Social-ecological vulnerability: from assessment to action

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Social-ecological vulnerability

From assessment to action

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GENERAL INTRODUCTION

General introduction

Social-ecological science to navigate wicked management problems

Human-nature interactions can be valued in different ways (e.g. MEA ecosystem services framework, MEA 2005; IPBES framework; biocultural approach to sustainability, Sterling et al. *in press*). All these perspectives acknowledge that people heavily rely on nature to satisfy a diversity of needs such as food, water, spirituality and other contributions to human well-being. Yet, the escalating speed of change of numerous direct and indirect drivers, including human demography and migration (Crist et al. 2017), markets and teleconnections (Seto et al. 2012) and climate change (Scheffers et al. 2016) are causing rapid environmental shifts toward undesirable states (Rockström et al. 2009), thus threatening the sustainability of those interactions contributing to well-being. More than ever, efforts are needed to foster sustainability.

From the local to the global scale, finding solutions to sustainability challenges posed by environmental degradation and changing patterns of use hinges in an understanding of the complex interactions between people and the natural environment (Young 2006). For example, fisheries sustainability requires considering the livelihoods of local communities (Allison & Ellis 2001) while accounting for the management of other potential sources of food such as agriculture or aquaculture (Golden et al. 2016) and cross-scale or off-site effects (Pascual et al. 2017b). Similarly, the conservation of marine ecosystems cannot be achieved without also considering the ultimate drivers of environmental change such as markets, governance, or culture and values (Cinner et al. 2016; Hicks et al. 2016a; Hughes et al. 2017). Securing both ecosystems and human well-being requires a balanced consideration of the social and ecological systems, and their multiple, complex interactions (Kates et al. 2001).

The recent attention to people, both as a factor affecting environmental state, and as an outcome through human well-being targets, is now well established in modern sustainability science and conservation (Mace 2014). The consideration of people as part of – rather than apart from, or on top of – ecosystems marks a paradigm shift from an overly simplistic representation of sustainability issues toward a more complex and holistic understanding of the interlinked dynamics of environmental and societal change (Fig. 1; Fischer et al. 2015).

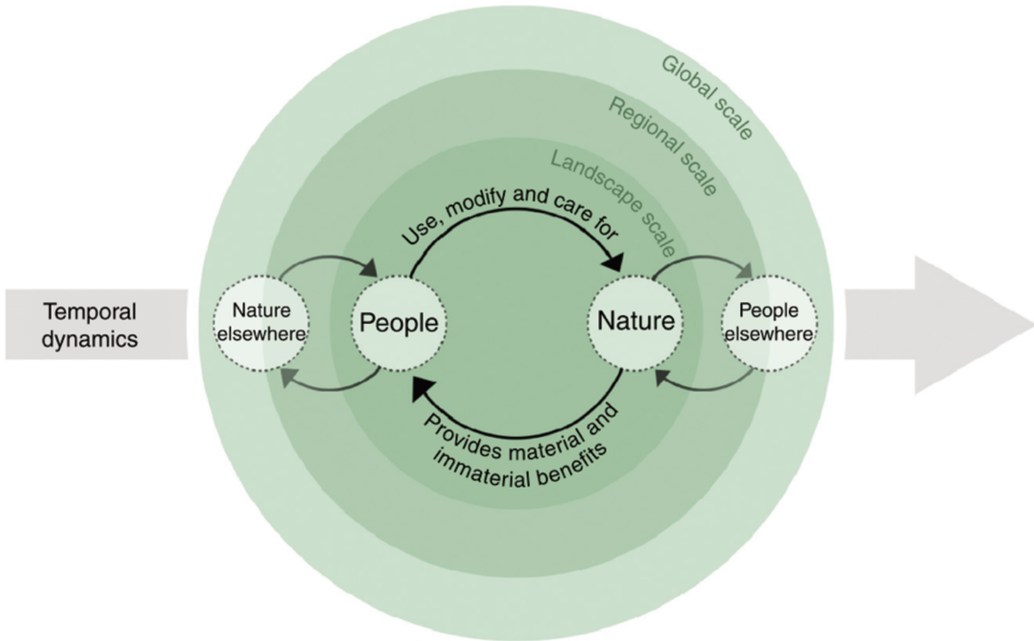

Figure 1: Conceptual representation of a social-ecological system (Fischer et al. 2015). While deriving elements of well-being from nature, humans alter, either directly or indirectly, these ecosystems. Therefore, humans are both dependent upon – and major drivers of – ecosystems and the benefits they generate. Interactions within social-ecological systems are a function of internal dynamics and their responses to external drivers that occur at various spatial and temporal changes.

The combination of social and ecological systems within the same model, often referred to as "social-ecological systems", has emerged as a prevalent concept in the growing transdisciplinary research, with each major intellectual lineage illuminating different – but complementary – aspects of the sustainability picture (Bousquet et al. 2015). The accumulated knowledge stemming from this research on social-ecological systems has provided significant theoretical and practical contributions to management and policy. For example, the need to embrace the inherent dynamics and uncertainty of social-ecological systems (Adger et al. 2005; Parrott & Meyer 2012), the importance of various disciplinary inputs (Leenhardt et al. 2015), the role of institutions and governance arrangements (Ostrom 1990) and the recognition that outcomes may involve various types of trade-offs (Daw et al. 2015) have been recognized as key pieces to foster sustainability. Uptake of the abundant literature on social-ecological systems has become apparent in the practitioners' recognition that management is a "wicked problem"– that is, a problem characterized by the elusiveness of a final resolution and the absence of panacea (DeFries & Nagendra 2017). For instance, the multiple, interacting linkages between poverty, resource-use, development and environmental condition need to be navigated carefully and make this issue unsuitable to one-size-fits-all solutions (Ostrom 2007, 2009; Cinner 2011).

However, even though social and ecological systems, their internal features and linkages are recognized as key intervention points for managing sustainability, and significant theoretical and empirical advances have been offered, finding solutions to sustainability challenges continues to be elusive and rarely achieved. From a management perspective, the reasons for this lack of success of current management practices can be broadly as follows: (1) falsely applying a tame solution to a wicked problem: approaches and tools currently available to practitioners are suited to simpler, more tractable systems and remain insufficiently nuanced to allow for planning management actions in specific contexts; (2) inaction from overwhelming complexity: prevalent concepts and approaches in social-ecological research lack of operational clarity and do not match practitioners' needs for straightforward, transferable, meaningful and scalable methods; and (3) absence of effective knowledge systems: arrangements that facilitate communication, translation and mediation across boundaries separating the knowledge and action communities are lacking (Cash et al. 2003; Leenhardt et al. 2015; DeFries & Nagendra 2017; Olander et al. 2017).

Putting social-ecological science into practice

Avoiding either oversimplification of a wicked problem or inaction from overwhelming complexity requires effectively harnessing relevant social-ecological knowledge for sustainability, and this calls for balancing precision (i.e., depicting a comprehensive picture of complex social-ecological systems) and feasibility (i.e., technical, financial, logistical capacity available to practitioners) while breaking the communication barriers between science and action (Fazey et al. 2005; Game et al. 2014; Olander et al. 2017). Therefore, efforts to link knowledge and action may greatly benefit from approaches that don't necessarily provide a comprehensive picture of complex social-ecological interactions, but illuminate key social-ecological dimensions relevant to the focal problem. For instance, identifying actions most likely to succeed across different future scenarios (often referred to as "strategic foresight"; Cook et al. 2014) entails different methods and tools than prioritizing where to take action at a minimum "cost" (often referred to as "systematic planning"; Margules & Pressey 2000) or quantifying the consequences and trade-offs of choosing amongst a set of alternative actions (often referred to as "structured decision making"; Gregory et al. 2012). Yet, practitioners are encouraged to integrate multiple approaches into their decision-making process in order to navigate through the full spectrum of challenges involved in the management of complex social-ecological systems. This requires mixing and matching tools from various disciplines. Despite some successful applications (e.g., Bryan et al. 2011; Schofield et al. 2013), this is generally difficult to achieve, owing to the often specific and rigid nature of the toolkit available to practitioners and the challenges of integrating multiple disciplines in practice (Schwartz et al. 2017). The development of easily transferable and flexible frameworks, methods and tools that enable tackling the many challenges of social-ecological sustainability at most management stages (scoping, planning and learning; UNFCCC 2011) and in most contexts thus holds great promises to break down barriers between knowledge and implement social-ecological based management (Sunderland et al. 2009).

Vulnerability assessment to bridge science and policy

Conceptual background

Vulnerability is a complex and multifaceted concept with interpretations that vary according to the system, the driver, the scale and field of application considered. In its most broad sense, vulnerability is the degree to which an entity is likely to experience harm from exposure to drivers, although many definitions have been proposed in the literature (Adger 2006; Adger et al. 2009a). The relevant entity may be any system, sub-system or component that composes a social-ecological system and a driver refers here to a press or pulse event that affects the entity of interest beyond the "normal" range of variability in which it operates (Turner et al. 2003).

In the context of resource sustainability and biodiversity conservation, this led to a proliferation of multidisciplinary applications that generally fall within two categories of conceptual models (Turner et al. 2003; Brugère & De Young 2015). The first category is rooted within the risk-hazard research lineage and focuses on the impact of drivers through entities' exposure and sensitivity ("dose–response") to this driver (White & Haas 1975; Burton et al. 1978). This "end-point" view of vulnerability (or *outcome* vulnerability) is often used to estimate the extent to which different exposure scenarios lead to changes in the state of an entity (generally places, sectors, activities, landscapes or regions), but does not acknowledge the crucial role of adaptation in shaping differential vulnerability outcomes. The second category, which has been largely influenced by the political ecology/economy schools of thought, perspective on vulnerability emphasizes the inherent state or conditions that make an entity susceptible to change (Hewitt 1983). Rather than being seen as a consequence, vulnerability is here an intrinsic property of the entity (generally a social entity like individuals, households, social groups, communities, livelihoods) that is determined by its capacity to attenuate the consequences of exposure to a particular driver. However, this *contextual* perspective on vulnerability tends to minimize the implications of environmental dynamics on the social system (for detailed discussions of the concept of vulnerability, see O'Brien et al. 2004, 2007; Adger 2006; Eakin & Luers 2006; Costa & Kropp 2012; Tonmoy et al. 2014; Brugère & De Young 2015).

A unified framework of vulnerability

Building from these conceptual developments, the Intergovernmental Panel on Climate Change (IPCC) (re)defined vulnerability as the function of a system's exposure to a driver, its sensitivity to such driver and its capacity to adapt to it (Fig. 2). This simple generic definition provides flexibility to allow different conceptual perspectives to integrate into this definition. For example, the exposure element tends to link well to the *outcome* perspective on vulnerability, and the sensitivity and adaptive capacity elements allow for an understanding of *contextual* vulnerability. In this generic yet encompassing conceptual model, the key elements of vulnerability are interpreted as follows:

1) **Exposure** designates the magnitude, frequency, duration and/or extent in which an entity is in contact with, or subject to, a driver of change (Kasperson et al. 2005).
2) **Sensitivity** describes the set of conditions and/or characteristics mediating its short-term propensity to be influenced following the exposure (Bousquet et al. 2015).
3) Exposure and sensitivity create potential impact of a stressor, which is fully experienced in the long-term depending on the entity's **adaptive capacity**. This last component includes present and future ability to implement effective and long-lasting responses to changes by minimizing, coping with, and recovering from the potential impact of a stressor (modified from Bousquet et al., 2015 and Cinner et al., 2013).
4) **Vulnerability** then results from the potential impact combined with adaptive capacity.

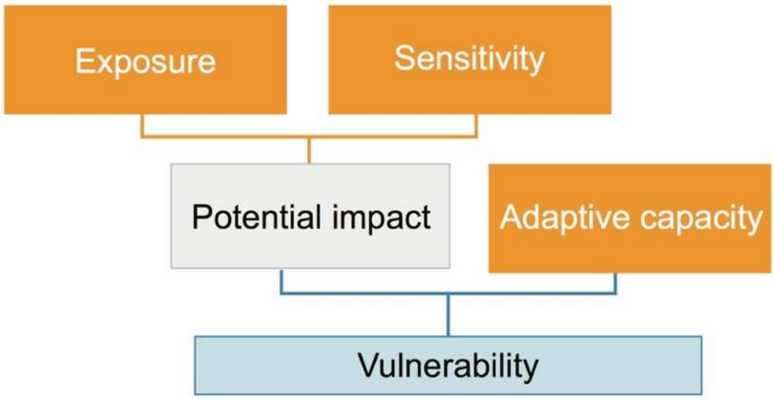

Figure 2: Conceptual framework of vulnerability as proposed by the IPCC. The combination of exposure and sensitivity creates the potential impact, which can be offset by adaptive capacity.

This vulnerability framework has provided the foundation for characterizing interactions between external drivers and internal system processes, and for estimating or ranking relative magnitudes of consequences for social and ecological systems resulting from exposure (or risk of exposure) to these drivers (Wilson et al. 2005; Johnson & Marshall 2007; Johnson et al. 2016). It also helped understanding the effects of major external drivers like climate change on social-ecological systems, enabling more informed and structured decision making (Marshall et al. 2010; Anthony et al. 2015).

In a management perspective, assessing each dimension of vulnerability provides a simple and flexible way to guide decision: if low vulnerability is the fundamental objective, then it can be achieved via actions to (1) reduce exposure, (2) decrease sensitivity, (3) enhance adaptive capacity, or a combination of those. Among the various ways that have been proposed to operationalize the vulnerability framework (Brugère & De Young 2015), indicator-based approaches (i.e., quantifying each dimension based on quantitative indicators) has emerged as a valuable method for designing interventions that can reduce vulnerability by modifying internal system properties in order to increase coping capacity to (often unmanageable) external drivers (Johnson & Welch 2009; Cinner et al. 2012a; Foden et al. 2013; Ekstrom et al. 2015). Broad strategies to address vulnerability (e.g., reducing exposure, decreasing sensitivity and enhancing adaptive capacity) are generally well proven as they result from the applied research literatures (e.g., sustainable livelihood approach, common-pool resources, etc.). For instance, in their vulnerability assessment of fishing communities to the impacts of climate change, Cinner et al. (2012) have proposed interventions focusing on strengthening community groups and investing in strong local institutions, which are direct inputs from Ostrom's and colleagues' work on commons and fisheries applications that have resulted from it (Ostrom 2009; Basurto et al. 2013). Other policy recommendations based on a vulnerability assessment include the development of social safety nets, adaptive management approaches or poverty reduction, which are core principles of the Sustainable Livelihood Approach (Allison & Ellis 2001; Allison & Horemans 2006). Although initially informed by and applied in social sciences, the many examples of vulnerability assessment in the ecological realm that emerged over the last years (Foden et al. 2013; Parravicini et al. 2014; Anthony et al. 2015; Okey et al. 2015; Johnson et al. 2016) and illustrate the ability of this framework to promote interdisciplinarity.

Social-ecological vulnerability

In practice, this tripartite framework can foster a social-ecological thinking, since, for each dimension of vulnerability, lexical analogies can be made between social and ecological systems (Table 1). Accordingly, this framing represents an opportunity to facilitate interdisciplinary research and enhance our understanding of conditions and processes leading to vulnerability in social-ecological systems.

Interdisciplinary vulnerability assessments (i.e., considering both social and ecological vulnerabilities) applied to natural resource management and biodiversity conservation have managed to incorporate both social and ecological scenarios and indicators. However, the many interpretations of vulnerability and its various scales and fields of application have led to a wide array of propositions regarding ways and means by which social-ecological vulnerability could be understood (Brugère & De Young 2015). One approach that seems to reach a consensus is to consider social-ecological vulnerability into a nested model where ecological vulnerability is the exposure dimension of the social vulnerability (Fig. 3; Cinner et al. 2013; Marshall et al. 2013, 2014).

Table 1: Key terms referred to in the context of social and ecological vulnerability.

General term Social equivalent Ecological equivalent	Definition	Example Social equivalent Ecological equivalent
Entity Social entity Ecological entity	System, subsystem or system component that compose the social-ecological system of interest.	Individuals, households, communities, institutions, sub-national sectors, societies. Populations/stock, species, ecological communities, habitats, ecosystems, eco-regions.
Driver Driver Driver	Press or pulse factor that affects an entity beyond the normal range of variability in which it operates.	Environmental degradation, socioeconomic or cultural changes, institutions and governance. Human population, markets, socioeconomic development, natural disturbance.
Exposure Exposure Exposure	Magnitude, frequency, duration and/or extent in which an entity is in contact with, or subject to, a stressor	Change in resource availability, gentrification, new institutional rules. Intensity of human use, nutrient input, change in biochemistry, frequency of natural disturbance.
Sensitivity Dependency Sensitivity	Conditions mediating the short term propensity to be influenced following the exposure	Level of importance for food, livelihood, employment, economy. Degree of specialization, dependence on environmental triggers, rarity.
Adaptive capacity Adaptive capacity/adaptability Recovery potential	Current and future ability to implement effective and long term responses to changes by minimizing, coping with, or recovering from the potential impact of a driver.	Learning and knowledge, social capital, diversity and flexibility, infrastructure, assets, agency. Environmental tolerance or thresholds, hysteresis, functional diversity, dispersal ability, evolvability.
Vulnerability Social vulnerability Ecological vulnerability	Degree to which an entity is likely to experience harm due to exposure, sensitivity and capacity to adapt to a stressor	-

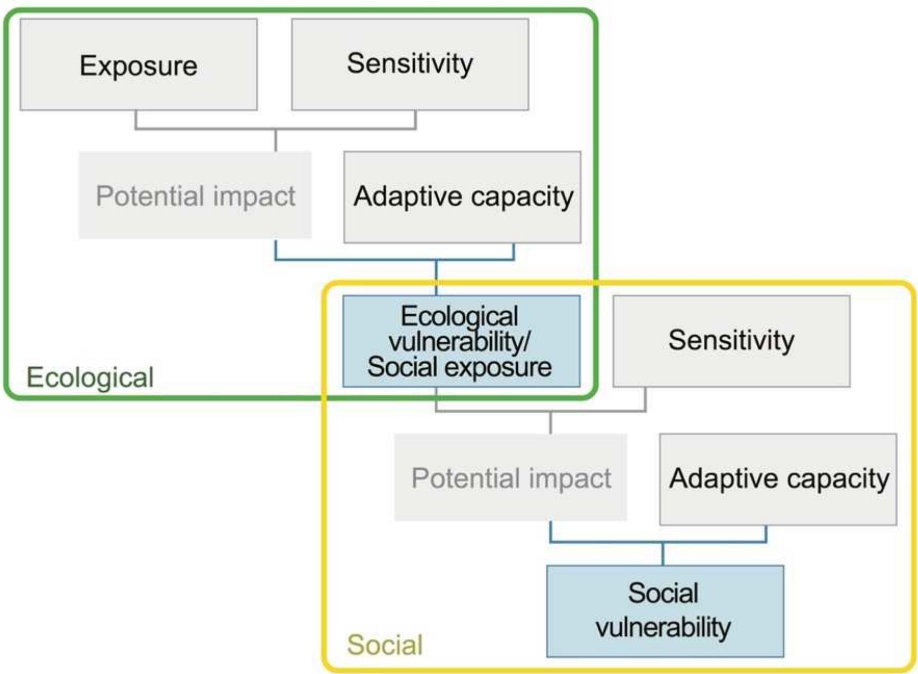

Figure 3: Conceptual framework of social-ecological vulnerability. Adapted from Marshall et al. (2010). Note that the dependency between social and ecological systems is unidirectional: there is no feedback from the social to the ecological system.

The tacit assumptions in this approach is that people and the natural environment are equally important and can be linked through the concept of ecosystem services: the social entity is indirectly vulnerable as the consequence of both its internal features (social sensitivity and adaptive capacity) and the modification of ecosystem goods and services delivery, availability or functional space on which they rely (social exposure) generated by environmental degradation (ecological vulnerability). This nested framework of social-ecological vulnerability (Fig. 3) has attracted the attention of scholars and development practitioners seeking to implement a multidisciplinary perspective on sustainability in a structured manner, and provided valuable insights to target interventions that balance human and environmental aspects (Cinner et al. 2013a, 2013b).

When looking at social, ecological and social-ecological vulnerability scholarships, policies and programs, it appears that the relationships between ecological and social vulnerabilities has been fairly well documented in regard to climate change or global environmental change (Bennett et al. 2016). As a consequence, the vulnerability framework has been greatly influential for guiding driver-oriented strategies at the national, regional and global levels (Johnson & Welch 2009), but there has been a distinct lack of application of the vulnerability framework in place-based management contexts. Yet, local managers could greatly benefit from the application of the vulnerability framework, especially with regard to drivers that can be directly addressed by local decision-makers (such as pollution, resource overexploitation, etc.). Indeed, social-ecological systems and the entities that compose it experience a broad array of multi-scalar and multi-temporal, social, political, economic and environmental changes to which they are potentially vulnerable. Given the special importance of such drivers, vulnerability assessments that better consider the whole range of drivers and leverages at stake, at the appropriate scales, will improve guidance for prioritizing conservation and advise management options. Congruent with the lack of operationalization to local-scale and multiple drivers is the absence of feedback from the social to the ecological system when dealing with social-ecological vulnerability (Fig. 3). This tends to constrain the social-ecological vulnerability framework into an anthropocentric approach, which impedes truly balanced management practice that treats social and ecological systems equally and as being interdependent (Binder et al. 2013). Finally, vulnerability assessments, especially of the social dimension, tend to focus on a single type of entities (e.g., fishing communities, functional group, tourism operators, economic sector, resource, etc.) while social-ecological systems, by essence, exhibit a diversity of entities that are interconnected and integral part of the dynamics.

The IPCC vulnerability framework and its extension (i.e., the social-ecological vulnerability framework) hold great promises to help practitioners integrate social and ecological dependencies and move toward a more holistic approach to management practice. However, its current application – without due account for non-climate-related drivers of change, feedbacks between social and ecological systems, as well as temporal changes, multi-driver (i.e., other than climate-related) and multi-entity – makes such approach poorly suited, or overly simplistic to address the needs of many practitioners. These are the gaps we here aim at bridging.

Objectives and thesis outline

Through three contrasting but complementary case studies (Box 1), this research project aims to develop new analytical approaches to help current management practices implement more holistic management of social-ecological systems. The central thread of PhD thesis is the in-depth examination of vulnerability as a potential boundary-crossing framework for linking science and policy and implement innovative and interdisciplinary management. In the light of the review of vulnerability applications and gaps described above, the seven chapters (Fig. 4) concentrate on addressing the two main following challenges:

1 - Making vulnerability relevant to local decision makers by downscaling vulnerability assessments and accounting for social-ecological interdependencies

The first part of this thesis focuses exclusively on the small-scale coral reef fishery of Moorea, French Polynesia (Box 1, Case study 1). Chapter I builds the case that the current marine spatial plan tool (a network of marine protected areas) has unlikely contributed to improve ecological outcomes, thus casting doubts on its capacity to meet its conservation and fishery management objectives. Chapter II describes a standardized but flexible approach that combines participatory mapping with socioeconomic approaches to generate a comprehensive map of fishing effort in Moorea's reefs, a critical input to improve spatial planning design (Parnell et al. 2010; Weeks et al. 2010). Based on this map of fishing pressure and other spatially-explicit social and ecological information related to fishing, social-ecological vulnerability in then mapped at a fine-scale in Chapter III. This approach explicitly considers interdependencies between the social and the ecological systems locally through ecosystem services delivery and use. Drawing from this spatial analysis of aggregated social-ecological vulnerability, and building on mature and extensive applied research in the field of fisheries management and poverty reduction, social-ecological vulnerability is then unpacked in Chapter IV and a decision-support tool is proposed to systematically address the underlying sources of unsustainability.

2 - Expanding the scope of vulnerability assessments to real-world management challenges

Whereas the first part of this thesis primarily explores how validated and well studied set of methodologies can be spatially applied to local human-nature dependencies to fine tune or guide spatial planning, the second part investigates the potential of vulnerability as a flexible tool to provide an even more holistic understanding of social-ecological systems that includes multi-driver and multi-sectors perspectives. Specifically, in Chapter V is presented an approach that integrates the temporal dimension into social-ecological vulnerability assessments to help communities and decision makers understand and plan for the effects of large scale or external drivers. Two key metrics (current vulnerability and vulnerability trajectory) following exposure to multiple drivers of change are mapped in Moorea, which provides decision makers with the information required to implement proactive and adaptive management, and also highlighted the importance of external forces in shaping social-ecological trajectories. Therefore, Chapter VI focuses on two major drivers of change in Chilean artisanal fishing communities (Box 1, Case study 2), namely markets and poaching. This required adapting the vulnerability framework to multiple drivers, which was achieved through the differentiation between general versus specific aspects of vulnerability. Building on these conceptual and methodological advances, an analytical framework enabling to reduce vulnerability in a multi-driver context is proposed. Finally, by assessing the vulnerability of nations' agriculture and fishery sectors to the impacts of climate change (Box 1, Case study 3), Chapter VII highlights how multiple entities can be incorporated into vulnerability assessments and yield practical recommendations that could improve coordination across multilateral policy initiatives.

Operationalization and measurements were essentially undertaken through quantitative/statistical downscaling approaches, integrating a great variety of information types and modelling techniques, and spanning a variety of temporal and spatial scales that are detailed and discussed in each chapter. The thesis concludes by highlighting the conceptual and practical contributions of this research to the knowledge on social-ecological systems, vulnerability and broader action-oriented research. This is followed by a discussion of some identified limitations and potential future research directions.

Box 1: Overview of the case studies.

Case study 1: Coral reef small-scale fishery in Moorea, French Polynesia

Interconnections between people and the natural environment are clearly apparent in Moorea, notably through the small-scale coral reef fishery, which supports the livelihood of hundreds of households but has shown shows some signs of unsustainability over the last decades (Leenhardt et al. 2016). The mixed economy in which people are imbeded, the recent urban development and intense exposure to globalization (Féral 2013) as well as recurent large-scale drivers of ecological (Kayal et al. 2012; Lamy et al. 2015a, 2015b), socioeconomic (ISPF 2014; Insee-ISPF 2016) and governance changes (Audras et al. 2016, 2017) make the current marine spatial planning revision particularly challenging. However, the large amount of pre-existing ecological and socioeconomic data available (Cressey 2015) and the clearly identifiable social-ecological boundaries make Moorea an indeal case for the in-depth examination of system-wide vulnerability of local human-nature interdependencies and their spatiotemporal dynamics.

Case study 2: Co-management of benthic resources in Chile

Chile has established a national Territorial User Rights for Fisheries (TURF) policy for benthic resources, which gave legal authority to assign collective exclusive access rights to artisanal fisher organizations. By 2016, there were around 550 TURFs decreed to fisher organizations, making Chilean artisanal fisheries the largest TURF system under one policy instrument. Yet, the problems confronting fisheries managers in Chile are reflective of the difficulties encountered in the management of marine resources worldwide: disputes over territorial rights between fishers are responsible for illegal fishing practices including poaching (Gelcich et al. 2005, 2012), and a strong reliance on a virtually unmanageable international seafood trade exposes the fishing organizations to the whims of new markets and demand (Gelcich et al. 2010; Castilla et al. 2016). Applying the vulnerability framework to examine the underlying sources of social vulnerability (Gelcich et al. 2017) provides a unique opportunity to scale-up TURF management in an increasingly complex and interconnected world.

Case study 3: World's food systems in a changing climate

Agriculture and marine fisheries are cornerstone sectors not only for food security and safety, but also for economic growth and employment worldwide. It is now clear that ongoing climate change will alter the functioning and productivity of agricultural and marine ecosystems these sectors depend on. In this context, (1) understanding linkages between climate impacts on ecosystems and the cascading consequences on human societies, and (2) ensure coordination across sectors so that decisions about one do not (unexpectedly) affect the other are key to identify future transformation pathways. Considering both sectors as different entities of a system, vulnerability is here used as a way to explore potential opportunities for synergies and identify trade-offs across multilateral climate policy initiatives.

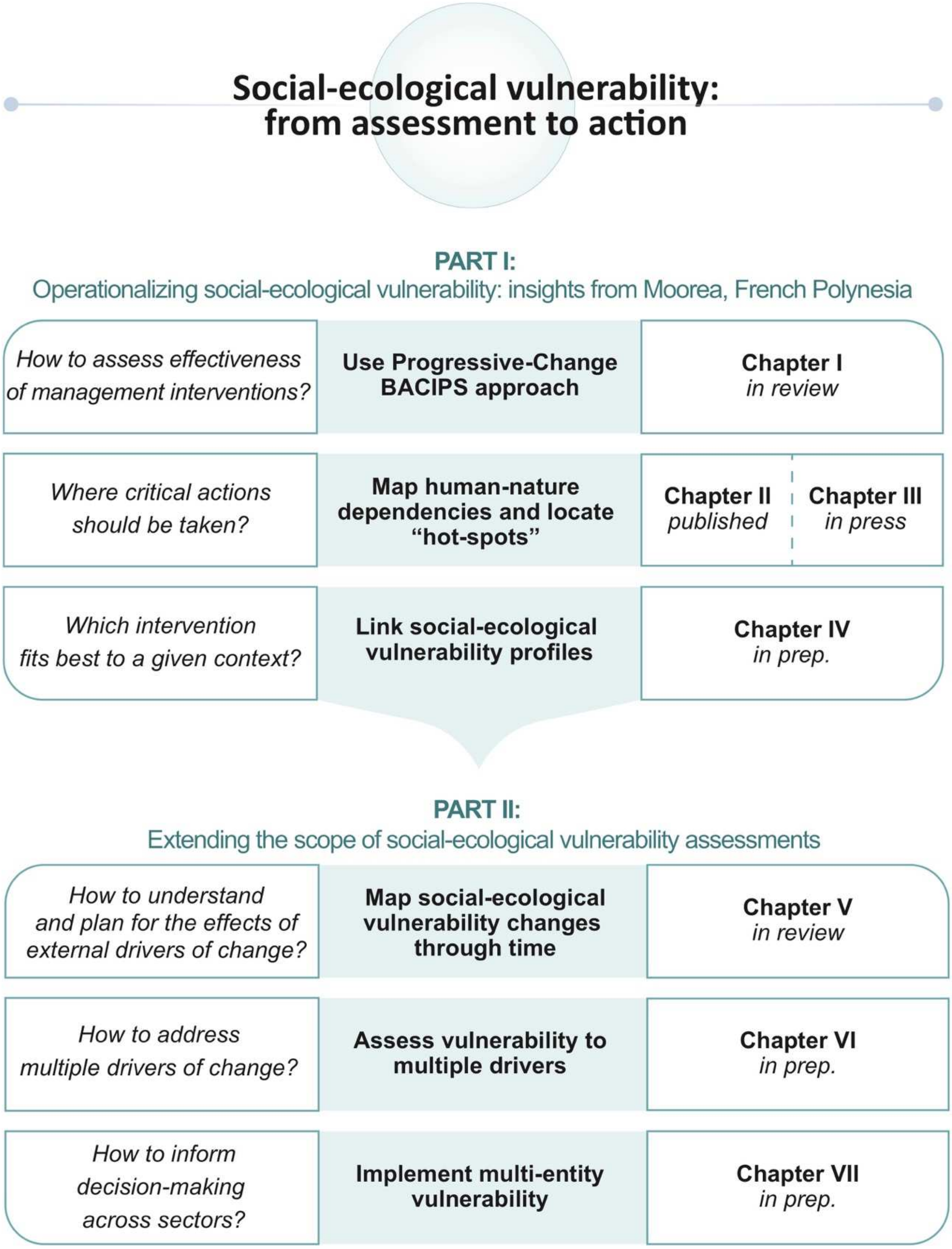

Figure 4: Synthesis of the approach developed in this PhD thesis. Practical questions are aimed to be addressed through quantitative, statistical and/or spatial modelling that blend a great variety of data types and methods. To each key question corresponds one or several chapter(s) published in, submitted or to be submitted to a peer-reviewed academic journal.

PART I: OPERATIONALIZING SOCIAL-ECOLOGICALVULNERABILITY

INSIGHTS FROM MOOREA, FRENCH POLYNESIA

Chapter I

Ecological evaluation of a marine protected area network: A Progressive-Change BACIPS approach

Status: In review

Journal: Biological Conservation

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Chapter I: Ecological evaluation of a marine protected area network: A Progressive-Change BACIPS approach

Abstract

Marine Protected Area (MPA) networks are potentially useful tools to manage trade-offs between conservation and fishing activities but their effectiveness must be evaluated to ensure they meet their conservation objectives. Past assessments have been criticized because they lack Before data and therefore cannot discern natural spatio-temporal variation and pre-existing differences from effects of the MPAs. Here, we used a Progressive-Change BACIPS approach to analyse the effects of a network of eight fully and moderately protected MPAs on fish communities in two coral reef habitats (lagoon and fore reef) based on a 12-year time-series of data collected Before and After the network's establishment on Moorea Island (French Polynesia). At the network scale, on the fore reef, density and biomass of harvested fishes increased by 19.3 and 24.8 %, respectively, in protected areas relative to controls. Most of this effect arose from fully protected MPAs; no significant effects were detected in moderately protected MPAs. Regardless of the protection level, no significant effects of MPAs were detected in the lagoon. Patterns were similar when analyses were conducted individually for each MPA. We suggest this lack of pronounced effects of protection is due to limited compliance and weak surveillance, although other factors such as the occurrence of a crown-of-thorns starfish outbreak and a cyclone may also have impeded the ability of the network to provide benefits or our ability to detect responses. Our results highlight the relevance of fully protected MPAs over moderately protected MPAs to achieve biodiversity conservation objectives, even in complex social-ecological settings, but also stress the need to enhance human and financial capacity to improve the ecological success of management actions.

Introduction

Marine Protected Areas (MPAs) are an important management tool to conserve or restore fish populations inside their borders (Kerwath et al. 2013; Lubchenco & Grorud-Colvert 2015) and export biomass to surrounding fishing grounds (Goñi et al. 2008; Harmelin-Vivien et al. 2008; Di Lorenzo et al. 2016). Past studies have shown that ecological effects of MPAs depend on MPA age (Claudet et al. 2008; Molloy et al. 2009; Friedlander et al. 2017), network design (Jupiter & Egli 2011; Green et al. 2014a, 2014b), species traits (Claudet et al. 2010) and degree of compliance (Guidetti et al. 2008; Campbell et al. 2012; Gill et al. 2017). The most compelling evidence for beneficial effects of MPAs arises from meta-analyses that synthesize data from many empirical studies (Côté et al. 2001; Micheli et al. 2004; Claudet et al. 2008; Gill et al. 2017).

Despite accumulated evidence suggesting far-reaching average benefits of MPAs, assessment designs of individual MPAs or MPA networks have substantive limitations that constrain evaluation of their effectiveness. MPA assessment remains a challenging task because most studies lack data from Before the establishment of the MPA and therefore cannot discern effects of the MPA from pre-existing differences (Guidetti 2002; Halpern et al. 2004; Osenberg et al. 2006, 2011). Even studies with Before samples tend to have a limited time-series, often only one survey, making it difficult to attribute any observed temporal trends to effects of the MPA or MPA network (Osenberg et al. 2006, 2011).

When Before data are available, the BACIPS (Before-After Control-Impact Paired-Series) assessment design (Stewart-Oaten et al. 1986; Osenberg et al. 1994; Stewart-Oaten & Bence 2001) provides a powerful tool to overcome many of the limitations of typical studies (e.g. Guidetti, 2002; Osenberg et al., 2011, 2006). Repeated assessments *before* enforcement provide an estimate of the spatial variability between the Control and Impact sites in the absence of an effect of the MPA. In its simplest application, a change in the difference (Δ) in density (or other response parameter) between the Control and Impact sites *after* the establishment of the MPA (i.e., Δ_{After} - Δ_{Before}) provides an estimate of the local effect of the MPA (see Stewart-Oaten and Bence, 2001 for a more detailed discussion of the BACIPS analysis and Osenberg et al., 2011, 2006 for a discussion of the BACIPS method applied to marine reserves). This step-change (from Δ_{Before} to Δ_{After}) is highly unlikely in most MPA systems. For example, enforcement may be gradual or the response of long-lived species may be slow to accumulate (e.g., Russ and Alcala, 2010). In such cases, the effect of the MPA may follow more complex dynamics (Babcock et al. 2010). Recently a more flexible approach, the Progressive Change BACIPS, was proposed (Thiault et al. 2017b) that allows quantification of various patterns of temporal change in addition to the traditional step-change (e.g. linear, asymptotic, sigmoid). Unfortunately, these BACIPS designs have rarely been used to assess MPAs (but see Castilla & Bustamante 1989; Lincoln-Smith *et al.* 2006; Claudet *et al.* 2006; Shears *et al.* 2006; Moland *et al.* 2013; Grorud-Colvert *et al.* 2014; Fletcher *et al.* 2015).

Here, we analyze a 12-year time series of data (which includes four years of Before data) to assess the effectiveness of a network of 8 fully and moderately protected MPAs (Horta e Costa et al. 2016) in Moorea, French Polynesia. We apply the Progressive Change BACIPS design to evaluate the pattern of response to the establishment of the MPA network. We assessed the effects of the MPA network as a whole, the sub-network of fully protected MPAs, the sub-network of moderately protected MPAs, and the individual MPAs.

Material and methods

Data collection

The MPA network in Moorea, French Polynesia was officially designated in October 2004, although establishment and enforcement required several additional years (Lison de Loma et al. 2008). MPAs were delimited inside the lagoon using buoys in September 2005. The first information campaign and police patrols were conducted in 2006. A second information campaign was initiated in 2007, and police monitoring was subsequently increased (and accomplished by hiring a local mediator/enforcement agent and purchasing a boat). We therefore consider January 1, 2007 to constitute the start of enforcement (Lison de Loma et al. 2008).

