote aggregation relationships. A “1” indicates a one-to-one relationship; an “+” indicates a one-to-many relationship. Concept names (first-tier variables) are indicated in the top part of the boxes; attributed variables (second-tier variables) are listed in the bottom part of the boxes and indicated by “C.” Process relationships (interactions) and outcomes are also given. All key first-tier (e.g., governance system = GS, see Figure 3) and most second-tier variables (e.g., GS1) presented in McGinnis and Ostrom (2014) and Basurto et al. (2013) are given, but reorganized following Hinkel et al. (2014) and Schlüter et al. (2014) in GS domain. We follow the Appendix 1 in Schlüter et al. (2014) and also present interactions (I) and outcomes (O). Different colours indicate different levels of biological and social organization.
rules (e.g., rules that define how operational rules are changed and enforced) and operational rules (i.e., rules that define specific management regulations, such as minimum-length limits or the amount of fish to be stocked) (Ostrom, 2005).

A Resource Unit (RU) is an individual unit of a CPR that is desired to be appropriated by A and that is the focus of management initiatives. Resource units are nested in Resource Systems (RS) who through ecological and other processes provide the flow of RUs to the social system (Hinkel et al., 2014). The CPRs (i.e., the RUs) are characterized by high rivalry in consumption due to subtractability and the difficulty of excluding specific actors (Ostrom, 2005). Subtractability may be a temporary characteristic of catch-and-release fisheries that are not extractive. For example, a rare trophy fish that is returned to the water (Arlinghaus, 2007; Hinkel et al., 2014) might remain invulnerable to fishing gear for a refractory period (Camp et al., 2015). The RUs can be individual fishes with certain characteristics (e.g., trophy size) or other resources desired by anglers such as fishing sites (Hinkel et al., 2014, 2015). Managers may also consider individual fish stocks as the RUs forming productive metapopulations in certain areas (e.g., various salmonid stocks that differ in timing of migration or have other characteristics of importance to managers and society; Schindler et al., 2015).

Interactions among GS, A, RS and RU are mediated by the broader first-tier variables of social, economic, and political Settings (S) and Related Ecosystems (RECO) within which the SES of recreational fisheries is embedded (McGinnis and Ostrom, 2014). The authors prefer to use the term RECO rather than ECO (Ecosystem) (McGinnis and Ostrom, 2014) to emphasize the relatedness property of the other ecosystems (e.g., terrestrial ecosystems who are often tightly linked to aquatic ecosystems in both lentic and lotic ecosystems; e.g., Pace et al., 2004) and to differentiate these from the fish ecosystems within the RS.

Of key importance for the dynamics of recreational fisheries, and indeed any SES, are the processes of appropriation or extraction (e.g., harvesting) and maintenance or provision (e.g., management by stocking) of CPRs (RUs). Here Interactions (I) (horizontal arrows) and Outcomes (O) (vertical arrows), as the seventh and eight first-tier variables of the SES framework, come into play (Figures 3 and 4; Table 2). Collectively, I and O represent dynamic processes, which feedback on the various compartments of the SES and are thus of fundamental intellectual interest. Ostrom’s framework (2007, 2009) considers I and O to be unified in a so-called action situation (see Figure 2 in McGinnis and Ostrom, 2014), which is the collection of social and environmental processes whereby I lead to O (Figure 5). Within any action situation, O can occur at the micro-level (e.g., a change in a regulation) or at the macro-level (e.g., overflowing, loss of resilience at the regional level). Outcomes are generally evaluated by the social system (Hinkel et al., 2014). In the institutional literature micro-level outcomes are sometimes referred to as fast feedbacks, while macro-level outcomes emerge from slow feedbacks (Ostrom, 2011). In the framework, micro-level outcomes are intermediate steps in action situations that link I through processes and feedbacks to macro-level O (Table 2; Figure 5). Various action situations maybe

<table>
<thead>
<tr>
<th>First-tier variable</th>
<th>Second-tier variable</th>
<th>Example from recreational fisheries</th>
</tr>
</thead>
<tbody>
<tr>
<td>Interactions (I) leading to outcomes (O)</td>
<td>11 – harvesting activities</td>
<td>The intensity of harvesting can strongly affect size structure and abundance of fishes (Post et al., 2002)</td>
</tr>
<tr>
<td></td>
<td>12 – information sharing</td>
<td>Affects effort distribution as well as the uptake of local information by managers</td>
</tr>
<tr>
<td></td>
<td>13 – deliberation process</td>
<td>Affects the level of trust and compliance by users to management actions</td>
</tr>
<tr>
<td></td>
<td>14 – conflicts</td>
<td>Failure to address conflicts leads to long-lasting issues and system dysfunction</td>
</tr>
<tr>
<td></td>
<td>15 – investment activities</td>
<td>Affects the ability to monitor and store information as well as harvesting effort</td>
</tr>
<tr>
<td></td>
<td>16 – lobbying activities</td>
<td>Affects the political pressure exerted on decision-makers and their decisions</td>
</tr>
<tr>
<td></td>
<td>17 – self-organizing activities</td>
<td>Could for example result in the illegal release of non-native fishes (Johnson et al., 2009) or cleanup of river banks</td>
</tr>
<tr>
<td></td>
<td>18 – network activities</td>
<td>Fosters the possibility that innovation spreads and becomes implemented</td>
</tr>
<tr>
<td></td>
<td>19 – monitoring activities</td>
<td>Affects the quality of knowledge about resource and angler states and the effectiveness of past actions</td>
</tr>
<tr>
<td></td>
<td>10 – evaluative characteristics</td>
<td>Defines which criteria are used to judge outcomes and how different criteria are weighed (Fenichel et al., 2013a)</td>
</tr>
<tr>
<td></td>
<td>01 – social performance measures</td>
<td>Relates to measures of angler satisfaction, economic impact, efficiency of resource allocation, equity and accountability of decisions and more general optimal social yield as well as resilience</td>
</tr>
<tr>
<td></td>
<td>02 – ecological performance measures</td>
<td>Relates to measures of biological sustainability, loss of evolutionary significant units and biodiversity, impacts in food webs and ecosystems as well as resilience</td>
</tr>
<tr>
<td></td>
<td>03 – externality to other SEs</td>
<td>Relates to unintended effects on other SES, for example the discovery of a threatened native fish might change conservation actions in other SEs</td>
</tr>
</tbody>
</table>
effects are often moderated by ecological processes. A related interactions and processes take place, whose
Hence, in any action situation, many relevant angling-information sharing and deliberative interactions are involved
(see Os are involved. For example, in the action situation
In any given action situation, multiple Is and micro-level appropriateness, monitoring, evaluation and sanctioning.
Macro-scale outcomes (social or ecological or system-level)

Figure 5. A conceptual sketch of an example action situation to differentiate among interactions, feedbacks, processes and outcomes from a social-ecological system (SES) perspective. Note that there will be several action situations operating jointly in a given SES, usually involving multiple interactions that through links and feedbacks lead to multiple micro-scale and ultimately macro-scale outcomes. While there are no pure ecological interactions in the Ostrom framework, ecology plays a fundamental role in shaping links, feedbacks and outcomes by mediating the effects of actor-centred interactions.

linked leading to what is known as “linked action situations” (Ostrom et al., 1994).

From an SES perspective the key aspects to understand are that action situations arise where (human-induced) I (Table 2) are mediated by links and feedbacks among the social and ecological compartments and lead to O (Figure 5). Action situations come in many forms, such as collective choice of rules, operational choice of rules, appropriation, monitoring, evaluation and sanctioning. In any given action situation, multiple Is and micro-level Os are involved. For example, in the action situation “monitoring,” monitoring and evaluation activities, information sharing and deliberative interactions are involved (see Table 2 for a full list of all interactions possible). Hence, in any action situation, many relevant angling-related interactions and processes take place, whose effects are often moderated by ecological processes. A focal action situation may also be affected by other action situations happening concurrently at different scales (Figure 2). For example, an evaluation action situation may be affected by a monitoring action situation because monitoring provides the input data for the evaluation of outcomes.

“Fast” micro-level outcomes (Figure 5) lead to the emergence of “slow” macro-level outcomes (O) at the regional (landscape) level, which are represented by the vertical arrows in Figure 3. For example, anglers enjoy or are repelled by certain micro-level outcomes (e.g., changes in catch rates, sizes of fish, crowding, harvest regulations), and management is similarly affected by such outcomes (Figure 2), particularly those assessed by monitoring activities (the outer loop in Figure 3). Anglers and the governance and management system absorb all micro-level outcomes – deliberately or unintentionally – which leads to further interactions in the social domain, resulting in changing participation rates, site choices, lobbying intensities, and potentially changes in preference for management tools (Figure 2). These changes ultimately affect policy and management choices, which in turn feedback to influence harvesting interactions and to alter the social or ecological systems (Figure 2). By including the outer loop, the framework explicitly acknowledges the process of learning and adaptation from evaluation of past actions and from the flow of rewards to the actors, which has both a loop directed at anglers (leading to anglers adapting their strategies) and a loop directed at managers (leading to managers adapting their responses; Figure 3). Correspondingly, the ecological system also consistently adapts and changes through compensatory responses, life-history changes and alteration of fish behaviour (Johnston et al., 2013; Alós et al., 2015a,b; Ward et al., 2016). At a dynamic equilibrium, three key macro-level outcomes (O) emerge, that either belong to the social domain (e.g., angler satisfaction, efficiency, equity, accountability of decisions), the ecological domain (e.g., regional overfishing, biodiversity impacts, biological sustainability) or to the system level (e.g., collapse or resilience), including spill-over effects to other SEs (Table 2). Note that some of the macro-level Os can emerge without any corresponding change in the ecosystem services that directly link to human well-being (Figure 1), e.g., due to an improved management decision-making process that is more accepted by stakeholders and perceived as generating equity (Table 2).

The presentation of Is and Os in Figure 3 deviates from Ostrom’s original figure in two important ways. First, as mentioned before, an external loop that represents monitoring and adaptive management (learning) is explicitly represented, which is part of an evaluation action interaction. Although monitoring, evaluation and adaptation is conceptually integrated in Ostrom’s framework through ten key Is
(Table 2), these dynamic issues are considered of such fundamental importance to effectively understand the dynamics of recreational fisheries (FAO, 2012; Hansen et al., 2015) that it was decided to include monitoring, evaluation and adaptation explicitly in the framework in Figure 3. Second, stylized interactions are represented horizontally among compartments (e.g., among A and GS) rather than being organized centrally between the social and ecological dimensions as previously conceptualized in the Ostrom framework or its adaptations (McGinnis and Ostrom, 2014; Hinkel et al., 2014). In the social domain, interactions include information sharing, deliberation processes, and lobbying activities (Table 2). In the central area between the social and the ecological domains where the social world is linked to RU and RS, Is represent appropriation (e.g., harvesting or site choices) and provisioning activities (e.g., management actions such as stocking) (Hinkel et al., 2014).

To apply the present version of the Ostrom framework to the analysis of recreational fisheries as SES, it is important to differentiate the meaning of interactions, feedbacks and links (Figure 6), which are all involved in process relationships in the SES (Figure 5). Links are simply (functionally—neutral) descriptive relationships between two players or groups of players, e.g., between one fish and an angler or between an angler population and managers in an agency (Figure 6). Direct feedbacks also exist among two players of the SES, but the relationship implies an interdependent, bi-directional process between them, e.g., fishing induces changes in the behaviour of an exploited fish, which in turn affects catchability by anglers (Figure 6). Processes can be social, ecological or social-ecological. Note that although 'interactions' is a common term in ecology, reserving the term interactions for the ten key social interactions in Table 2 is appropriate in SES analysis because the interdependence of actors and actor groups, institutional responses and social and ecological outcomes all arise through the effect of socially-induced and human-constructed interactions (e.g., activities such as harvesting, monitoring, lobbying). These ten interactions are mediated via a biophysical system and the complex processes happening in the ecological realm.

Figure 6. Representation of the main components of the social-ecological system (SES) of recreational fisheries outlining six key first-tier variables of the Ostrom framework (unbolded abbreviations, RU = resource units, RS = resource system, A = actor, GS = governance system, S = social, economic and political setting, RECO = related ecosystems, note the omission of I = interactions and O = outcomes shown in Figure 3) as well as the key players of the SES abbreviated by bold capitals. The figure represents more details as to the main units of analysis to visualize all possible links among the various players (abbreviated by small letters). The bi-directional nature of the links shows feedbacks.
Table 3. First and second-tier variables of a social-ecological system (SES) along with examples from recreational fishing, including, where available and appropriate, key references. The presentation follows McGinnis and Ostrom (2014) but is adapted in several ways to meet the specifics of the framework shown in Figure 3. Selected second-tier variables are slightly adapted in wording and clarified, and the outcomes extended. Interactions and outcomes are given in Table 2. The first capitals S, GS, etc., are the first-tier variables shown in Figure 3.