The network consists of five fully protected MPAs and three moderately protected MPAs (Fig. 1; Lison de Loma et al. 2008). MPAs within the network were classified using the Regulation-Based Classification System for MPAs (Horta e Costa et al. 2016). Other areas are open to fishing but subject to general restrictions, such as species size regulations on the north shore.

Fish communities and benthic assemblages were sampled from 2004-2015, with sampling surveys (which we refer to as “dates”) conducted once in 2004 (during the dry season), twice each year from 2005-2009 (during both the dry and wet seasons), and once each year thereafter (during the wet season). Thus, our dataset consists of five sets of surveys from the Before period (i.e., prior to January 2007) and 12 sets of surveys from the After period (i.e., after January 2007). We refer to each set of surveys as a date, even though data were obtained over an approximately week-long period.

At each MPA and Control site we sampled one location on the fore reef and two locations in the lagoon. Fishes were identified to species and enumerated along 3, 25 x 2 m underwater belt transects at each location. Total length of each fish was estimated to the nearest centimeter for isolated fish, and mean length was estimated for schools of fish.

We also quantified the density of Crown Of Thorns Starfish (COTS, *Acanthaster planci*), a coral predator, as well as the cover of live coral and algae. Coral and algal cover were estimated using a point-intercept transect method, using the 25-meter line that was deployed for fish transects. A total of 50 points were used, spaced equally along each transect. All point contacts were done after the fish surveys to minimize disruption to the fish assemblage.

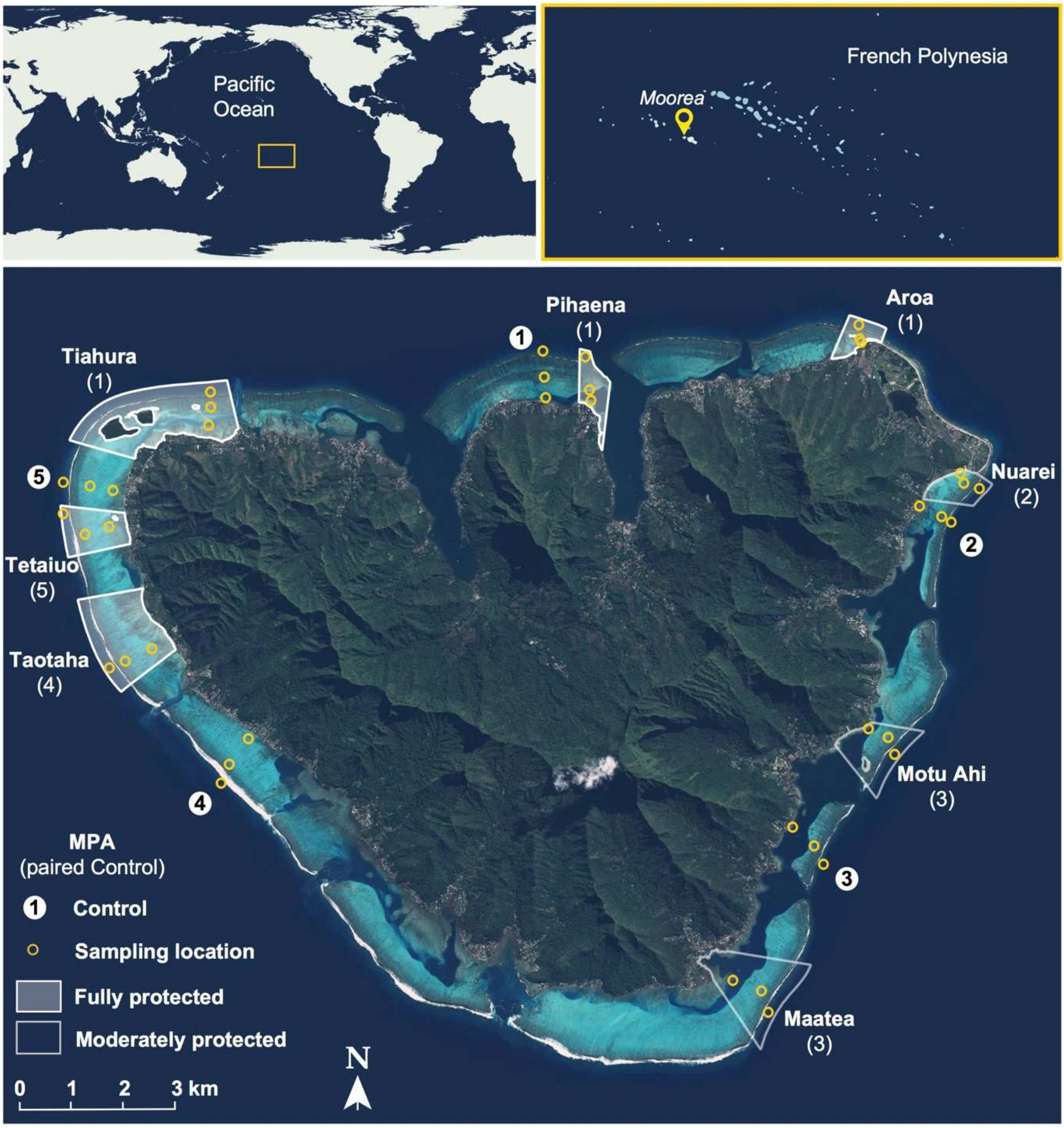

Figure 1: Network of MPAs implemented in the coral reefs of Moorea Island, French Polynesia. The network consists of five fully protected MPAs (on the north and west shores) and three moderately protected MPAs (on the east shore) (Regulation-Based Classification system for MPA, Horta e Costa et al. 2016), as well as five Control areas. Each Control is numbered; numbers in parentheses for MPAs refer to the Control site to which it was paired. Sampling at each MPA and Control was conducted in two distinct habitats (two locations in the lagoon and one location on the fore reef) five times Before and 12 times After implementation, allowing us to perform a Progressive-Change BACIPS analysis.

The Progressive-Change BACIPS design and analysis

We categorized fish species as harvested or non-harvested based on local expert knowledge, and converted all lengths to wet mass (g) using species-specific length-mass relationships (Kulbicki et al. 2005). Sharks, rays and pelagic species were omitted from the analyses because the transects were not designed to count those highly vagile species. Within each date, site and habitat, data from the transects were averaged (i.e., 3 transects for the fore reef sites and 6 transects for the lagoon sites). We then determined the difference, Δ, between the MPA and its Paired Control site (see Fig. 1) after log-transformation (Lison de Loma et al. 2008):

$$\Delta_{P,i} = \ln (N_{MPA,P,i} + a) - \ln (N_{Control,P,i} + a) \qquad (1)$$

where N was the average target fish density or biomass (across the three or six transects) at either the MPA or Control site, during the i^{th} date in the P^{th} period (P=Before or P=After), and 'a' was added to avoid taking logarithms of zero. We used the smallest value of 'a' possible, by assuming it represented the addition of one fish to one of the n (n=3 or 6) transects (i.e., a = 1/n for analyses of density, and a = [mean mass of one fish]/n for analyses of biomass).

We evaluated whether protection had an ecological effect by assessing if the difference in density or biomass, Δ, changed from Before to After the establishment of the MPAs. Instead of a step-change in Δ, we expected a more complex transition in Δ during the After period for two reasons: (1) continuous increases in fish densities have been observed up to 25 years after protection (Russ & Alcala 2004; Babcock et al. 2010; Coll et al. 2013), and perhaps more importantly, (2) enforcement of the MPAs in the network was gradual. Therefore, we applied a Progressive-Change BACIPS by competing four models: step-change, linear, asymptotic and sigmoid models (Thiault et al. 2017b). The magnitude of the response of fish to protection (hereafter refered to as effect size) was then measured based on the predictions of the best-fit model (highest AICc score) at t=2015, which corresponds to the last year in our dataset.

We assessed the effect of (i) the whole network (i.e., the island-wide effect of the MPA network), (ii) the two sub-networks (i.e., the network of fully protected and moderately protected MPAs, respectively) and (iii) each individual MPAs using all $\Delta_{P,i}$ calculated for each pair of sites (i.e., eight pairs for each date). To avoid pseudo-replication, and because we expected that each MPA would have its own response, we fitted mixed models with MPA as a random effect for network- and sub-network-scale analyses.

We explored the consequence of implementing a BACIPS design by comparing effects of individual MPAs measured using the Progressive-Change BACIPS with those obtained using the more commonly used Control-Impact approach. For each possible comparison, effect sizes from the Control-Impact approach were calculated using data from the last sampling date (t=2015) with confidence intervals based on spatial variation among transects. These means and associated 95% CIs were then plotted against effects sizes and 95% CIs from the Progressive-Change BACIPS. All analyses were performed using the R statistical software (R Core Team 2014).

Results

Fishes were generally more abundant on the fore reef compared to the lagoon (Fig. 2). Coral and algal cover changed dramatically over the period of study on the fore reef following a Crown Of Thorns Starfish (COTS; *Acanthaster planci*) outbreak between 2007 and 2012 and cyclone Oli in 2010: as COTS increased, coral cover decreased by more than 90% between October 2006 and March 2010, and algae increased by 34% (Fig. 2). These temporal changes highlight the importance of designs that include Control sites and Before data.

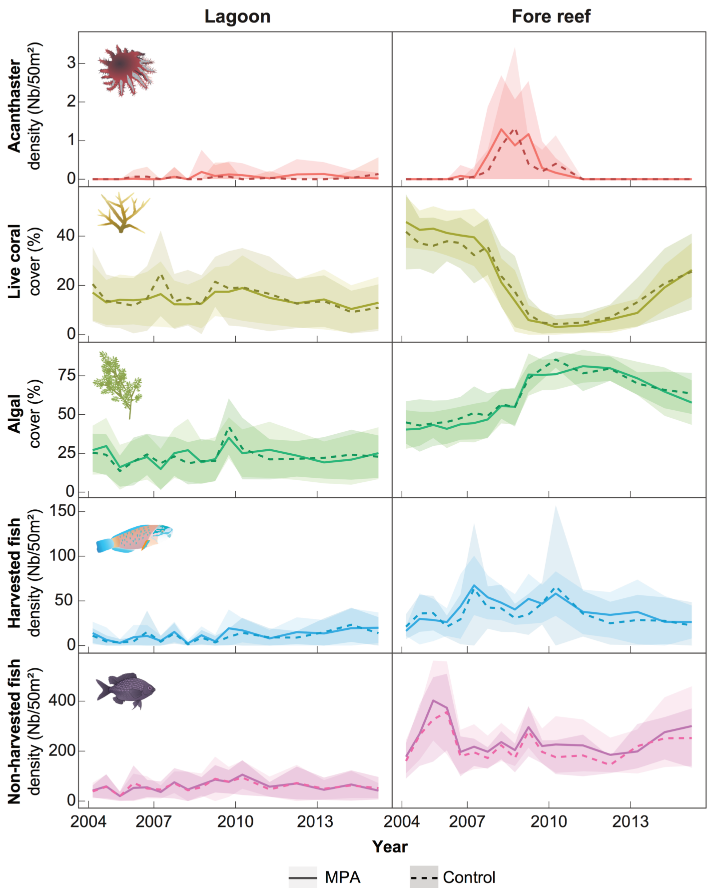

Figure 2: Temporal variation in key ecological components of the studied coral reef. From top to bottom: density of *Acantasther planci* (red), living coral cover (yellow), algal cover (green), density of harvested (blue) and non-harvested fish (purple) in the lagoon (left column) and fore reef (right column) during the study. Lines are the mean of the eight MPA (solid line) and five Control (dashed line) sites and ribbons represent standard deviations (the darker ribbon corresponds to the Controls).

The pattern of change in response to protection provided approximately equal support for the step-change model (asserting an immediate shift in density in the MPA relative to the Control site) and the linear model (in which the difference increased linearly with time since protection). In no case was the asymptotic or sigmoid model better supported by the data. In 37.5% of cases, the likelihood (ω) of the second best-fit model was comparable to that of the best-fit model (i.e., $\omega_{\text{best-fit}} - \omega_{\text{second best fit}} \leqq 10\%$). For consistency, we only present results derived from the best-fit model.

On the fore reef, harvested fish biomass increased significantly (by 24.8 %) at the network scale (Fig 3); the increase in density was similiar in magnitude (19.3 %), although not significant. Density and biomass of harvested fish on the fore reef increased significantly in fully protected MPAs (by 43.2 % and 31 %, respectively), but not in moderatley protected areas (Fig. 3). At the individual MPA-scale, effects on harvested fishes on the fore reef were positive in 15 out of 16 comparisons, although only one of the 15 was significant (Pihaena MPA), and the one case of a negative response (density in Maiata) was also significant.

No effect of protection was detected on harvested fish communities inside the lagoon (Fig. 3), regardless of the scale (whole network, sub-network, or individual MPA) or metric (density or biomass) considered.

Effects on non-harvested fishes were generally smaller in magnitude and more often negative (Fig 3), as might be expected because these fishes are not harvested and may be negatively affected by an increase in the density of harvested fishes (e.g., due to competition or predation). For example, density and biomass of non-harvested fishes did not respond significantly in either the lagoon or fore reef at either the network-scale or sub-network-scale. At the MPA-scale, there were slightly more increases than decreases (20 vs. 12), although only two of these were significant (Pihaena and Maatea) and both represented decreases in fish density (or biomass) following protection.

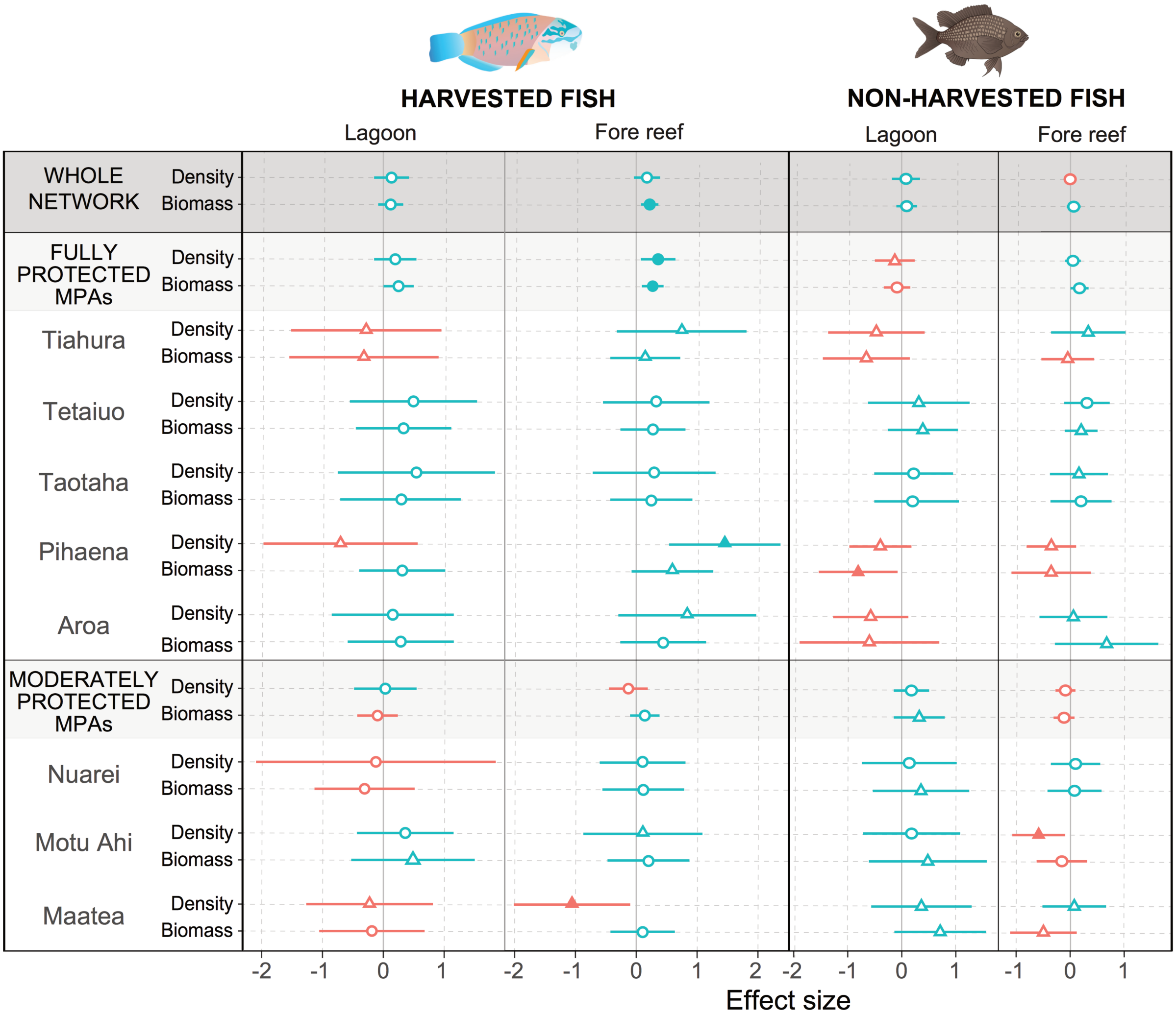

Figure 3: Effect sizes measured at the whole network scale, at the fully protected MPAs and moderately protected MPAs sub-networks scale, and at the MPA-scale, in the lagoon reef and fore reef, for density and biomass of targeted and non-targeted fishes. Tiahura, Tetaiuo, Taotaha, Pihaena and Aroa are fully protected MPAs; Nuarei, Motu Ahi, and Maatea are moderately protected MPAs. Effect sizes are expressed as the log-ratio of the density or biomass in the MPA relative to its Control (Δ) as predicted by the best-fit model at t=2015. Changes by a factor of 2 thus correspond to values of -0.7 (halving) or 0.7 (doubling) over the 9-yer period of protection. Red color represents negative effects and blue color represents positive effects; filed symbols indicate that the 95% confidence interval of the effect does not overlap zero, whereas open symbols indicate non-significant effects. Shapes indicate the best-fit model: step-change (circle) and linear (triangle). There were no cases in which the asymptotic or sigmoid model was best supported by the data.

Overall, effect of individual MPAs as measured by our Progressive-Change BACIPS approach were significant in 4 cases out of 64. Although the pattern was similar using the Control-Impact approach, the number of significant effects increased from 4 to 18, because effects tended to be larger in magnitude (i.e., more dispersed) and confidence intervals tended to be smaller (Fig. 4).

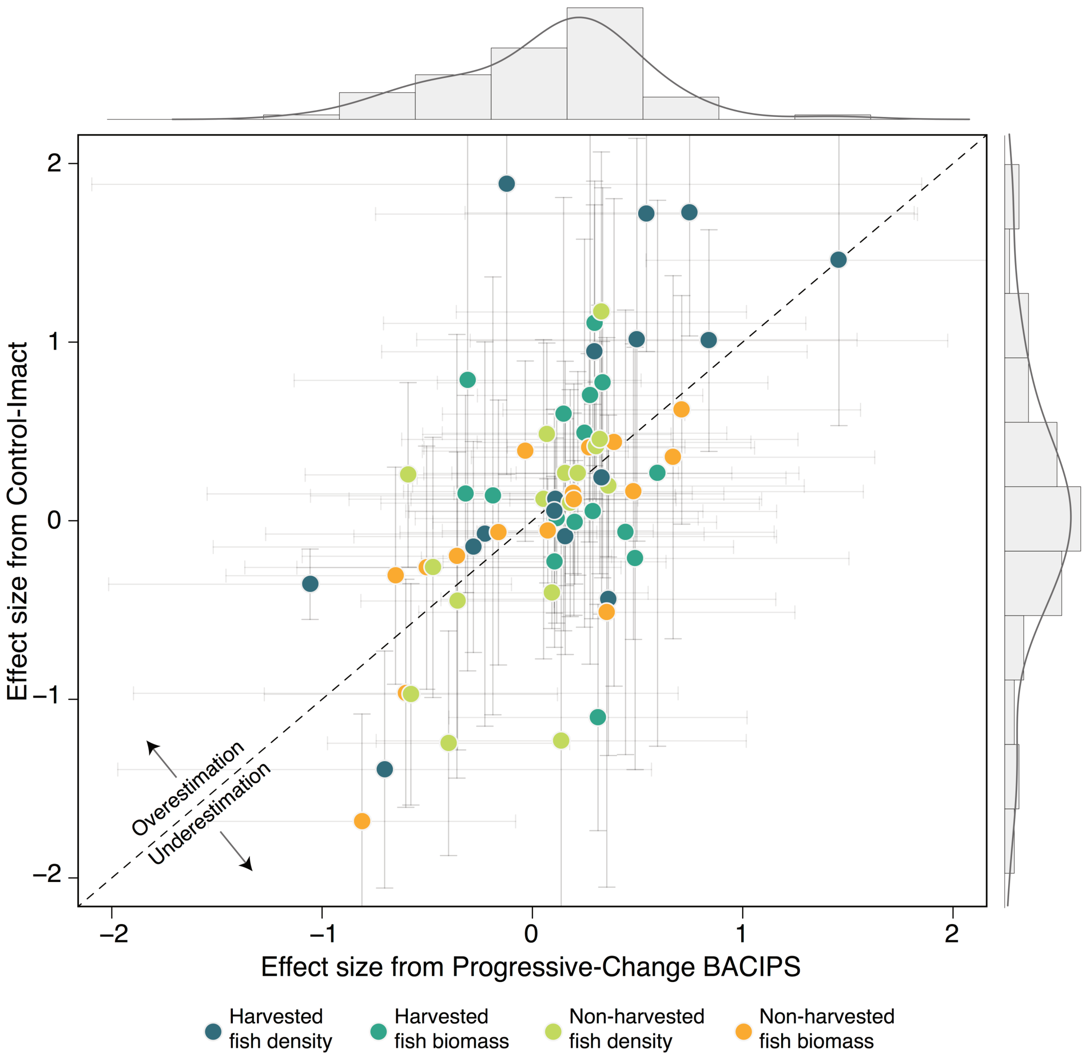

Figure 4: Comparison of effect sizes as estimated by the Progressive-Change BACIPS approach and the more traditional Control-Impact method. Each point represents an effect-size (+/- 95% confidence intervals) measured at the MPA-scale and colors indicate the variable considered. The 1:1 line shows where the two methods provide equal effect sizes; deviation from this line indicates situations where the Control-Impact method underestimates and overestimates the effect size relative to the more robust Progressive-Change BACIPS approach used in this study. Histograms indicate the distribution of effect sizes along the x and y axes.

Discussion

The establishment of the MPA network in Moorea provided positive – but limited – ecological benefits. Network-wide, harvested fish biomass had increased 24.8% on the fore reef eight years after the start of enforcement. Importantly, this positive, network-scale effect was mostly driven by responses in the fully-protected MPAs (+30% and +43.2% in biomass and density, respectively). Indeed, no positive effect was detected in moderately protected MPAs in either habitat. The absence of a demonstrable increase in areas that were not fully protected is consistent with a recent meta-analysis that compared ecological benefits conferred by various classes of MPAs (Zupan et al. *in prep.*).

The response in fully protected MPAs, although the largest effect we detected, was relatively small compared to effects documented in other published MPA studies, in which harvested organisms were generally 2-3 times more dense inside MPAs compared to fishing grounds (Halpern 2003; Claudet et al. 2008, 2011). The lower effect measured in this study could be due to a combination of reasons, including (i) overestimation of the effect size in previous studies due to the lack of data Before the establishment of the MPA, (ii) limited compliance and enforcement, (iii) limited statistical power, and (iv) dramatic habitat disturbance following enforcement due to the COTS outbreak and cyclone.

Although there have been hundreds of assessments of MPAs, very few include data from Before the establishment of the MPA and even fewer have multiple surveys from Before. MPAs are often strategically implemented in sites with higher densities than surrounding areas. The absence of Before data precludes the incorporation of these initial (and potentially large) differences, which may then become confounded with effects of MPAs. As a result, studies that lack Before data may overestime the benefits of MPAs. Indeed, it has been suggested that up to half of the commonly observed "increase" in density inside MPAs is due to these pre-existing differences (Osenberg et al. 2006, 2011). The BACIPS approach circumvents these problems, and likely leads to smaller, but more accurate, estimates of effect sizes.

Limited public appreciation about the benefits of MPAs and an understaffed management team may have limited compliance, and therefore ecological effectiveness (Gaspar & Bambridge 2008; Gill et al. 2017). Inside the lagoon, the absence of significant effects of protection on harvested fishes may also be explained by poaching and limited enforcement (Guidetti et al. 2008; Edgar et al. 2014). For example, surveillance reports made by the local mediator (Gaspar & Bambridge 2008) and surveys completed by local experts (Appendix A; Table S1) suggest that enforcement (was heterogeneous, being mostly limited to the north shore, where the largest beneficial effects were observed (Fig. 3). Furthermore, the lagoon habitat is highly fished at night with light attractors, and surveillance is non-existant at night. We observed the largest effects of protection on the fore reef, which is less accessible, and more hazardous. As a result, the fore reef is likely to experience less night poaching than the lagoon (Thiault et al. 2017a). The marine spatial management plan of Moorea is currently being revised to better to engage local communties and foster better compliance.

Limited statistical power might have prevented us from detecting effects, especially at the MPA-scale. The power of a BACIPS design is determined by the number of sampling dates and the degree of spatio-temporal variation in density and biomass, which is driven by sampling error as well as true spatio-temporal variability (Osenberg et al., 1994). In our dataset, this error term (reflected by the size of confidence intervals in Fig. 3) was large. Using these estimates of variation, we conducted power analyses and found that we were unlikely to detect a 100% increase in density and biomass in the MPA (Appendix A; Table S2). Thus, we had limited power to detect effects previously documented at the

MPA-scale. Interestingly, error varied systematically among the two habitats we sampled: it was smallest on the fore reef and greatest in the lagoon (Appendix A; Table S2). This may result from the greater habitat heterogeneity inside the lagoon (Galzin 1987a). As a result of this variation in power, we were better able to discern effects of the MPA on the fore reef than in the lagoon.

Finally, the whole island underwent severe natural disturbances during the time frame of our study. An outbreak of Crown Of Thorns Starfish (COTS, *Acanthaster planci*), followed by a cyclone in 2010 occurred on the fore reef. This resulted in a dramatic (90%) decline in live coral cover and an increase in macro-algae (Fig. 2). Shifts in microhabitats have led to changes in the composition of reef-associated fishes, but after a time lag of several years (Adam et al. 2011; Lamy et al. 2015b; Han et al. 2016). These types of dramatic temporal changes can cause problems with some types of assessments (e.g., Before–After comparisons). BACIPS, in theory, can handle such regional phenomena because the Control site will reflect the effects of the regional processes (i.e., the natural disturbance) but not the local factors (i.e., MPA). Surveys confirmed that similar impacts were simultaneously observed at the MPA and Control sites (Kayal et al. 2012; Lamy et al. 2015b), suggesting that COTS and cyclone effects are not confounded with possible MPA effects and underlining the importance of implementing BACIPS designs for future MPA assessments.

By applying a BACIPS design, our goal was to more effectively quantify the benefits of MPAs. Interestingly, Osenberg et al. (2011) suggested that when sites are sampled Before, the pre-existing differences are somewhat reduced (i.e., because the Before period allows investigators to select sites that are, *a priori*, more similar to one another). Indeed, our results support this interpretation: effect sizes from a Control-Impact comparision were comparable in magnitude as the effects from the BACIPS analyses (Fig. 4). However, there were two important differences. Firstly, the variation in the effects was greater for the Control-Impact analyses, presumably because initial differences between sites added to the variation in the Control-Impact effects (relative to the effects quantified with BACIPS). Secondly, the confidence intervals on the effects were smaller for the Control-Impact estimates, likely because Control-Impact studies only capture spatial variation while BACIPS captures spatio-temporal variation. These results not only suggest that Control-Impact studies might overestimate effects, but that they might also give a false sense of confidence in the estimates because they fail to incorporate temporal variance.

Our results have been communicated to some local community members and to local administrations to inform the ongoing revision of the marine spatial management plan in Moorea. We believe the Progressive-Change BACIPS approach used here could lead to more rigorous evaluation of management interventions in other settings.

Chapter II

Combining participatory ad socioeconomic approaches to map fishing effort in small-scale fisheries

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Chapter II: Combining participatory and socioeconomic approaches to map fishing effort in small-scale fisheries

Abstract

Mapping the spatial allocation of fishing effort while including key stakeholders in the decision making process is essential for effective fisheries management but is difficult to implement in complex small-scale fisheries that are diffuse, informal and multifaceted. Here we present a standardized but flexible approach that combines participatory mapping approaches (fishers' spatial preference for fishing grounds, or fishing suitability) with socioeconomic approaches (spatial extrapolation of social surrogates, or fishing capacity) to generate a comprehensive map of predicted fishing effort. Using a real world case study, in Moorea, French Polynesia, we showed that high predicted fishing effort is not simply located in front of, or close to, main fishing villages with high dependence on marine resources; it also occurs where resource dependency is moderate and generally in near-shore areas and reef passages. The integrated approach we developed can contribute to addressing the recurrent lack of fishing effort spatial data through key stakeholders' (i.e., resource users) participation. It can be tailored to a wide range of social, ecological and data availability contexts, and should help improve place-based management of natural resources.

Introduction

Small-scale fisheries, which have been defined as those "traditional fisheries involving fishing households (as opposed to commercial companies), using relatively small amounts of capital and energy, relatively small fishing vessels (if any), making short fishing trips, close to shore, mainly for local consumption" (FAO 2007-2016 n.d.), provide an iconic example of the intricate links between people and nature. Food and capital accumulation through fishing and selling of marine products are important for food security and poverty alleviation, especially in developing countries (Béné 2006; Béné et al. 2007; Daw et al. 2011a). Less tangible benefits such as well-being, and individual and collective cultural identity also make small-scale fishing strongly embedded in the lifestyle of many fishing communities (Tunstall 1969; van Ginkel 2001; Pollnac & Poggie 2008; Urquhart & Acott 2014). Their characteristics, compared to large-scale fisheries, have often been advanced by academics as key aspects of their sustainability (Jacquet & Pauly 2008; Carvalho et al. 2011). However, in most countries, issues including conflicts with industrial fisheries (Bennett et al. 2001; DuBois & Zografos 2012), open-access to fisheries (Stobutzki et al. 2006) or use of destructive fishing methods and poverty traps (Cinner 2009, 2011; Shester & Micheli 2011), have led many small-scale fisheries to be exploited beyond sustainable levels (Pomeroy 2012). Securing fisheries' and livelihoods' sustainability, and preventing or escaping from social-ecological traps can only be achieved with sound fisheries management (Allison & Ellis 2001; Cinner & McClanahan 2006).

Management of small-scale fisheries requires knowledge to make decisions about where, when, to whom and to which extent resources should or should not be allocated (Berkes et al. 2003; Ostrom 2007; Basurto et al. 2013). One of the overarching challenges of current fisheries science is that data related to the human-nature interactions are difficult to integrate into tools that can effectively guide decision-making (Kittinger et al. 2013; Leenhardt et al. 2015). Albeit advocated as a critical input for policymakers and managers (Berkes et al. 2001), the spatial distribution of resource use (hereafter referred to as fishing effort) is no exception due to the often diffuse and informal nature of the fisheries, the variety of motivations to fish among individuals (e.g., to eat, to sell and/or for pleasure), the diversity of strategies regarding gear, habitats and species caught, and a common lack of human, technical and financial resources for data collection and processing (Zeller et al. 2007). Such complexity in assessing spatial distribution have made conventional quantitative assessments of fishing effort such as fleet registers, catch declarations, sales notes and individual tracking from vessel monitoring systems relatively uncommon in small-scale fisheries (but see (Stelzenmüller et al. 2008) for counterexample).

Going beyond conventional fisheries assessment methods requires alternative approaches that better incorporate the human dimension while coping with the inherent complexity of small-scale fisheries (McConney & Charles 2008; Kittinger 2013). At the local scale, academics and practitioners have already begun to integrate social components into spatial assessments of fishing effort (Jones et al. 2008; McCluskey & Lewison 2008). Such methods include interview data and quantitative participatory processes to better understand fishing intensity at particular sites (Wynne & Côté 2007; Daw 2008), individual or collective description of the value of fishing areas (Levine & Feinholz 2015; Ramirez-Gomez et al. 2015), focal follows (Aswani & Lauer 2006) and self-reporting diaries (Albert et al. 2015). These approaches have the advantage of generating a great amount of spatial information about linked provisioning and cultural services (Brown et al. 2012), can yield information about fishing practices at high temporal and spatial resolution, and facilitate gathering of additional data such as local ecological knowledge. However, for this information to scale in coverage and produce reliable fishing effort estimates at the fishery level, these approaches require large sample sizes and appropriate sampling designs, which are rarely achieved due to logistical constraints (but see (Leopold et al. 2014; Albert et al. 2015)). Therefore, obtaining reliable information on fishing effort through active participation of fishers (from now on referred to as direct participatory approaches) may be only applicable if some particular conditions are met, or if significant financial, human and time investments are committed, which unfortunately is generally not the case in small-scale fisheries.

When participatory methods are difficult to implement or when the outputs are uncertain, national socioeconomic statistics (hereafter referred to as population censuses) and other sources of large-scale, non-fishery-related, information, may represent a key contribution to fisheries spatial pattern assessments. In the same way that taxonomic or environmental surrogates are used to depict the spatial patterns of other – unknown – aspects of biodiversity (Grantham et al. 2010), socioeconomic approaches based on social surrogates can help to fill the lack of fisheries data in small-scale fisheries. Previous studies have used proxies based on distance to fishing ports or accessibility points (Mazor et al. 2014), population density (Ban et al. 2009) or number of boats (Sala 2002; Stewart et al. 2010) to predict the spatial allocation of the fishing effort. However, relying only on such fairly coarse proxies for place-based management purposes can be misleading as it assumes that fishers' spatial behavior is random and only driven by the distance to their place of departure (e.g., port, settlement, accessibility point), which is unlikely to be the case in most contexts (Daw 2008; Metcalfe et al. 2016).

Here, we propose a standardized but flexible approach that addresses the difficulties of obtaining accurate spatial fishing effort allocation data in small-scale fisheries, by linking participatory and socioeconomic approaches to model fine-scale fishing effort distribution. The approach requires the combination of fine-scale representation of fishers' spatial preference for fishing grounds (estimated through a participatory approach) with fishing capacity (estimated through a socioeconomic approach). We tested our integrated approach in the context of the small-scale coral reef fishery of Moorea, French Polynesia, which shares a number of important features with an array of other coral reef small-scale fisheries (Leenhardt et al. 2016).

Methods

Theoretical approach for mapping relative fishing effort

Spatial fishing effort allocation is considered here at the fishery level in terms of overall patterns of distribution. It is analyzed considering two components, namely (i) the fishing suitability and (ii) the fishing capacity (Fig 1). Here, fishing suitability refers to the suitability of fishing grounds. It can be represented spatially using quantitative participatory approaches involving direct or indirect representation of fishers' spatial preference, depending on the ability of practitioners to engage fishers in the participatory process. Fishing capacity designates the overall ability of the fishery to extract resources in a given area. Socioeconomic approaches can provide large-scale and continuous estimations of this aspect of fishing effort, but remain too coarse to be used at local scales. The rationale of this approach is therefore to combine the in-depth knowledge from the fishing suitability analysis and the broader scale fishing capacity analysis as a way to scale the coverage of participatory methods to determines where the fishing effort concentrates.