<table>
<thead>
<tr>
<th>Second-tier variable</th>
<th>Narrative example from recreational fisheries</th>
</tr>
</thead>
<tbody>
<tr>
<td>S1 – Economic development</td>
<td>Exerts influence on participation rates, initially increasing them and later decreasing due to competition with alternative leisure activities among other structural reasons (Arlinghaus et al., 2015a)</td>
</tr>
<tr>
<td>S2 – Demographic trends</td>
<td>Aging reduces fishing participation rates due to physical and health constraints (Murdoch et al., 1996; Arlinghaus et al., 2015a) and there are general intergenerational effects on the relevance of angling as a lifestyle activity and interspecific and intraspecific competition</td>
</tr>
<tr>
<td>S3 – Political stability</td>
<td>Affects socio-political priorities, how society deals with environmental impacts and the development of governance systems and governance capacity</td>
</tr>
<tr>
<td>S4 – Other governance systems</td>
<td>Drives higher-order management decisions related to water and socio-political climate in which fishing is fostered or constrained</td>
</tr>
<tr>
<td>S5 – Markets</td>
<td>Affects impacts on the environment as well as availability of fishing opportunities and competition among leisure activities and with commercial fisheries (Abbott and Fenichel, 2013)</td>
</tr>
<tr>
<td>S6 – Media organizations</td>
<td>Shapes public values with reference to the consumptive use of natural resources (Manfredo, 2008; Riepe and Arlinghaus, 2014b) and the way and speed that anglers communicate</td>
</tr>
<tr>
<td>S7 – Technology</td>
<td>Affects catchability and the communication channels anglers use (social media, Martin et al., 2014)</td>
</tr>
<tr>
<td>GS1 – Governmental organizations</td>
<td>Management agencies play an important role in shaping legislation, regulation and monitoring activities in some countries (Daedlow et al., 2011)</td>
</tr>
<tr>
<td>GS2 – Nongovernmental organizations</td>
<td>NGOs may engage in lobbying and in some countries, such as central Europe, are the main managers of wild-living fish stocks given common property rights (Daedlow et al., 2011)</td>
</tr>
<tr>
<td>GS3 – Operational rule</td>
<td>Harvest control rules and base regulations determine the ability of managers to manage fish mortality and achieve other outcomes (Abbott and Fenichel, 2013; Gwinn et al., 2015)</td>
</tr>
<tr>
<td>GS4 – Collective choice rule</td>
<td>Fosters compliance and local self-organization if users retain the right to devise their own rules and engage in enforcement and graduated sanctions (Daedlow et al., 2011)</td>
</tr>
<tr>
<td>GS5 – Constitutional choice rule</td>
<td>Defines who is entitled to use and manage resources and may range from public fishing rights regimes to private property (Daedlow et al., 2011)</td>
</tr>
<tr>
<td>GS6 – Policy area</td>
<td>A country may focus strongly on fisheries as a commercial activity and devise policies that marginalize recreational fisheries or vice versa. For example, the Common Fisheries Policy of the EU is strongly commercially oriented where anglers are considered, if at all, a driver of fish population change that needs to be accounted for mainly to improve stock assessments for commercial quota setting</td>
</tr>
<tr>
<td>GS7 – Geographical scale</td>
<td>May lead to mismatch of ecological issues and governance structures (Arlinghaus, 2006a)</td>
</tr>
<tr>
<td>GS8 – Population</td>
<td>Population sizes and the composition define the impacts on resources and who engages in fishing (Post et al., 2008; Arlinghaus et al., 2015a)</td>
</tr>
<tr>
<td>GS9 – Regime type</td>
<td>Ranges from polycentric governance systems in central Europe to centralized governance systems in North America, which will strongly affect the way management and governance is institutionalized (Daedlow et al., 2011)</td>
</tr>
<tr>
<td>GS10 – Property rights structure</td>
<td>Relates to the full set of rights ranging from alienation, management, exclusion, withdrawal and access rules (Daedlow et al., 2011)</td>
</tr>
<tr>
<td>GS11 – Network structure</td>
<td>Networks among angler groups, as well as among anglers, managers, scientists and policy-makers are decisive for local and regional outcomes (Arlinghaus et al., 2015b)</td>
</tr>
<tr>
<td>GS12 – History of continuity</td>
<td>Shapes the governance systems, transactions and processes involved in policy making through path-dependencies</td>
</tr>
<tr>
<td>GS13 – Monitoring and Sanctioning</td>
<td>All the processes involved in assessing the status of stocks (Lester et al., 2003; Hansen et al., 2015) as well as regulatory enforcement mechanisms (Walker et al., 2007)</td>
</tr>
<tr>
<td>A1 – Number of relevant actors</td>
<td>Management effectiveness declines in multi-stakeholder environments, particularly when there are few overlapping goals (Arlinghaus, 2005)</td>
</tr>
<tr>
<td>A2 – Socioeconomic attributes</td>
<td>Socio-economic status will affect social identity and relatedly the acceptability of social norms</td>
</tr>
<tr>
<td>A3 – History or past experience</td>
<td>Sets the base for traditional ecological knowledge</td>
</tr>
<tr>
<td>A4 – Location</td>
<td>Conflicts between residents and non-residents may be affected by place attachment to particular localities (Arlinghaus, 2005)</td>
</tr>
<tr>
<td>A5 – Leadership</td>
<td>Social capital to make use of windows of opportunity to change traditional rules and regulations</td>
</tr>
<tr>
<td>A6 – Norms, cognitions (beliefs, attitudes) and emotions</td>
<td>Managers are strongly affected by local social norms, for example when deciding about stocking (van Poorten et al., 2011). Norms (both personal and social) are also extremely important for local self-governance and may undermine well-intended formal regulations (Cooke et al., 2013).</td>
</tr>
<tr>
<td>A7 – Ecological knowledge of SES and mental models</td>
<td>Defines how stakeholders and anglers perceive cause-and-effect relationships and in turn which management strategies to pursue (or prefer) (Gray et al., 2015)</td>
</tr>
<tr>
<td>A8 – Importance of resource</td>
<td>Strongly drives incentives to either overharvest or conserve, particularly in the absence of substitutes (Dorow et al., 2010)</td>
</tr>
<tr>
<td>A9 – Technologies available</td>
<td>Determine monitoring quality and harvesting efficiency</td>
</tr>
<tr>
<td>RU1 - Mobility</td>
<td>In recreational fisheries where mobility of fishes is high the potential exists to move beyond boundaries of governance structures, reducing the incentives for local conservation</td>
</tr>
<tr>
<td>RU2 – Growth, mortality and reproduction</td>
<td>Vital rates (growth, survival, recruitment) cumulatively drive standing stock and ability to recover from overexploitation or other impacts (Johnston et al., 2013)</td>
</tr>
<tr>
<td>RU3 – Interdependence among resource units</td>
<td>Intraspecific and interspecific processes are important in maintaining food webs and are often driven by size in freshwater fish stocks (de Roos and Persson, 2013)</td>
</tr>
<tr>
<td>RU4 – Economic and social value</td>
<td>Provides a catalyst for management conflicts, but also incentives for conservation and regulatory compliance (Arlinghaus, 2005)</td>
</tr>
<tr>
<td>RU5 – Number of units</td>
<td>Drives the ability to allocate among competing users and ecological density-dependence</td>
</tr>
</tbody>
</table>

(Continued on next page)
(Figures 5 and 6). Interactions (the horizontal arrows in Figure 3) once initiated thus spread through the network of links mediated through feedbacks and have no precise location in Figure 6, whereas links and bi-directional feedbacks can be located explicitly using arrows between two players (Figure 6). To give a concrete example: one angler as A is linked to one particular fish as RU when the angler attempts to capture that fish in a harvesting interaction within an appropriation action situation. The micro-level outcomes (e.g., capture or not) that emerge from the harvesting interaction are mediated through (various) links and feedbacks, which lead to (bi-directional) processes induced by the catching attempt, such as site choice by the angler, his or her choice of gear, or the fish’s behavioural reaction to the angling attempt. Cumulatively across all anglers, a macro-level outcome (e.g., stock status after a fishing season) will emerge. While purely ecological feedbacks (i.e., processes) in the ecosystem exist, purely ecological interactions do not, because from a SES perspective, only (human-induced) interactions lead to outcomes (Figure 5). Put differently: interactions and feedbacks are separate concepts, but they join to form process-relationships in action situations that ultimately lead to outcomes experienced and evaluated by humans (Figure 5).

**Second-tier variables.** So far the six major structural components of a recreational fisheries SES (GS, A, RU, RS, I, O) as well as embedding the SES in a broader context (S, RECO) were highlighted, leading to eight structural components. These are the first-tier categories following Ostrom’s terminology (Figure 4). Key explanatory factors driving the processes by which interactions lead to outcomes for each of the six categories are included in the SES framework as “second-tier” variables nested within the “first-tier” structures (Table 3; Figure 4). The key assumption is that the configuration of the second-tier variables defines the functional properties of the process relationships within the SES. Among-case explanatory power related to effects of second-tier variables on system outcomes is a central open question in SES studies in general (Hinkel et al., 2014). Put differently: lower-tier variables are logical subdivisions within higher-order elements of the framework and are thought to be fundamentally important for dynamics and outcomes of SES by affecting the way interactions lead to outcomes through processes characterized by links and feedbacks (Hinkel et al., 2014, 2015). For example, second-tier variables associated with an RS may include characteristics such as the size of the resource system (e.g., the total water bodies in a particular fishery), clarity of system boundaries (e.g., how many lakes belong to one management unit and is access open to all or not) or the productivity of the system (e.g., what is the biologically sustainable yield or the renewal rate of the population after harvesting). The productivity of the system will interact with the size of the regional angler population to affect the pattern of overharvesting (Hunt et al., 2011; Post and Parkinson, 2012), i.e., the second-tier variable is assumed to affect the process and ultimately the dynamics and outcomes of the system in very characteristic ways, but whether this is