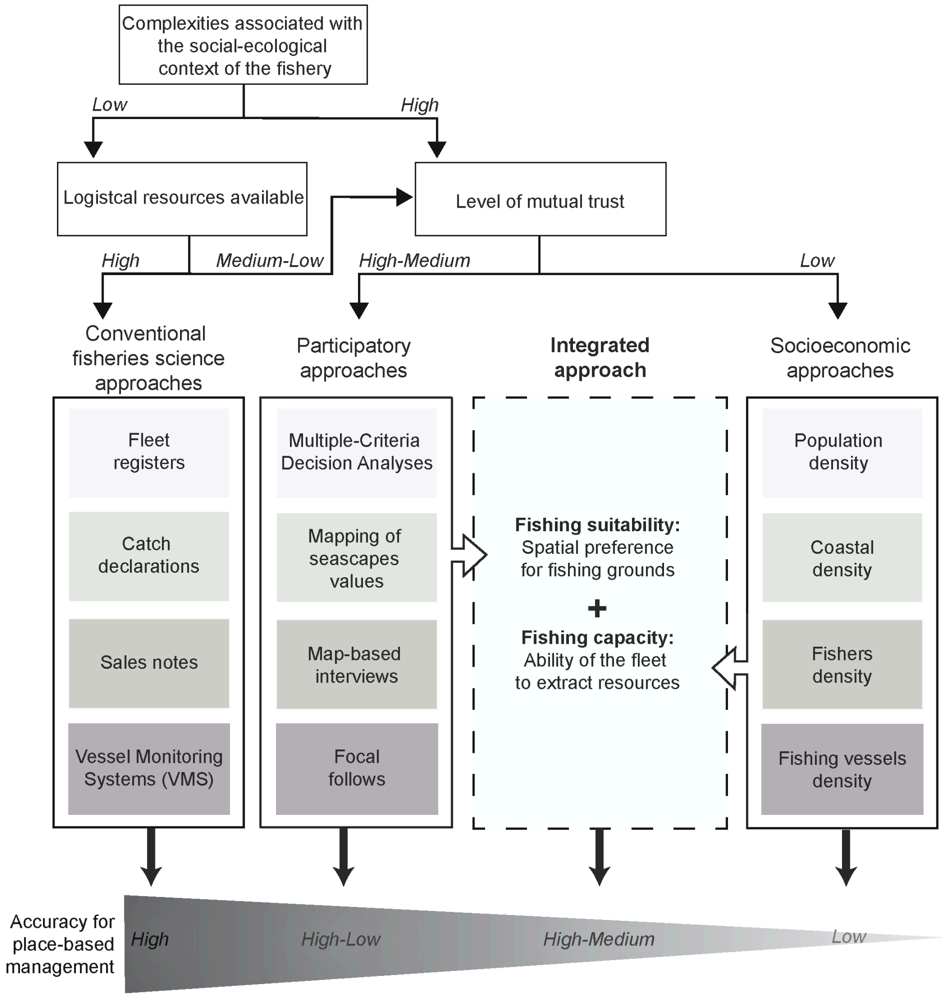

Figure 1. Conceptual flowchart for selecting the best approach to map fishing effort according to availability of three critical factors to be considered by practitioners, namely the complexity of the social-ecological context, the availability of human and financial resources and the degree of cooperation possible with local fishers (i.e., mutual trust level). Techniques commonly used in each type of approach are indicated in boxes. The accuracy of each approach for place-based management (i.e., reliability of the gathered information, level of accuracy/resolution achieved and add-on information gathered during data collection) is provided. Although providing the most accurate estimates of fishing effort, fisheries approaches are unlikely to work in most small-scale fisheries due to the inherent complexity of the social-ecological context and the recurrent lack of logistical resources. Depending on the degree of participants' engagement in the participatory process, information gathered through participatory approaches can be either highly (e.g., using self-reporting diaries and map-based interviews) or moderately (e.g., collective mapping of seascapes values and weightings of spatially-explicit criteria through Multiple-Criteria Decision Analysis) relevant for place-based management. Socioeconomic approaches rely on the extrapolation of social surrogates such as total or coastal population density, fisher or vessel density and may therefore fail to represent fine-scale patterns of the fishing effort (i.e., low accuracy for place-based management). The approach we present here proposes to combine the ability of participatory approaches to map fishers' spatial preference (i.e., fishing suitability) with the power of socioeconomic approaches to estimate the fishery's ability to extract resources (i.e., fishing capacity) and create fine-scale information on the spatial distribution of the fishing effort.

Mapping fishing suitability

A wide array of participatory approaches have successfully described fishers' spatial preferences at high resolution through a variety of direct quantitative mapping techniques (Scholz et al. 2011; Yates & Schoeman 2013). However, such direct mapping approaches require access to and a high degree of cooperation with local fishing communities and hence rely on deeply rooted, and notoriously hard to control, factors such as historical (dis)trust between scientists and fishers, organizational capacity of the fishers and accuracy of fishers' answers. They are also difficult (and expensive) to scale in coverage and enable statistical generalization to the overall fishery.

In contexts where direct participatory mapping methods are difficult to conduct, indirect approaches that quantify the relative importance (weight) of criteria involved in fishing ground selection (e.g., habitat, depth or marine traffic activity) can facilitate the mapping of fishing suitability. Mapping spatially-explicit criteria can be achieved in many ways, depending on the criteria considered, the logistical resources and biophysical context. For instance, acoustic systems can yield high-resolution images of the seabed but are costly and not suitable for large areas (Kenny et al. 2003; Di Maida et al. 2011). Remote sensing has proven accurate and cost-effective for mapping habitat-related criteria (e.g., geomorphologic zones and substrate types) (Knudby et al. 2007). Finally, other methods requiring less technologies are suitable to spatially represent cost-related criteria (e.g., distance to nearest port) (Kavadas et al. 2015). It also requires appropriate methods to consider uncertainties and multiple value judgments at stake in the decision-making process. A variety of Multiple-Criteria Decision Analysis (MCDA) methods has been developed for solving multiple-criteria decision-making problems and computing criteria weights. Although criteria weights can be directly assigned by the decision-maker (e.g., weighted ranking method), it is acknowledged that using ranks to elicit scores through mathematical formulas is more reliable because decision-makers are more confident about the ranks of some criteria than their weights (Figueira et al. 2005). Well-accepted weighting methods based on rankings include the ratio method (Winterfeldt & Edwards 1986), the Analytic Hierarchy Process (AHP; (Saaty 2008))and the Measuring Attractiveness by a Categorical Based Evaluation Technique (MACBETH; (Bana e Costa & Vansnick 1994)) (for comparison of these methods see (Bell et al. 2001; Delle Site & Filippi 2009)).

Mapping fishing capacity

The literature on systematic marine conservation planning (which is closely linked to fisheries management and marine spatial planning literatures (Weeks et al. 2014)) provides a classification through which various levels of fishing capacity resolution can be structured hierarchically by progressively adding more detailed information (Ban et al. 2009; Weeks et al. 2010). In its simplest form, fishing capacity can be estimated and mapped as the population density extrapolated onto the water to a given distance of influence using density decay. This straightforward approach relies on the implicit assumption that the proportion of fishers is evenly distributed within the study area, which is not the case in most contexts (Daw et al. 2011b). One way of gaining accuracy is to restrict the extrapolation process to the coastal population when fishers live close to the shore. Nevertheless, the latter approach still assumes that the fisher population is proportionately spread within the overall population. A step forward is hence to consider the number of fishers based on socioeconomic characteristics (i.e., using population censuses, which are collected from the entire population). In the case where such information is available, fishers may be identified based on their principal and secondary declared livelihood activities. Once fishers are identified and located, fishing capacity can be estimated by (linear or other) distance function of the estimated number of fishers to home ports, accessibility points or markets rather that the entire coastline. An even finer resolution can be added by also integrating boat ownership. Fishing capacity can thus be approximated as a function of fishing vessels density within a radius that depends on the type of fishing vessel.

Mapping relative fishing effort

Fishing suitability determines where the fishing capacity is distributed. Hence, fishing suitability can be used as a weighting factor of the fishing capacity (or its transformation) to create a map of relative fishing effort.

Application to a case study

Moorea's coral reef fishery

We applied this approach to assess the small-scale fin-fish fishery of Moorea island, French Polynesia (Fig 2), which is acknowledged as very challenging to assess (Leenhardt et al. 2016). More than 3/4 of Moorea's land area consists of uninhabitable volcanic peaks. As a consequence, the 17,000 inhabitants are mainly concentrated along a coastline of just over 60 km long and ancient villages occupying small valleys (Insee-ISPF 2012). This particular arrangement is probably an important factor explaining why the marine environment and its use, mostly fishing, are still strongly embedded in the livelihood and lifestyle of the local population despite a recent switch from a rural to an urban economy (due to the proximity to the main island Tahiti and establishment of several hotels; (Féral 2013)). 23% of the adult population still derives some or all of its subsistence and/or income from marine resources, with 35% of households engaged in a fishery-related activity (Insee-ISPF 2012). Consumption surveys conducted in 136 households have highlighted the critical importance of unreported catches due to self-consumption and shares among family or other village members into the total catches (Yonger 2002). Catches that still go through conventional sales channels remain hard to assess due to the absence of a market on the island. Instead, fishes are sold by the roadside, generally close to fishers' houses, making the entire coastline a potential landing/selling area. Such ubiquitous, diffuse, and atypical features make landing surveys and direct observation unsuitable to provide reliable information regarding spatial patterns of the fishery (Leenhardt et al. 2016).

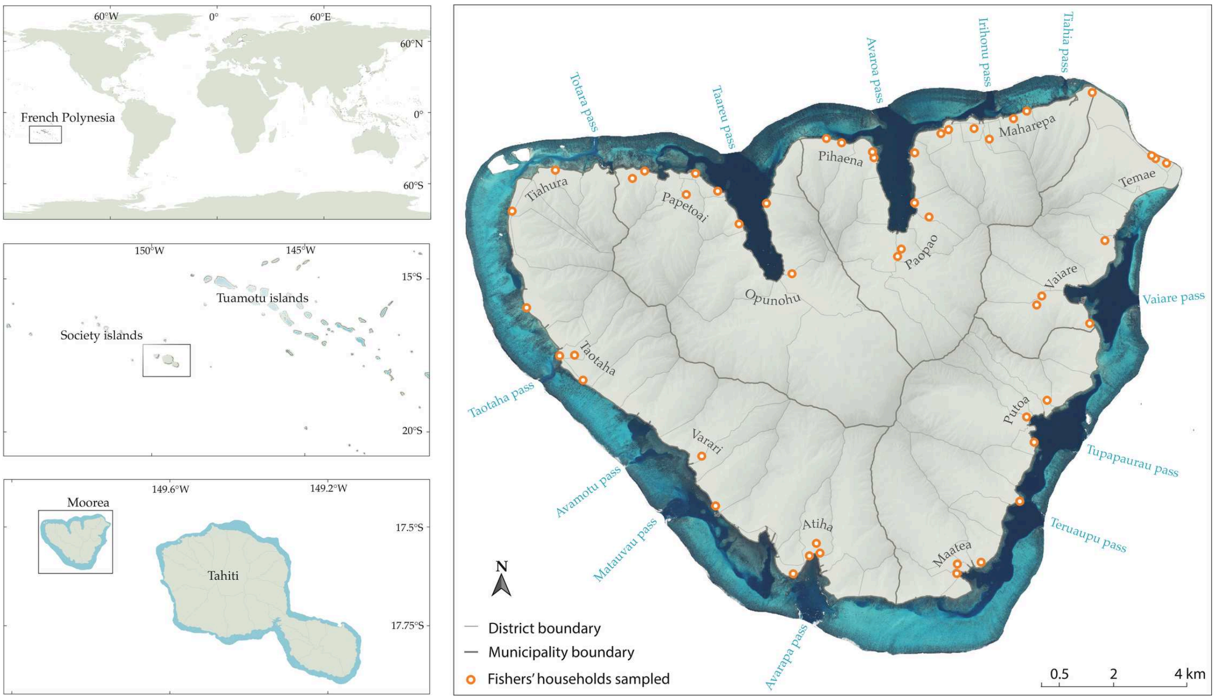

Figure 2. Map of Moorea Island, French Polynesia. Orange circles represent the location of households surveyed to quantify criteria and sub-criteria weights. Thick lines denote municipality boundaries and thin lines district boundaries. Key place names are indicated either in blue (reef passages) or in black (villages).

A spatially-explicit management plan (*Plan de Gestion de l'Espace Maritime, PGEM*), including a network of eight permanent Marine Protected Areas (MPAs), was officially established in 2004, although its actual implementation was only achieved in 2007. During the ten year long planning process prior to the implementation of the management plan, fishing grounds and fishing effort were not properly considered due to the absence of adequate data during this period (Walker 2001). As a consequence, a significant number of fishers still question the legitimacy of this management plan today, which may – at least partly – explain the weak compliance of users with fishing regulations (Gaspar & Bambridge 2008) and the unclear effect of MPAs on marine resources (Chapter I). In addition, previous planning processes have created distrust among stakeholders and widespread participatory approaches, such as direct mapping of fishing grounds or self-reporting diaries, are unlikely to succeed (but see (Walker & Robinson 2009) for application of a direct participatory approach to map general fishing areas at three locations around Moorea).

Mapping fishing suitability

In order to map fishing suitability of the overall fishery, we applied an indirect mapping method following five main steps. First, we first identified biophysical and cost-related criteria (and sub-criteria) considered by local fishers when making decisions about where to go fishing in the long run based on a literature search, six key informant interviews and five pilot interviews conducted with fishers. They included Distance to the shore (0-400m, 400-1,000m and >1,000m), Depth (0-3m, 3-8m and >8m), Distance to the closest pass (0-250m, 250-1,000m and >1,000), Slope of seafloor (flat, medium and steep), Substrate of seafloor (coral, algae and sediment). Attention was paid not to include too many criteria and sub-criteria, in order to avoid confusion and reducing lack of focus among interviewees.

Second, we selected survey participants by implementing a non-stratified random sampling from among all households of the island, in the aim of gathering the overall fishers' spatial preferences and avoiding social-, cultural- and gear-related bias. Overall, 51 coral reef fishers (i.e., household members present at the time of our visit that declared to have fished over the last two weeks) who accepted to participate to the survey (acceptance rate = 96.2%) had a mean age of 32.7 years (min=12, max=60, SD=9.76), were native of French Polynesia (98%) and mostly male (94.6%).

Third, we conducted a survey that included a ranking exercise based on the Analytic Hierarchy Process (AHP) decision-making methodology (Saaty 2008) to measure sub-criteria weights identified in step 1 (Appendix B; S1 File). AHP uses a multi-level hierarchical structure of criteria and sub-criteria that are pairwise compared by participants (here fishers) to derive criteria weights and estimate the consistency of the judgments (called consistency ratio). The difference in importance between each pair of criteria and sub-criteria was indicated by fishers on a 4-point scale (1 = same, 2 = slightly higher preference, 3 = higher preference, 4 = major preference) for which we assigned scores with intervals of three to fit with the 10-points scale used in the AHP methodology (Saaty 2008).

Fourth, we calculated an aggregated weight for each sub-criterion as its average weight among fishers, weighted by the judgment consistency ratio of their response.

Finally, we represented aggregated sub-criteria weights spatially using high resolution maps of each criterion and sub-criterion. Space borne imagery was used along with geolocated acoustic depth measurements and seafloor data, to predict and map Depth, Slope of seafloor and Substrate of seafloor criteria and sub-criteria (Collin & Hench 2015). Areas where the models did not perform satisfactorily (i.e., where depth exceeded 12m or where the water was turbid) were discarded from the analysis. Using spatial processing, we derived the Distance to pass and Distance to shore criteria and sub-criteria based on coastline and reef crest maps extracted from the same satellite imagery. For each criteria map, every 5 x 5m cell was assigned its corresponding aggregated sub-criteria weight. We then summed all criteria maps to obtain the fishing suitability map (FS), whose cells' value potentially ranged from a low of 0 to a high of 1. Additional information on the creation of the spatial data can be found in Appendix B S2 File.

Mapping fishing capacity

In Moorea, most coral reef fishers start their fishing trip from the closer access point from their home, but there is no reliable estimate of their number, the intensity of their fishing activity and their location around the island. The number of households and their dependence on marine resources (see below) were thus used as a proxy of the fishing capacity. A dependence on marine resources index, *D*, was calculated for each district (n=69, Fig 2) from population census data (Insee-ISPF 2007) and based on established protocols (Campaner 2010; Cinner et al. 2012a) (Eq. 1):

$$D = \frac{F}{F+NF} \times \frac{N}{F+NF} \times \frac{U}{N} \qquad (1)$$

where *F* is the number of households having at least one member who declared fishing as its primary or secondary livelihood activity; *NF* is the number of households having at least one member who declared non-fishery-related occupation as its primary or secondary livelihood activity; *U* is the number of households having at least one member having no activity, whether primary or secondary; and *N* is the total number of households. The first term in Eq. 1 captures the ratio of fishery-related activities to the overall livelihood activities within the district. The second term captures the extent to which households engaged in fisheries also engage in non-fishery livelihood activities. The third term captures the degree to which livelihood activities determine the subsistence of the other – inactive – members. The second and third terms thus decrease the level of dependence when many households are engaged in both occupational categories, and when inactive people represent a small portion of the district population, respectively. Although it could be argued that the absence of inactive household members in a district (U=0) may lead to null dependence on marine resources (D=0) – even in households only engaged in fishing – such configuration does not exist in our data set.

We then mapped the households' dependence on marine resources by locating each household and assigning them their corresponding district-level level of dependence on marine resources (D). Finally, household density, weighted by the level of dependence on marine resources, was extrapolated onto the lagoon using linear decay to map fishing capacity (FC). The underlying assumptions are that (i) fishing capacity is high in areas with high household density and dependence on marine resources, and (ii) coral reef fishers in Moorea all travel at the same maximum distance from their home. This assumption is reasonable given that fishing trips never exceeded a couple of hours including travel time from home to sea access point. Based on responses given during preliminary interviews with six key informants, we fixed the distance at which fishers could fish at 2 km around individual households.

Mapping relative fishing effort

The predicted fishing effort (FE) was finally calculated in each cell according to the following formula:

$$FE_c = FC_c * FS_c \quad (2)$$

where FE_c, FC_c and FS_c are respectively the predicted fishing effort, fishing capacity and fishing suitability at the 5 x 5m cell c.

All statistical and spatial analyses were implemented in the R statistical software version 3.2.2 (R Core Team 2014) using the {rgdal} package (Bivand et al. 2016).

Ethics statement

We followed the Code of Ethics adopted by CRIOBE and validated by the Ethics Committee of the CNRS. Accordingly, fishers involved in the study were informed about the purpose of the questionnaire as well as data use and diffusion. We obtained verbal consent from participants prior to conducting surveys. If provided, we also recorded personal contact information to facilitate restitution of results to participants. Population census data were provided through a memorandum of understanding that CRIOBE has with the *Institut des Statistiques de la Polynésie française* (ISPF) and adhered to the CRIOBE Code of Ethics for research involving people."

Results

General patterns regarding fishing ground selection were successfully described despite the great diversity of fishing practices and fishers' profiles. The three sub-criteria preferred by fishers (i.e., greater weights) are coral substrate (0.17 ± 0.03 95% CI), short distance to reef passages (0.13 ± 0.04 95% CI) and steep bottoms (0.09 ± 0.03 95% CI) (Table 1). Because they often combine the highest ranked sub-criteria, reef passages' edges appear as one of the most suitable fishing ground, despite a general exposure to strong current and high exposure to waves (Fig 3a). Fishers ranked higher sub-criteria generally associated with high fish abundance (e.g., coral substrate, steep rocks) and low travel cost (short distance to the shore significantly ranked higher than large distance). Another important type of fishing area for fishers includes fringing reef covered by hard corals (Fig 3a). Households highly dependent on marine resources for food and/or livelihoods are spread around the island (Fig 3b). The dependency on marine resources island-wide is variable among the 69 districts (Fig 3b) with district-level levels of dependency varying from 0 (Temae, where no household had members engaged in fishing) up to 0.23 (Maatea), 0.25 (Taotaha) and 0.28 (Putoa). Lagoon areas located in front of most dependent populated areas have higher levels of fishing capacity, while remote areas display low levels of fishing capacity (Fig 3c).

Table 1: Averaged sub-criteria weights (+/- 95%CI) obtained from AHP exercises performed with local fishers to estimate their preference for fishing grounds. The three sub-criteria ranked higher are indicated in bold.

Criteria	Sub-criteria	Weight +/- 95%CI
Substrate of seafloor	**Coral**	**0.166 +/- 0.028**
	Sediment	0.074 +/- 0.023
	Algae	0.029 +/- 0.004
Distance to reef passage	**0-250m**	**0.126 +/- 0.038**
	250-1000m	0.041 +/- 0.01
	>1000m	0.026 +/- 0.003
Slope of seafloor	**High**	**0.087 +/- 0.026**
	Low	0.057 +/- 0.014
	Medium	0.039 +/- 0.01
Distance to shore	0-400m	0.067 +/- 0.029
	400-1000m	0.036 +/- 0.01
	>1000m	0.027 +/- 0.009
Depth	3-8m	0.066 +/- 0.014
	>8m	0.038 +/- 0.014
	0-3m	0.038 +/- 0.012

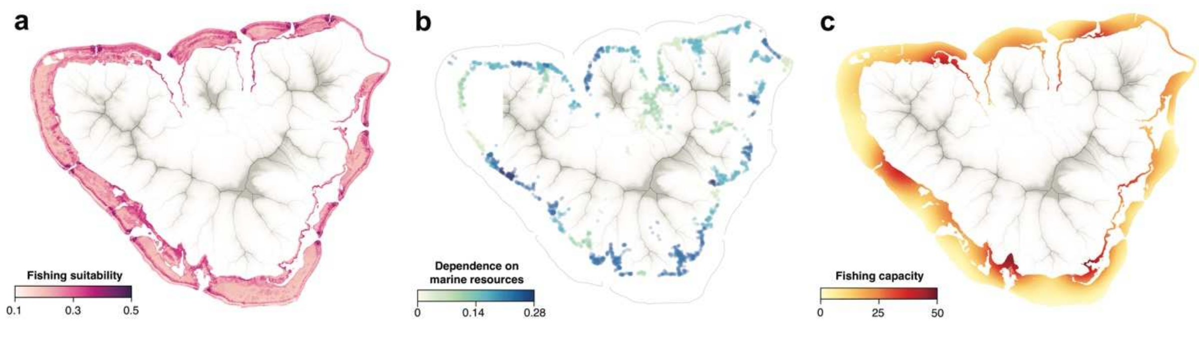

Figure 3: Information used to calculate the two components of the predicted fishing effort: fishing capacity and fishing suitability. (a) Spatial representation of sub-criteria weights measured using the AHP methodology. See Appendix B S2 File for additional information on the approach used to map sub-criteria weights. (b) Households' dependence on marine resources. Dots represent households and colors indicate their district-level level of dependence. (c) Fishing capacity, calculated using the cumulated distance to households within a 2-km radius, weighted by their level of dependence on marine resources.

On average, predicted fishing effort decreased with distance to coast (mostly due to fine-scale, within-reef habitat heterogeneity) and was variable along the coast (mostly because of varying household density and dependency on marine resources) (Fig 4a). Near-shore areas and reef passages generally displayed relatively higher levels of predicted fishing effort, whilst lower relative levels remained far from the shore (outer reef and remote lagoon areas) and in shallow, flat and less complex areas. Highest levels of predicted fishing effort were found in Taotaha, Papetoai, Maharepa, Atiha (Fig 4b-e) and on the fringing reef of Maatea. Varari also displayed high levels of predicted fishing effort, while the South point and the North-West side of the lagoon (Tiahura) were among the least exposed to fishing pressure according to our model.

Discussion

Failure of resource management strategies to achieve the triple bottom line of social, environmental and economic sustainability is often related to the lack of reliable information on spatial patterns of fishing effort (Berkes et al. 2001) and the weak ability (or the lack of opportunity) of key stakeholders (e.g., resource users) to affect the outcome of the decision-making process (Lynham et al. 2017). The approach proposed here tackle these issues by integrating participatory approaches with socioeconomic approaches, thereby scaling in coverage the spatial distribution of fishing effort in a cost-effective manner.

Associating various stakeholders (i.e., policymakers, managers, scientists and fishers) in the decision making requires a reflexive analysis of the context of production and use of data on fishers' spatial preference. Direct participatory approaches to map fishing suitability (the first component of the fishing effort) are only well suited when trust among stakeholders is high and power relations balanced, because of three main challenges. The first is ethical, since the consequences of transferring very specific information on fishing areas beyond traditional boundaries (i.e., to science and management) are difficult to control and can be detrimental to resource users (Maurstad 2002). The second refers to the modalities of participation: which people have spoken and what knowledge and techniques are then revealed while others may remain hidden? The third is linked to the formatting of knowledge into databases that can lead local populations to be dispossessed of their knowledge in the profit of development or management ideologies (Agrawal 2002; Maurstad 2002).

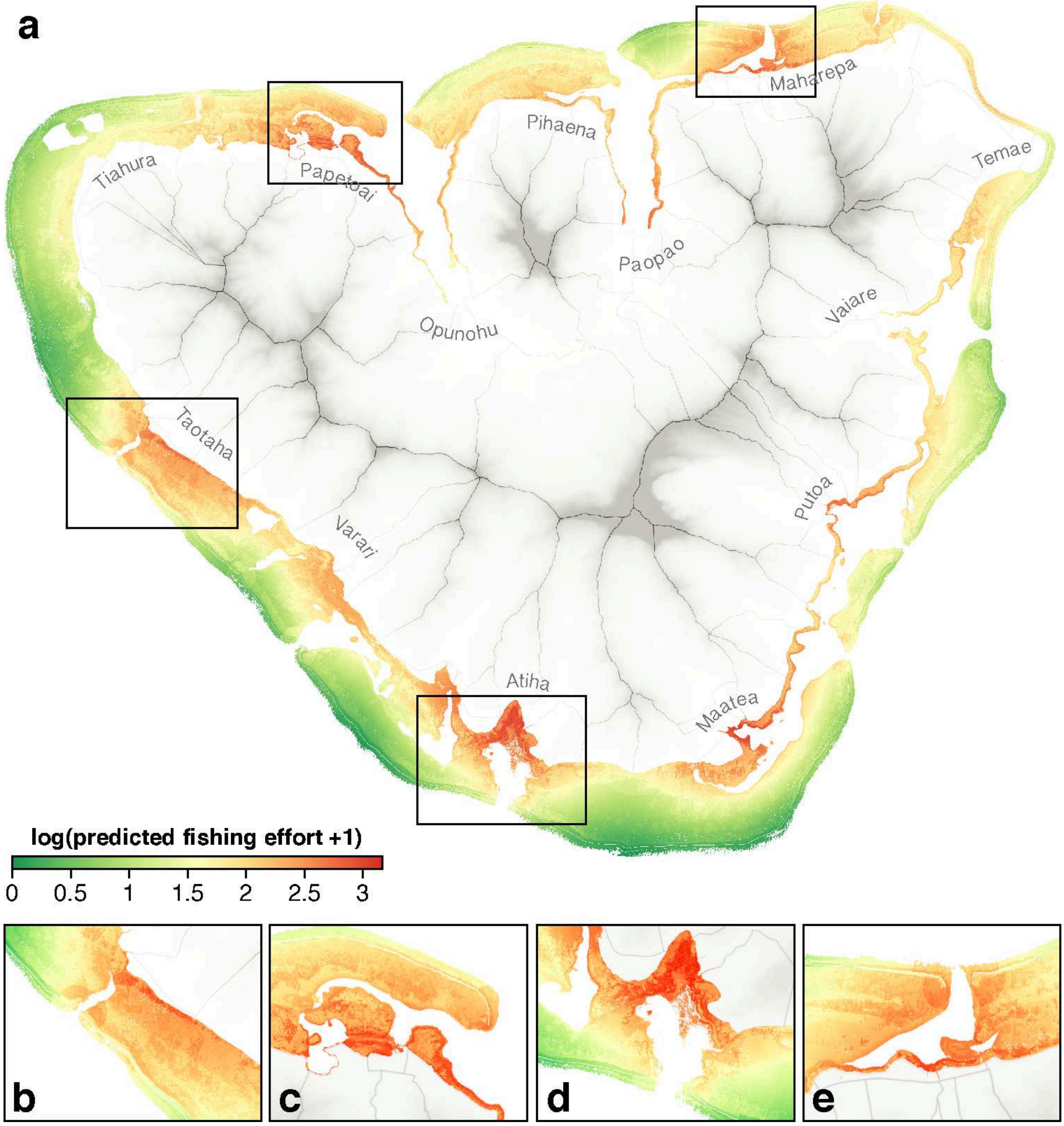

Fig 4. Spatial variation in predicted fishing effort. (a) Island-wide analysis highlights a high level of fine-scale spatial variation in predicted fishing effort. Highly exposed areas include (b) Taotaha, (c) Papetoai, (d) Atiha and (e) Maharepa villages.

With this is mind, and given the complex context in Moorea, we adopted an indirect participatory approach to quantify fishers' preference for spatially-explicit criteria. The set of pairwise qualitative comparisons developed in the AHP (Saaty 2008) **enabled us to overcome trade-offs made by fishers when making decisions about where to go fishing.** Importantly, this process is advantageous as it does not require participants to directly identify specific places of interest, as is the case with direct participatory mapping methods. Rather, respondents are only asked to broadly compare preferred criteria and sub-criteria **in relation to each other**. This difference in approaches is fundamental because it enables respondents to feel more secure, protecting them from sharing highly sensitive information with strangers (interviewers) and keeping specific fishing grounds a secret. It also avoids biased information related to illegal behavior like poaching. **To our knowledge, an AHP analysis has not yet been used directly with fishers to identify key factors driving the spatial preference of fishers, although a related approach was recently applied by Kavadas et al.** (Kavadas et al. 2015) in their study of the Greek artisanal fishing fleet where expert judgment was used.

Based on our experience we suggest that AHP is likely to be a good methodology to investigate spatial preference with fishers, particularly when the initial level of mutual trust is low, when only coarse weight estimation of the criteria that characterize fishing areas are needed and when the required sample would be large (e.g., when fishing practices are highly diverse). In our case, performing such a simple and non-intrusive exercise helped us to establish dialogue, trust and cooperation with interviewed fishers. We therefore believe that this method can contribute to complementing and facilitating the other – more sensitive and difficult to implement, but also more accurate – direct participatory mapping approaches. Besides, contrary to more demanding participatory approaches (Brenier et al. 2013), the AHP approach is simple and standardized enough to be taken up entirely by the local stakeholders (fishers, policymakers and/or managers) and enable continuous, regular and long-term monitoring of the predicted fishing effort. This may in turn enhance participation and increase sample size, ultimately improving spatial preference estimates and allowing the investigation of within-community differences in fishing ground features (e.g., spatial segregation per gender, gear or other factors) and temporal assessments (i.e., seasonal variations) in highly complex fisheries settings.

General patterns regarding fishing ground selection were successfully described despite the great diversity of fishing practices and fishers' profiles. Exposure to less than favorable conditions (short distance to pass) was ranked second in the overall sub-criteria. Indeed, local fishers have long been aware of the ecological importance of reef passages, which bridge the lagoon and the ocean, are often times important spawning aggregation sites, and may serve an important passage for many fishes that enter the lagoon at dawn to feed during the day and exit at sunset (Galzin 1987a, 1987b). Such features make reef passages of particular interest for experienced fishers (i.e., skilled sailors with good knowledge about currents and lunar cycles).

We obtained fishing capacity (the second component of the fishing effort) through a combination of household density and level of dependence on marine resources. Although aggregated at the district level, areas displaying the highest levels of dependence on marine resources highlighted in this study (Atiha, Maatea, and Taotaha) correspond to the main fishing villages described in the literature (Yonger 2002). Areas displaying the lowest level of fishing capacity were found in Temae, which is known for being highly embedded into salaried employment and tourism activities (Insee-ISPF 2007). Some intermediate configurations were also identified (e.g., Pihaena and Maharepa). A benefit of considering both household density and dependence on marine resources resides in overcoming conventional approaches that constrained fishing capacity estimation to some particular villages, which limits the generalization of results to surrounding areas. Practitioners conducting fishing capacity assessments at the system-scale (e.g., including several villages or a continuous urban area) are often confronted with a lack of fisheries data and may turn to coarse proxies such as population density to estimate the fishing capacity (Ban et al. 2009). The method described here offers a more nuanced view of the fishing capacity, releasing the investigator from major assumptions such as the spatially-homogeneous distribution of fishing households along the coast. Integrating our novel index of dependence on marine resources into household density extrapolation on lagoon waters added a new level of fishing capacity resolution that bridged "coastal population density" and "number of fishers".

In the context of fisheries management, the lack of information on spatial patterns of resource use makes decisions for marine spatial planning more difficult and potentially at odds with the underlying social-ecological configuration of the system. Fishing capacity roughly indicates where resource extraction is likely to be high (or low), and can therefore fail to provide information fine enough to differentiate areas in patchy and heterogeneous configurations. Combined with spatial preferences (i.e., fishing suitability), it provides insights on the extent to which fishers interact with the marine environment. In our case study, for instance, one might expect the South point of the island to be highly exposed to fishing because it is located close to the fishing villages of Maatea and Atiha. However, with the exception of the fringing reef, this part of the lagoon is mainly composed of shallow sandy bottoms, which are generally of low interest for local fishers and thus shows a remarkably low level of predicted fishing effort. Hence, integrating spatially explicit data on fishers' preference for fishing grounds (i.e., fishing suitability) adds a crucial level of accuracy in the appreciation of the fishing effort over space, aligning fishing effort mapping within the local context.

Our approach provides relative fishing effort; it does not provide absolute number of boats or biomass caught. While such absolute estimates are more desirable, in their absence, relative fishing effort maps can be used as a systematic decision support tools to represent resource use or opportunity cost (i.e., the cost of management to fishers) (Pressey & Cowling 2001; Naidoo et al. 2006).

Co-construction of fishing effort maps not only inform marine spatial planning, but can also provide a valuable way to engage stakeholders early and continually in the planning process. Individual or collective restitution of research outputs (maps) to interviewees indeed provides a simple and powerful opportunity to develop proximity between fishers and scientists, to strengthen relationship building, and to foster cooperation for future projects, hence contributing to both stakeholder participation and empowerment (Pomeroy & Douvere 2008). Further, an effective stakeholder engagement process will enable a proactive fine tuning of the marine spatial planning process; it has the potential to enhance the reliability of the maps and to promote their acceptance amongst the population.

Here we presented the first attempt at quantifying and mapping fishing effort around Moorea. The similar distribution of fishing effort inside and outside marine protected areas (Appendix B; S3 File) indicates that opportunity costs have not been considered when they were designed, which may explain low support among the population (although they therefore represent a random subset of Moorea's fishing grounds).

Conclusion

The standardized and flexible approach we developed here can produce baseline spatial patterns of resource use. This baseline information can then be used to establish a dialogue among stakeholders to provide guidance for fisheries management and environmental conservation policies and to initiate standardized temporal assessments of fishing activity when conventional fisheries methods are not suitable. Although we focused on a small-scale fishery operating in a coral reef ecosystem, the portfolio of direct and indirect participatory and socioeconomic approaches we presented makes our fishing suitability capacity mapping framework suitable for other types of fisheries.

Chapter III

Mapping social-ecological vulnerability to inform local decision making

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Chapter III: Mapping social-ecological vulnerability to inform local decision making

Abstract

An overarching challenge of natural resource management and biodiversity conservation is that relationships between human and nature are difficult to integrate into tools that can effectively guide decision-making. Social-ecological vulnerability offers a valuable framework for identifying and understanding important social-ecological linkages, and the implications of dependencies and other feedback loops in the system. Unfortunately, its implementation at local scales has hitherto been limited, due at least in part to the lack of operational tools for spatial representation of social-ecological vulnerability. Here, we develop a method and demonstrate its utility for mapping social-ecological vulnerability using information on human-nature dependencies and ecosystem services at local scales within the context of the small-scale fishery of Moorea, French Polynesia. Our approach produced a spatial analysis that reveals social-ecological vulnerability hotspots that highlight focal areas for management intervention. The results can also inform decisions about where biodiversity conservation strategies are likely to be more effective, and how social impacts from policy decisions can be minimized. This study provides a new perspective on human-nature linkages that can inform efforts to manage for sustainability at local scales. Our approach delivers insights that are distinct from those provided by the emphasis on a single vulnerability component (e.g., exposure), and demonstrates the feasibility and value of operationalizing the social-ecological vulnerability framework for policy, planning and participatory management decisions.

Introduction

People benefit from ecosystems by harvesting food, earning income, gaining protection or deriving social and cultural meaning – all of which are the underpinnings of human well-being (MEA 2005). While deriving elements of well-being from natural systems, humans alter, either directly or indirectly, these ecosystems. Therefore, humans are both dependent upon - and major drivers of - ecosystems and the services they generate (Fischer et al. 2015). Intensification of human activities in many parts of the world has led to growing concerns about the sustainability of ecosystem services, and the consequential implications for human well-being (MEA 2005; Worm et al. 2006). This has driven an increasingly mainstream realization that effective management of human activities must be viewed not just as a goal for biodiversity conservation, but as an essential foundation for food security, community well-being and sustainable development (Naeem et al. 2016).

Effective management that reconciles resource use and conservation of ecosystems is no small challenge (Game et al. 2014; Gelcich & O'Keeffe 2016). Balancing short-term needs with longer-term requirements for sustainability, while managing multiple uses, accounting for legacy issues, and integrating the dynamic and complex relationships between human and nature through space and time, requires a more complex and integrative approach than is normally used in biodiversity conservation and natural resource management.

Conventionally, biodiversity conservation and natural resource management has focused on reducing the pressures that affect ecosystems (Salafsky & Margoluis 1999; Prugh et al. 2010; Burke et al. 2011). In the language of vulnerability (Marshall *et al.* 2010), this is a focus on *exposure*. Understanding exposure is important, but ignoring *sensitivity* means we make the implicit assumption that all entities of a system have equal response to the stressor, which is likely to be incorrect in most cases. Ignoring differences in *sensitivity* (i.e., set of conditions and/or characteristics mediating short-term propensity of a system to be influenced following exposure; Bousquet *et al.,* 2015) can result in an underestimation or overestimation of the *potential impact*, or level of threat. It is also important to consider *adaptive capacity* (i.e., the system's latent ability to implement effective and long-lasting responses to changes by minimizing, coping with, or recovering from the potential impact of a stressor; Cinner *et al.* 2013; Bousquet *et al.* 2015). Therefore, an approach that incorporates sensitivity and adaptive capacity, as well as exposure, enables practitioners and decision-makers to consider key aspects of a system that determine vulnerability to a stressor of interest.

Formalized and promoted by the Intergovernmental Panel on Climate Change (IPCC), the vulnerability framework has proven to be a robust method to understand the key determinants of system responses to external influences, like climate change (Marshall et al. 2010). This conceptual model of vulnerability has also been used to understand risk and response options in both ecological and social systems and has provided the framework for vulnerability assessments at local, regional and global scales (Johnson & Marshall 2007; Acosta-Michlik & Espaldon 2008; Allison et al. 2009; Fazey et al. 2010; Nelson et al. 2010; Cinner et al. 2012a; Foden et al. 2013; Parravicini et al. 2014; Okey et al. 2015; Pandey & Bardsley 2015; Brugère & De Young 2015; Mcleod et al. 2015; Johnson et al. 2016).

The vulnerability model also has potential as a modular framework for representing linkages in social-ecological systems, and thus to examine the implications of dependencies and other feedback loops in the system where ecosystem services are vulnerable to drivers such as climate change and extreme weather events (Marshall et al. 2013). This linked social-ecological vulnerability framework has proven valuable for large scale conservation and adaptation analyses (Cinner et al. 2012a, 2013a). However, the utility of this approach for local-scale management and planning decisions remains undemonstrated. Key challenges for application of the social-ecological vulnerability framework at the local scale include accounting for interdependencies between social and ecological sub-systems, and representing outputs in a comprehensive and understandable way for local planners and managers.

Here, we demonstrate that the linked social-ecological vulnerability framework can be used to characterize and explore the interactions between people and nature at scales relevant to most community decision ns and management interventions. Specifically, we operationalize the linked social-ecological vulnerability model to produce a map of social-ecological vulnerability that will enable planners, resource managers and communities to examine spatial variation in vulnerability, to explore the different sources of vulnerability at a fine spatial scale, and inform local scale decision-making.

Materials and methods

Conceptual model of social-ecological vulnerability

We characterize the ecological and the social components of vulnerability, and provide distinct nomenclature as follows (refer to Fig. 1). *Ecological exposure* refers to the magnitude, frequency, duration and/or extent to which the ecological sub-system is subject to a driver of change. This driver can be biophysical (e.g., climate change) or socioeconomic (e.g., globalization). In the case where the driver of interest is a direct human use (e.g., fishing), ecological exposure may be substantially determined by the strength of reliance on ecosystem services, also termed resource dependency (Marshall et al. 2007). In the context of social vulnerability, this is represented by *social sensitivity*. Ecological sensitivity in cases where the main driver is direct human use is an indication of the propensity of the ecological system to be directly influenced following exposure to that human use. The combination of ecological sensitivity with ecological exposure creates a potential impact that is buffered by *ecological adaptive capacity* through ecological mechanisms. The overall *ecological vulnerability* of the ecological sub-system resulting from the combination of ecological exposure, ecological sensitivity and ecological adaptive capacity determines its (in)ability to provide ecosystem services, which ultimately translates into the social sub-system as *social exposure*. The combination of social exposure and social sensitivity determines the potential impact of an environmental change. Moderated by the *social adaptive capacity* (i.e., the current and future ability of the social system to cope with, minimize, react and/or adapt to the potential loss of ecosystem services), it then translates into *social vulnerability* (i.e., the eventual degree to which the social system is expected to be affected by the loss of ecosystem services).

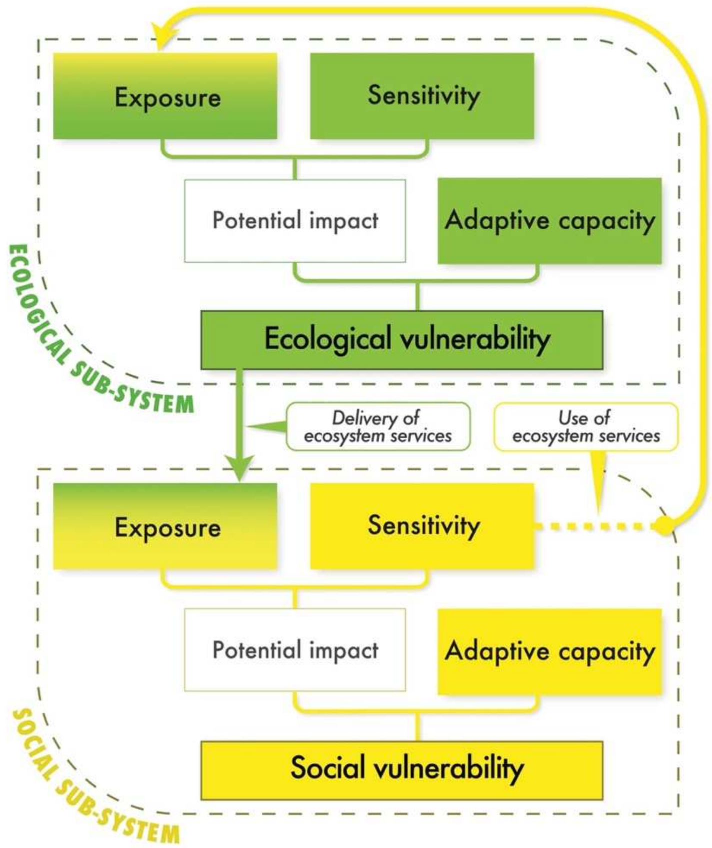

Figure 1: Conceptual social-ecological vulnerability framework used to link human-nature dependencies (adapted from Marshall et al. 2010 and Cinner et al. 2013)**.** The ecological sub-system comprises the entire ecosystem, or a subset of it (e.g., the resource), while the social sub-system includes people interacting with this ecological sub-system (e.g., resource users). Vulnerability, whether ecological or social, is the result of the combination of exposure to a stressor and current inherent feature of the system (sensitivity and adaptive capacity). Ecological vulnerability and social vulnerability are interdependent: the social sub-system affects ecological exposure directly (through ecosystem services use) or indirectly, and ecological vulnerability affects social exposure through ecosystem services delivery.

Operationalizing and applying the framework at a local scale

Overview

We operationalized the social-ecological vulnerability framework from a conceptual model into a quantifiable and spatially-explicit set of indicators for the whole coastal social-ecological system of Moorea, French Polynesia (Leenhardt et al. 2017). Small-scale coral reef fishing provides important livelihood and social benefits to local communities in Moorea, but also has shown signs of being unsustainable over the last decades (Leenhardt et al. 2016). We use Moorea as a case for development of our approach because it has many features common to reef-dependent communities on small island states throughout the tropics, and because of the practicability of compiling the data necessary for the model. While our primary purpose was to demonstrate feasibility of operationalizing social-ecological vulnerability, we expect that many aspects of our approach are transferable to other settings.

Vulnerability is here assessed in the specific context of resource-resource user interactions. Households are considered entities of the social sub-system; and finfish resource assemblages – which represents the vast majority of biomass locally fished (Leenhardt et al. 2016) – are considered as the entity of the ecological sub-system. Ecological vulnerability therefore refers to vulnerability of the resource to human use (fishing), while social vulnerability refers to households' vulnerability to the loss of ecosystem services provided by fishing (mostly provisional and cultural services) (Marshall et al. 2010; Cinner et al. 2013a). Since, in our case, social and ecological vulnerabilities are interdependent – both conceptually and in practice – we refer to the overall results as linked social-ecological vulnerability.

Deriving spatially-explicit components of vulnerability

To address the inherent spatial nature of vulnerability at the local scale, we collected and processed data from 2007 to derive spatially-explicit indicators of the components of social-ecological vulnerability covering every household of the island (social vulnerability) and the entire reef surrounding it at a 5m resolution (ecological vulnerability). All data were collected within the same time window of six months, with no major internal or external driver of change occurring in between.

Ecological exposure: We used a previously published model of spatial patterns of the fishing effort around Moorea (Thiault et al. 2017a) to map ecological exposure. This predicted fishing effort model combines an index of household dependency on marine resource (see method to calculate social sensitivity below) with fine-scale spatial representation of fishers' preference for fishing grounds to predict fishing effort at each reef cell (see Thiault et al. (2017a) for more details).

Ecological sensitivity and adaptive capacity: In the context of this study, ecological sensitivity and ecological adaptive capacity both depend on intrinsic features of the species present in the assemblages that are expected to be affected by, and recover from, fishing. These intrinsic features include natural mortality rate, maximum age, habitat range, annual fecundity or strength of aggregation behavior (Reynolds et al. 2001), which altogether determine the intrinsic growth rate. Hence, we developed an intrinsic resilience index, termed **ecological resilience** ER, as a combined measure of the combination of ecological sensitivity and ecological adaptive capacity:

$$ER_{\mathrm{a}} = \sum_{i=1}^{s_{\mathrm{a}}} \left(1 - v_{\mathrm{i,a}}\right) \times n_{\mathrm{i,a}} \qquad (1)$$

where ER_{a} is the ecological resilience of the target fish assemblages at the site a, $v_{\mathrm{i,a}}$ is the normalized (scaled between 0 and 1) intrinsic vulnerability to fishing index (Cheung et al. 2005) of the species s in the assemblages at the site a, and $n_{\mathrm{i,a}}$ is the log-transformed and normalized (scaled between 0 and 1) abundance of the species s in the assemblages of site a. Cheung et al.'s index of intrinsic vulnerability has been widely accepted as a suitable indicator of the intrinsic vulnerability (i.e., the opposite of intrinsic ecological resilience) of fish species to fishing and is currently acknowledged as the best estimate of sensitivity and adaptive capacity to fishing to date (Graham et al. 2011). We calculated ecological resilience for 57 sites where fish count data were available and then predicted values at unsampled sites using spatially-explicit predictors extracted from space-borne imagery (depth, slope, coral cover, sediment cover, distance to the shore, distance to the closest pass and rugosity of the seafloor) in boosted regression trees (BRT; Elith et al. 2008) to generate a continuous map of ecological resilience. While the use of fish count data allowed us to depict the actual state of the fish assemblage (determined by ecological characteristics, fishing and other drivers), the fine resolution (5m) of our habitat- and access-related predictive variables enabled us to capture natural spatial variability. The BRT model showed a high predictive performance, explaining 90% of the total deviance using a cross-validated procedure. Therefore, and although temporal variability in fish assemblages (e.g., moon cycles) is not taken into account here, this map represents a snapshot of ecological resilience within Moorea's reefs that can reliably support the decision-support purpose of the final results.

Social exposure: In Moorea, fishing is characterized by short trips (a few hours) in small boats (if any) originating in front of people's houses (Leenhardt et al. 2016), which restricts the potential fishing grounds to areas adjacent to households. We therefore measured the social exposure as the average ecological vulnerability within a buffer area of 2 km radius around households, which is the distance of reference to estimate the potential fishing area in Moorea (Thiault et al. 2017a).

Social sensitivity: Using data from an island-wide survey conducted on the entire population of Moorea (Insee-ISPF 2007), we calculated social sensitivity at the district-level (69 districts with roughly 80 households each) according to the following formula:

$$\mathrm{S} = \frac{\mathrm{F}}{\mathrm{F+NF}} \times \frac{\mathrm{N}}{\mathrm{F+NF}} \times \frac{\mathrm{U}}{\mathrm{N}} \tag{2}$$

where S is the sensitivity metric within a distric, F is the number of households in this district having at least one member who declared fishing as its primary or secondary livelihood activity; NF is the number of households in this district having at least one member who declared non-fishery-related occupation as its primary or secondary livelihood activity; U is the number of households in this district having at least one member having no activity; and N is the total number of households in the district (Thiault et al. 2017a).

Social adaptive capacity: Based on the same population census data, and following practices used in similar contexts (McClanahan et al. 2008a; Cinner et al. 2012a, 2015; Bennett et al. 2014a), we measured eleven indicators that we grouped into five components: (1) spatial mobility, (2) material assets, (3) occupational mobility; (4) attachment; and (5) education. Each indicator was assigned a score between 0 and 1, depending on whether they contributed (1) or not (0) to the overall adaptive capacity. The final social adaptive capacity index was calculated at the district-level from the average of the indicators weighted by their relative importance (measured by a pool of local experts following the analytic hierarchy process (AHP) methodology; Saaty 2008). The full list of indicators and their associated weights is provided in Appendix C; S1 Table 1.

Our set of composite indicators therefore captures key aspects of social-ecological vulnerability in the context of fishing and enables us to focus on major and known drivers of vulnerability. Nevertheless, we acknowledge that other features – such as characteristics of connectivity among reefs (Jones et al. 2009) and practices associated with local ecological knowledge (McMillen et al. 2014) are not included in this first iteration of our model. We have instead focused on known important features of the system for which we had island-wide coverage to ensure we could produce an operational mapping tool that substantially advanced the information available to decision makers. Additional information on the methods used to map components of vulnerability are given in Appendix C.

Mapping vulnerability

Many approaches have been developed to quantify vulnerability. Some of these are effective for retaining discrete variables, such as clustering and multivariate analyses (Sietz et al. 2011; Cinner et al. 2012a; Foden et al. 2013; Kok et al. 2016), which can help identify the specific contributions to vulnerability. However, more complex systems spanning social and ecological domains, require a more composite approach where variables are integrated into indices. This enables spatial analyses and the production of maps that are often foundational to management and decision-making in real-world complex systems such as coral reefs. Therefore, we adopted a commonly-used approach that combines vulnerability components into a single index of vulnerability (Adger & Vincent 2005; Allison et al. 2009; Marshall et al. 2010; Cinner et al. 2012a, 2013a; Hughes et al. 2012; Parravicini et al. 2014; Johnson et al. 2016). To combine the disparate metrics of the vulnerability components we log-transformed data to reduce the effect of rare, extremely high values and rescaled them between 0 (lowest value) and 1 (highest value). We then applied an additive model with equal weighting among components (Allison et al. 2009; Cinner et al. 2012a; Hughes et al. 2012) to calculate and map social and ecological vulnerabilities (Eq. 3-4).

$$EV_{\mathrm{c}} = EE_c - ER_c \tag{3}$$

$$SV_{\mathrm{h}} = SE_h + SS_h - SA_h \tag{4}$$

where EV_c, EE_c and ER_c are ecological vulnerability, ecological exposure and ecological resilience, respectively, at the cell c, and SV_h, SE_h, SS_h, SA_h are the social vulnerability, social exposure, social sensitivity and social adaptive capacity, respectively, of household h. This model assumes that all components have the same importance and add up to determine the overall vulnerability (i.e., components were not weighted). In order to test the robustness of these assumptions, we conducted an uncertainty analysis by investigating the range of possible vulnerability outputs under alternative assumptions (i.e., different transformation types, aggregation formula and unequal weighting scheme). A detailed description of the methods and results is provided in Appendix C.

In addition to maps of linked social-ecological vulnerability, we summarized social and ecological vulnerability in units that were locally relevant. Social vulnerability was summarized at the scale of municipalities (n=5), which is the unit at which most decisions are taken, and ecological vulnerability was summarized per habitat types (n=6), which is a common stratification for resource management.

All statistical and spatial analyses were implemented with R software (R Core Team, 2015). We fitted BRT using the dismo package (Hijmans et al. 2016) and performed other spatial analyses with raster (Hijmans et al. 2015) and rgdal (Bivand et al. 2016) packages.

Results

The uncertainty analysis shows that, in our case, model assumptions (i.e., data transformation, aggregation formula and components' weighting) have limited effect on the spatial patterns of social and ecological vulnerability (Appendix C). It also supported the validity of representing vulnerability as integrated indices for spatial analysis and mapping. Therefore, we present only the measure of vulnerability presented above, distinguishing low, intermediate and high vulnerable areas.

A striking feature of our results in the strong spatial heterogeneity of social-ecological vulnerability in Moorea (Fig. 2a). We found strong variability in social vulnerability among municipalities across the entire island, and among households within municipalities, with a tendency toward unimodal distribution at both scales (Figs. 2 and 3). High levels of social vulnerability are found where low social adaptive capacity overlaps with high social exposure and sensitivity. Highly vulnerable households are spread around the island (Fig. 2), but a significant proportion are concentrated in the municipality of Papeotai (Fig. 3). Areas with lowest social vulnerability are confined to only one district: Temae village, Northern Teavaro (Fig. 2d).

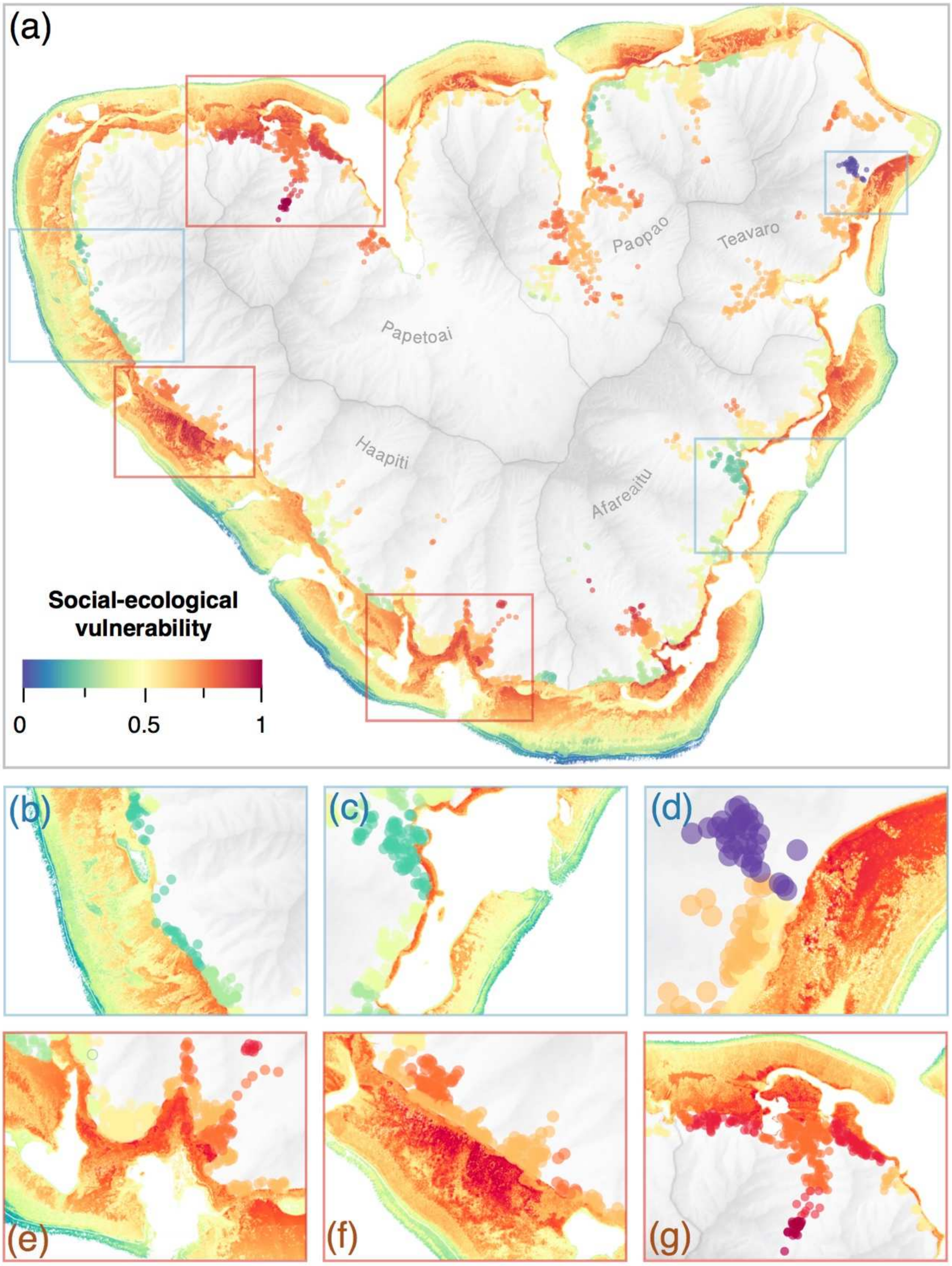

Figure 2: Maps of linked social-ecological vulnerability. (a) Island-wide linked social-ecological vulnerability. Locations with low social-ecological vulnerability include (b) Gendron; Haapiti, (c) Ahi; Afareaitu and (d) Temae; Teavaro. Locations with high social-ecological vulnerability include (e) Papetoai; Papetoai, (f) Taotaha; Haapiti and (g) Atiha; Haapiti. Reef pixels show ecological vulnerability, while households (circles) show social vulnerability.

Ecological vulnerability had a more even distribution. The highest ecological vulnerability was generally located close to shore, where target fish assemblages are both highly exposed to fishing (a key feedback loop) and poorly resilient. As a consequence, fringing habitats (i.e., both reef and sand) display the highest levels of vulnerability. Ecological vulnerability is particularly high in front of Papetoai, Taotaha, Atiha and Maatea villages (Fig. 2e-g). In contrast, lowest ecological vulnerability is found in remote and deep areas dominated by a coral substrate (Fig. 2b-d). Those areas are generally found in passes and on the fore reef (Fig. 3).

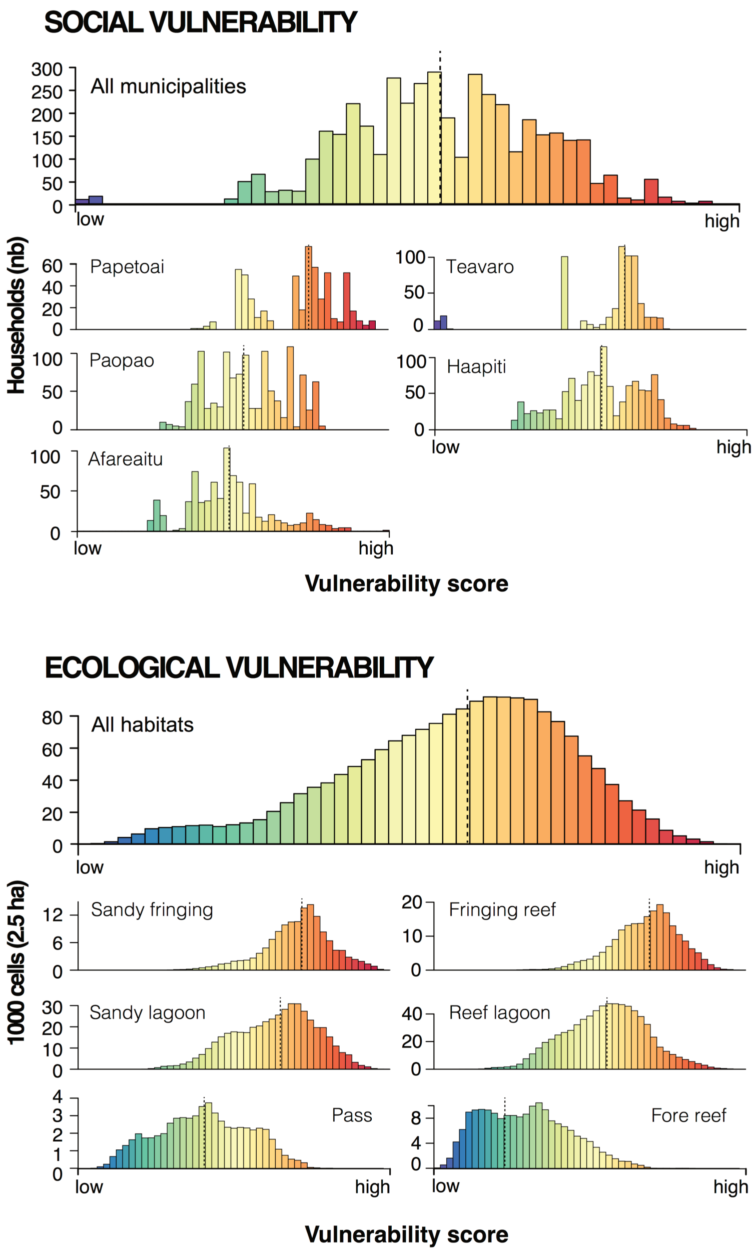

Figure 3: Distribution of social and ecological vulnerability at the system scale and for each municipality (social vulnerability) and habitat (ecological vulnerability). Medians are represented by dotted lines.

Reef areas located in front of socially vulnerable households are generally highly vulnerable (Fig. 2e-g). In some cases, however, such spatial correlation between social and ecological vulnerabilities did not occur. This is for instance the case in Temae, where high levels of ecological vulnerability are located in front of weakly vulnerable households (Fig. 2d), suggesting a decoupling between the social and natural sub-systems.

Discussion

Applying integrative, place-based and theoretically-sound social-ecological systems research is needed to add practical value to management processes and improve their social and ecological outcomes (Leenhardt et al. 2015; Leslie et al. 2015). By exploring human-nature dependencies in Moorea through the lens of social-ecological vulnerability we show how integrative, interdisciplinary research that includes both social and ecological data may be synthesized to yield a better understanding and visualization of linked social-ecological systems. We demonstrated that widely promoted conceptualizations of dynamic interactions in complex systems, such as vulnerability, can also be operationalized to support local-scale decisions using readily-obtained data in complex social-ecological systems such as Moorea. Our operational approach demonstrates the feasibility of developing an integrative synoptic tool that can be used to communicate complex processes and outcomes to decision-makers and stakeholders interested in both ecological and social sustainability.

Importantly, our analysis highlights that assessments based only on one or two key drivers of vulnerability can inform the selection of alternative management decisions. In the case of Moorea, both fringing reefs and reef passes were highly exposed to fishing. If managers and planners base their decision solely on this information (i.e., reducing ecological exposure), they would tend to consider them as both a priority for intervention and would dedicate the same amount of management effort in both habitats. Yet, due to intrinsic features, and potentially lower exposure to other stressors (Leenhardt et al. 2017), we show that the resilience of fish assemblage to fishing is greater in passes than in fringing reefs. Therefore, if exposure and resilience to fishing are considered together (i.e., ecological vulnerability), practitioners would give greater priority to fringing reefs than to passes as they are more vulnerable to fishing.

Many of the data presented in this study have never been mapped and combined together before – especially at such fine scale – and yet, such information provides an important knowledge foundation for marine spatial planning, and can lead to more targeted policy and management interventions. Indeed, the social-ecological vulnerability maps generated in our approach provide insights into where human-nature dependencies are likely to be the most unsustainable (or "social-ecological vulnerability hotspots"). By operationalizing key features of social-ecological vulnerability in spatial terms, we have demonstrated that vulnerability can be a valuable, integrative variable for optimizing marine spatial planning and systematic conservation (e.g., using decision support tool such as Marxan; Ball & Possingham (2000)). By integrating social and ecological vulnerability, our approach also empowers decision-makers to identify opportunities for maximizing conservation outcomes (e.g., protecting most vulnerable areas) while minimizing socio-economic costs. In the case of Moorea, this could be achieved by implementing conservation measures in front of communities with low (e.g., Fig. 2d) rather than high (e.g., Fig. 2g) social vulnerability.

Overlaying social and ecological vulnerabilities in Moorea reveals that social-ecological vulnerability hotspots are mainly observed in three villages: Atiha, Taotaha and Papetoai (Fig. 2e-g). In these villages, combined factors such as low adaptive capacity and high environmental degradation may trigger future social-ecological traps (Cinner 2011) and therefore merit immediate management action. In Moorea, such measures could have the form of banning fishing techniques with high impact (e.g., large nets) during species aggregation events, forbidding night fishing using spear guns to reduce ecological exposure and improve resilience and initiating programs to increase social adaptive capacity: e.g., strengthen social capital, create spaces for learning and sharing, improve material assets and developing sustainable agriculture or tourism activities to reduce social sensitivity. The above are some of the opportunities for action to reduce vulnerability that our tool can reveal when applied to a real decision-making context.

Through the process of mapping social-ecological vulnerability it is possible to evaluate the likely effectiveness of different potential management strategies. For instance, in Moorea, certain management strategies are likely to be more effective in some places than others. Areas where socially vulnerable households are situated adjacent to highly ecological vulnerable areas were typical in Moorea of situations where households have low sensitivity and high adaptive capacity (Appendix C), which make them potentially less likely to be affected by the implementation of ecologically efficient, but socially costly, management tools like marine protected areas (Christie 2004). Temae (Fig. 2d) may therefore represent an opportunity to demonstrate the social-ecological benefits of such tools and induce a pro-conservation behavior that can initiate new pathways for the development of conservation initiatives (Castilla 1994; Gelcich et al. 2010). Although the aggregation limits the ability to determine the causal drivers of vulnerability across our analysis, the components of the aggregated index of vulnerability can also be examined separately to better tailor management interventions (Appendix C). Through our analysis, local decision-makers in Moorea now have the tools to discern the importance and urgency of addressing and mitigating linked social-ecological vulnerability through appropriate interventions if they are to sustain fishing as a foundation of subsistence and culture.

Our model measures overall vulnerability with good accuracy and reasonable robustness (Appendix C). We have explicitly focused on known key drivers of vulnerability in Moorea for this study (i.e., resource degradation arising from overexploitation as the main driver of ecological exposure, and human dependence on fishing as the main driver of social sensitivity; Fig. 1), which had the primary purpose of demonstrating the feasibility of operationalizing social-ecological vulnerability for decision-making in fisheries-dependent coral reef systems such as Moorea. This work lays the foundations for more nuanced analyses that include others drivers (Bennett et al. 2016), or for adaptation for application in different contexts.

This study advances the application of vulnerability and social-ecological system theory to local contexts, using empirical spatial data. The social-ecological vulnerability framework focuses – by design – on simple interactions between people and nature through a single ecosystem service of particular importance to local communities. However, human-nature relationships are often complex (Daw et al. 2016), and relatively simple representation of relationships and feedbacks between ecological and social sub-systems may obscure information that is important to understanding system dynamics, key drivers or opportunities for effective intervention. Our model demonstrates the feasibility and utility of an operational tool for mapping social-ecological resilience. Future work building on this approach can expand the set of drivers and linkages between the various components of social-ecological vulnerability. This would further improve the ability of stakeholders and decision-makers to identify sources of vulnerability and discover a wider range of options for building resilience of coral reefs, and the people who depend upon them.

Mapping social-ecological vulnerability offers an approach for decision-makers to identify which areas are the most (or least) socially and/or ecologically vulnerable – and why – and thus help target management actions. Because of the standardized nature of the social-ecological vulnerability framework, this approach allows direct and fine scale comparisons that enable decision-makers and stakeholders to measure and map change through time. Such information may provide additional indications on the direction (positive or negative) and nature of change (key sources of vulnerability affected) in particular locations, which can inform efforts to identify management priorities (Halpern et al. 2015a) and improve our understanding of the consequences of multiple stressors on human-nature dependencies. This approach and its future iterations are likely to be applicable in a variety of other places where human activity represents both an important threat to the ecosystem and an invaluable source of benefits for local communities. Operational tools like the one developed here provide the basis for an improved understanding of human-nature interactions that can underpin improved biodiversity protection and sustainable development in the world’s coral reef areas.

Chapter IV

A vulnerability-based approach to foster synergies in the management of social-ecological systems

Status: In preparation

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Chapter IV: A vulnerability-based approach to foster synergies in the management of social-ecological systems

Abstract

Biodiversity conservation, social equity and poverty reduction should be tackled together to sustain natural resource use effectively. However, clear operational schemes are needed to maximize synergies through management interventions. Here we present a four-step procedure to identify interventions which can promote synergies, based on (1) identification of key social and/or ecological vulnerability profiles; (2) selection of potential interventions; (3) choice of a portfolio which reduces tradeoffs; (4) confrontation of interventions portfolio with institutional and community partners. We applied this four-step procedure to the coral reef small-scale fishery of Moorea, French Polynesia. As a result of the framework, a portfolio of interventions combining technical measures and temporal closures in the lagoon habitat, permanent closures on the fore reef, and targeted coastal development actions is recommended to maintain ecosystem functioning and foster resilience while advancing human well-being. Managing social-ecological systems through the lens of linked social and ecological vulnerabilities enables to embrace a broader range of potential policy levers that better cope with complexities of social-ecological systems.

Introduction

The recent recognition of the interdependent relationships between people and nature has shifted the focus through which we conceptualize natural resource management to one where people and nature are intimately intertwined in social-ecological systems (Ostrom 2009), pressing practitioners to adopt a holistic view of these resource systems. From a place-based management perspective, effectively confronting the challenges associated with social-ecological systems requires to understand and address the complex interdependencies between social and ecological sub-systems (Game et al. 2014; DeFries & Nagendra 2017).

To date, two main standpoints coexist on the best pathways to reach sustainable use of natural resources. One considers that well-designed ecosystem-level management will maintain ecosystem functioning and hence benefit human communities through increased flows of ecosystem services supply (Leslie & McLeod 2007; Anthony et al. 2015; Selkoe et al. 2015). This “ecological entry-point” thus sees conservation benefiting human communities indirectly as it produces improved socioeconomic benefits through more sustainable yields or other add-on benefits such as protected area tourism arrangements. A second approach that can be qualified of “social entry-point”, places emphasis on broader development challenges where livelihoods improvements and poverty alleviation eventually lead to environmental sustainability through more sustainable practices and lower use intensity (Allison & Ellis 2001; Béné et al. 2007).

These approaches have matured and evolved relatively independently, each providing significant and unique insights about successes and failures of management and policy in a complex social-ecological context. In practice, these different views remain loosely and superficially linked. Here, we advocate for a better integration targeted to the needs of each situation. The question we aim to address is not whether one approach is better than the other, but to what extent these differing conceptual views and the management solutions they advocate can, in fact, be integrated into a united framework.

Cross-fertilization among social and ecological approaches to natural resource management and conservation requires addressing the challenges of interdisciplinary communication to develop operational schemes for thinking systematically about multiple solutions and their consequences on each component of the social-ecological system (Leenhardt et al. 2015). One possibility is to understand and unpack the key dimensions of the resource social-ecological system to identify the leverage points and design an appropriate set of potential actions that create synergies and achieve social-ecological sustainability.

Vulnerability, which integrates external threats (exposure) and internal sources of resilience (sensitivity and adaptive capacity), has recently emerged as a transversal theme in the scientific literature on social-ecological systems (Bousquet et al. 2015). Operationalizing this concept to socioeconomic (Cinner et al. 2012a; Hughes et al. 2012; Marshall et al. 2014; Mora et al. 2015) and ecological (Johnson & Marshall 2007; Foden et al. 2013; Mcleod et al. 2015) issues has enabled to provide practical and context-grounded management and policy recommendations. Since it can used in both social and ecological disciplines, the concept vulnerability also is relevant to bridge the communication gap between those disciplines and assess linked social-ecological vulnerability – that is, integrating social and ecological vulnerabilities in a nested model through the exposure components (Marshall et al. 2010; Cinner et al. 2013a, 2013b; Thiault et al. 2017c). In the context of the commons and local human-nature interdependencies, ecological vulnerability refers to the vulnerability of the resource unit (e.g., water, wild food, landscape) to use by the resource users (e.g., farmers, fishers, tourist-operators), while social vulnerability refers to the vulnerability of the resource users to use-induced resource degradation. Assessing local-scale human-nature interdependencies and common resources through the lens of social-ecological vulnerability hence enables to identify critical components contributing to vulnerability and can therefore reveal key leverage points for improving social-ecological sustainability. Here, we present a novel operational framework that considers the interactions between key dimensions of social and ecological vulnerabilities to derive a portfolio of recommended management interventions.

We apply this framework to a spatially-explicit example from a coral reef small-scale fishery in French Polynesia.

A framework for linking social and ecological vulnerabilities to management interventions

Ecological vulnerability can be operationalized by combining resource's exposure and resilience to use by humans, while social vulnerability can be measured by considering threats to users' well-being caused by resource decline (social exposure) and internal sensitivity (resource dependency) and adaptive capacity (determined by knowledge, infrastructure, flexibility, social capital and agency). Bellow, we present a four-steps framework that draws together several well-established perspectives and management practices, and can accommodate any social-ecological vulnerability configuration.

Step 1: Identify key vulnerability dimension(s) to address

The framework first guides practitioners to independently measure each dimension of social-ecological vulnerability of human-nature dependencies (Marshall et al. 2010) in the specific context of their case study. The dimensions can then be linked to allocate ecological and social sub-systems to one of four quadrants (hereafter referred to as "vulnerability profiles") that inform on the key vulnerability dimensions to address (see Foden *et al.* 2013 for an example of vulnerability profiles in the context of ecological vulnerability to climate change).

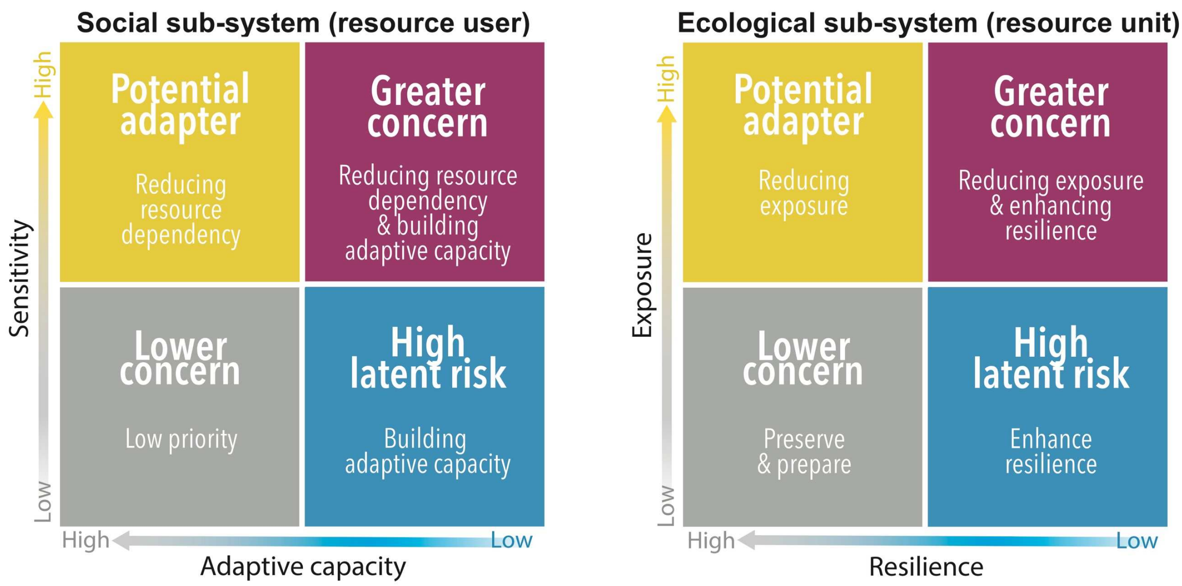

Figure 1: Ecological and social vulnerability profiles and associated management targeted responses. Each ecological or social vulnerability profile is identified through the combinations of exposure and resilience gradients, or sensitivity and adaptive capacity gradients, respectively.