<table>
<thead>
<tr>
<th>Second-tier variable</th>
<th>Narrative example from recreational fisheries</th>
</tr>
</thead>
<tbody>
<tr>
<td>RS6 – Equilibrium properties</td>
<td>Stock enhancement and variable angling effort lead to constantly changing equilibrium stock sizes and fish size compositions (Post et al., 2008)</td>
</tr>
<tr>
<td>RS7 – Predictability of system dynamics</td>
<td>Stock-recruitment is highly stochastic leading to natural variation in fish abundances (Myers et al., 1999)</td>
</tr>
<tr>
<td>RS8 – Storage characteristics</td>
<td>Due to the portfolio effect of variable populations and individuals there is the ability to buffer against environmental shocks, or alternatively, the homogenization of stocks may lead to a loss in resilience (Schindler et al., 2015)</td>
</tr>
<tr>
<td>RECO2 – Pollution and impact patterns</td>
<td>Relates to all anthropogenic factors that affect lakes, rivers and fish stocks and that are not caused by external influence (agriculture, industry)</td>
</tr>
<tr>
<td>RECO3 – Flows into and out of focal SES</td>
<td>Relates to the connectivity of ecosystems, some of which might be fostered by fish ladders and other constructions</td>
</tr>
<tr>
<td>Table 3. (Continued)</td>
<td></td>
</tr>
</tbody>
</table>
the case across a set of different SES is an open empirical question (Leslie et al., 2015). Table 3 lists key second-tier variables following McGinnis and Ostrom (2014) that are proposed to be general across all SESs and overall decisive drivers of outcomes in coupled SESs, followed by interpretative examples from recreational fisheries, and Figure 4 positions first- and second-tier variables in the appropriate nested structure. One can obviously further expand the structure to include third-tier variables as well (Basurto et al., 2013; McGinnis and Ostrom, 2014), and it is highly likely that these third-tier variables will also exert systematic effects. Due to space limitations the authors refrain from a further reflection here. Ultimately, choosing the various tiers to include and the appropriate level of aggregation strongly depends on the specific case, the key research question and on the variables believed to be driving the system. From the perspective of generality of the SES framework, it is however recommended always following the list of variables currently proposed to establish a cohesive body of work that will allow the relative importance of each second-tier or third-tier variable to be learned through meta-analysis of accumulated case studies (Thiel et al., 2015).

Deciding which second-tier (and by the same token third-tier) variables are generally important across SESs is of ongoing scientific discussion as are further refinements to the Ostrom SES framework, and the choice must be dictated by the specific case systems, the research question, available data and the diagnostic approach chosen by the analyst (Hinkel et al., 2014, 2015). There are other issues that need sorting by the scientific community. For example, the Ostrom framework is ambiguous about which specific second-level tiers apply to individuals, groups or organizations, or whether the second-tier variables of the Actors category (e.g., mental model) apply to individual actors, actor groups or organizations (see Hinkel et al., 2014 for an insightful discussion). Moreover, Ostrom did not as yet consider individual-level (i.e., psychological) determinants of human behaviour, which have been the focus of much human dimensions research in recreational fisheries (Arlinghaus, 2004) and are a key aspect of Hunt et al.’s (2013) SES framework for the social dimension of recreational fisheries. For example, while norms occur in Ostrom’s collection of key second-level tiers in the A domain, norms can be a characteristic of a group (i.e., a social norm or the norms of a management agency) or an individual (i.e., a personal norm) (Vaske and Manfredo, 2012). All key second-tier variables that are considered of fundamental importance to recreational fisheries are listed in Table 3, and hierarchical structures and relationships as well as aggregation (e.g., all individual fish aggregate to form a fish population), attribution (encompassing a “has—a” logic, such as a one-to-one or a one-to-many relationship) and subsumption (encompassing a “is—a” logic, e.g., an organization is an actor group) categorizations are outlined in Figure 4 following Hinkel et al. (2014). Moreover, key individual-level cognitions (values, attitudes, beliefs), emotions and mental models are explicitly mentioned as second-tier variables of the Actors group to represent all key determinants of human behaviour. The authors maintained Hinkel et al.’s (2014) attribution of these psychological traits to characterize groups and organizations (e.g., a fisheries NGO has a certain mental model of how the ecological relationships are perceived and what a desirable end state of management should be, Figure 4). Clearly, this is work in progress with future adaptations expected as the relative importance and practicability of selected second-tier variables becomes better understood. Although the framework captures what is considered to be the most important first and second-tier variables driving any recreational fishery, the framework should be further developed as the scientific community engages in its application to specific cases. The reader is referred to Schlüter et al. (2014) and Hinkel et al. (2014) for examples in recreational fisheries and approaches to the problem of choosing specific variables and processes when formalizing (Hinkel et al., 2014), diagnosing (Hinkel et al., 2015) or modelling recreational fisheries as SESs (Schlüter et al., 2014). For a general assessment of challenges in SES modeling, see Schlüter et al. (2012) and Levin et al. (2013).

4. Operational management framework

To apply the analytical framework described in the previous section, the adaptive management (AM) feedback represented by the outer loop in Figure 3 has to be operationalized. Therefore, an operational management framework is presented to guide selection of suitable management actions through experiential learning (be it academic or not). Given pervasive uncertainties that stem from a range of non-linear interactions between recreational fishers and fish stocks (Carpenter and Brock, 2004; Biggs et al., 2009; Hunt et al., 2011) and the inability to precisely forecast fisheries due to the complexity of the natural and social world (Schindler and Hilborn, 2015), an operational management framework must emphasize flexibility and adaptability and be designed to help identify “optimal” management responses through planned interventions (i.e., management experiments) at the system level (Pope et al., 2014; Hansen et al., 2015). Although quantitative models that incorporate all known fish-angler interactions and associated uncertainties, are, in principle, able to identify optimal regulations at the landscape level (e.g., Mapstone et al., 2008; Johnston et al., 2010), many remaining uncertainties preclude us from predicting the real-life effects of certain policies and regulations.
Therefore, one has to design closed-loop testing of management interventions (e.g., different stocking rates in different fisheries) under real life conditions to sort among the various potential management tools available to the manager – a process known in the environmental sciences as AM (Walters, 1986). The reason for engaging in some form of AM for recreational fisheries is simple: there will always be unpredictability in the dynamics and behaviour of complex SES, yet management decisions must be made. Accordingly, the core of the onion-like operational management framework proposed here (Figure 7) revolves around AM as a management principle facilitated by the decision-making principle of structured decision-making, which is highly recommended in multi-stakeholder environments (Walters and Hilborn, 1978; Walters 1986; Irwin et al., 2011; FAO, 2012; Hansen et al., 2015).

4.1. Active versus passive adaptive management

Adaptive management can be operationalized in active and passive ways (Figure 8; Walters, 1986; Williams, 2011a,b). Active adaptive management (AAM) is purposeful experimentation with policy options to test the outcomes in real fisheries by conducting whole-ecosystem experiments, ideally in a replicated fashion (Boxes 3 and 4). This AAM is the research-oriented form of
adaptive management, which is meant to improve understanding of critical uncertainties about processes inherent in the SES of recreational fisheries (Box 3). Hence, AAM is strictly speaking a component of the analytical research framework introduced in the previous section (the outer loop, see Figure 3).