For the ecological sub-system, the resource unit is considered to be of "greater concern" if they qualify as highly exposed and weakly resilient to human use. As first priority, it requires prompt intervention to improve resilience and reduce exposure. "Potential adapter" are highly exposed, but highly resilient, and may therefore be abler to recover from the use under focus, although reducing the use intensity is preferable to avoid undesirable outcomes. Conversely, resource unit at "high latent risk" has both low resilience and exposure, so human use may not be an immediate threat but preventive measures and interventions designed to improve resilience may be applied to ensure assemblages can withstand potential increase in use intensity (and other threats) in the future. Finally, a resource unit of "lower concern" has high resilience and is exposed to low use intensity. Although not of immediate concern, it could become vulnerable if exposed to increased levels use in the future (Fig. 1).

The above illustrates the four profiles of ecological vulnerability, but their equivalent can be identified for the social sub-system. Resource users of "greater concern" can include users with high sensitivity and low adaptive capacity, therefore requiring actions focusing on both adaptive capacity building and resource dependency reduction. Likewise, "potential adapters" include users highly sensitive and at the same time highly adaptable, therefore requiring approaches focusing on decreasing resource dependency. Resource users with low sensitivity and adaptive capacity are at "high latent risk", thus rather requiring livelihood and development approaches aimed at building adaptive capacity. Managers and decision-makers should aim keeping "lower concern" users in this state (Fig. 1).

Step 2: Screening of candidate interventions

Place-based management must first be adapted to the specific features of the sub-system on which the interventions is implemented to avoid unsuccessful or undesired outcomes. For instance, interventions aimed at reducing use intensity such as closures or input/output controls will have simply no effect if ecological exposure (i.e., human use) is already low. Although this statement may seem obvious, many examples related to opportunistic implementation of conservation plans exist – both in marine and terrestrial realms – and have resulted in lower (if any) benefit than expected (Leenhardt et al. 2013; Venter et al. 2017). A worst-case scenario is that the intervention conducts to negative outcomes because some components of the sub-system have been neglected. For instance, within-sector diversification programs to improve social adaptive capacity often overlook the associated increased resource dependency caused by the economic investment in new equipment (Allison & Horemans 2006). Commonly used interventions can have different impacts on ecological and social vulnerability profiles, respectively (Table 1).

Table 1: Typology of interventions to manage resource-resource user dependencies, and implications for social and ecological vulnerability profiles. Symbols indicate the effect of interventions (● rather positive; ⊗ rather negative; ○ no effect) on each vulnerability profile (greater concern; potential adapter; high latent risk; lower concern; see Fig. 1). Interventions a-e: interventions with an ecological entry-point. Interventions 1-8: interventions with a socioeconomic entry point. Shaded boxed indicate direct effects on the sub-system under focus (Step 2). Indirect positive effects of ecologically-focused interventions on the social sub-system arise from more stable or increased ecosystem services delivery, while socially-focused interventions may have an indirect positive effect on the associated ecological sub-system indirectly through reduced or more sustainable use. Conversely, negative indirect effects of ecologically-focused interventions on the social sub-system arise from reduced access to ecosystem services, while socially-focused interventions may have a negative indirect effect on the associated ecological sub-system through increased or less sustainable use (Step 3).

	Type of intervention	Ecological sub-system	Social sub-system	Statement of evidence
Ecological entry-point	**a –** Mitigate other sources of cumulative impact	● ○ ● ●	● ● ● ●	Reducing resource exposure to other threats without affecting the use under concern may improve ecological resilience through better ecosystem functioning
	b – Nature-based solutions and ecological engineering	● ○ ● ○	● ● ● ●	Can play a role in mitigating damage to, or improving, natural habitats and increasing ecosystem service supply. Does not represent an opportunity cost to users.
	c – Permanent closure	● ● ● ●	⊗ ● ⊗ ●	By preventing impact from direct use, permanent closures enhance resilience of the resource unit. However, this comes with important social costs and thus requires high social adaptive capacity to avoid resistance from resource user.
	d – Capacity reduction	● ● ○ ○	● ● ● ●	Well-designed capacity reduction measures allow resource recovering by reducing resource use (temporally, in space, etc.) or access to resource users, which reduces ecological exposure but does not necessarily improve ecological resilience. Opportunity cost depends on the design, regulations and other parameters, but are generally lower than permanent closures.
	e - Output control	● ● ○ ○	⊗ ● ⊗ ●	Reduces the overall amount of resource use but requires pre-existing high level of social adaptive capacity to be effectively implemented. Requires high economic and administrative investment to be implemented effectively.
Social entry-point	**1 –** Livelihood diversification	● ● ○ ○	● ● ● ○	Diversification reduces the risk of livelihood failure by spreading it across more than one income source. Reduces resource dependency and enhances the ability to exit the sector, but depends on community aspirations.
	2 – Behavioural Change	● ● ● ○	● ● ● ○	Improve food security and overall well-being by incentivizing shifting diets toward more nutritive or more resilient resources or more diverse resource sources. Incentivize targeting invasive species can enhance ecological resilience in the case of wildlife harvesting.

3 – Insurance Scheme	●●●○	●●●○	Provide a safety-net in case of environmental perturbation that may causes a loss of ecosystem services supply.
4 - Social capital Building	●●●○	●○●○	Increases the resilience and stability of users, enhances bargaining power in relation to traders/market, improves logistics and access to information, manage risk through collective action and facilitates the use of sustainable practices.
5 – Assets strengthening	○○○○	●○●○	Direct improvement of adaptive capacity components. A prerequisite to the implementation of livelihood diversification strategies.
6 - Market system improvement	●●⊗○	●○●○	Improved values chains, marketing, eco-labeling and good relations with middleman can create higher and/or more stable income for fishers. May induce a shift in fishing practices that may displace fishing effort to previously unexposed areas.
7 – Within-sector diversification	⊗○⊗○	⊗⊗●○	Flexibility to move across use strategies is important for the adaptive capacity but may push part-time users into full-time operations to repay loans and to earn an adequate return on the increased investment. May also increase pressure on resource.
8 – Capacity enhancement	⊗⊗⊗○	⊗⊗●○	Improves use efficiency, increases incomes across users only if the resource remains relatively under-exposed and highly resilient. May increase resource dependency due to economic investment. May induce distal impacts on adjacent resources.

Step 3: Derive interventions portfolio

Implementing actions on one sub-system without carefully considering the potential indirect effects on the associated sub-system can lead to unexpected trade-offs and seriously erode their effectiveness and ultimately give rise to uncertain and potentially damaging ecological and/or social consequences. There is literature on synergic management interventions occurring on associated sub-system. For example, some livelihood diversification initiatives have direct positive consequences on users (i.e., improved adaptive capacity and reduced sensitivity) and indirect positive effects on the resource (i.e., reduced use due to lower resource dependency) (Allison & Ellis 2001). However, there are also indirect negative impacts induced by inappropriate interventions. For example, capacity enhancement often implemented to improve use efficiency and income promote the escalation of use intensity, which can be damaging on weakly resilient resource ("high latent risk" and "greater concern" vulnerability profiles) (Khalilian et al. 2010; Lubchenco et al. 2016). Thus, an understanding of the indirect effect of each management intervention on the linked sub-system facilitates the identification of a set of potential actions that avoids trade-offs and promotes synergies in social-ecological systems. Table 1 summarizes how various types of commonly used interventions implemented on one sub-system may have indirect effects on the associated sub-system, and how this can be interpreted in terms of ecological and social vulnerability profiles.

Step 4: Adapt interventions portfolio to local context

The feasibility and viability of any particular intervention heavily rely on the institutional, cultural and economic specificities in which the resource-resource user system is embedded (Ostrom 1990). For instance, the improved incomes induced by increased tourism activity often brought forward to underline the socioeconomic advantages of protected areas may only be experienced if local communities are willing and able to be involved in this sector (Okazaki 2008). In addition of being rarely completely effective and carrying with it some risk of unexpected consequences, other interventions such as ecological engineering (e.g., reseeding and habitat restoration) are highly costly and may therefore not be a realistic option in many cases. Similarly, insurance schemes, which enable resource users to remove use pressure from resource following environmental perturbation, thus avoiding damaging repercussions on the resource, require the resource users to be clearly identified and structured, which is not necessarily the case in some contexts.

This vulnerability-based framework thus consists of four steps eventually leading to the selection of synergic management interventions for both the ecologic and social sub-systems (Figure 2).

Application to a case study: a small-scale coral reef fishery

The recognition that people and nature are intimately intertwined in social-ecological systems is particularly true for small-scale fisheries (Kittinger et al. 2013). Here, we further explore this vulnerability-based management framework by applying it to coral reefs and the associated small-scale fishery of Moorea, French Polynesia, where fishing activity represents both an important threat to the ecosystem and an invaluable source of benefits for local communities that needs to be addressed with appropriate management interventions. Therefore, we consider linked social-ecological vulnerabilities in the specific context of fish (the resource unit) and fishers (the resource users) interactions. The Moorea case study is also relevant because it is characterized by a broad spectrum of social and ecological configurations that greatly vary through space, potentially allowing to examine all combinations of social and ecological vulnerability profiles.

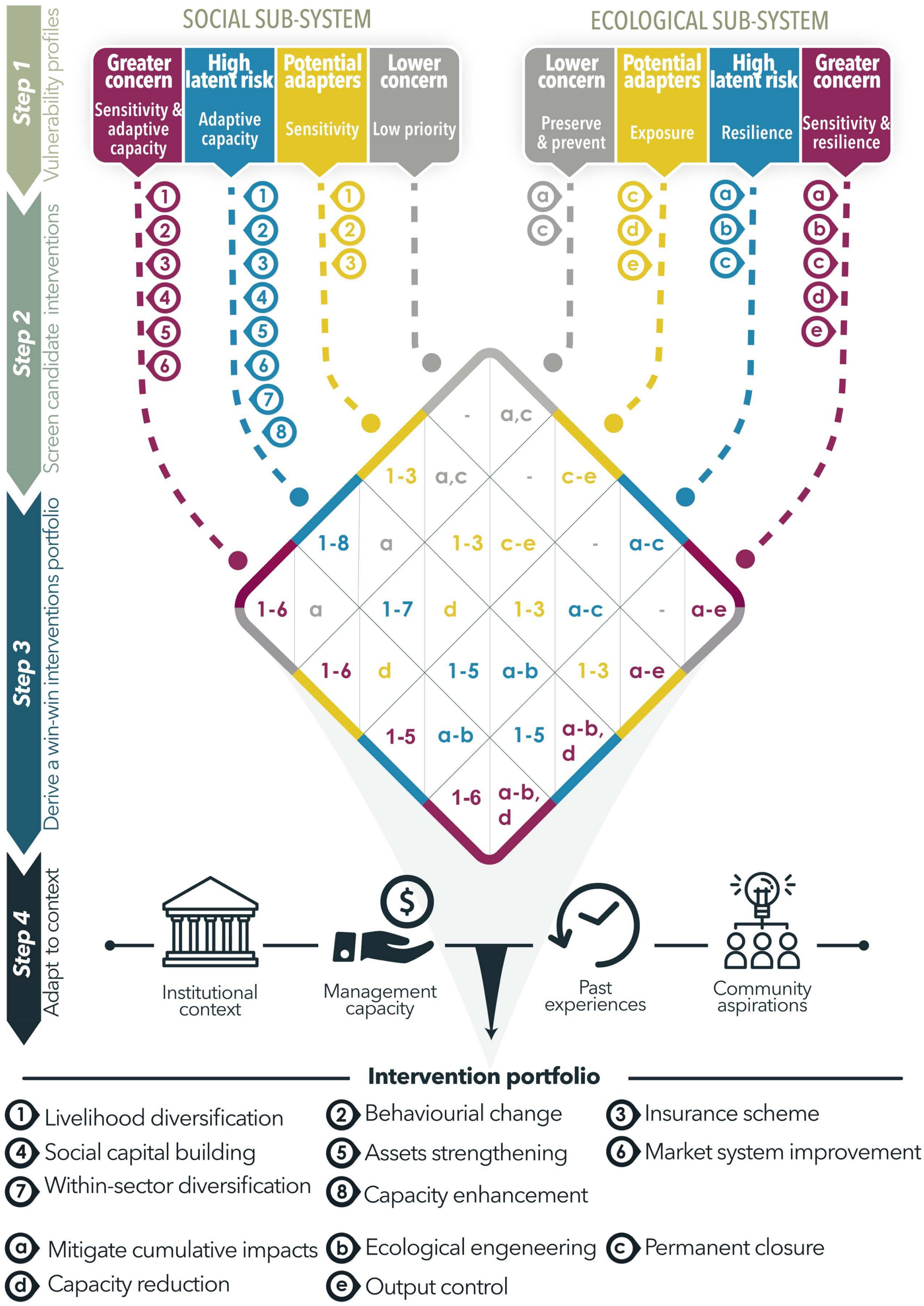

Figure 2: Flowchart illustrating the key steps of the vulnerability-based management framework. Step 1: identify the key vulnerability driver(s) to address through social and ecological vulnerability profiles. Step 2: for each sub-system, determine a set of potential interventions to reduce each sub-system' driver(s) of vulnerability. Step 3: Consider the vulnerability profile of the associated sub-system and determine a win-win portfolio of potential interventions that avoids trade-offs and promotes synergies. Step 4: Ensure the feasibility and durability of the interventions portfolio by putting identified interventions into a broader institutional, logistical, socio-cultural and historical context.

We used fish surveys, remote sensing imagery and a socioeconomic census of households to measure and map each dimension of social-ecological vulnerability at a whole-of-island scale and determine vulnerability profiles (Step 1). A detailed description of the methods is provided in (Thiault et al. 2017c). Briefly, the fish surveys conducted on 57 sites allowed to quantify ecological resilience based on the combination of intrinsic vulnerability to fishing and density of each targeted fish species present in the surveyed assemblage. The formula provided a resilience index (normalized between 0-1) whose spatial distribution was then predicted for every 5 x 5m reef cell using spatially-explicit predictors extracted from space-borne imagery in a boosted regression trees model. The socioeconomic census provided a district-level (n=69, roughly 200 households per district) social adaptive capacity index based on eleven quantitative indicators and a social sensitivity index based on the proportion of households with member engaged in fishery, non-fishery and neither occupation. The social sensitivity index was then used to weight extrapolation of household density which, combined with fishers' spatial preferences, provided a 5 x 5m resolution map of predicted fishing effort that was used to map ecological exposure (Thiault et al. 2017a). Based on these four continuous and spatially explicit measures of social-ecological vulnerability dimensions, we represented social and ecological vulnerability profiles using bivariate maps (Fig. 2).

Then, building from the lessons of past successes (and failures) of various small-scale fisheries management practices found in the literature, we detailed how each general type of management intervention identified in Table 1 are operationalized in the context of small-scale fisheries (Step 2; Table 2). This allowed us to examine directly on the map the potential management intervention at each location around the island (Step 3; Table 2 and Fig. 2).

Implementing the vulnerability-based management framework in the context of Moorea's small-scale fishery revealed that the current conservation strategy is not aligned with the approaches suggested by our framework. For example, the fore reef generally showed low vulnerability, as it is highly resilient and weakly exposed to fishing. Our framework suggests that such configurations may be an opportunity for the development of fully protected areas because such ecologically efficient – but socially restrictive – measures are easier to implement and less socially costly for the households associated with this part of the reef. Yet, despite the relatively large permanent marine protected area fisheries closure system (20% of the total reef area), the fore reef only represents 7.7% of the total reef area protected, while the closed lagoon areas are sometimes located in front of poorly adaptive households (Fig. 2; see Chapter I for map of Moorea's MPA network). Therefore, the design and relevance of this conservation strategy is questionable.

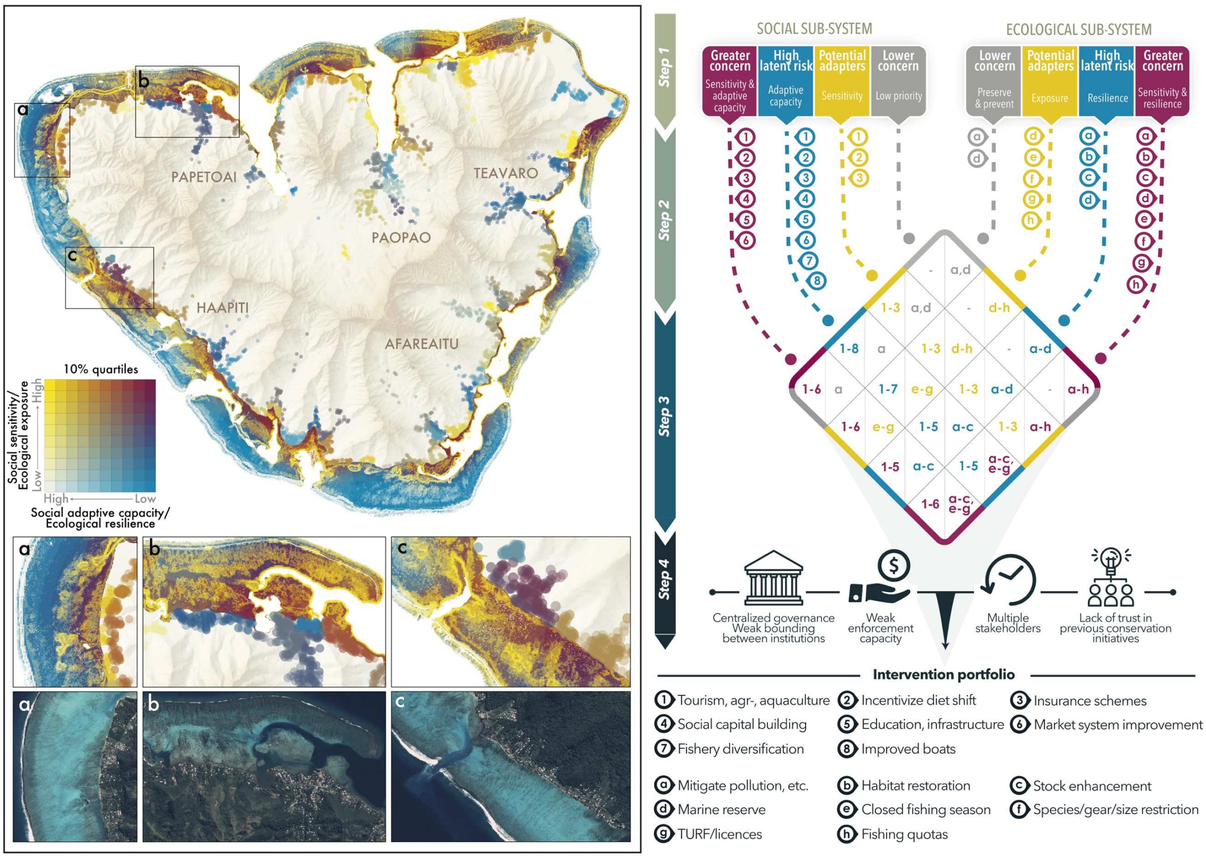

Figure 3: Application of the vulnerability-based management framework to the small-scale coral reef fishery of Moorea, French Polynesia, using spatially-explicit dimensions of social and ecological vulnerability profiles. Because households (users; represented by dots) mostly depend on adjacent reefs (resource unit) for provision and cultural services, combinations of social and ecological vulnerability profiles are directly observable on the map.

In some cases, however, other fisheries management strategies are in accordance with the recommendations of our framework. Size and species regulations, which are especially recommended to reduce fishing effort when the associated households are of "greater concern", have been implemented in the reefs off Papetoai (8.7% of the total reef area), one of the places in Moorea where households are the more vulnerable. This specific set of rules have been decided during the planning process (2000-2004) following claims from important local figures of the island (Audras et al. 2017). Although participation of a well-defined set of stakeholders is a prerequisite for effective and equitable management, they are sometime difficult to reach and include in the management process (Lynham et al. 2017). This example illustrates how conducting such spatially detailed social-ecological vulnerability assessments may help prioritize and increase the likelihood of success of interventions.

Our framework suggests that development strategies such as investments in market-based interventions, insurance schemes and livelihood diversification may be needed to improve social adaptive capacity of households of "greater concern" and "high latent risk". Although the former two may not be appropriate for Moorea due to the absence of conventional sales channels and the difficulty to identify fishers (Leenhardt et al. 2016), the latter can be achieved by developing tourism and creating incentives to develop sustainable agriculture. In high social sensitivity areas (Fig. 2 b-c), it is essential to development strategies that do not make local communities more dependent on reef-based resources that are already highly vulnerable. Island-scale incentives for motorized boats or new fishing gear are therefore not appropriate in our case.

Table 2: Typology of interventions to manage resource-resource user dependencies in the context of a coral reef fishery, and implications for social and ecological vulnerability profiles.

	Type of intervention	Examples of interventions in the context of small-scale fisheries
Ecological entry-point	**a -** Mitigate other sources of impact	Integrated coastal zone management (ICZM) Cumulative impact on marine ecosystem
	b - Ecological engineering	Artificial reefs Active habitat restoration Restocking and stock enhancement
	c - Permanent closures	Marine reserves, no-take zones, fully protected areas
	d - Capacity reduction	Temporal closures/closed seasons (fishing taboos) Restriction on species Size restrictions (protect young, protect breeders) Input control (licenses, TURFs) Gear regulations (minimum mesh size, gear restriction)
	e - Output control	TACs Output rights
Social entry-point	**1 -** Livelihood diversification	Provide land for agriculture or aquaculture Develop sustainable tourism
	2 - Behavioural change	Incentivize diet shifts (new target species)
	3 - Insurance schemes	Corporate insurer Government or informal insurances
	4 - Social capital building	Fisheries cooperatives Associations and other organizational forms
	5 – Assets strengthening	Access to health Education Infrastructure Information
	6 - Market system Improvement	Strengthen relations among actors Upgrade value chains Improved market terms
	7 - Within-sector diversification	New gear Alternative fishing techniques
	8 - Capacity enhancement	Improved boats Subsidizing credit for the purchase of outboard motors

Several isolated actions such as the provision of municipality-owned land to low income households to develop agriculture are likely to have improved social adaptive capacity and reduce social sensitivity locally in Moorea. Yet, such municipality-led initiatives have never been coordinated with agencies in charge of fishery management and marine spatial planning. Moving to less resource-focused actions, and better incorporating other institutions and agencies into planning and decision-making will represent a significant shift in how such fisheries management and resource conservation are approached in Moorea, and is likely to greatly improve the coherence of the spatial planning. Our framework provides a basis for understanding the local context and then prioritizing pragmatic actions to truly manage small-scale fisheries as social-ecological systems.

Application of our novel framework to Moorea reveals that current small-scale fisheries management strategy may be ill-fitted to the great spatial heterogeneity of Moorea's social-ecological system. We suggest that this could be improved by a micro-scale and integrated approach to coral reef management that (1) clearly distinguishes actions between locations, and particularly between the lagoon and the fore reef; (2) focuses less on resource to account more specifically for the existing social-ecological features of each area; (3) embrace a broader range of potential policy levers and (4) better coordinates with other local agencies (tourism, urbanism and environment) for a more coherent integration of conservation, fisheries management and development.

Our spatially-explicit application of the vulnerability-based management framework to Moorea illustrates its ability to inform (and even be incorporated into) marine spatial planning. The intensity of data collection and analysis required for our case study was high because vulnerability issues arising at the micro-scales are most relevant for this topic and study site. However, relatively simple, robust semi-quantitative approaches to assess vulnerability have been developed at a range of scales and data availability contexts (Johnson et al. 2016). We therefore believe that this approach can also be undertaken at larger scales (up to the national scale) and may be applicable to a wide range of complex social-ecological systems.

Conclusion

There are multiple paths to ensure food security, biodiversity conservation and economic development. Practitioners and decision makers should not be locked into a single approach *a priori*, whether it be focused on the resource (ecological entry-point) or the users of this resource (social entry-point). Rather, new management practices must embrace the complexity of social-ecological systems and reap the benefits of the lessons learned by these independent, but complementary, approaches. This intuitive, relatively easy to use and broadly replicable vulnerability-based management framework illustrates how complex social-ecological science and vulnerability can be put into practice to guide managers and decision-makers toward more holistic practices and produce win-win, synergic, opportunities that benefit people and the environment. We hope that its application will facilitate the development of better designed, and more balanced, management strategies and encourage a shift in how managers and policy-makers approach common pool resources problems.

PART II: EXTENTING THE SCOPE OF SOCIAL-ECOLOGICAL VULNERABILITY ASSESSMENTS

Chapter V

Mapping social-ecological vulnerability trajectories in response to multiple drivers

Status: In review

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Chapter V: Mapping social-ecological vulnerability trajectories in response to multiple drivers

Abstract

Managing local social–ecological interactions playing out against a backdrop of larger-scale dynamics are ubiquitous challenges to natural resource management and biodiversity conservation. Social-ecological vulnerability assessments are highly relevant for place-based management and can help prioritize and target management actions. We present an approach that integrates the temporal dimension into social-ecological vulnerability assessments to help communities and decision makers understand and plan for the effects of large scale or external drivers. We map current states and temporal changes in linked social-ecological vulnerabilities following exposure to multiple drivers of change in Moorea, French Polynesia. Nearly 23% of households and 13% of the reef area show low and decreasing vulnerability. However, 6% of households and associated reefs displayed high and increasing levels of vulnerability. By mapping social-ecological dependencies in space and time we provide decision makers with the information required to identify and prioritize management interventions. Our approach emphasises the importance of spatial heterogeneity resulting from the interactions between local and large-scale processes, and supports proactive and adaptive management by characterising trajectories of the system in response to important drivers of change.

Introduction

Conservation and natural resource management are endeavors to influence human behavior to achieve a range of objectives related to the condition of an ecosystem or a subset of its components (species, habitats, etc.) while increasing or maintaining human well-being. They involve decisions about the dynamics of what are invariably complex social-ecological systems (Cumming et al. 2016). Interactions within social-ecological systems are a function of internal dynamics and their responses to external drivers which can be social (e.g., shifting socioeconomic settings and governance of tenure) and/or biophysical (e.g., occurrence of natural disturbance) (Ostrom 2007; Fischer et al. 2015; Bennett et al. 2016). Understanding the key drivers and most influential internal linkages of these systems is a critical foundation for effective decision-making, yet too often conservation decisions are based on overly simplistic, fully internalized or overly vague representations of social-ecological systems (Game et al. 2014).

The concept of vulnerability has emerged as a valuable approach for understanding the effects of major external drivers like climate change on social-ecological systems, enabling more informed and structured decision making (Marshall et al. 2010). Its application has provided the foundation for characterizing interactions between external drivers and internal system processes, and for estimating or ranking relative magnitudes of consequences for social and ecological sub-systems resulting from exposure – or risk of exposure – to these drivers (Wilson et al. 2005). The vulnerability framework has also been instrumental in designing interventions that can reduce vulnerability by modifying internal system properties in order to increase resilience to (often unmanageable) external drivers (Cinner et al. 2012a; Foden et al. 2013). The utility of the vulnerability concept has prompted recommendations that it be used in a nested model to capture key features of the dynamics of linked social-ecological systems (Marshall et al. 2010; Cinner et al. 2013a), but has yet to be applied at local scales.

In order to capture complex realities and effectively guide place-based management, the operational application of social-ecological vulnerability faces four main challenges (O'Brien et al. 2004; Adger & Vincent 2005; Notenbaert et al. 2013). First, the interdependences between the natural and the human systems are widely acknowledged from a conceptual standpoint, but remain difficult to parameterize. Second, vulnerability is heterogeneous, as pressures on households, and accessibility and use of natural resources vary through space and among segments of the society (Cinner et al. 2015). Third, external drivers that shape vulnerability operate at different spatial and temporal scales (Fischer et al. 2015), making it difficult to represent vulnerability with a single snapshot. Fourth, similar magnitudes of vulnerability (or changes in vulnerability) can be caused by different drivers (Bennett et al. 2016) while identifying causal relationships are important to reveal and prioritize opportunities for intervention.

Here, we illustrate how the social-ecological vulnerability framework can be used to address these challenges and represent system-scale changes in human-nature dependencies across time. The operationalization of this standardized framework allowed us to map linked social and ecological vulnerabilities and track how spatial patterns changed in response to co-occurring drivers (socioeconomic, governance and biophysical). We show how conducting impact assessments through the lens of social-ecological vulnerability can provide insights about interactions between local and large-scale drivers that can ultimately help identify and prioritize management interventions.

Material and methods

Study site

The island of Moorea, French Polynesia, presents an interesting case study to examine the response of local social-ecological interdependencies to global drivers of changes. Key features include (i) a mixed economy, with dependent communities relying on a combination of salaried employment and subsistence linked, at least partly, to coral reef fishing (ISPF 2002; Leenhardt et al. 2016); (ii) recent urban development and intense exposure to globalization (Féral 2013); and (iii) exposure to recurent large-scale drivers of ecological, socioeconomic and governance changes (Lamy et al. 2015a; Leenhardt et al. 2015, 2016) (Fig. 1).

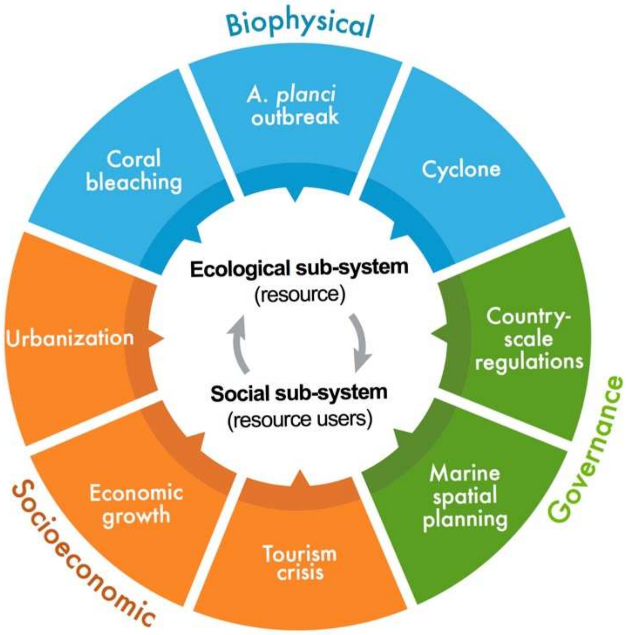

Figure 1: Like many other coral reef social-ecological systems, Moorea is shaped by biophysical (e.g., cyclones, coral bleaching events, crown-of-thorns starfish outbreaks), socioeconomic (e.g., urbanization, population growth, socioeconomic crisis) and governance (e.g., country-scale fishing regulations and marine spatial planning) drivers. The response of social-ecological interactions that play out against this backdrop of various drivers is uncertain, owing to complex interactions between external drivers and internal system processes. A temporal analysis of internal properties and interdependencies between social and ecological sub-systems through the lens of vulnerability may enable to understand the consequence of these drivers on social-ecological dynamics.

Mapping social-ecological vulnerability

Here, we characterize the social-ecological vulnerability of the human-nature interactions focusing on small scale coral reef fishing, which is the main form of natural resource use in Moorea. Linkages between the ecological and the social sub-systems are therefore viewed through the lens of resource dependency (flow from social sub-system to ecological sub-system) and fishing opportunity (flow from ecological sub-system to social sub-system). Ecological vulnerability is here considered at the scale of the targeted fish assemblages and results from the combination of exposure and resilience to fishing (where resilience is here the combination of adaptive capacity and sensitivity). Social vulnerability originates from sensitivity (i.e., fishing dependency), adaptive capacity and exposure of households to the loss of fishing opportunity. Methods for modelling and mapping each component are summarized in Table 1 and detailed in Chapater III.

Table 1: Description of the components of social-ecological vulnerability and overview of the methods used to their modelling and mapping. See Chapter III for a detailed description of the methods.

Sub-system	Vulnerability component	Description	Modelling and mapping
Social	Exposure	Households' exposure to the loss of fishing opportunity	Average ecological vulnerability of households' fishing grounds, defined as a 2 km radius around households
	Sensitivity	Households' dependence on marine resource for livelihood	District-scale index based on households engaged in a fishery- and non-fishery-related activity and level of employment
	Adaptive capacity	Households' ability to minimize, react and adaptive to the loss of fishing opportunity	District-scale index based on the weighted sum of five indicators (mobility, livelihood diversity, place attachment, education and material assets)
Ecological	Exposure	Exposure of the target fish assemblages to fishing pressure.	Household density, weighted by social sensitivity index, extrapolated onto lagoon waters and then combined with fishers' preference for fishing ground
	Resilience (sensitivity and adaptive capacity)	Ability of the target fish assemblage to recover following exposure to fishing.	Resilience index combining fish density and species intrinsic vulnerability index based on fish count data. Extrapolation of index distribution to unsampled areas using spatially explicit predictive variables.

Social vulnerability (SV_h) of each household h, and ecological vulnerability (EV_i) of each 5 x 5m cell of reef i were calculated as follows:

$$SV_h = SE_h + SS_h - SAC_h \quad (1)$$

$$EV_i = EE_i - ER_i \quad (2)$$

where SE_h, SS_h and SAC_h are social exposure, social sensitivity and social adaptive capacity of the household h, respectively, and EE_i and ER_i are the ecological exposure and ecological resilience of the reef cell i, respectively. Social vulnerability was calculated at the household level, but our metrics of social sensitivity and social adaptive capacity were used at the district-level (n=68) for confidentiality reasons, which avoided identification of individual household's response. In doing so, we assumed that the coarse-scale value was evenly distributed across all households within that district.

Spatiotemporal vulnerability analysis

To understand the temporal changes in vulnerability, we obtained data on each component of vulnerability described above spanning a five-year time period (between 2007 and 2012). This period represents a time of significant exposure to significant drivers of social-ecological change, including a cyclone (Lamy et al. 2015a), a coral predator *Acanthaster planci* outbreak (Kayal et al. 2012), a continuous urban development (Féral 2013), a country-scale socioeconomic crisis (IEOM 2014) and the implementation of a spatial marine plan (Aubanel et al. 2013) (Fig. 1). Therefore, our analyses provide maps of social-ecological vulnerability *before* (in 2007) and *after* (in 2012) the occurrence of the drivers. Following Halpern *et al.*'s (2015) approach, we then mapped and combined two key metrics at a whole-of-island scale: the current vulnerability and the vulnerability trend. While the former apprises on the current state of each area in the system, the latter provides information on the location, magnitude and direction of change induced by external drivers. We therefore identified areas of high (above the 75% quartile) and low (bellow the 25% quartile) current vulnerability and assessed if it increased or decreased during the time span of the study. In addition to combining maps of current vulnerability and vulnerability trajectories, we also aggregated results by locally relevant social and ecological units: municipalities (social vulnerability) and habitats (ecological vulnerability), respectively. The two representations – mapped and aggregated by units – are complementary. The former can both show spatial heterogeneity of processes at stake and their trajectories over time. The latter allows managers to target specific social or ecological units.

All statistical and spatial analyses were implemented in the R statistical software (R Core Team, 2015) using the {raster} (Hijmans et al. 2015) and {rgdal} (Bivand et al. 2016) packages.

Results

Changes in social-ecological vulnerability between before and after exposure to multiple drivers of change

Island-wide, ecological vulnerability decreased by 9.3%, with more than 70% of the total reef area experiencing decreased ecological vulnerability between 2007 and 2012 (Fig. 2a). Habitats that experienced an increase in ecological vulnerability (roughly one third of Moorea's reefs) were located on the fore reef and, to a lesser extent, on passes and channels.

Social vulnerability decreased by 20.3%, although patterns were highly heterogeneous around the island (Fig. 2a). Approximately 88% of the households experienced decreased social vulnerability between 2007 and 2012, while the remaining 12% experienced an increase in vulnerability over the same period.

High current ecological vulnerability levels are found where high ecological exposure overlaps with low ecological resilience. Hotspots of ecological vulnerability are located all around the island, generally close to shore in shallow areas with sandy and flat bottoms or in passes (Fig. 2b). Low levels of ecological vulnerability are located further from the coast and outside passes, most often in deep areas with high coral cover.

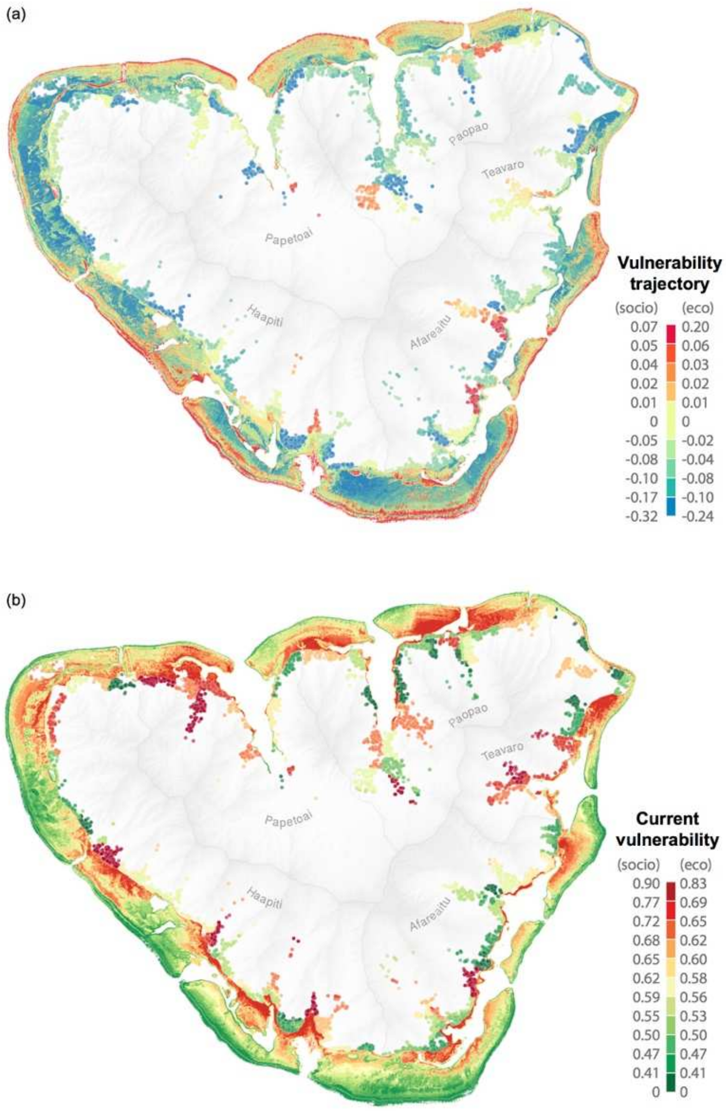

Figure 2: Social-ecological vulnerability in Moorea. (a) Social-ecological vulnerability changes from before (2007) to after (2012) the occurrence of multiple drivers of change, expressed as the log-ratio of vulnerability values over the two periods. (b) Social-ecological vulnerability as of 2012. Circles located on land represent individual households and cells surrounding the land represent the reefs; colors are assigned to 10-quantiles in the data.

Social vulnerability is highly heterogeneous and highly vulnerable households are not necessarily located in front of ecologically vulnerable areas (Fig. 2b).

Incorporating state and trends of social-ecological vulnerability

Between 2007 and 2012, reef areas with highest vulnerability have generally become less vulnerable, while least vulnerable habitats have experienced increased vulnerability. Low vulnerability following a decreasing trend (13% of the reef area) are located mostly on the fore reef and passes (blue patches in Fig. 3). Habitats with high vulnerability following an increasing trend (6% of the total reef area) are generally located in sandy and fringing reef areas (red patches in Fig. 3).

Households with low social vulnerability following a decreasing trend (23% of households) are mostly located in Paopao and Afareaitu (blue circles in Fig. 3). Households of high vulnerability following an increasing trend (6% of households) can be found in each municipality but represent 13.5% of households in Teavaro. They are among the most vulnerable households of Moorea (red circles in Fig. 3). Papetoai has the highest proportion of highly vulnerable households of the island, but most of them have experienced a decrease in vulnerability over the five-year period of this study (orange points in Fig. 3).

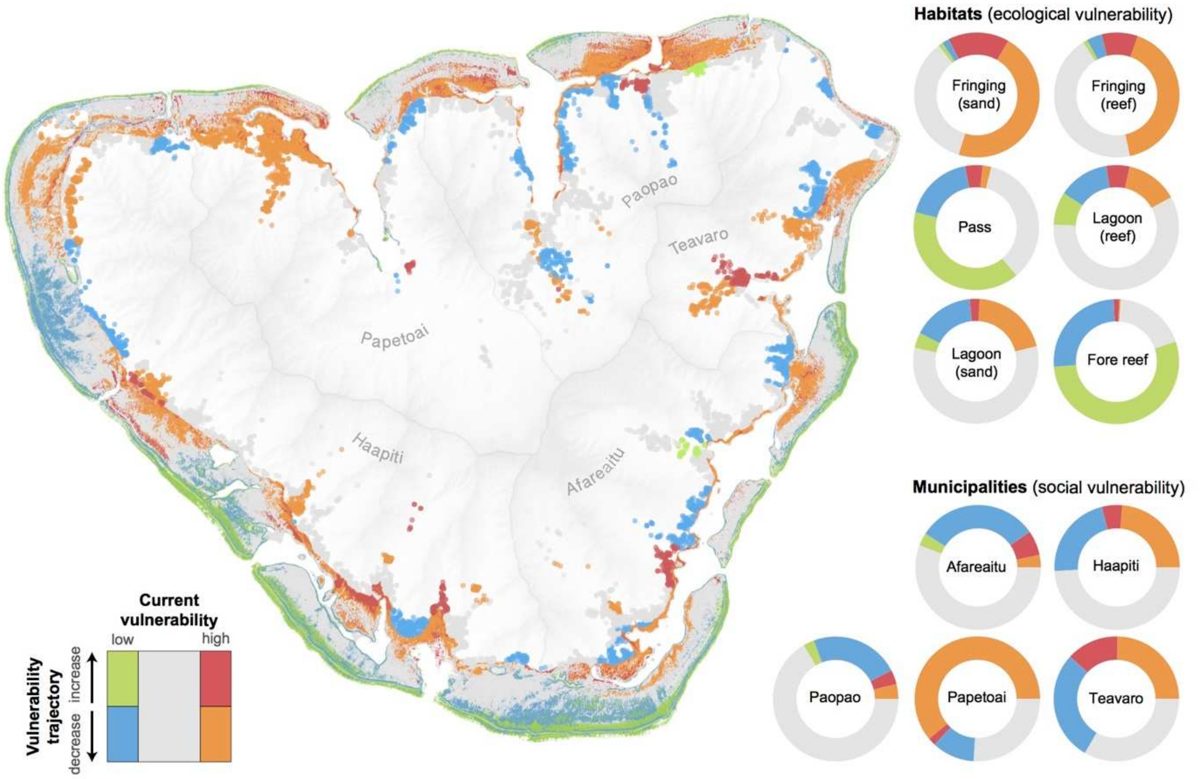

Figure 3: Combination of current social-ecological vulnerability and social-ecological vulnerability trajectories. They include areas with combinations of the highest (top quartile) and lowest (bottom quartile) vulnerability and increasing and decreasing vulnerability over the 5-years time span. Results are summarized for each natural habitat (ecological vulnerability) and municipality (social vulnerability).

Discussion

Managing social-ecological systems for sustainability requires actions that seek synergies but recognize tradeoffs between ecosystem processes and states and human well-being (Adams et al. 2004). However, managing for place-based social-ecological sustainability is not a small challenge because social-ecological interdependencies of human activities and the flow of nature's contributions to people are shaped by interactions, feedbacks and drivers that can be highly variable in space and time and across scales (Cumming et al. 2006, 2016; Fischer et al. 2015; Pascual et al. 2017a).

The approach we have adopted in this study addresses these challenges at three levels. First, we linked social and ecological vulnerabilities through exposure and sensitivity to explicitly recognize that people are integral components of social-ecological systems and that people both affect and respond to ecosystem processes. Second, we mapped current vulnerability for identifying spatial relationships between social (sociological profiles) and ecological (habitat heterogeneity) vulnerability components, and for identifying priority areas for management intervention. Third, we uncovered vulnerability trajectories to provide insight about changes over time and provide the foresight required to identify and implement strategic management actions.

Combining current vulnerability levels with past vulnerability trends is indeed highly relevant to inform decisions about the type(s) and location(s) of management interventions. Overlaying those two key metrics reveals areas of high and increasing vulnerability. This integrative view of the system enables policymakers and community leaders to identify areas of greatest concern for sustainability, and thus to focus efforts to planning, prioritizing and implementing management actions that can reduce vulnerability by targeting relevant components of vulnerability. Such system view of management can help achieve improvements in ecosystem condition while maintaining household livelihoods by revealing a greater range of intervention options. In general, programs that build and sustain natural and social capital, and broaden the range of livelihood opportunities reduce the social-ecological vulnerability (Allison & Ellis 2001; Allison & Horemans 2006; Béné 2006; McClanahan et al. 2008b).

We interpret the vulnerability trajectories observed in our case study as the combined consequences of four major external processes that occurred during the period of this study. First, a crown-of-thorns sea star (*Acanthaster planci*) outbreak, followed by a cyclone, caused significant decline of coral cover and altered composition of the coral assemblages on the fore reef, with cascading effects on target fish assemblages (Adam et al., 2011; Lamy et al., 2015; Lamy et al., 2015). This explains the decrease in ecological resilience observed on the fore reef and passes. Second, the marine spatial plan established on the ground in 2007 may have induced a general increase in ecological resilience within the lagoon, through species and size regulations and the ban of destructive fishing gear (e.g., long narrow-meshed nets) and damaging development practices (e.g., embankments' construction). Third, the global financial crisis that affected the entire economy of the island may have conducted households engaged in wage activity to increase coral reef fishing to mitigate economic impacts (increased sensitivity) (Insee-ISPF 2012; IEOM 2014). Finally, island-wide development and increased "urban" population largely due to the rapid increase in the number of non-native migrants (mostly native of France and generally wealthier and more educated) (Féral 2013; IEOM 2014) may have been important drivers of the observed increased adaptive capacity of the island (+18.6%, mostly through improved material assets and education), thus potentially avoiding vulnerability to increase (at an island-scale) following the socioeconomic crisis. However, since our district-level metrics of sensitivity and adaptive capacity do not enable to track vulnerability changes at the household-scale, the proportion of households that experienced increased vulnerability in Moorea is likely to be underestimated. Despite this caveat, and although we cannot demonstrate causalities (several other islands with similar social-ecological features would have been needed as controls), these observations are in line with other studies that suggest that lagoons are affected by socioeconomic drivers while the state and processes of fore reefs and passes are mostly shaped by biophysical drivers (Leenhardt et al. 2017). Management efforts should therefore focus on actions that can build ecological resilience of the lagoon ecosystems to future drivers of human use, while developing strategies to build social resilience to unmanageable changes that are likely to affect the fore reef and passes with more intensity.

We showed that in Moorea social and ecological vulnerabilities decreased on average between 2007 and 2012. However, the direction and magnitude of the changes varied greatly according to the municipalities or natural habitats. The great temporal variability and the important spatial heterogeneity highlighted in this study stress the need for implementing social-ecological long-term monitoring programs that are tailored to management and community decision information needs. We demonstrated the feasibility of implementing a truly adaptive management approach (Schultz et al. 2015) in Moorea to increase resilience and reduce vulnerability of the entire social-ecological system over the longer term. However, future work could better incorporate data on other components of social vulnerability that are important features of sustainable social-ecological systems (e.g., agency, values and inequity; Hicks et al., 2016). This would better reflect the actual impact of external drivers of changes on the social sub-system and its possible consequences on the ecological sub-system. Future efforts in this area would be particularly relevant to systems where sociocultural aspects of social-ecological linkages are strong and where perturbations are frequent and pressing.

Conclusion

Our approach illustrates how to derive a more integrated understanding of the condition and trajectories of linked social-ecological systems for use in real-world decision-making. It allows for a detailed view of the direction (positive or negative), magnitude, location and consequences of change in both the social and ecological sub-systems. It offers early warnings of negative trends and allows projections of future states that can proactively guide adaptive management actions. Our approach to mapping social-ecological vulnerability and its change over time is particularly relevant for spatial planning and management actions that need to address the complex and dynamic interplay between human and nature. Through this approach, decision makers have a tool to better understand their system's dynamics and guide actions that can build local resilience to global changes.

Chapter VI

Addressing market volatility and illegal fishing in territorial user right fishery policies: insights from multi-driver social vulnerability

Status: In preparation

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Chapter VI: Addressing market volatility and illegal fishing in territorial user right fishery policies: insights from multi-driver social vulnerability

Abstract

In response to the failure of numerous top-down approaches at sustaining small-scale fisheries, many governments are engaging fishers into Territorial User Rights for Fisheries (TURFs) co-management systems. Although supported by well-grounded and tested research on internal self-governance of commons, these approaches tend to neglect the importance of external drivers of change, which can critically undermine efforts to manage for fisheries sustainability. Here, we present a novel application of the IPCC framework of vulnerability that enables to identify key interventions for building social resilience to two major drivers of fisheries worldwide: markets volatility and illegal fishing. We ground the approach in 42 fishing organizations along the coast of Chile and show contrasting levels of exposure, sensitivity and adaptive capacity to these two drivers, even though they are ruled by the same policy instrument. There is a pressing need to integrate TURF regimes into broader management frameworks that include livelihood-, market- and social network-based approaches and practices in order to secure and enhance the contribution of fisheries to coastal communities in the future. Our multi-driver social vulnerability approach provides practitioners with the information required to tailor management interventions to each specific context.

Introduction

Small-scale fisheries are experiencing sustainability challenges in many parts of the world (Pomeroy 2012; Kittinger et al. 2013), thus threatening the livelihood of millions of people worldwide (FAO 2014a). The repeated failure of many single-stock and top-down approaches has prompted widespread calls for innovative policies and programs to enhance the sustainability of small-scale fisheries (Kittinger et al. 2013). Aid agencies, donors, policy-makers, and governments around the world are now actively considering the implementation of rights based approaches, with the hope of dramatically improving ocean stewardship and eliminating resource overexploitation (Castilla & Fernandez 1998; Beddington et al. 2007; Castilla & Gelcich 2008; Gelcich et al. 2010; Lubchenco et al. 2016). In particular, Territorial Use Rights for Fisheries (TURF) systems – which consist of assigning management and exclusion rights spatially to a defined group of users – are now widely advocated and embraced as a mean of resolving internal coordination problems (through incentives for collective profit) whilst facilitating institutional fit (by adapting government policies for efficient resource use with local context) (Castilla & Defeo 2001), all of which contribute to the creation or strengthening of conditions leading to the sustainable use of marine resources (Ostrom 1990; Castilla 2010a; Epstein et al. 2015). Yet, the simple creation of territorial-based right incentives does not automatically solve resource use problems, because other factors can also strongly affect the fisheries exploitation status (Gelcich et al. 2006; Cinner et al. 2012b; Aburto et al. 2014).

Among the variety of direct and indirect drivers experienced by fishing communities worldwide, markets and illegal fishing have the strongest and most widespread influences on management outcomes. Increased integration of small-scale fisheries into the global economy can be a boon for income generation, but predominately exposes resource users to the whims of markets and demand volatility (Berkes et al. 2006; Crona et al. 2016). Access to new markets and demand often coincide with exposition to unmanageable price fluctuations, ultimately impacting the livelihood of resource users (Béné & Doyen 2000). In TURFs, responsibilities and associated costs of surveillance for preventing illegal fishing are generally shifted to the fishers themselves. Hence, the capacity (and willingness) of fishers to enforce their own marine user rights can differ greatly according to context (Davis et al. 2015; Nguyen Thi Quynh et al. 2017). Absence of explicit recognition of these two key drivers in territorial rights-based policies can jeopardize the long-term viability of TURFs.

Here, we propose an approach for identifying effective management levers to allow TURF governance structures to build social resilience in the face of multiple drivers (market volatility and illegal fishing in the current case). We used the IPCC framework for analyzing social vulnerability (Marshall et al. 2010). The predominant focus of vulnerability research, policy and practice has been mostly on single driver (generally climate-related). Consequently, many studies have approached the measurement of vulnerability (i.e., the result of high exposure, high sensitivity and low adaptive capacity) as the combination of one specific and external component (exposure) and a suite of general intrinsic features (aggregated into sensitivity and adaptive capacity). Considering general intrinsic features is important, but ignoring specific conditions and factors that may be useful in confronting a given driver may overlook the possibility that the properties leading to reduced or increased vulnerability differ across drivers. We thus adapted the vulnerability framework to assess vulnerability to multiple drivers by accounting for general (sensitivity and general adaptive capacity) and specific (exposure and specific adaptive capacity) aspects of vulnerability (Table 1, Methods and SI Fig. S1). We used a combination of >400 semi-structured interviews with organization leaders and fishermen and market-related data (Fig. 1 and SI Table S1) to evaluate the vulnerability of 42 Chilean fishing organizations to market volatility and illegal fishing. We then show how the underlying source(s) of vulnerability can be systematically addressed in a TURF policy context.

Table 1: Vulnerability dimensions and associated general and specific components. Weights indicated are defined so that each dimension of vulnerability has a cumulative weight score of one (see Appendix D; Table S1 for details at the indicator-level).

Dimension / Aspect / Component	Weight
Exposure	
Specific (illegal fishing)	
1. Level of illegal fishing in TURF	1
Specific (markets)	
2. Resource price volatility	0.5
3. Resource price trend	0.5
Sensitivity	
General (resource dependency)	
4. Dependency on fishing	1
Adaptive capacity	
General	
5. Learning and knowledge	0.08
6. Diversity and flexibility	0.08
7. Infrastructure	0.08
8. Material assets	0.08
9. Social capital and trust	0.08
10. Agency	0.08
Specific (illegal fishing)	
11. Conflict resolution mechanism	0.125
12. Surveillance effectiveness	0.125
13. External support	0.125
14. Internal support	0.125
Specific (markets)	
15. Relationship with the middleman	0.25
16. Gear diversity	0.25

Results and discussion

We provide the most comprehensive vulnerability assessment to date of coastal communities to the impacts of both direct (illegal fishing) and indirect (market volatility) drivers. Vulnerability is unevenly distributed with a surprisingly high range of variation across fishing communities in Chile, regardless of the driver considered, suggesting that both intrinsic and extrinsic social features should be accounted for when planning for the implementation of TURFs (Fig. 2). Answers of both insiders and outsiders ranged from total compliance to total non-compliance with rules. Dependency on marine resources for livelihood greatly varied among study sites (80% +/- 24).

Fishing organizations vulnerable to one driver are not necessarily vulnerable to the other one (Table 1, Fig. 2). Specific sources of vulnerability cannot be inferred by general sources of vulnerability (Table 1 and SI Figs. S1-2). A focus on a single driver of change – which is a common practice in vulnerability assessments – may be problematic because it fails to recognize the multifaceted nature of impacts at local social–ecological systems levels and may result in undermined policy outcomes. This stresses the need of adopting a community-centered (rather than solely problem-centered) approach, focusing on community-relevant drivers (Bennett et al. 2016).

Distinguishing the contribution of general and specific aspects of vulnerability can help identifying communities that disproportionately requires context-specific mitigation actions (Fig. 3A). Our approach for discriminating between general and specific aspects (detailed in Methods) revealed that three organizations, namely Laguna Verde (region V), Larraquete (region VIII) and Los Molles (region V) were more vulnerable to illegal fishing than the global average, while only one (San Marcos, region I) was relatively more vulnerable to market fluctuations (Fig. 3B).

Figure 1: Chile is at the forefront of establishing TURFs for small-scale fisheries management. **A-C-** In 1991, Chile established a national TURF policy for benthic resources, which gave legal authority to assign collective exclusive access rights to artisanal fisher organizations. By 2016, there were ~550 fully operational TURFs decreed to fisher organizations in different biophysical and socioeconomic settings (Gelcich et al. 2017), making Chile the largest TURF system under one policy instrument. That macro-level (national) institutional constraint remaining consistent across fishing organizations, Chile provides a unique opportunity to examine the spatial heterogeneity among fisheries under a TURF regime. **D-** Location of the 42 fishing organizations sampled along the Chilean coast (only the South, in light grey, wasn't sampled because TURFs are not common).

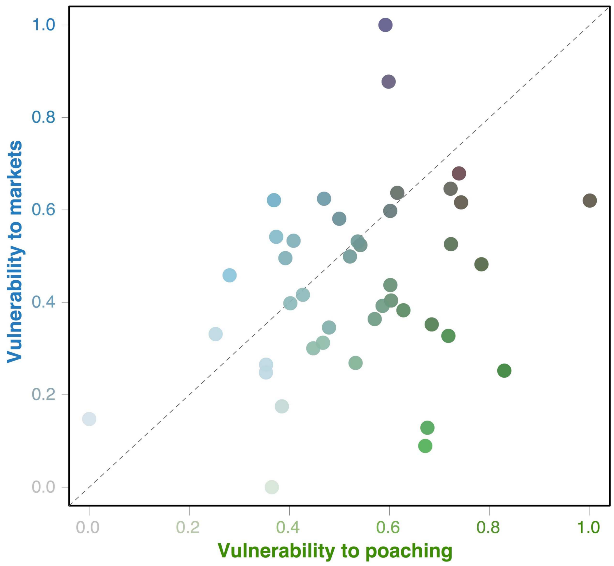

Figure 2: Bi-plot showing the great variability of vulnerability among Chilean fishing communities to markets and illegal fishing. Note that deviation from the 1:1 line does not necessarily indicate greater vulnerability to a driver relative to the other due to the distribution of the specific aspects of vulnerability may differ across drivers.

Vulnerability assessments are useful to identify management actions and inform policies aimed at mitigating linked social and ecological impacts of direct or indirect drivers (Allison et al. 2009; Cinner et al. 2012a; Hughes et al. 2012; Johnson et al. 2016). Markets and illegal fishing are major indirect and direct drivers of small-scale fisheries worldwide (Berkes et al. 2006; Agnew et al. 2009; Castilla 2010b; Ernst et al. 2013; Crona et al. 2016). Reducing social vulnerability to these drivers is critical for TURFs sustainability (Castilla et al. 2016; Gelcich et al. 2017; Nguyen Thi Quynh et al. 2017). Addressing general adaptive capacity may include, for example, livelihoods diversification (Torell et al. 2017), approaches to poverty alleviation (Allison & Ellis 2001), investments in infrastructure and material assets (McClanahan et al. 2008a), or social capital building (Marin et al. 2012). Market-specific interventions include improved market governance through certification schemes, improved information of price changes or diversification within the fishery to better respond to changing demand. Interventions targeted at reducing illegal fishing interventions range from improving enforcement subsidies (Sumaila et al. 2016b) and training programs for fishers (Akella & Cannon 2004) to TURFs spatial design improvement to clarify boundaries (Day et al. 2012) and communication to TURF members about benefits of enforcing and complying with the rules (Davis et al. 2015). We identified 22 potential policy levers to address general and specific aspects of vulnerability in the context of global markets volatility and illegal fishing (Table 2).

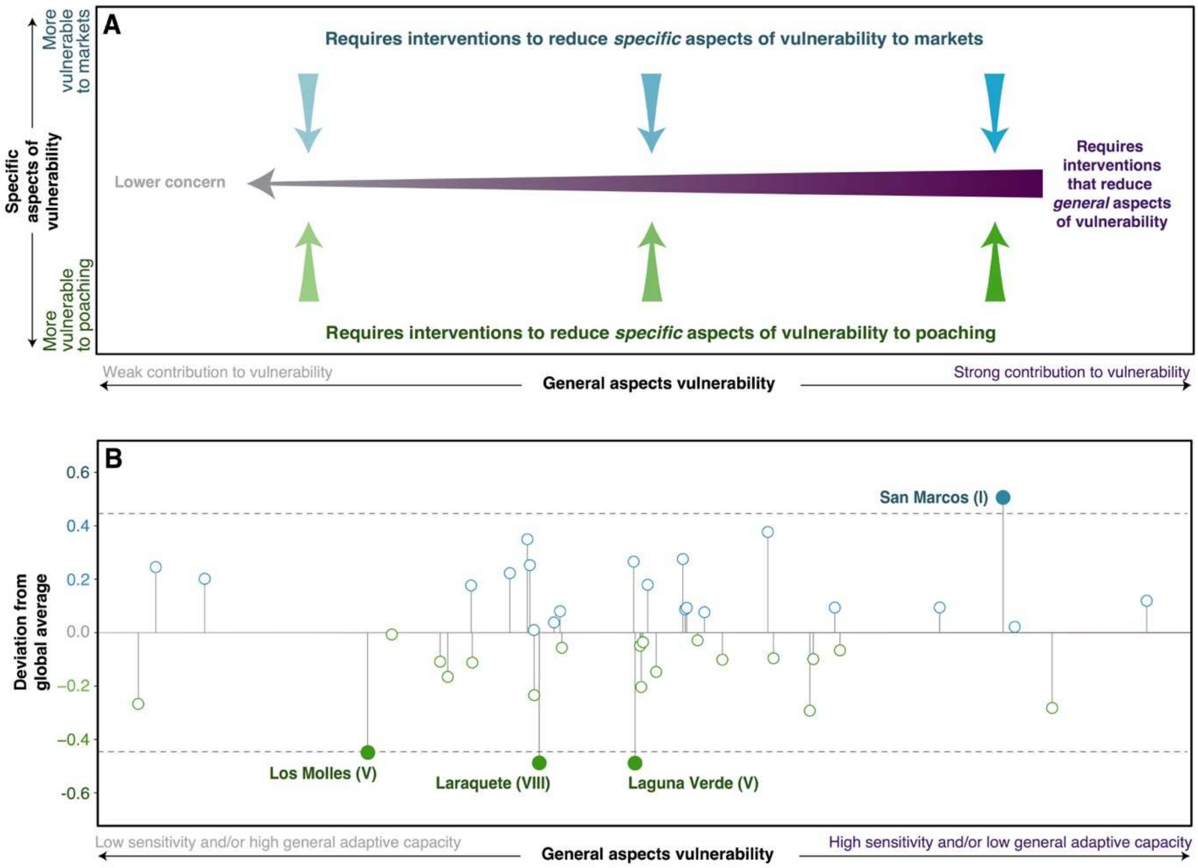

Figure 3: Identifying interventions based on general and specific aspects of vulnerability. A- Theoretical model indicating pathways toward reducing vulnerability through general and/or specific aspects of vulnerability. B- Application to the Chilean case study. Deviation from global average is calculated for each site as the absolute difference between vulnerability to markets and vulnerability to illegal fishing, readjusted so that the global average equals zero. Therefore, deviation is determined by specific aspects of vulnerability (i.e., exposure and specific adaptive capacity) and outliers to the global average (i.e., outside the 95% confidence interval, represented by dotted lines) are considered as being relatively more vulnerable to markets (blue) or illegal fishing (green) compared to the overall sites. See Table 2 for strategies targeting different aspects of vulnerability.

Here, we showed how applying the vulnerability framework in a multiple-driver context can help identifying general and/or driver-specific leverage points. We recommend that governments considering rights-based approaches and research on these programs explicitly integrate vulnerability assessments. This will represent a substantial departure from how most policies conceive TURF systems, and implementing them effectively will require improved partnerships with fishers, local and national institutions, social scientists, NGOs, and donors.

Table 2: Examples of strategies that can be taken to reduce the different aspects of vulnerability.

Aspect Component to address	Strategies
General	
Sensitivity	1. Occupation/livelihood diversification
	2. Linkages to other economic sectors (e.g., tourism)
Adaptive capacity	3. Networks reconfiguration
	4. Infrastructure improvement
	5. Poverty reduction/increase inputs
	6. Capacities and health status enhancement
	7. Spaces for sharing knowledge
	8. Invest in formal education and literacy
	9. Social security/safety nets
Specific (markets)	
Exposure	10. Stabilize market prices
	11. Eco-labelling and certification schemes (stabilize demand)
Adaptive capacity	12. Information systems on prices or changes in demand
	13. Diversification within the fishery (gear & species)
	14. Access to credit (to invest in new gear)
	15. Bargaining/negotiating power improvement
Specific (illegal fishing)	
Exposure	16. Enforcement subsidies
	17. Training programs for fishers/wardens
Adaptive capacity	18. Design for simple and clear boundaries
	19. Logistical support (boat, rangers, technology)
	20. Conflict resolution mechanisms training
	21. Institutional system improvement to undertake proceedings against non-compliance
	22. Information about benefits of enforcement

Methods

Study sites and data collection

In 2014, we conducted a socioeconomic survey in 42 fishing organizations along the Chilean coast (Fig. 1a). In order to capture various social-ecological contexts, fishing organizations were randomly selected to cover all coastal regions between Arica (North) and Los Lagos (South), spanning a 2,600 km coastline where most fishing organizations concentrate. At each study site, we conducted two different semi-structured interviews: one with organization leaders (n=42) and another one with fishermen from these organizations (n=396). All interviews were conducted in Spanish by trained interviewers.

Conceptualizing vulnerability to multiple drivers

To operationalize the IPCC framework in the context of illegal fishing and market fluctuations, we differentiate between general and specific aspects of vulnerability (SI Fig. S1). General aspects of vulnerability include shared properties that make a system more or less vulnerable, regardless the driver considered. Conversely, specific aspects of vulnerability refer to the features rendering a system more or less vulnerable in the particular context of a driver. Hence, exposure is by essence a specific aspect of vulnerability while sensitivity, which in the context of this study refers to resource dependency (Marshall et al. 2007), is here considered as a general aspect of vulnerability. Finally, adaptive capacity is treated as the combination of general and specific aspects.

Data and analysis

Each of the three vulnerability dimensions (i.e., exposure, sensitivity and adaptive capacity) is composed of either a specific aspect (exposure), a general aspect (sensitivity) or a combination of both (adaptive capacity) (SI Fig. S1). Each aspect is decomposed into 16 components in the context of illegal fishing and markets (Table 1). Based on our survey, we created 20 indicators to quantify each vulnerability component (SI Table S1). Exposure to markets, which was not captured by our survey, was obtained from the undersecretary of fisheries (SUBPESCA). Indicators were then normalized between 0-1, so they could be combined and compared. For each driver, each of the three dimension of vulnerability has a cumulative weight score of one (Table 1). The relative contribution of each of the indicator to this weight depends on the total number of such indicators analyzed under a particular component nested in a particular aspect (aspects have a weight of 1 for exposure and sensitivity and 0.5 for adaptive capacity). To aggregate indicators into components, aspects and dimensions, we used the TOPSIS method, which ranks the alternatives according to their relative distance to extreme values (for a description of the method in a vulnerability context, see Parravicini et al. (2014)). We then examined the correlation between each vulnerability aspect using Spearman's rank correlations (SI Fig. S2). In order to distinguish the contribution of general aspects from that of specific aspects, we calculated the absolute difference between vulnerability to illegal fishing and vulnerability to markets and used the global average as the reference for what is equally vulnerable to illegal fishing and markets. Then deviation from this global average was used to measure the contribution of specific aspects of vulnerability, with outliers (i.e., sites outside 95% confidence interval) being more vulnerable to one driver in particular due to specific aspects. All statistical analyses were performed using the R statistical software (R Core Team 2014).

Chapter VII

Human vulnerability to the impacts of climate change on world's food systems

Status: In preparation

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Chapter VII: Human vulnerability to the impacts of climate change on world's food systems

Abstract

Climate change alters the world's ecosystems, and thus threatens the ecosystem services that sustain humanity. Yet, consideration of climate change impact on multiple food production systems remains a poorly articulated issue, owing to a lack of cross-sector and interdisciplinary analyses, thus impeding the implementation of a truly holistic approach to the management of food systems. Here, we compiled global climate projections for 2100 and country-level socioeconomic data to evaluate human vulnerability to the impacts of climate change on the world's major food production systems: agriculture and marine fisheries. We show that (1) no country will be immune from the impacts of climate change on food production systems; (2) a significant proportion of the world's population (mostly located in the tropics) will experience the worst hardship associated with concomitant high vulnerability of agriculture and fisheries; and (3) climate-induced vulnerability can be massively reduced but require coordinating efforts to reduce greenhouse gas emissions at the global level while addressing intrinsic drivers vulnerability through capacity building and integrative low-carbon development pathways. Our findings provide a context for the design of more integrated policy action to tackle global changes and highlight the value of fostering positive feedback loops across multilateral policy initiatives to move onto more sustainable pathways.

Main text

A fundamental concern of science and policy agenda is the impact of climate change on world's ecosystems and the cascading consequences on human societies. Agriculture and marine capture fisheries represent cornerstone sectors that sustain humanity's food security, but also economic growth and employment worldwide (FAO 2017). Those food production systems heavily rely on current climatic conditions and are thus directly affected by ongoing climate change. The impacts of future climatic conditions on agriculture and marine fisheries are expected to be widespread, complex, geographically variable, and mostly unfavorable (Cheung et al. 2010; Lobell et al. 2011; Mora et al. 2013, 2015; Stock et al. 2017), raising serious concerns for food security worldwide. However, the degree to which human societies are likely to be impacted by climate-induced changes on food systems ultimately depends on the balance between this exposure to environmental change, human dependency on altered goods and services, and social adaptability, all of which may vary depending on the sector considered. Understanding linkages between human vulnerabilities to climate impacts across sectors is key to identify potential transformation pathways and target coherent policy levers that maximize synergies. Unfortunately, a number of challenges including sectorial silos and lack of fisheries and agricultural research integration have hampered cross-sectorial global assessments of human vulnerability to the impact of climate change on food production systems.

Drawing on the IPCC framework of vulnerability (Marshall et al. 2010), we modelled human vulnerabilities to climate change impacts on agriculture and marine fisheries for 225 and 185 countries/states/territories, respectively (hereafter "countries"), by considering *exposure* of their food systems to climate-induced changes, socioeconomic *sensitivity* to impacted goods and services as well as *adaptive capacity*. Specifically, we projected changes in productivity of agriculture and marine ecosystems for 2100 under three greenhouse gas emission scenarios (RCP 2.6, RCP 4.5 and RCP 8.5) (*exposure*) and combined it with dependency of countries on each sector for food security, economy and employment (*sensitivity*) and capacity of countries to respond to climate impacts through assets, governance and economic flexibility (*adaptive capacity*) to generate a comprehensive index of vulnerability (Methods and Appendix E Figure 1). In contrast to previous global studies on vulnerability that are focused on a single sector (Allison et al. 2009; Mora et al. 2013, 2015; Blasiak et al. 2017; Ding et al. 2017), our approach seeks to uncover the potential synergies in policy actions that can be derived from the spatial co-occurrence of various vulnerability drivers to reduce the vulnerability of food systems globally.

Under future climate projections, tropical areas will disproportionately be exposed to fewer suitable days for agriculture by 2100 due to changing air temperature, soil moisture and solar radiation, particularly in Latin America, Central Africa and South Asia (Fig. 1). Those areas are also highly dependent on agriculture for employment, food and/or revenue. Conversely, all countries will experience changes in ocean conditions (i.e., temperature, pH, productivity and oxygen), although areas at higher latitudes (North Europe, Russia, North America) – where dependence upon seafood is relatively low – will undergo the most important changes under future climate (Fig. 1).

In the context of agriculture and, to a lesser extent, fisheries, sensitivity is negatively correlated with adaptive capacity (Appendix E; Figure 2), indicating that most sensitive countries generally have the lowest capacity to adapt to the climate impacts on food systems. Therefore, the potential impact (i.e., the combination of exposure and sensitivity) of climate change on agriculture and fisheries will be exacerbated in low latitudes, where most developing countries with weak ability to respond to and recover from the climate change lie. Drivers of vulnerability generally merge to create a "perfect storm" that leads many of most vulnerable countries to climate change impacts on agriculture to also be the most vulnerable to climate impact on their fisheries (Fig. 2).

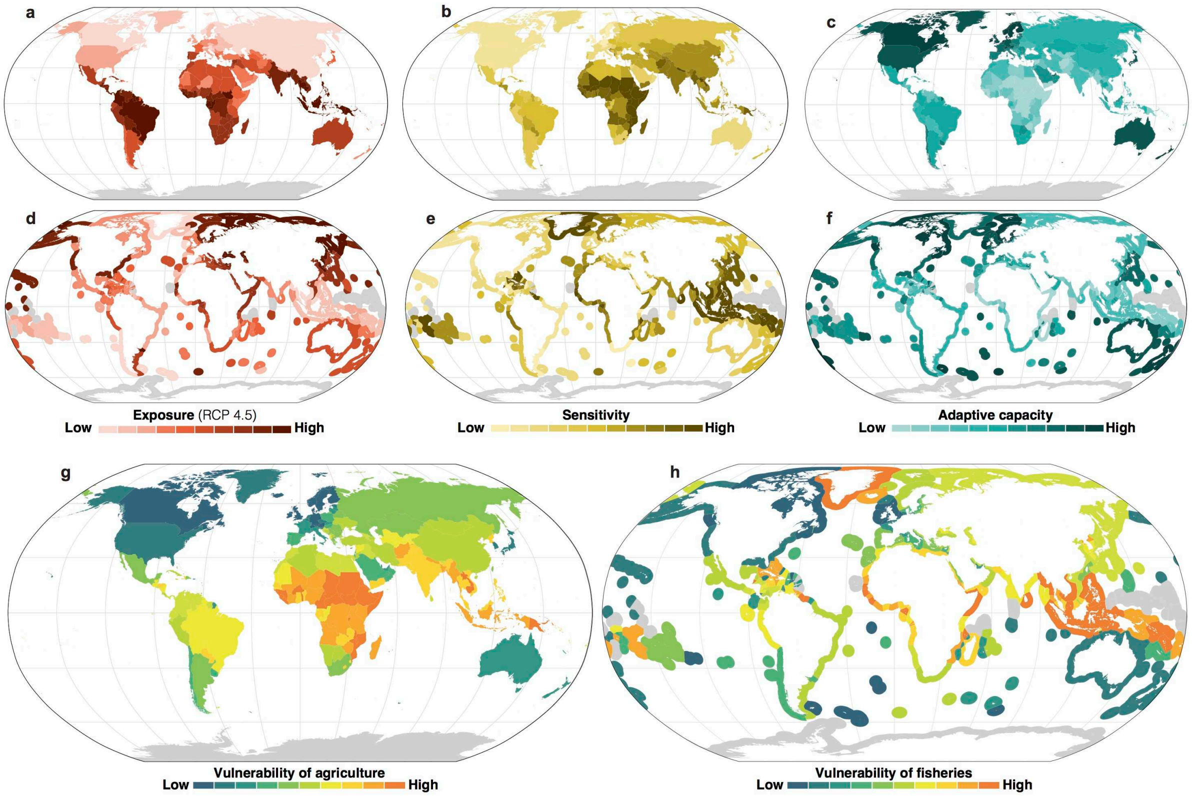

Figure 1: Vulnerability of countries to future climate change. a-f, Maps of vulnerability dimensions used to assess vulnerability to projected climate conditions under RCP 4.5 emission scenario. Average changes in suitable days for agriculture and in ocean condition within EEZs were used to estimate exposure of agriculture and fishery, respectively. Sensitivity on each sector is a composite metric of dependence for food, job and revenue. Adaptive capacity depends on assets and governance status and is identical for each sector. g-h, Countries' vulnerability obtained by combining of exposure, sensitivity, and adaptive capacity. Vulnerability of agriculture is represented on land; vulnerability of fishery is represented within EEZs. Grey indicates area removed from the analysis because at least one indicator was missing (<0.1% of world's population).