Passive adaptive management (PAM; Figure 8) evaluates the outcomes of ongoing management tools and actions. While such an approach cannot reap the benefits of the full inference space that AAM opens (FAO, 2012; Hansen et al., 2015), it still allows various management tools to be judged in light of pre-defined (abstract) goals and (measurable) objectives (Barber and Taylor, 1990). PAM has many fewer demands on ongoing monitoring efforts as long as some base monitoring information, e.g., on catches, is collected. Hence, the operational management framework proposed here is both a component of the scientific approach to recreational fisheries as well as a tool to identify operationally viable management strategies in practical fisheries management that are acceptable to stakeholders. The conditions for putting AM into place are particularly good in freshwater recreational fisheries where there is ample opportunity for replication and testing of spatially varying management tools (given the many lakes and river sections present in a landscape) and where there is often “natural” variation in key policies of interest (e.g., variation in size-limits in different water bodies). Adaptive management is however currently underutilized in recreational fisheries and suffers from inappropriate monitoring methods and scales (Hansen et al., 2015). For example, despite more than hundred years of reliance on both harvest regulations and fish stocking in recreational fisheries (Arlinghaus et al., 2016), there are only a handful of intervention experiments that analyse their outcomes with appropriate replication compared to unmanipulated controls. There is thus the need for more ecosystem scale experimentation or for more passive analysis of ongoing replication to learn about expected outcomes of even the most basic management tools regularly used in recreational fisheries.

4.2. Implementing adaptive management in the context of an ecosystem and precautionary approach to resilience

Conceptually, AM is nested within and affected by overarching fisheries-management principles such as the ecosystem approach to fisheries (EAF), the precautionary approach (PA) and resilience thinking (Biggs et al., 2012; FAO, 2012; Pope et al., 2014), which are all designed to deal with uncertainty while avoiding irreversible ecological and social change (Figure 7). In fact,
because one source of pervasive uncertainty is the biological impact of recreational fishing and of fisheries management on the ecosystem and biodiversity, an AM approach constitutes an explicit means to respond to the demands of the (risk-averse) ecosystem approach and precautionary approach to fisheries (FAO, 2003; Arlinghaus and Cowx, 2008). Moreover, AM is a key component of the emerging principle of resilience, defined as a system able to reorganize and adapt to change while sustainably maintaining the delivery of basic functions such as the cultural service of angling (Figure 7; FAO, 1996, 2003; Pope et al., 2014; Hansen et al., 2015). These overarching principles set constraints of what AM can test or explore. For example, it would be incompatible with the ecosystem and precautionary approaches to test large scale releases of non-native fishes in an attempt to learn how fisheries react because such releases would likely cause irreversible impacts. Adaptive management is also at the core of the emerging normative framework of resilience (Chapin et al., 2010; FAO, 2012), which acknowledges multiple objectives and sources of knowledge, multiple ecological services of interest, the critical importance of feedbacks and key system variables, the possibility for undesired regime shifts and the need for continuous learning and adaptation to iteratively approach an “optimal” management solution in the long-term (Walters, 1986; Ludwig et al., 1993; Biggs et al., 2012; Pope et al., 2014; Hansen et al., 2015; Schindler and Hilborn, 2015). Accordingly, from a resilience perspective, it would be unthinkable to test management actions on larger scales that erode ecological and social diversity because conservation of diversity is considered a key component of a resilient system (Biggs et al., 2012).

Figure 9. Adaptive management of renewable natural resources such as fish, often characterized as ‘learning by doing’, is a formalized iterative process that acknowledges uncertainty and manages by increasing knowledge of the system through monitoring, feedback and revision of objectives and the means to achieve (dynamic) objectives. Structured decision-making (gray circles), a term sometimes confused with adaptive management, is an organized and transparent approach to the decision process for identifying and evaluating alternatives and justifying complex decisions (see Figure 7, Irwin et al., 2011); however, structured decision-making does not necessitate the iteration and consequential higher order learning (white circles) inherent in adaptive management (modified from Allen et al., 2011 and FAO, 2012).
To be successful, AM depends on continuous monitoring of feedbacks and system outcomes (Hansen et al., 2015), which is also an important precondition to understand a coupled SES and to manage for resilience to external disturbances (Figure 9). While the conceptual underpinnings for AM are straightforward, its practical implementation offers substantial challenges in terms of financial and human resources, monitoring demands and socio-political support (Halbert, 1993; Walters, 2007; Allen and Curtis, 2005). Hence, it would be an illusion to think that most recreational fisheries will ever engage in rigorous, experimental AAM as originally conceptualized by Walters and Hilborn (1978) and Walters (1986) (Figure 8). But there are many less demanding forms of PAM that still seem superior than non-adaptive approaches for steering recreational-fisheries management (Figure 8). In particular the circular, rigorous, open, and inclusive management process that AM advocates (Figure 9) is of core importance for successful fisheries management, and because this process may be implemented with a range of data sources, including qualitative reasoning, it may be less resource-heavy than initially appears. See Boxes 3 and 4 to learn about examples of active management projects conducted on sustainable stocking in Canada and Germany.

Implementing AM with structured decision-making (Ihde et al., 2011) depends on identifying various potentially suitable management directions and tools a priori and possibly testing for their effects by evaluating system outcomes (FAO, 2012; Hansen et al., 2015; Arlinghaus et al., 2016). This implementation involves difficult decisions as to which management tools to consider a priori (e.g., stocking, habitat management, harvest regulations, effort controls), at what configuration (e.g., which densities to stock and which sizes) and against which set of evaluative criteria to pre-screen suitable tools. Decision-trees tailored to the life-history of a given species and modelling can help in identifying principally useful management directions (e.g., whether harvest regulations are more suited than stocking; see, e.g., Welcomme, 2001; Cowx, 1994; Lorenzen, 2005; Ihde et al., 2011; FAO, 2012; Arlinghaus et al., 2016). One particularly useful modelling philosophy is the management strategy evaluation framework, which was successfully applied to several recreational fisheries in the United States and Australia (e.g., Sainsbury et al., 2000; Mapstone et al., 2008; Ihde et al., 2011; Irwin et al., 2011).

To guide the choice of direction to test, considering general principles of resilience and risk-averse environmental management - in particular the EAF and the PA (Figure 7) - are useful. In this context, the EAF is characterized as “to plan, develop and manage fisheries in a manner that addresses the multiple needs and desires of societies, without jeopardizing the options for future generations to benefit from the full range of goods and services provided by ecosystems” (see FAO, 2003 for details). The first step is to accept that ecosystem-level impacts are possible through recreational fishing, rather than discounting such effects as has happened in the past (Arlinghaus, 2006a; Arlinghaus and Cowx, 2008). Next, rather than focusing only on target species, a broader ecosystem perspective shall then be used in routine assessments and evaluations of alternative management options (Pope et al., in press), including risk analysis in the cycle of AM prior to initiating action. The EAF principle thus supplements the narrow, “piscicentric” perspective on a single target species or a single fishery that is currently prevalent among many recreational fisheries stakeholders and managers (Arlinghaus and Cowx, 2008; Pope et al., in press). Having said this, it is recognized that in some situations focusing on a target-species may be both appropriate and practical to meet stakeholder demands, and this perspective is valid as long as any planned intervention has no wider and irreversible ecosystem-level effects. Overall, the EAF is a principle to account for ecosystem processes in the formulation of fisheries-management measures (Sissenwine and Murowaski, 2004). The EAF hence emphasizes an evolution of fisheries management rather than a revolution as sometimes perceived (Mace, 2004; Rice, 2011).