Nevertheless, these results can be interpreted as an empowering finding because this congruence of vulnerability drivers can provide a useful starting point for directing current and future synergistic climate change adaptation strategies policy. First, and although the geographical distribution of exposure varies little according to the RCP scenario, overall consequences on both agricultural and marine ecosystems can be greatly reduced if measures to reduce greenhouse gas emissions are taken rapidly. Indeed, under a business-as-usual emission scenario (RCP 8.5), almost the entire world's human population (95.5%) are projected to be directly exposed to high levels of change on at least one food production system by 2100, while 43.15% and 2.7% will fall into this category under moderate and highly successful emission cuts, respectively (Fig. 3; see details on categories of vulnerability dimensions in Methods). This change in levels of exposure could have serious repercussions on the number of people at greater risk from climate impact on food systems. Indeed, given current levels of sensitivity and adaptive capacity, around 205 million and 2.3 million could be considered highly vulnerable to climate-induced impacts on agriculture and fisheries, respectively, with 1.5 million of the poorest people that may experience the worst hardship associated with concomitant high vulnerability through both sectors under RCP 8.5. However, the number of people considered as highly vulnerable through climate impacts on agriculture will be halved (113 million) under RCP 4.5 and lowered to 22 million under RCP 4.5; while no country will combine extremely high exposure, high sensitivity and low adaptive capacity in the context of fishery under the both of these emission scenarios (Fig. 3). Although the consequences of climate change cannot be avoided in some regions of the world, they have the potential to be dramatically reduced if actions to cut of greenhouse gas emission are taken rapidly.

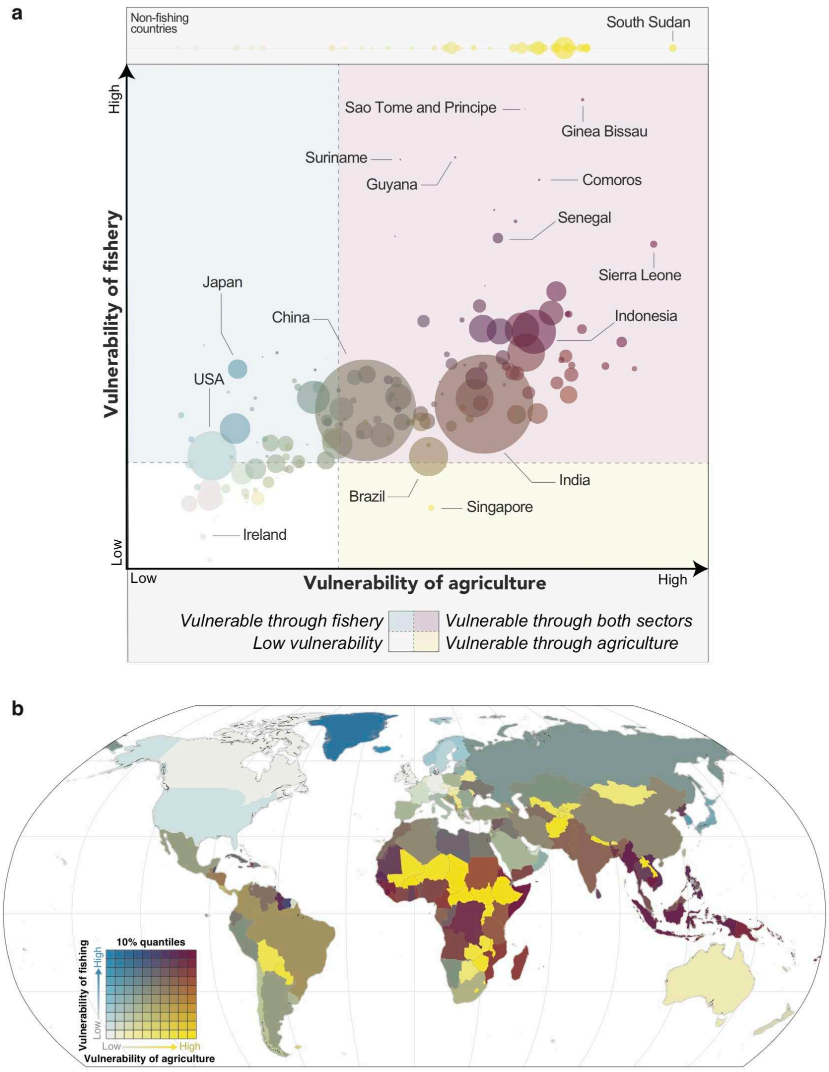

Figure 2: Cumulative countries' vulnerabilities to climate impacts on fisheries and agricultures future climate on agriculture and marine fisheries. a, Bi-plot showing the relationship between human vulnerability to the impacts of climate change on agriculture and fisheries. The median of each vulnerability delimits quadrants and are only indicative. Bubble size represents each country' population. b, Bivariate map showing linked vulnerabilities (of agriculture and fishery) for each country. Colour key is the same for both panels.

Pathways for reducing exposure to the impacts of climate change through reduced greenhouse gas emission should include global action and be long-lasting as achieving (~RCP 4.5) or even surpassing (RCP 2.6) the Paris Agreement targets have the potential to massively reduce human vulnerability to climate change on food systems (Fig. 3).

Adaptive capacity can be enhanced in a more direct and timely fashion through action at national and sub-national levels, and further supported by regional and global partnerships. Country-scale improvements in adaptive capacity components such as wealth, governance effectiveness and economic diversification will enable to implement practical actions to adapt and mitigate the effect of climate change on both agriculture and fisheries simultaneously. Specific technical and policy interventions deriving from these improvements could entail, for instance, developing sustainable intensification of agriculture and fishing practices especially in low-exposed areas, fostering climate-safe agricultural practices, improving land/resource management, increasing economic resilience to climate change, and strengthening national and local institutions (Johnson & Welch 2009; Vermeulen et al. 2012; Garnett et al. 2013; Lipper et al. 2014; Costello et al. 2016). Such adaptation strategies should be prioritized in most vulnerable countries (Figs. 1-2).

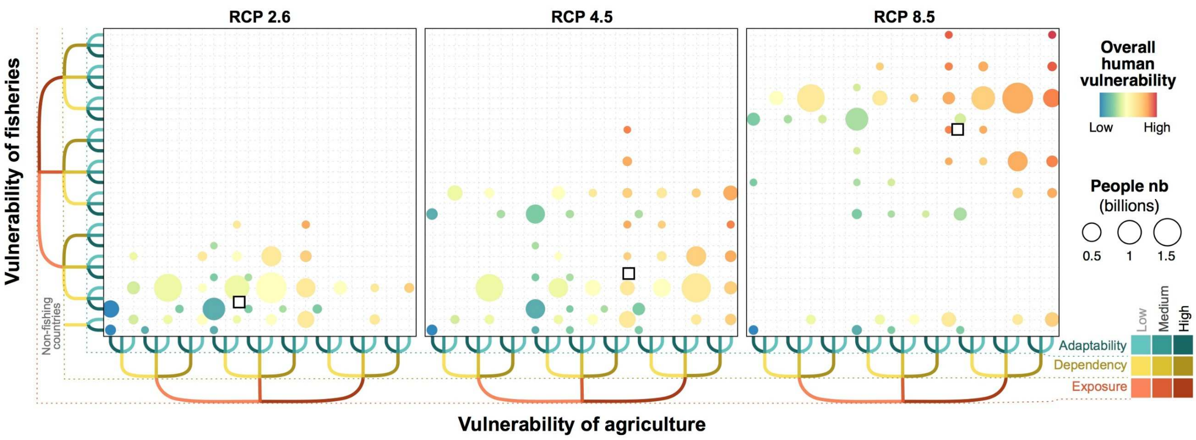

Figure 3: Change in vulnerability of humans to projected climate conditions according to different emission scenarios. This plot shows the total number of people likely to be vulnerable through exposure to changing number of suitable days for agriculture (vulnerability of agriculture) and ocean biogeochemistry (vulnerability of marine fishery) according to RCP 2.6 (left panel), RCP 4.5 (middle panel) and RCP 8.5 (right panel) emission scenarios. Black squares indicate the centroid for each scenario. Approach to categorize vulnerability dimensions as low, medium or high are described in the Methods.

Reducing countries' reliance on agriculture and fisheries sectors (i.e., decreasing sensitivity) can also affect the overall vulnerability, particularly in highly sensitive countries (Fig. 1). However, while a reduction in exposure or increase in adaptive capacity may find broad acceptance and could benefit both sectors, a reduction in sensitivity is a less clear cut as it potentially involves trade-offs or leakages from one sector to another. For instance, a reduction in the workforce employed in one sector (e.g., reducing the number of fishers) may lead to loss of livelihoods and greater unemployment, or movement into other, potentially climate-sensitive, sector like agriculture. Similarly, reducing meat consumption may increase demand for fish. Reducing production (i.e., decreasing economic dependency) will increase food insecurity, both locally (country-scale) and globally. Recently, UN Sustainable Development Goals have been adopted by the parties with the specific objectives of eliminating poverty, improving food security and reducing inequalities among countries. Pathways towards these goals will greatly help at reducing sensitivity and enhance adaptive capacity, and frameworks enabling to consider the interactions between Sustainable Development Goals (Nilsson et al. 2016) should be used to maximize synergies and minimize trade-offs among these goals (Singh et al. 2017).

Our cross-sector application of the vulnerability framework in the context of climate change reveals how the critical issue of trade-offs is at the nexus of socioeconomic development and environmental objectives. It provides some insights for implementing more integrative approaches across multilateral policy initiatives, including the United Nations Framework Convention on Climate Change (UNFCCC) and the United Nations Sustainable Development Goals. With well targeted policies, agriculture and fisheries can move onto more sustainable pathways, resulting in decreased risk in food-insecure regions in the short term while contributing to reducing climate change as a global threat over the longer term.

Methods

Vulnerability dimensions

Each vulnerability dimension (exposure, sensitivity and adaptive capacity) was evaluated using a set of quantitative indicators at the country-level. The exposure of countries to the impacts of climate change on agriculture and marine fisheries was calculated based on three different representative concentration pathways (RCPs), which provided insight into exposure levels in the case of highly successful reduction of greenhouse gas emissions (RCP 2.6), more modest emissions reductions (RCP 4.5), and a continued increase in carbon emissions (RCP 8.5) at 2100 (van Vuuren et al. 2011). The average change in suitable days for plant growth under projected change in temperature, soil moisture and solar radiation (Mora et al. 2015) was used as the exposure for the agriculture sector. Similarly, exposure of marine fisheries was determined as the average absolute change in temperature, pH, productivity and oxygen (Mora et al. 2013) within each country's Exclusive Economic Zone (EEZ) under the assumption that the magnitude of changes of co-occurring multiple stressors translates linearly into impacts on marine catch and associated human populations. This is a reasonable assumption given the accumulated evidence that changes in each of these parameters affects a variety of physiological, biological and ecological mechanisms (Scheffers et al. 2016) that will modify the productivity and catch composition, ultimately affecting the associated capture fishery. Sensitivity was assessed by combining the country-scale contribution of each sector to economy (revenue dependency), employment (job dependency) and food security (food dependency). Percentage of GDP contributed by agricultural (FAO 2014b) and seafood landings (Pauly 2007) revenue were respectively used as metrics of economic dependency to agriculture and fisheries, percentage of the workforce employed by agriculture (FAO 2014b) and fisheries (Teh & Sumaila 2013) were respectively used to measure job dependency on agriculture and fisheries sectors, and percentage of net primary production appropriated by people (Imhoff et al. 2004) and fraction of consumed animal protein supplied by seafood (FAO 2014b) were used as indicators of food dependency to agriculture and fishery, respectively. Finally, adaptive capacity was quantified for each country by combining standardized indicators of assets, governance status and flexibility. Per capita GDP (CIA 2015) was used as a measure of countries' overall assets while a set of governance indicators including voice accountability, political stability, government effectiveness, regulatory quality, rule of law and control of corruption was used to estimate governance status (The World Bank Group 2015). Countries' export diversification (IMF 2014) was used to quantify economic flexibility. Therefore, our indicators of adaptive capacity reflect the overall countries' ability to facilitate existing of agriculture and fisheries as well as their capacity to obtain food from elsewhere. Figure 1 in Appendix E provides a summary of vulnerability dimensions and their corresponding indicators.

Missing data

Main data sources (Appendix E ;Table 1) allowed to estimate vulnerability of 97.8% of the world's population. Territories and dependencies with missing data were assigned their sovereign's values, which allowed to raise the total proportion of the population represented to 98.4%. Finally, the remaining 1.6% was imputed using boosted regression trees (i.e., boosted regression trees were used to predict each individual indicator using all other indicators), with the exception of a few areas (<0.1% of total population) for which one indicator (change in the number of suitable days for agriculture) was not imputed as it could not be treated as a regression problem (i.e., it depends on future climatic conditions rather than current countries' socioeconomic and governance indicators). Overall, our dataset covers 225 and 185 countries/states/territories for agriculture and for fishery, respectively, thus providing the most comprehensive assessment to date of vulnerability due to climate change impacts on global food systems

Aggregated vulnerability index

In order to combine each vulnerability dimension (exposure, sensitivity and adaptive capacity) into a single measure of vulnerability per sector and per emission scenario, we first normalized all the indicators (Appendix E; Table 1) to a scale ranging from 0 (lowest contribution to vulnerability) to 1 (highest contribution to vulnerability), so they could be compared. Then, each indicator was aggregated into its corresponding vulnerability dimension by calculating the unweighted average of the standardized indicators. Finally, we used the TOPSIS method (Technique for Order Preference by Similarity to an Ideal Solution), which expresses vulnerability as the relative distance to the positive (all dimensions = 0) and negative (all dimensions = 1) ideal solutions in the Euclidean space. Each vulnerability dimension had a weight of 1, assuming that they contributed equally to vulnerability (see (Parravicini et al. 2014) for a detailed application of this approach in a vulnerability context).

Categories of vulnerability dimensions

We categorized each country as having "low", "medium" or "high" exposure, sensitivity and adaptive capacity (Fig. 3). In the context of agriculture, each country was categorized as "low", "medium" or "high" whether they experience <10%, 10 to 30% or >30% reductions in suitable plant growing days, while cumulative changes in ocean condition were divided into three equal bins to classify countries with "low", "medium", and "high" exposure of marine fisheries was determined. For dependency – whether on agriculture or fisheries – the three categories were determined if their cumulative percentages in those three sub-components (job, revenue and food dependency) ranged from 0% to 33% ("low" dependency), >33% to 66% ("medium" dependency), or>66% ("high" dependency). The three adaptive capacity indicators were normalized (0-1), added together and the resulting composite index of adaptive capacity was then divided in three equal bins to indicate countries of "low", "medium", and "high" categories.

Caveats and limitations of the approach

Our study provides the most comprehensive analysis of the relative vulnerabilities of countries to the impacts of climate change on agriculture and marine fisheries at a global-scale. However, there are several caveats about our approach and methodology that are important to acknowledge. First, we emphasise that our country-scale, policy-specific vulnerability metrics, does not aim to quantify the impacts but rather, to identify key dimensions driving vulnerability, thus enhance understanding of how each sector and their dependent societies will experience the consequences of climate change, so facilitating action to support human well-being. A limitation of this study is related to tele-coupling and increased globalization that make developed countries dependent on developing countries for agriculture products (MacDonald et al. 2015; Tombe 2015) wild fish provision (Sumaila et al. 2016b), resulting in distal exposures of climate changes that are potentially underestimated (exposure) by our approach. Although quantifying tele-coupling falls outside the scope of this study, this aspect remains reflected in the socioeconomic variables used to calculate sensitivity (e.g., % of NPP appropriated by people and fish protein as a proportion of all animal protein), which provide general estimates of international trade and enable comparability across countries, but mask the distal effect that climate change may have on importing countries. Importantly, others sectors, including freshwater fisheries and aquaculture also have significant importance globally (McIntyre et al. 2016; Sumaila et al. 2016a) and may interact with agriculture and fisheries in many ways. Thus, further development in cross-sectorial global vulnerability assessments will benefit from the inclusion of others sectors to provide a more comprehensive picture of the vulnerability of global food systems to the impact of climate change. Finally, adaptive capacity and sensitivity are based on a static view of the socioeconomic and institutional features, while those aspects are also highly dynamics. Further refinements of the method could incorporate population and development trajectories into assessment of the impacts of climate change on food systems.

GENERAL DISCUSSION

General discussion

Operationalizing the vulnerability framework

Conservation and sustainable management of natural resources are endeavors to influence human behavior to ensure the persistence of an ecosystem or a subset of its components in a way that benefits – directly or indirectly – to human well-being. They involve the many challenges of decision-making in a complex and dynamic world, and hence require a transdisciplinary perspective (Leenhardt et al. 2015). Yet, significant barriers continue to hinder research efforts that integrate expertise across fields of study and meet practitioner's needs. Among these barriers is the difficulty of balancing a complex picture of social-ecological systems with the establishment of straightforward, transferable, meaningful and scalable methods that can be readily understood and implemented by policy-makers and practitioners (Blythe et al. 2017b; Olander et al. 2017). Tremendous theoretical and practical progresses have been made in respect to our understanding of resilience, common-pool resources, and complexity within social-ecological systems (Ostrom 1990, 1999, 2009; Berkes et al. 2003; An et al. 2005; Folke et al. 2005; Chapin et al. 2010; Parrott & Meyer 2012; Nyborg et al. 2016; Hughes et al. 2017). Similarly, we understand better the role of, and linkages between ecological condition, resource dependency, social adaptive capacity and how successful policy interventions depend on and influence those aspects (Béné et al. 2007, 2016; Cinner & Bodin 2010; Mills et al. 2011; Armitage et al. 2012; Cinner et al. 2013b; McClanahan et al. 2015). And yet, translation of this extensive knowledge into operational tools that match practitioners' needs remains incomplete.

Drawing from the extensive research efforts made in diverse disciplines (Allison & Ellis 2001; Berkes et al. 2003; Basurto et al. 2013), and applying them to address the wicked problems of managing complex social-ecological systems (DeFries & Nagendra 2017) have been the essence of this research project. We focused on the development of innovative and interdisciplinary analytical approaches to link science and policy, and ultimately achieve more effective management. The particularly contrasting case studies used in this research project have allowed for a unique examination of the interactions between ecological and social systems, and to develop operational approaches for more holistic, systems-based approaches to management.

The use of the Progressive-Change BACIPS approach in Moorea (Chapter I) helped us to shed light on the ecological effects of the existing management plans, at the source of conflicts between managers and users. In Chapter II we showed how the use of mixed methods combining different data sources allowed us to map fishing effort in the challenging context of Moorea where fishing is diffuse among inhabitants and along the coast. Using this information within a vulnerability-framework (Chapter III) we mapped resource-user dependencies, to highlight focal areas for management interventions in Moorea. Then, in Chapter IV, we developed a framework combining spatial social and ecological vulnerabilities to recommend interventions portfolio in those focal areas, specifically targeting each social and ecological conditions within each management spatial unit. In Chapter V we showed how vulnerability assessments in two different times can be used to capture combined changes in social-ecological systems in response to direct and indirect drivers. This approach precluded us to identify the specific response of the social-ecological system to each of the drivers, therefore, in Chapter VI, we developed a framework to disentangle the effect of direct and indirect drivers on the vulnerability of communities involved in the management of common-pool resources through TURFs, scaling-up our vulnerability assessments. Finally, in Chapter VII, we assessed how countries are vulnerable to the effects of climate change on agriculture and fisheries globally to show how vulnerability assessments can be suited in a cross-sectorial management context.

Operationalization, adaptation and extension of the vulnerability framework to various scales and contexts required innovative approaches that have both conceptual and practical implications, thus contributing to improve our understating of social-ecological interactions and assisting practitioners to apply social-ecological science in real-world contexts.

Methodological implications

As strategic management of the human-nature relationship has nerver been more important, there is a pressing need to develop "approaches that include social and biophysical data, and explicitly assess human-environmental interactions, taking into account their dynamic nature", also known as "integrated social-ecological assessments" (Kittinger et al. 2014). Tremendous advances have been made as concerns the development and use of social and ecological indicators. Broadly speaking, indicators describe what exists, and in doing so, they define what is important. Hence, appropriate frameworks are needed to determine and articulate social and ecological indicators in a meaningful and accurate way. When available and well-designed, social and ecological vulnerability indicators (i.e., exposure, sensitivity and adaptive capacity) can represent different but complementary facets of social-ecological systems (Cinner et al. 2013a). This provides justification to our effort to identify indicators describing these different dimensions, and to compile them into aggregated measures of relative vulnerability (Chapters III-VI) or into a more nuanced understanding of the specific contributions to vulnerability (Chapters IV and VI).

Although widely used, aggregated vulnerability analyses at any scale invariably make assumptions on transformation type, aggregation formula and weighting that are difficult to verify on the ground (Adger & Vincent 2005; Allison et al. 2009; Cinner et al. 2013a; Parravicini et al. 2014; Aretano et al. 2015; Himes-Cornell & Kasperski 2015). Despite recognition that inference and management decision based on (vulnerability) models require an understanding of the uncertainty induced by these assumption in model outputs, a surprisingly few number of studies actually considered this issue (but see Allison et al. 2009 and Cinner et al. 2013b for efforts to deal with some of these issue). The uncertainty analysis performed in Chapter III enabled model uncertainty to be quantified and accounted for, which allowed for the first time model interpretation to be based upon a number of models (with different combinations of factors) rather than one model that is hypothesized to be the best based on lack of sound evidence.

Mapping and combining key indicators at the appropriate scale is particularly powerful not just to translate a complex phenomenon into simple measures, but also for stakeholder engagement and policy recommendations. Until recently, mapping of relevant ecological indicators has lagged behind spatial assessments of classically used socioeconomic variables. New developments in spatial and statistical modelling techniques of biophysical indicators (Knudby et al. 2007, 2011) have opened a window of opportunity to better integrate quantitative social and ecological data spatially (Stephanson & Mascia 2014). Building on these advances, Chapters II-V combined a great variety of data types (interviews, socioeconomic surveys, satellite imagery, underwater visual surveys, etc.), modelling techniques (complex decisions analysis, machine learning for regression problems, GIS procedures, etc.), which allowed for a truly systematic and balanced spatial social-ecological assessment at fine-scale. This innovative approach marks a departure from the classically used spatial approaches that tended to focus either on the impacts of human use on ecosystems (Halpern et al. 2008, 2015a; Selkoe et al. 2009; Tulloch et al. 2015) or on the benefits humans derive from the ecosystems (Yates & Schoeman 2013; Hashimoto et al. 2014; Ramirez-Gomez et al. 2015). In this view, the adaptation of the social-ecological vulnerability framework to resource-user interactions (Chapters III-V) illustrates how social and ecological data can be considered and linked together, and shows how interdisciplinarity can be implemented to improve the representation of primary direct interactions occurring within social-ecological systems (Kittinger et al. 2012; Fischer et al. 2015).

The conceptual and methodological advances presented here provide complementary insights with previous interdisciplinary and quantitative studies on the importance of integrating social and ecological data (Christie et al. 2005; McClanahan et al. 2008a; Cinner et al. 2012a; Koehn et al. 2013; Kittinger et al. 2014; Le Cornu et al. 2014). Indeed, evidences from the present work suggest that adoption of a more integrative and context-grounded lens of human-nature dependencies provides information that might otherwise be missed. Chapter II shows how ignoring key aspects of a system (in this case "fishing suitability") may far be oversimplified and result in flawed estimations of the fishing footprint, ultimately affecting planning decisions. Further, in Chapter III, we demonstrated how focusing solely on fishing effort (the focus of Chapter II) to identify management priorities – a common practice in spatial planning (Ban & Klein 2009; Ban et al. 2011; Hamel et al. 2017) – could fail to represent what is truly at stake because some areas are more resilient than others, or because adjacent households might not have the same adaptive capacity to fishing restrictions.

Social and ecological systems intertwine with one another and create spatial patterns, but these interactions are also dynamic. Beyond the development of an improved approach to link social and ecological data and incorporate spatial patterns, this study shows how vulnerability can provide a better understanding of social-ecological changes over time as a result of exposure to multiple drivers. First, the Before-After analysis performed in Chapter V yields, for the first time, key insights on the system-scale responses of human-nature interactions following major external and interacting drivers of change. As an increasing number of practitioners seek to measure the effect of management interventions on both social and biophysical components (Gurney et al. 2014), temporal assessment of social-ecological vulnerability dimensions provides new perspective for assessing the impact of management interventions on the social-ecological system. Second, the present work demonstrates how multiple drivers can be explicitly included into vulnerability assessments (Chapter VI). More than an operational fine-tuning of the IPCC framework of vulnerability (Marshall et al. 2010), this stresses the need for considering vulnerability as specific to the drivers of changes (Adger 2006; Gallopín 2006; Adger et al. 2009b; Cinner et al. 2012a, 2013a).

Management implications

Overall, this PhD thesis demonstrates how consideration of social and ecological systems, their interaction, their spatial distribution and the backdrop of multiple drivers against which they play out are amenable to concrete management interventions and policy actions, and may ultimately represent an opportunity to move away from tame solutions to wicked management problems (DeFries & Nagendra 2017).

Vulnerability is often considered as a unifying theme of theoretical and practical research on social-ecological systems management. The various tools based on (social-ecological) vulnerability introduced in the various chapters of this thesis, and particularly in Chapters IV and VI, support this statement as they put together a variety of key lessons learned from various disciples, practices and research lineages in a structured and transparent way. For example, insights gained from the Sustainable Livelihood Approach as applied to fisheries (Allison & Ellis 2001) enabled to target livelihood-based solutions to address underlying sources of social vulnerability (Chapters IV and VI). Similarly, a large part of policy lever recommended by the social-ecological vulnerability-based management framework (Chapter IV) are drawn from the Ecosystem Approach to Fisheries (Garcia & Cochrane 2005).

The resulting key outputs obtained through the diverse case studies (i.e., current or future vulnerabilities, the vulnerability trajectory and the vulnerability profiles) enabled to (1) identify where management should focus in priority (social, ecological or social-ecological hotspots); (2) understand the responses of linked social-ecological vulnerabilities following significant exposure to drivers of change; and (3) derive detailed opportunities for action to address key drivers of vulnerability, and reduce overall vulnerability while fostering synergies over trade-offs, thus leading to more effective and equitable management strategies (Table 1).

Whether vulnerability assessments should be used at early or late stages of the management process is sometimes debated (Adger 2006; Cash et al. 2006; Gallopín 2006; Smit & Wandel 2006; Metcalf et al. 2015). Most core chapters in this research place vulnerability assessments at the outset of a project design and scope (e.g., "assessment" and "planning" components of the United Nations Framework Convention on Climate Change, UNFCCC, adaptation process). For instance, we identified "social-ecological hot-spots" in Chapter III with the goal to identify places of first concern for local managers and decision-makers (Table 1).. Similarly, Chapter IV uses social-ecological vulnerability assessments in order to determine a set of practical management actions based on the extensive work in the applied social-ecological research. In addition, the set of policy actions that can be derived from vulnerability assessments are extended to multiple drivers in Chapter VI, and Chapter VII – which is largely based of projections of future climate scenario – also place vulnerability assessment in the "assessment" and "planning" stages of the adaptation process (Table 1). Chapter V, on the other hand, is a first attempt to operationalize social-ecological vulnerability assessment both at the early ("assessment" and "planning") and late phases ("monitoring and evaluation") of the management process. It provides for the first time a dynamic understanding of spatial variation in vulnerability (Maynard et al. 2015) that may be used to understand the effect of external drivers of change, implement adaptation measures and/or monitor and evaluate the effect of adaptation or mitigation measures (Table 1).

Overall, findings are context-specific (Thiault & Claudet 2016) but the contributions of the present work to address social-ecological systems management as a wicked problem are generic (Table 1).

Perspectives

This thesis project focused on examining vulnerability in a quantitative manner. Most avenues for future research directions relate to how indicators are selected and combined to either capture exposure, sensibility or adaptive capacity.

Fine-tuning indicators to socio-cultural contexts

A diversity of social data and principles such as equity, power, legitimacy and agency are relevant to manage for sustainability (Hicks et al. 2016b). Similarly, cultural perspectives encompassing values, knowledges, and needs (Sterling et al. *in press*) are particularly important to evaluate resource use and human well-being, in particular at local scales. While these issues remain critical for research and practice, here we focused specifically on how practitioners can provision management and policy with adequate social and ecological data, presenting guidance that is generalizable across different scales, contexts, and levels of institutional capacity. Future efforts to incorporate culturally-grounded indicators and key social principles in social and social-ecological vulnerability analyses would greatly benefit initiatives to foster interdisciplinarity and establish more effective management and policy interventions.

Table 1: General contributions of this PhD thesis to addressing wicked problems faced by managers.

Management question	Management challenge	Contribution of the present work
How to assess effectiveness of interventions?	Natural spatiotemporal variability may hinder true effect of intervention.	Progressive-Change BACIPS analysis allow isolating the effect of interventions from other sources of variation; this require appropriate monitoring design (Chap. I).
Where critical actions should be taken?	Mismatch in ecological and social data; feedbacks between social and ecological systems; spatial heterogeneity.	Linking social and ecological vulnerabilities enables to represent key human-nature interactions; depending on objectives, interventions can focus on reducing either or both social and ecological vulnerability (Chap. II & III)
How to adapt management of a social-ecological system in a constant change?	Multiple drivers lead to a variety of trajectories that may make a given intervention unsuitable in the future.	Analysis of vulnerability spatiotemporal trajectories highlights places that experienced decreased and increased vulnerability and provides system-scale insights on the consequences of multiple co-occurring drivers (Chap. V).
What actions are likely to most efficiently achieve a given objective in a given context?	Important spatial heterogeneity of social and ecological components leading to uncertain outcomes.	Vulnerability-based management can help avoid trade-offs among management actions in a structured manner; it can also promote institutional interplay to include multiple institutions (Chap. IV & VI).
How to design interventions in a highly inter-connected world?	Institutional misfit can lead to a mismatch between drivers of change and actions made locally.	Impact of large-scale drivers can be attenuated through coherent actions locally; focusing on various impact pathways, and differentiating between *general* versus *specific* aspects of vulnerability in a multiple driver context enables more coherent decision making (Chap. VI).
How to coordinate multi-sector decision making?	Services from various ecosystems are not factored into decisions about single sectors.	Cross-sector vulnerability assessments help acknowledging trade-offs and identifying potential for synergies (Chap. VII).

Accounting for vulnerability's multidimensionality

A recurrent caveat of quantitative vulnerability analyses is the lack of empirical justification to determine how the three key dimensions (exposure, sensitivity and adaptive capacity) should be aggregated into a single index of vulnerability. Indeed, it is often stated that each dimension has equal importance in driving vulnerability (Belliveau et al. 2006; Adger et al. 2009a; Marshall et al. 2010). This has resulted in the creation of composite vulnerability metrics based on equal weights among vulnerability dimensions and a choice between an additive or multiplicative aggregation formula that is rarely soundly justified. In this thesis, for instance, two different formulas were used: multiplicative (Chapters III-V) or the TOPSIS method (Technique for Order Preference by Similarity to an Ideal Solution; Chapters VI-VII). Yet, most of the time, the underlying assumptions made (e.g., "vulnerability is constrained by the dimension with the lowest value", etc.) cannot be verified, which reflects the lack on information of the system and on how vulnerability dimension interact. Consequently, aggregated vulnerability metrics may provide a false sense of what is really at higher risk and where management should be directed in priority. Such potentially biased information with regard to aggregated vulnerability outputs (whether social, ecological or social-ecological) are shared with the large body of literature on cumulative impact assessments on ecosystems (Halpern et al. 2008; Selkoe et al. 2009; Korpinen et al. 2012; Maxwell et al. 2013; Micheli et al. 2014; Korpinen & Andersen 2016). Drawing on analytical advances in this field of research (Stock & Micheli 2016), we performed in Chapter III the first quantification of uncertainty underlying an aggregated model of vulnerability to date. Although outputs were demonstrated as robust in our case, further empirical research efforts are needed to effectively and reliably investigate cause-effect relationships, non-linearity and potential tipping-points between vulnerability dimensions (Bunce et al. 2010; Bennett et al. 2014b; Foley et al. 2015; Selkoe et al. 2015). In parallel, analysts seeking to implement quantitative vulnerability assessments should systematically state the assumptions of their models and follow good-practices guidelines (Cinner et al. 2013b) to ensure the reliability and transparency of their models' outputs.

Toward more integrative vulnerability assessments

This research has highlighted how quantitative, interdisciplinary vulnerability assessments can be relevant for planning, management and policy. However, in the light of the caveats and assumptions highlighted above, outputs from such models be greatly complemented. In particular, other formal modelling approaches like agent-based modelling (An et al. 2005; Bousquet et al. 2005; An 2012) could help dealing with complex dynamics (e.g., non-linearity, emergent and dynamic behavior, cross-scale linkages) and, by doing so, would provide the option to build a layer of complexity over the basic IPCC vulnerability components with complementary considerations and perspectives that might have been missed otherwise. In addition, coupling with institutional diagnostic approaches (Ostrom 2007, 2009; Basurto et al. 2013; Blythe et al. 2017a) could represent a great opportunity to work through complexity by incorporating governance processes and outcomes. These research areas still remain under explored, but progress toward the development of hybrid methodologies and exploratory modelling are needed to make vulnerability assessments more inclusive and relevant in the future, and ultimately help support the things that we care for and wish to sustain.

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APPENDICES

Appendix A (Chapter I)

Table 1: Results of the surveys performed with 10 local experts on their perception of the surveillance effort of each MPA. 1- inexistant surveillance, 5-high surveillance effort.

	Tiahura	Pihaena	Aroa	Nuarei	Motu Ahi	Maatea	Tetaiuo	Taotaha
exp1	3	4	2	3	3	2	2	2
exp2	3	4	2	3	2	2	1	1
exp3	3	4	2	2	2	2	2	2
exp4	4	4	3	4	4	3	3	3
exp5	3	3	3	2	2	2	2	2
exp6	3	4	2	3	2	2	2	2
exp7	4	3	3	3	2	2	2	2
exp8	4	5	4	4	4	4	4	4
exp9	5	3	3	4	3	3	4	4
exp10	2	4	2	2	2	2	2	2
Average	3.4	3.8	2.6	3	2.6	2.4	2.4	2.4

Table 2: Probability of detecting a change in delta equal to 0.7 (i.e., a 100% increase in density or biomass in response to MPA establishment for harvested and non-harvested fishes) in two habitats.

	Lagoon				**Fore reef**			
	Harvested fish		Non-harvested fish		Harvested fish		Non-harvested fish	
	biomass	density	biomass	density	biomass	density	biomass	density
Tiahura	47.2	46.9	82.3	75.5	97.9	51.4	99.9	91.3
Tetaiuo	68.7	44.8	93.3	68.3	97.6	66.6	100	99.5
Taotaha	63.9	42.4	82.7	87.4	80.6	48.0	93.0	99.0
Pihaena	82.0	42.1	83.9	96.6	92.4	50.3	86.7	99.5
Aroa	59.1	50.0	47.4	86.8	80.4	58.7	68.3	96.3
Fully protected	99.2	92.2	99.0	99.5	100	99.4	100	100
Nuarei	63.8	16.4	78.8	58.5	83.6	77.7	96.8	98.6
Motu Ahi	65.6	67.1	56.5	56.5	82.1	66.4	98.9	99.2
Maatea	65.6	61.2	72.1	70.2	96.9	57.4	97.6	97.7
Moderately protected	97.2	65.1	97.1	96.7	99.7	96.0	100	100
Entire network	100	97.9	100	100	100	100	100	100

Appendix B (Chapter II)

S1 Table: List of Moorea's districts and corresponding dependency on marine resource. High value: high dependency; low value: low dependency. District IDs can be found at http://ispf.pf.

The file can be downloaded at the following link: https://ndownloader.figshare.com/files/8405447

S2 Table. Outputs of the AHP ranking exercise. Code for columns A-O is criteria_subcriteria. Inconsistency score: 0-high inconsistency; 1-high consistency.

The file can be downloaded at the following link: https://ndownloader.figshare.com/files/8405462

S1 File. Survey questionnaire used to quantify fishers' preference for fishing grounds. The ranking exercise is based on the Analytic Hierarchy Process (AHP) decision-making methodology and enables to measure the importance of sub-criteria weights in fishers' fishing ground selection.

Fisher ID : ……………………… Interviewer: …………………………
Date: … / … / …….