Where knowledge about system dynamics is insufficient, the EAF also calls for precautionary recreational fishery management measures that minimize ecological risks in light of dynamic and difficult-to-predict recreational angler responses (Arlinghaus and Cowx, 2008). Precaution is also a key element of resilience, which emphasizes maintaining options for the future. Thus, resilience and the PA are proposed as final guiding principles in AM processes in recreational fisheries (Figure 9). The precautionary approach is not to be confused with the precautionary principle originally emanating from environmental law and policy. The latter emphasizes that any risk is “too much” and often results in delay or even constrains any fisheries management decisions (Peterman, 2004). The PA, in contrast, recommends explicitly taking environmental risk into account during decision-making (Peterman, 2004). Relatedly, the absence of data should not be a reason for postponing actions, especially considering that inaction itself is a management decision, with its own potentially costly consequences. In reality, the benefits of further data collection associated with testing novel management actions may quickly outweigh the costs of current actions and assessments (Samuelson and Zeckhauser, 1988; Hansen
they are traditionally popular such as continuous stock-socially problematic management approaches, even if can help to identify and avoid ecologically risky and will probably lead to more conservative decisions and (Hickley and Chare, 2004). A key point is that, if faced with considerable uncertainty and risks and different possible actions, actions should be chosen to give priority to conserving the biological productivity over the long term rather than satisfying short-term social demands (Peterman, 2004). These actions can involve setting safety margins based on clearly articulated limits or target reference points, such as how much fishing mortality or effort to tolerate, but this requires that these baselines can actually be measured (which is often impossible in data-poor recreational fisheries, Lester et al., 2014; de Kerckhove et al., 2015). Activities that greatly modify communities and food webs (e.g., by pervasively altering the size and age structure of stocks leading to increased variability of stocks, van Kooten et al., 2010; Hsieh et al., 2010) and that lead to strong social conflicts should be thoroughly reviewed and the risks and costs vs. benefits properly evaluated (Francis et al., 2007) (Figure 9). Usually, given trade-offs between social and ecological benefits and among several ecosystem services (Pope et al., in press), EAF and PA enter the AM planning process by determining “risk-averse” boundaries and the choice of principally useful management strategies (Garcia, 1994; FAO, 1996; Peterman, 2004; Fenichel et al., 2008). Consequently, the chosen actions tend to be those that promise to not irreversibly affect or modify the natural ecosystem (e.g., avoiding the release of non-native genotypes). Running enclosed artificial put-and-take fisheries may be an exception (Hickley et al., 2004) as long as the spread of fishes or other organisms to natural ecosystems can be minimized (Hickley and Chare, 2004).

Following the operational framework just described will probably lead to more conservative decisions and can help to identify and avoid ecologically risky and socially problematic management approaches, even if they are traditionally popular such as continuous stocking of fish into self-reproducing stocks (Hühn et al., 2014). Thereby, AM becomes a key component of the increasingly popular management goal of resilience (Walker et al., 2004; FAO, 2012; Pope et al., 2014; Hansen et al., 2015). Resilience is particularly useful as a goal in situations where management deals with difficult-to-predict CAS in data-poor situations under large uncertainty and risk. Resilience to abrupt change is not only a desired endpoint, but also a management philosophy. A number of key resilience principles have been recently proposed as a guide to managing ecosystem services sustainably in multi-stakeholder environments (e.g., diversity, connectivity, polycentric governance, see Biggs et al., 2012 for details). To be resilient to external disturbances and shocks recreational fisheries have to be able to adapt to new situations, defined as the ability to change the system to keep it within current development trajectories (Walker et al., 2004), or were needed transform into a fundamentally new development trajectory (Folke et al., 2010). To do so necessitates that the governance and management systems develop a so-called adaptive capacity (Folke et al., 2002), which among other things demands having up-to-date information about the functioning of the SES—the core of what AM is expected to deliver—and flexibility and ability to learn (Hansen et al., 2015). To avoid a common misunderstanding: resilience is only a desired endpoint if the “sustained provision of the cultural ecosystem service” (Chapin et al., 2010) of recreational fisheries is currently deemed satisfactory and the system not vulnerable to collapse. Otherwise, the objective might be to transform the system into a new state by reducing resiliency, but this again demands knowledge of how to do so, to be generated by AM. Key principles for managing the resiliency of ecosystem services have been presented elsewhere (Biggs et al., 2012, for an application to recreational fishing, see Pope et al., 2014; Hansen et al., 2015; Pope et al., in press), but the main aspects will be reviewed in Section 6.

Adaptive management is critically dependent on a base level of monitoring. Although sustaining monitoring over long periods of time maybe perceived as daunting and impractical for freshwater landscapes, the monitoring needs do not necessarily depend on data hungry approaches (Lester et al., 2014; de Kerckhove et al., 2015) and may even be conducted with anglers’ diaries (Arlinghaus et al., 2015b) or other comparatively easy means of long term catch monitoring (Hansen et al., 2015). What matters more is that the correct variables (e.g., critical slow ones) are monitored at the right spatial-temporal scales (Hansen et al., 2015). Additionally, to be able to actually learn from purposely planned interventions one must have some form of before-after design, ideally paired with unmanipulated control systems to identify the effects of management interventions (Hansen et al., 2015). Freshwater ecosystems in landscapes come with many natural replicates (lakes, rivers), which is why the implantation of AM seems particularly suitable and double in freshwater fisheries. To assist management organizations to implementation AM, it may be worthwhile to establish agency extension services in
cooperation with universities, which are trained in the frameworks presented here and elsewhere (Lester et al., 2003) and in data-poor assessment and monitoring methods (Lester et al., 2014; de Kerkhove et al., 2015; Hansen et al., 2015). Such developments would increase the prospects for recreational fisheries to successfully implement AAM and thereby be empowered to change, adapt, and potentially to transform when confronted with new challenges (e.g., climate change, altered social values, invasion of novel organisms).

4.3. Inter- and transdisciplinarity

Putting the frameworks presented above in place when studying complex adaptive SES demands interdisciplinary cooperation (Levin, 1998, 1999), often involving modeling for data integration and synthesis (Schlüter et al., 2012; Levin et al., 2013). At the very least, insights, assumptions, theories, models, and methods from the social and ecological sciences need to be integrated, because it is otherwise impossible to study crucial feedbacks among the social and ecological components of a SES in a rigorous way. Hence, SES studies, as well as AM experiments, are more than multidisciplinary.

There has been great confusion about the similarities and differences among multidisciplinary and interdisciplinary research in fisheries and other fields of environmental science, which is why some clarification is in order (Arlinghaus et al., 2014a). Most previous research in recreational fisheries has been monothematic focused on just one scientific discipline and paradigm (e.g., human dimensions research rooted in social-psychology). The few projects or

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**Box 3. Implementing a scientific active adaptive experiment to understand key uncertainties in the social-ecological system of angling for rainbow trout in British Columbia, Canada**

The freshwater recreational fishery of British Columbia, Canada, is highly valued with the primary target species being rainbow trout (*Oncorhynchus mykiss*). The species is native to this region, but many lakes are managed as a culture-based fishery completely dependent on stocking. Anglers focus on wild and stocked lake populations of this species across approximately 3,800 lake populations, of which 800 are stocked using one of several wild genotypes, which are reared to fry or yearling stages in hatcheries before stocking primarily in formerly fish-free lakes. The fisheries are distributed across a large and spatially variable landscape with substantial climate and productivity gradients and a patchily distributed angler population. The angler population is diverse in preferences, including willingness to travel. This landscape-scale variability in ecology, and angler behaviour presents challenges to fisheries management in their role of ensuring quality fisheries and wild stock conservation. In support of freshwater fisheries management in British Columbia, a group of academic fisheries ecologists, social scientists and modellers have partnered with government and non-profit fishery managers to undertake a landscape scale active adaptive management program involving coordinated ecological and human dimensions data collection, paired with experimental manipulations of stocking rates and harvest regulations. The key research question was how to optimize stocking and harvest policies across this large and heterogeneous landscape with the twin goals of benefiting the diverse angler population and conserving wild stocks. Natural and social scientists and modellers jointly developed the experimental program from the outset to ensure a coordinated effort to identify and measure the key processes within and between the ecological and social elements of this complex SES. The ecological and social processes are being modelled within a spatially explicit SES to explicitly examine the feedbacks and patterns in fishing quality and angler effort that emerge from this dynamic system. Briefly, the modelling approach is to characterize angler-fish population dynamics in lakes that vary in productivity, embedded in a matrix of alternate fishing locations, connected by road networks and the spatial distribution of human settlements. The angler population is segmented into types based on large-scale surveys, and their behaviour across the landscape and alternate opportunities are determined through discrete choice experiments applied to a sample from the British Columbia angler licence data base. Empirical results from stocking rate and harvest regulation experiments are being used to ground-truth model predictions. Some of these results have been completed in the meantime, showing that after controlling for access and travel costs stocking does not lead to a stable increase in fishing quality because anglers homogenize catch qualities to a regional average due to substantial among-lake mobility (Wilson et al., 2016; Mee et al., 2016). These results are being integrated into spatial SES model, which will be used as an expert system to assess the viability of various management policy alternatives to ensure sustainability of the fishery, provide quality fishing opportunities to the diverse angler population, and to conserve wild stocks.
 Unless otherwise specified, the terms “researcher” and “researchers” in this paper refer to all the people involved in research, including scientists, managers, and policy makers. In this context, the term “research” is used to refer to a systematic investigation of a problem or phenomenon that is typically conducted to develop a new understanding or to improve existing understanding on a particular topic. The term “researcher” is used to refer to any person who engages in research, regardless of their specific role or position within a research project.

**What is the main focus of the research?**

The main focus of the research is to understand how interdisciplinary research can be used to address complex issues in fisheries science. The authors argue that interdisciplinary research is important because it allows scientists to address complex issues by bringing together knowledge from different disciplines. However, they also note that interdisciplinary research can be challenging because it requires researchers to work together and to understand each other’s perspectives.

**What methods are used in the research?**

The authors use a combination of qualitative and quantitative methods to assess the effectiveness of interdisciplinary research. They also use case studies to illustrate how interdisciplinary research can be used to address complex issues in fisheries science.

**What are the main findings of the research?**

The main findings of the research are that interdisciplinary research can be effective in addressing complex issues in fisheries science. The authors also note that interdisciplinary research can help to identify gaps in knowledge and to develop new research questions.

**What are the implications of the research?**

The implications of the research are that interdisciplinary research can be a powerful tool for addressing complex issues in fisheries science. However, the authors also note that interdisciplinary research requires a lot of effort and collaboration to be successful.

**What are the limitations of the research?**

The limitations of the research include the fact that it is based on case studies and may not be generalizable to all contexts. Additionally, the research may not capture all the nuances of interdisciplinary research.
Box 4. Transdisciplinary research on sustainable fish stocking in Germany

Some of the authors were involved in designing and running a five-year transdisciplinary research program called Besatzfisch (stocked fish, www.besatz-fisch.de) in Germany. Angling clubs are fishing rights holders in Germany, and any changes to the governance and management of local fisheries depend on active decisions by local angling clubs. Stocking is a contested issue, whose success or failure depends on a range of social, ecological and evolutionary factors (Box 1; Lorenzen et al., 2012; Hühn et al., 2014). To learn about successful and unsuccessful stocking practices, as well as associated genetic and other ecological risks, a team of researchers partnered with 18 angling clubs in Lower Saxony, Germany. In close collaboration with the anglers and managers in the angling clubs, the research team developed an active adaptive stocking experiment involving radical stocking density treatments of northern pike (Esox lucius L.) and common carp (Cyprinus carpio L.) into angler-managed gravel pits. Pike served as an example of a naturally recruiting species (Hühn et al., 2014), while carp served as an example of a non-naturally recruiting but highly desired and often-stocked species (Arlinghaus et al., 2015b). Experiments were planned and conducted in 24 angler-managed gravel pits. Workshops were used for developing goals and objectives as well as hypotheses about stocking outcomes and allocation of treatments to study lakes. The stocking outcomes were monitored jointly through a series of workshops creating opportunities for learning and reflexive processes. Anglers participated in fish surveys and also completed angling diaries for monitoring carp catches, which are very difficult to enumerate with other gears than angling. Results were regularly discussed in workshop settings, where joint interpretation of findings and adaptation of future club policies occurred. Large-scale surveys of club anglers and workshop attendees were used to understand attitudes, norms, and other human dimensions related to stocking and to parameterize utility functions driving angler behaviours (Arlinghaus et al., 2014b; Gray et al., 2015; Fujitani et al., 2016). These functions were incorporated into bioeconomic models designed as a learning tool for examining stocking policies (Arlinghaus et al., 2015b). Moreover, the researchers tested different variants of pedagogical interventions, comparing the transdisciplinary cooperation with standard teaching of scientific content through a power point-based seminar. Results showed that stocking failed to have additive stock effects when the fish stocks naturally reproduce (exemplified by pike; Hühn et al., 2014) and that integration of anglers in the jointly conducted active adaptive stocking experiments was instrumental to changing ecological knowledge, beliefs, and behavioral intentions about stocking, effects retained when measured 10 months later (Fujitani et al., unpublished data). Details about this project can be found in Arlinghaus et al. (2015b) and in a documentary film (www.youtube.com/watch?v=sFMvsz4YuY).

(scientific and traditional ecological), (7) contextualization of research results (i.e., place-based results), (8) production of different types of knowledge (i.e., target, system and transformative knowledge, Hirsch Hadorn et al., 2006), and (9) societal as well as standard scientific impact. Box 4 describes an example of transdisciplinary recreational fisheries research in the context of optimizing stocking policies.

In transdisciplinary science stakeholders (broadly including managers, anglers, NGOs) are actively involved in the research process, for example by co-defining research questions, generating data, and the interpretation and evaluation of research outcomes. Although stakeholder involvement may be perceived as daunting by some researchers, and will have substantial transaction costs, involving stakeholders through transdisciplinary fisheries research can be highly beneficial and lead to rapid uptake of research results by managers and stakeholders (Arlinghaus et al., 2015b). For example, stakeholders can help in shaping the sustainability science problem that is to be studied and can justify research needs, co-determine management goals and objectives (which in turn affect research questions and approaches), represent traditional ecological knowledge, facilitate the collection of novel types of data inaccessible to scientists (e.g., through smartphones in citizen science projects; Pappenfuss et al., 2015) and legitimize and democratize management decisions that emerge from research results (Arlinghaus et al., 2014a; Ruppert-Winkel et al., 2015).

Transdisciplinary research places particularly high demands on the project team and involved stakeholders and can only function if both parties share a willingness to engage in a long-term collaborative effort. Given enough time and funding such processes may nevertheless succeed (Box 4), especially if the research/management problem has high relevance for both stakeholders and scientists, incentivizing collaboration over multiple years. For such initiatives to increase, university
programs need to be more strongly tailored to promote and support inter- and transdisciplinary work (Arlinghaus et al., 2014a). Furthermore, hiring decisions by agencies must also emphasize selecting candidates with the skills to collaborate on inter- and transdisciplinary projects. Although transdisciplinary science is again no panacea and clearly has significant costs, the authors believe it has large untapped potential in studies on recreational fisheries as complex adaptive SESs, for example within the context of citizen science (Jordan et al., 2015).

5. Conclusions

Recreational fisheries are examples of complex adaptive SESs. Studying and managing recreational fisheries as complex adaptive SESs may thus offer general insights of relevance to the emerging field of SES science. To promote such research, two interrelated frameworks for the analysis and adaptive management of recreational fisheries were presented. The frameworks documented in this article have been inspired by Ostrom’s (2007, 2009) landmark proposal for the analysis of the sustainability of SESs and by adaptive control theory; they are meant to foster an integrative view on recreational fisheries acknowledging all types of relevant feedbacks: within and among the social and ecological systems, interactions within different angler groups; from anglers to managers; and from the fisheries sector to non-fishing related agencies and other stakeholders (Ward et al., 2016). The frameworks also foster a perspective that sustainable fisheries require an AM system (outer feedback loop in Figure 3) consisting of monitoring and assessment tools (Walters, 1986) and transparent, socially defined evaluation criteria to judge outcomes (Fenichel et al., 2013a). Overall, the two frameworks presented serve three main purposes. First, they help the analyst to think about the key structural components making up a given SES and the relevant units of analysis. Second, given a particular research question and assisted by a diagnostic approach that needs to be developed for each case application (see Hinkel et al., 2015 for a full account) the frameworks help identify key links and social-ecological feedbacks that are likely to drive system outcomes in action situations. Thereby they, thirdly, help in guiding selection of theory and hypotheses to inform empirical studies in the case of data gaps or inform meta-analytical approaches to data integration and modelling. Finally, the frameworks constitute a communication tool that will help integration of disparate data and knowledge in interdisciplinary research teams. It is the hope of the authors that the summaries presented here can be an inspiration for the next decade of research and management of recreational fisheries as complex adaptive SESs.

The key management question to be solved is how to steer and manage a complex adaptive SES of recreational fisheries toward a desired state given the multitude of unpredictable feedbacks and outcomes. A central tenet proposed here and elsewhere (Biggs et al., 2012) is that one must focus on managing relevant feedbacks to keep a recreational fishery in a desired state and prevent it from flipping into an undesired one, i.e., achieve resilience. The new proposal emerging from an SES perspective of managing feedbacks and processes explicitly, rather than merely focusing on resource states as is typical for fisheries management, is both a fundamentally new perspective and challenge because managers are used to managing site or fishery-specific outputs generated by localized interactions (e.g., yield, catch rate, angler satisfaction) or simply fish population states (e.g., spawning stock biomass) in selected ecosystems. As mentioned before, however, it is an illusion that one would ever have enough monitoring information for each of the hundreds or thousands of lakes and rivers to justify reference point-based “resource state management” at the level of individual lakes or river sections (Post et al., 2002; de Kerckhove et al., 2015). Therefore, the authors and others (Lester et al., 2003; Mahon et al., 2008) propose a shift to a process- and principle-based landscape management system that is fundamentally about managing feedbacks and fostering empowerment, self-organization and in general “enabling” processes rather than command-and-control, but this can only be put in place if recreational fisheries scientists start producing relevant knowledge about macro-scale system dynamics emerging from micro-scale interactions, rather than focusing on local lakes or river sections. To this end, it is proposed that future science and management of recreational fisheries as complex adaptive SESs can benefit from refocusing on three key areas of concern (Biggs et al., 2012; Pope et al., 2014; Hansen et al., 2015):

1. **Identify and manage feedbacks.** A key focus includes understanding and in turn managing (pervasive) feedbacks between recreational fishers and fish stocks in addition to paying attention to traditional metrics such as optimal social yield (OSY; Johnston et al., 2010). Positive (amplifying or destabilizing) feedbacks may for example result in ever increasing stocking levels to meet ever-increasing angler expectations (Johnson and Staggs, 1992) that may in turn prove catastrophic for recreationally exploited fish stocks (van Poorten et al., 2011). To give another example, inverse density-dependent catchability — a
Depensatory mechanism (Post et al., 2002, 2008) - may interact with unresponsive recreational fishing effort to cause widespread collapse of recreationally exploited fish stocks across a landscape (Hunt et al., 2011). When such feedback dynamics are known, there is the need for a modified management approach by for example reducing the responsiveness of managers to angler dissatisfaction or by educating anglers about realistic catch rate expectations. The hope is then that the system would self-organize without a need for annual stock assessments and be resilient, once the perverse feedback is addressed. Negative (stabilizing) feedback loops are also possible, e.g., when anglers remain attracted to low abundance and low catch-rate fisheries near urban centres, thereby reducing fishing effort in remote areas (Post et al., 2002; Johnston et al., 2010). Managing this feedback loop may entail effort limitations or strategic use of high stocking rates in selected fisheries to "lure" anglers away. While negative feedback loops may increase stability, they are not necessarily desirable and then need to be understood and managed explicitly. Larrosa et al. (in press) present a generic framework that can help us categorize and think particularly about unintended feedbacks. They distinguish (1) flow unintended feedbacks when pre-existing feedbacks are enhanced or dampened (examples above); (2) deletion unintended feedbacks; and (3) addition unintended feedbacks when interventions, respectively, remove or add actors or links to the SES structure. Application of this typology to recreational fisheries in the future may help us navigate through the challenges of understanding and managing feedbacks.

(2) Understand and manage critical slow variables and associated processes: to avoid potentially drastic system shifts when a recreational fishery is in a satisfactory state demands careful attention to critical slow (i.e., low turnover rate) variables. These critical variables involve, for example, spawning habitat, ecotone diversity, genetic and other forms of biodiversity, human values, and institutions (rules in use). All of these variables may be slow in their turnover, but they are key ingredients determining the future trajectory of an SES and therefore, require particular management attention in recreational fisheries (Biggs et al., 2009; Carpenter et al., 2011), and environmental management in general when the aim is to manage for resilience (Biggs et al., 2012). A focus on documented or likely critical slow variables and associated key processes to safeguard an SES in a desired state will also help justify monitoring efforts and strategies (Hansen et al., 2015).

(3) Understand and manage diversity at all levels: because diversity provides the raw material on which future innovation is based and often contributes to stability and resilience (Biggs et al., 2012), maintaining diversity across all levels constitutes a third and final key component of sustainable and resilient fisheries (Schindler et al., 2015). For example, it has been shown that "one-size-fits-all" regulations applied across a landscape produce suboptimal outcomes (Carpenter and Brock, 2004) and that erosion of the biocomplexity of salmon stocks reduce productivity (Schindler et al., 2015). It has also been shown that introgression of foreign genes and homogenization of gene pools through stocking has negative effects on a stock's productivity (Chilcote et al., 2011) and that habitat homogenization will lead to homogenized fish faunas as well as a loss of desired species (Craig, 2016). The importance of variability extends to age groups and size structure (Hsieh et al., 2010) and involves the need to manage genetic biodiversity as well as diversity in cultural values and institutions. The portfolio effect proposes that a system’s robustness and productivity will be maximized when diversity is maintained (Schindler et al., 2015). The ramifications of this are profound, because the complex adaptive features of any SES mean that selection will reduce variability in a self-organized fashion, for example when anglers stock fishes from non-local sources, which leads to population mixing, or when small angling clubs merge with others to form large management organizations. Hence, it is important to manage with an intent to maintain some base level of diversity to ensure multiple options exist for the future, i.e., to retain capacity to adapt to new situations, both in the ecological and social domains of the coupled SES.

Many ideas presented here on the value and necessity of considering recreational fisheries as complex adaptive SESSs are theoretical and lack an empirical contrast to simpler scientific and management approaches. Future research is needed to test some propositions presented here (e.g., to focus on managing feedbacks rather than states). Integrated social-ecological models (Schlüter et al., 2012) as well as simulation-based management strategy evaluation models are tools that can help resolve these uncertainties, as long as institutional, social, and ecological dynamics are all rigorously accounted for in the modelling framework. Because most contemporary fisheries science is quantitative, it is predicted that quantitative social sciences will most readily interface with
traditional fisheries biological models to foster the understanding of the dynamics of recreational fisheries as CAS. Model predictions should be closely interfaced with AAM in the real world to test processes and functional relationships and observe system outcomes directly. Such experiments, informed by the frameworks presented here, will be particularly fruitful in freshwater recreational fisheries and hence something to be strategically pursued in the future in larger multi-investigator and multi-stakeholder teams.

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