Ranking exercise

2.1 Substrate

If we only consider seafloor bottom, which criteria is the most important when going fishing in general? 1 - Same importance, 3 - Moderate importance, 5 - Strong importance, 7 - Very strong importance, 9 - Extreme importance

Coral					Algae			
9	7	5	3	1	3	5	7	9

Sediment					Algae			
9	7	5	3	1	3	5	7	9

Coral					Sediment			
9	7	5	3	1	3	5	7	9

2.2 Depth

If we only consider depth, which criteria is the most important when going fishing in general?
1 - Same importance, 3 - Moderate importance, 5 - Strong importance, 7 - Very strong importance, 9 - Extreme importance

0-3m					> 8m			
9	7	5	3	1	3	5	7	9

3-8m					> 8m			
9	7	5	3	1	3	5	7	9

0-3m					3-8m			
9	7	5	3	1	3	5	7	9

2.3 Slope

If we only consider slope, which criteria is the most important when going fishing in general?
1 - Same importance, 3 - Moderate importance, 5 - Strong importance, 7 - Very strong importance, 9 - Extreme importance

Low					High			
9	7	5	3	1	3	5	7	9

Medium					High			
9	7	5	3	1	3	5	7	9

Low					Medium			
9	7	5	3		3	5	7	9

2.4 Distance to coast

If we only consider distance to coast, what is the most important when going fishing in general?
1 - Same importance, 3 - Moderate importance, 5 - Strong importance, 7 - Very strong importance, 9 - Extreme importance

0-400m					> 1000m			
9	7	5	3	1	3	5	7	9

400-100m					> 1000m			
9	7	5	3	1	3	5	7	9

0-400m					400-1000m			
9	7	5	3	1	3	5	7	9

Fisher ID : ………………………. Interviewer: …………………………
Date: … / … / …….

2.4 Distance to reef passages

If we only consider distance to reef passages, which criteria is the most important when going fishing in general? 1-Same importance, 3-Moderate importance, 5-Strong importance, 7-Very strong importance, 9-Extreme importance

0-250m					> 1000m			
9	7	5	3	**1**	3	5	7	9

> 1000m					250-1000m			
9	7	5	3	**1**	3	5	7	9

0-250m					250-1000m			
9	7	5	3	**1**	3	5	7	9

2.5 Criteria

All criteria considered together, which is the most important when going fishing in general? 1 - Same importance, 3 - Moderate importance, 5 - Strong importance, 7 - Very strong importance, 9 - Extreme importance

Depth					Dist shore			
9	7	5	3	**1**	3	5	7	9

Depth					Dist pass			
9	7	5	3	**1**	3	5	7	9

Depth					Slope			
9	7	5	3	**1**	3	5	7	9

Depth					Substrate			
9	7	5	3	**1**	3	5	7	9

Dist shore					Dist pass			
9	7	5	3	**1**	3	5	7	9

Dist shore					Slope			
9	7	5	3	**1**	3	5	7	9

Substrate					Dist shore			
9	7	5	3	**1**	3	5	7	9

Slope					Dist pass			
9	7	5	3	**1**	3	5	7	9

Substrate					Dist pass			
9	7	5	3	**1**	3	5	7	9

Substrate					Slope			
9	7	5	3	**1**	3	5	7	9

1 Additional remarks

………
………
………
………

S2 File. Extended methods. Creation of the spatial information for mapping criteria and sub-criteria.

Satellite data was used along with acoustic depth measurements and seafloor data, to respectively predict depth and habitat composition at any 5 x 5m pixel of the lagoon (see details bellow). Areas where the models did not perform well (i.e., deep areas and/or turbid areas) were removed from the analysis. Coastline and reef crest were also extracted from the satellite image using GIS procedure. Five criteria, each divided into three sub-criteria, were finally identified following preliminary interviews with fishers and represented spatially based on the digital spatial models described above (Figure 1-6).

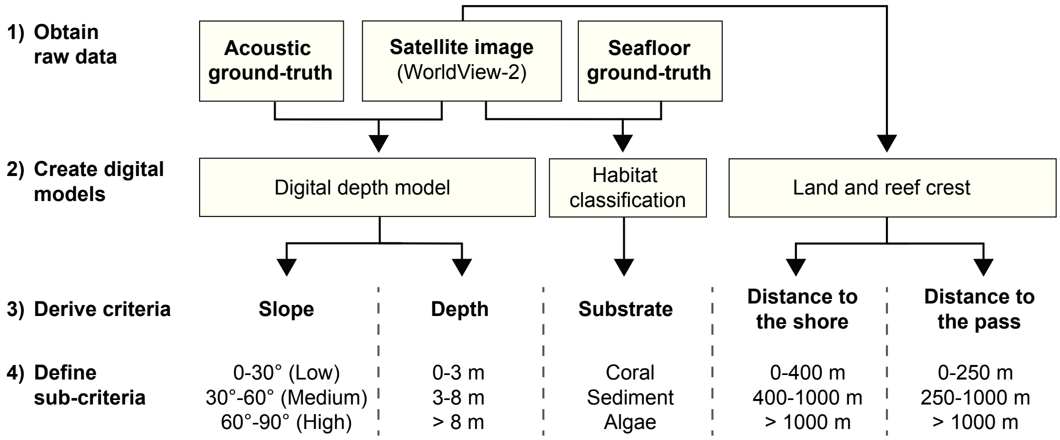


Figure 1: Flow diagram illustrating the process used to create the spatial information used to map fishing suitability. Details are given bellow for each criteria.

Depth was mapped using a combination of accurate sonar soundings and continuous spaceborne imagery. A total of almost 16,000 soundings spanning the first 20 m water depths was collected in January 2011 with a small boat provided with a combo 200-kHz echosounder / 12-channel GPS receiver. In parallel, a very high resolution Pleiades-1 imagery composed of four multispectral bands (blue, green, red and near-infrared with 2-m pixel size) and one panchromatic band (0.5 m pixel size) was acquired on 23 June 2014. Following geometric, radiometric corrections and pansharpening procedure (see Collin and Hench 2015 for further details), the 0.5-m visible multispectral dataset was used to train and validate a neural network model standing for the bathymetric model. Very high agreements (r=0.89, R^2=0.8 and RMSE=2.44 m) between predicted and actual ground-truth were found out so as to build a digital depth model (DDM) of Moorea bound by 0 and 20 m. The DDM was resampled at 5-m spatial resolution and turbid areas or areas below 12 meters (19.5 % of the total reef area), were removed from the analysis because of the growing uncertainty with depth regarding the substrate (Figure 2).

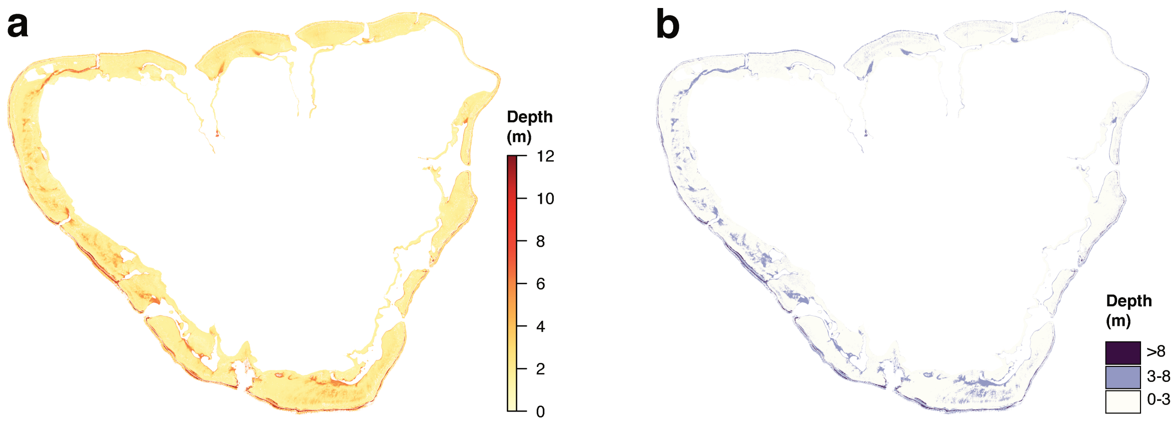


Figure 2: Maps of "depth" criteria. (a) Continuous scale. (b) Categorized according to the sub-criteria boundaries.

Distance to the shore was calculated in meters and expressed as the minimum distance of each 5 x 5 m pixel to the nearest coastline (Figure 3). This criterion expresses the minimum distance fishers must travel to fish in this pixel. Since the fore reef in only accessible by the passes, we used the r.cost.full function in GRASS to force the distance calculation to account for the reef crest.

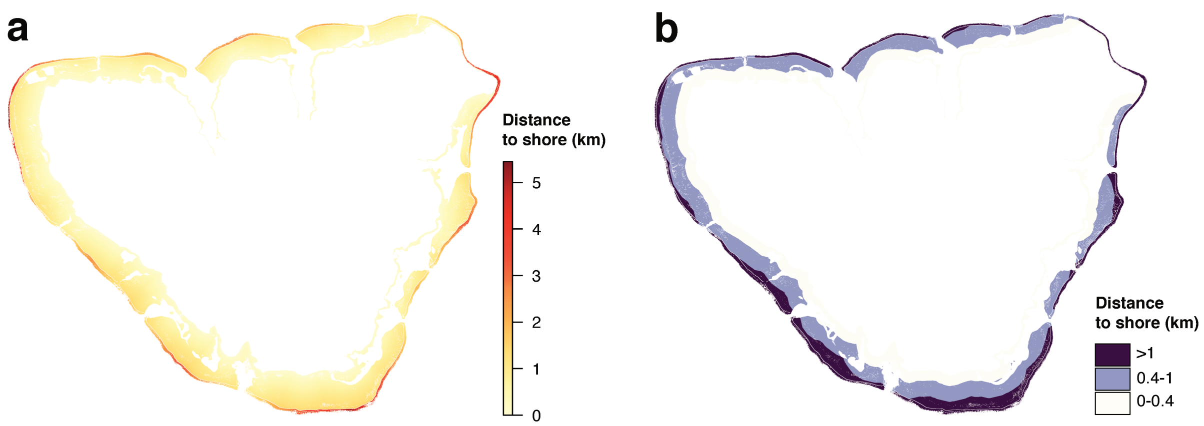


Figure 3: Maps of the "distance to shore" criteria. (a) Continuous scale. (b) Categorized according to the sub-criteria boundaries.

Distance to the pass was calculated in meters and expressed as the minimum distance of each 5 x 5 m pixel to the nearest pass (Figure 4).

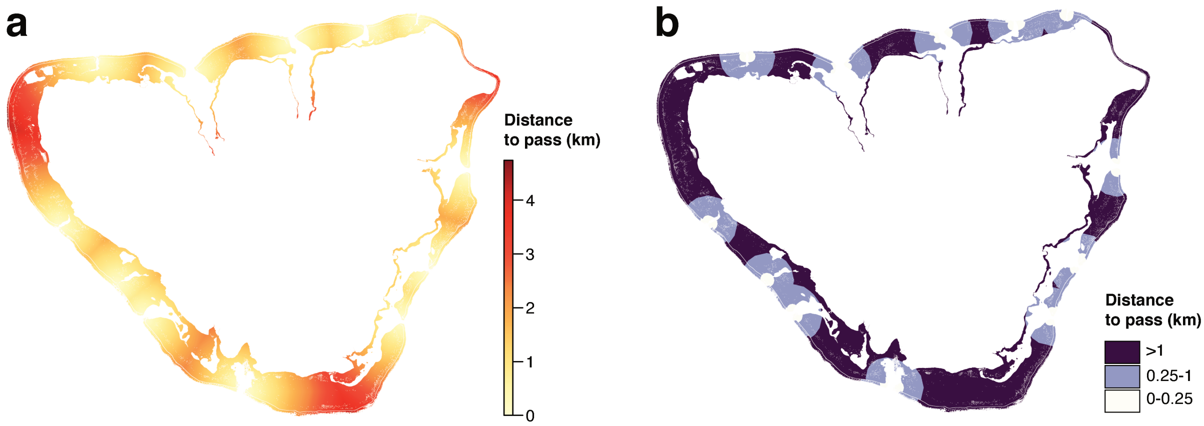


Figure 4: Maps of the "distance to pass criteria". (a) Continuous scale. (b) Categorized according to the sub-criteria boundaries.

Substrate were mapped using a synergy of punctual photoquadrats and seamless spaceborne imagery. An array of 897 geolocated ground-truth pictures were gleaned from June 2010 to November 2011 using a small boat adapted to navigate in very shallow waters. Following a thorough examination of the pictures, a gamut of 22 benthic classes were discriminated so as to categorize the pictures. The spectral signature of each of the 22 classes was computed using the associated pixels contained into the corrected 2-m multispectral Pleiades-1 dataset. A very high coefficient of agreement (kappa=0.91) was derived from the confusion matrix. Because of the straightforwardness of this paper objective, we merged the 22 classes into 4 super-classes: deep water, sediment, algae, hard corals (Figure 5). Deep water pixels were removed (see Depth). Due to the uncertainty regarding substrate sub-criteria, we calculated each pixel's score using the following formula:

$$S = \frac{S_C \times P_C}{100} + \frac{S_A \times P_A}{100} + \frac{S_S \times P_S}{100} \quad \text{(Eq. 1)}$$

where S_C, S_A and S_S are sub-criteria scores for coral, algae and sediment, respectively; PC, PA and SS are the probability of the substrate to be to encounter coral, algae and sediment in %.

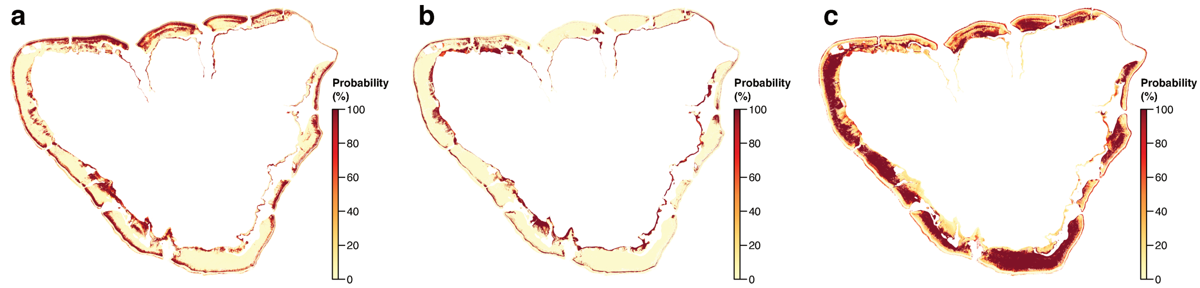


Figure 5: Maps of the "substrate" criteria. Probability of the substrate to be (a) coral, (b) algae and (c) sediment.

Slope was calculated in degree and expressed as the greatest drop in bathymetry among the 8 neighboring pixels. We computed the slope based on the bathymetry resampled at 20-m resolution is order to account for slope variation rather than undesirable fine scale variation in slope that could have arisen from the presence of numerous bommies or error in the bathymetry model (see Depth section).

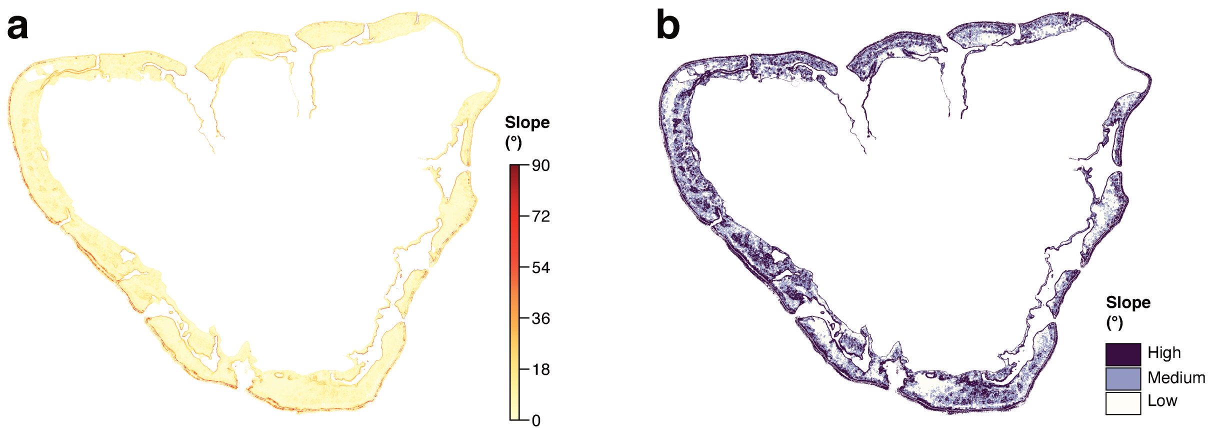


Figure 6: Maps of the "slope" criteria. (a) Continuous scale. (b) Categorized according to the sub-criteria boundaries.

Reference

Collin A. and Hench J. 2015. Extracting shallow bathymetry from very high resolution satellite spectral bands and a machine learning algorithm. *International Council of the Exploration of the Sea (ICES)* CM/N:24.

S3 File. Additional analyses. Predicted fishing effort inside and outside Moorea's current marine protected areas.

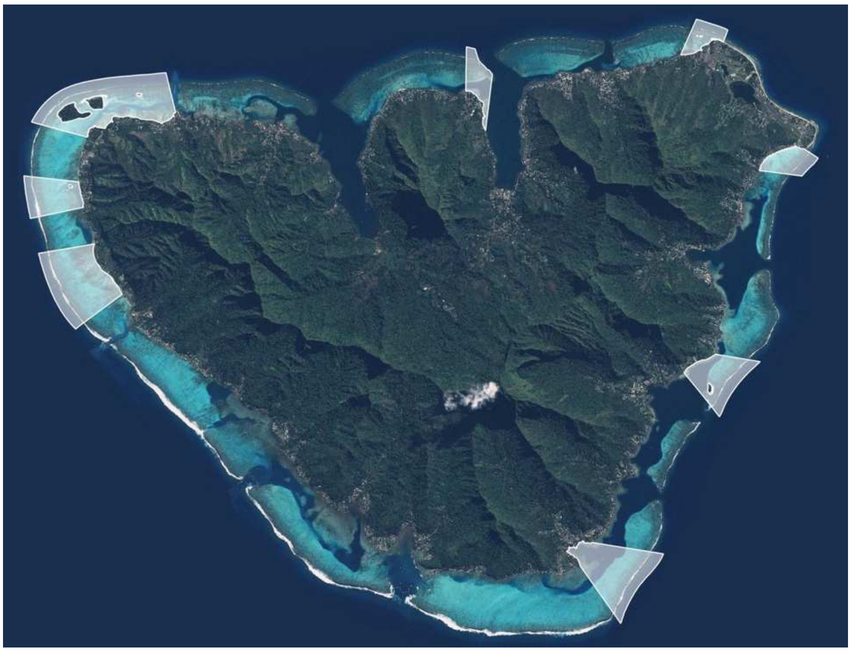


Figure 1: Map of Moorea showing the location of the eight marine protected areas.

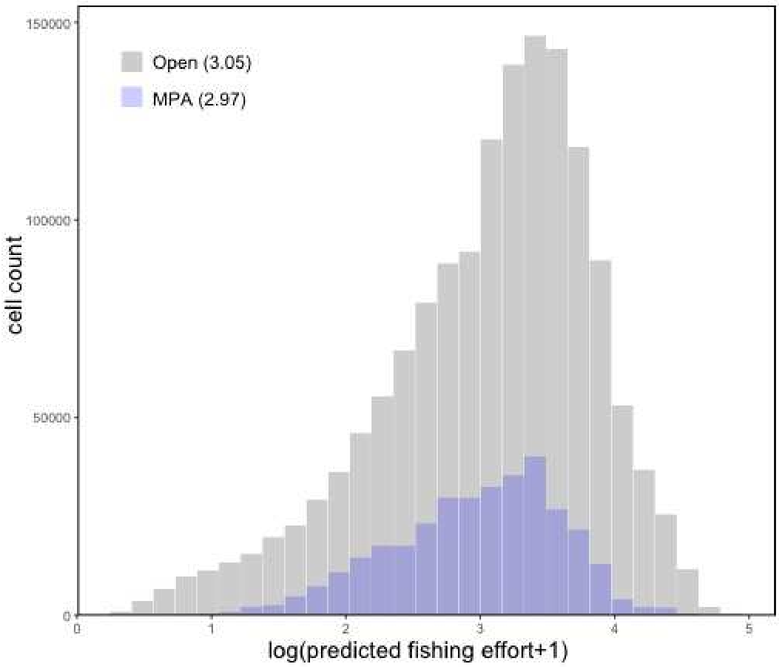


Figure 2: Distribution of predicted fishing effort inside (MPA) and outside (open) marine protected areas (means in parentheses).

Appendix C (Chapter III)

Appendix S1: Extended methods on the creation of the spatially-explicit indicators of social-ecological vulnerability components.

Ecological vulnerability

Ecological exposure

We used used a previously published model of spatial patterns of the fishing effort around Moorea (Thiault et al. 2017a) to map ecological exposure. This proxy-based model combines an index of household dependency on marine resource (see Section 2.2) with fine-scale spatial representation of fishers' preference for fishing grounds to predict fishing effort at every 5 x 5m cell within Moorea's reef. The map of ecological exposure is presented in Figure 1.

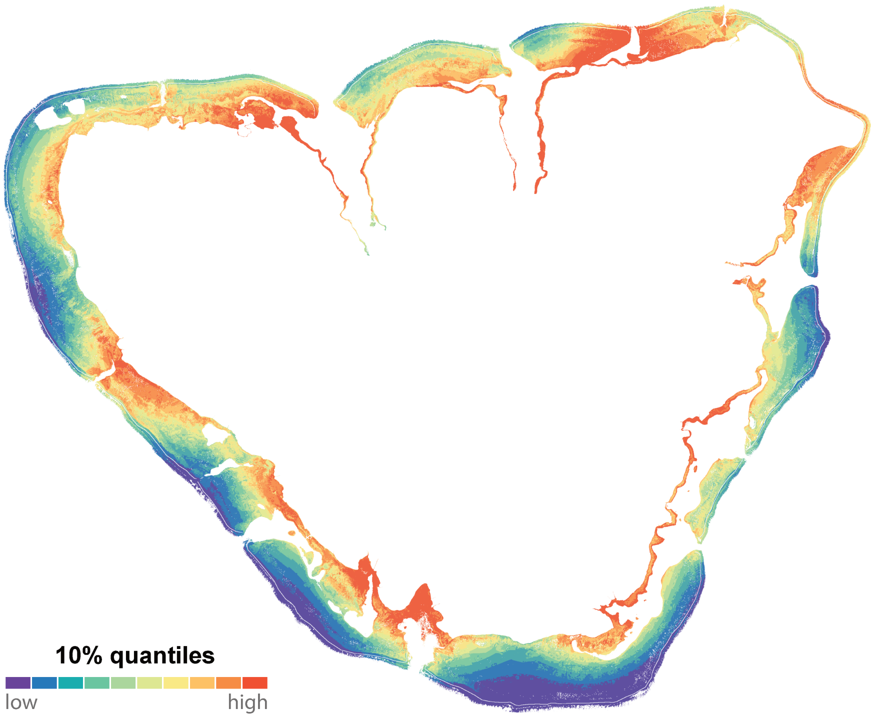


Figure 1: Map of ecological exposure to fishing.

Ecological resilience

To map ecological resilience, we first developed a resilience index (R) as a measure of the resilience of the fish assemblage to fishing using the following formula (Eq. 1):

$$R_{\mathrm{a}} = \sum_{i=1}^{S_{\mathrm{a}}} \left(1 - v_{\mathrm{i,a}}\right) \times n_{\mathrm{i,a}} \tag{1}$$

where R_{a} is the resilience of the fish assemblage at the site a, $v_{\mathrm{i,a}}$ is the normalized (scaled between 0 and 1) intrinsic vulnerability to fishing index (Cheung et al. 2005) of the species *s* present in the fish assemblage at the site *a*, and $n_{\mathrm{i,a}}$ is the log-transformed and normalized abundance (scaled between 0 and 1) of the species *s* present in the fish assemblage at the site *a*. Cheung et al.'s index of intrinsic vulnerability to fishing takes account of eight life-history parameters (maximum body length, age at first maturity, von Bertalanffy growth parameter K, natural mortality rate, maximum age, geographic range, annual fecundity and strength of aggregation behavior). Using this simple metric, we were able to capture some previously unexplored aspects of fish assemblage resilience (i.e., inverse of intrinsic vulnerability) to fishing over a variety of habitat configurations sampled using 2 x 25m line transects (a=57, Fig. 2).

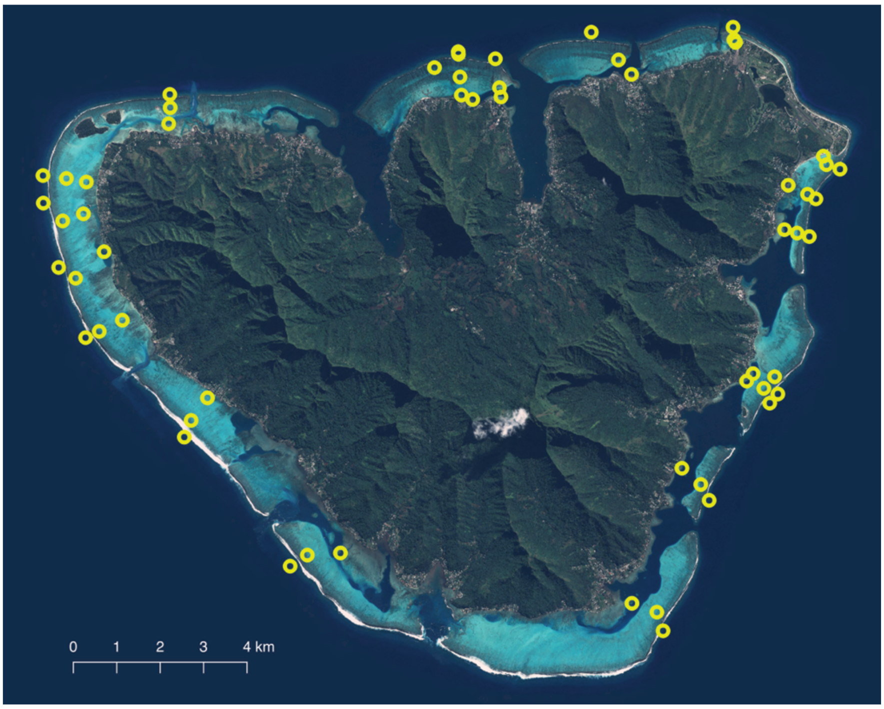


Figure 2: Location of sampling sites used to calculate the ecological resilience index.

Then, the resilience index (R) was predicted at unsampled sites using seven spatially-explicit predictors extracted from space-borne imagery (depth, slope, coral cover, sediment cover, distance to the shore, distance to the closest pass and rugosity of the seafloor; see (Thiault et al. 2017a) and Collin & Hench 2015 for additional information on the modelling of spatially explicit predictors) in boosted regression trees (BRT), although we acknowledge that resilience may be explained by a number of other environmental factors that we were not able to cover here (e.g., hydrodynamic conditions, source-sink dynamics). BRT are considered as a robust technique which can handle interacting factors and non-linearity (which were expected to occur in our case) without overparametrizing (Elith et al. 2008). The total number of trees was determined using k-fold cross validation and we set all other parameters according to Elith et al. (2008)'s guidelines (tree complexity, *tc*=4; learning rate, *lr*=0.0005 and a bag fraction of 0.8). The BRT model showed a high predictive performance, explaining 90% of the total deviance using a cross-validated procedure. The relationship between observed and predicted resilience was remarkably high (R^2 = 0.812). BRT outputs consisted of a map of resilience of the target fish assemblage to fishing at every 5 * 5m cell of the lagoon. The map of ecological resilience is presented in Figure 3.

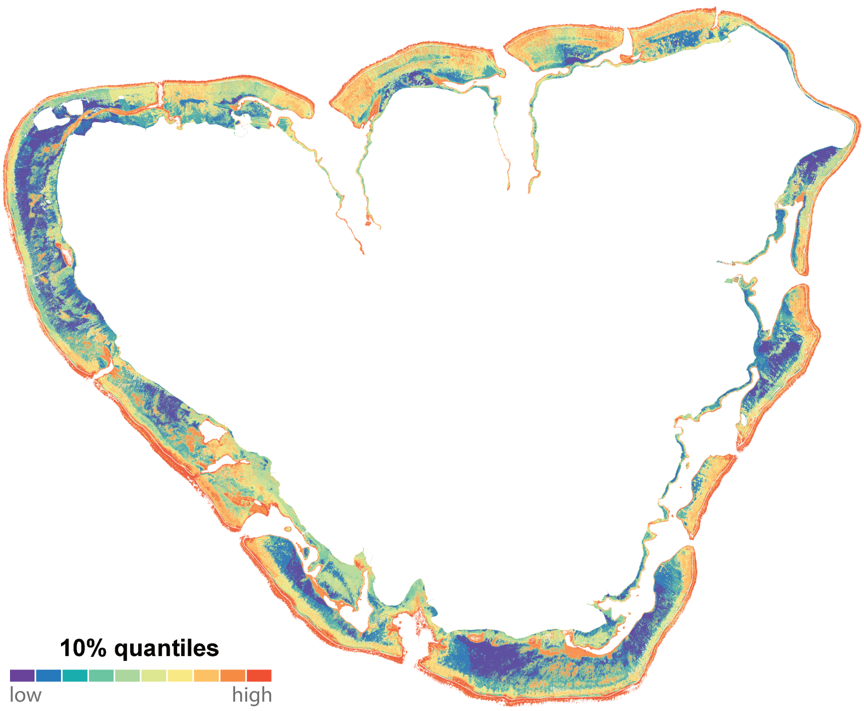


Figure 3: Map of ecological resilience to fishing.

Social vulnerability

Social exposure

In Moorea, fishing is characterized by short trips (few hours) starting in front people's houses (Leenhardt et al. 2016), which limits the potential fishing grounds to areas adjacent to households. We therefore measured the exposure of individual households as the arithmetic mean of the ecological vulnerability of cells located within a buffer area of 2 km radius, which is the distance of reference to estimate the potential fishing area in Moorea (Thiault et al. 2017a). A consequence of this approach is that it considers households located in the mountains to have smaller fishing grounds than those located on the coast, which can be interpreted as the cost of displacement from the household to the fishing ground. The map of social exposure is presented in Figure 4.

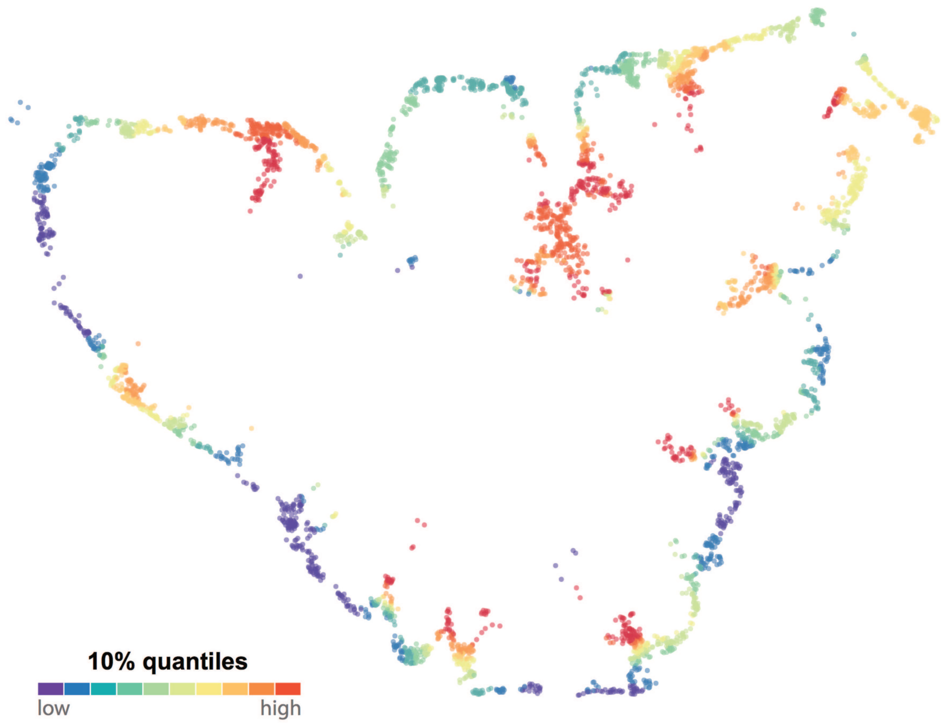


Figure 4: Map of social exposure.

Social sensitivity

Using data from an island-wide survey conducted on the entire population of Moorea (Insee-ISPF 2007), we calculated social sensitivity as a function of (1) the proportion of households engaged in fisheries (subsistence or for sell), (2) the extent to which household engaged in fisheries also engage in non-fishery occupations and (3) the fraction of inactive people in the population (Eq. 2):

$$S = \frac{F}{F+NF} \times \frac{N}{F+NF} \times \frac{U}{N} \quad (2)$$

where S is the sensitivity metric, F is the number of households having at least one member who declared fishing as its primary or secondary livelihood activity; NF is the number of households having at least one member who declared non-fishery-related occupation as its primary or secondary livelihood activity; U is the number of households having at least one member having no activity, whether primary or secondary; and N is the total number of households. This metric of sensitivity (to the loss of fishing opportunity) was calculated at the district-level (n=69 with roughly 80 households per district; see Thiault et al. (2017a) for detailed information on the method). Therefore, social sensitivity was mapped for each individual household, with sensitivity values corresponding to their district-level score of sensitivity. It is worth noting that this index of social sensitivity is also used to map ecological exposure (see section 1.1). Therefore, higher social vulnerability due to high sensitivity (i.e., marine resource dependency) leads to high levels of exposure of the resource to fishing, in turn increasing ecological vulnerability. The map of social sensitivity is presented in Figure 5.

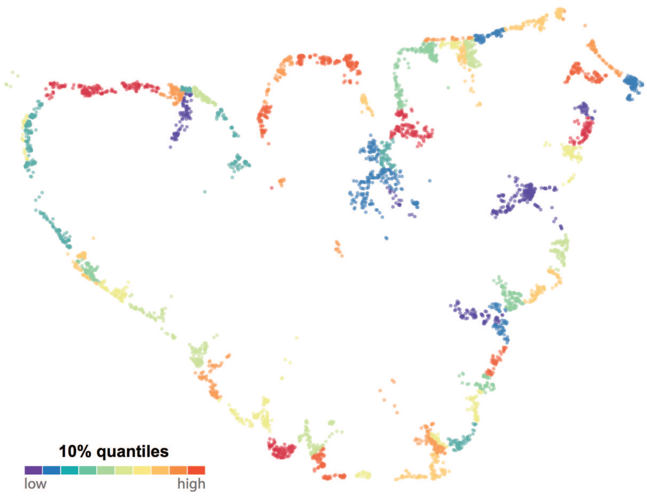


Figure 5: Map of social sensitivity.

Social adaptive capacity

Using the same population census data as for quantify social sensitivity, and following practices used in similar contexts (McClanahan et al. 2008a; Cinner et al. 2012a, 2015; Bennett et al. 2014a), we measured eleven indicators grouped into five components of social adaptive capacity: (1) spatial mobility, (2) material assets, (3) occupational mobility; (4) attachment; and (5) education (Table 1). Then, a pool of experts composed of marine scientists (n=2), anthropologist (n=1), managers (n=2), fishery service (n=1), policymakers (n=1) and national statistics experts (n=2) made pairwise comparisons of the importance of the eleven indicators following the analytic hierarchy process methodology (AHP, Saaty 2008). The average of the weightings was used to calculate social adaptive capacity for each household as the weighted sum of the eleven indicators (scaled between 0 and 1). Although each household can be localized individually, their actual adaptive capacity values have been aggregated and mapped at the district level due to confidentiality reasons. The map of social adaptive capacity is presented in Figure 6.

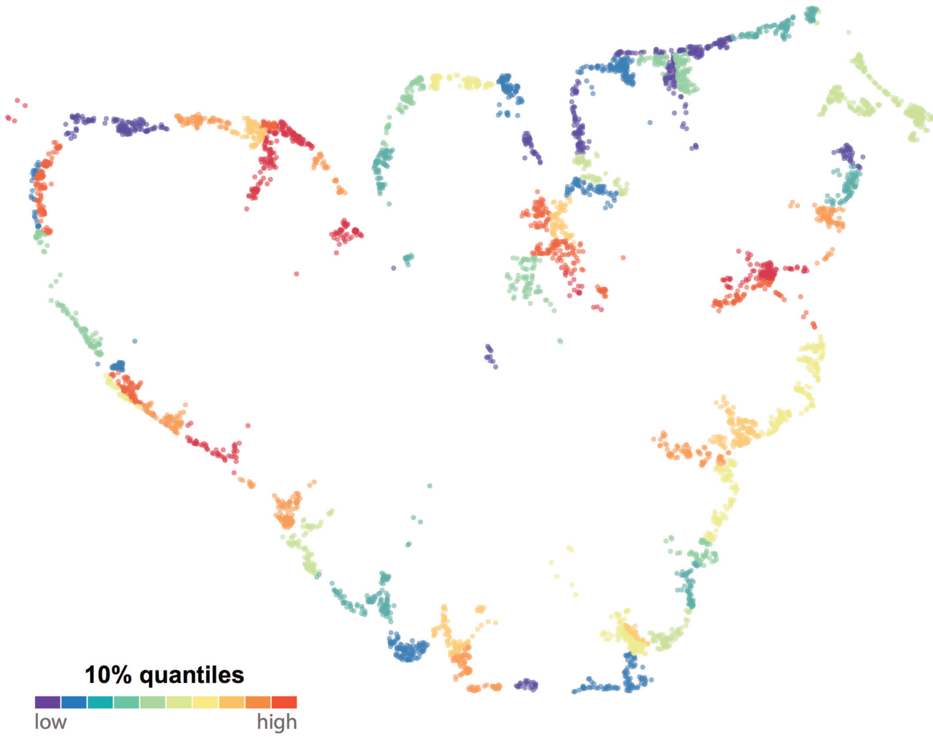


Figure 6: Map of social adaptive capacity.

Table 1:Indicators of social adaptive capacity and corresponding average weight measured by experts.

Component *Description*	**Household-level indicator**	**Bounding**	**Weight (*w*)**
Spatial mobility *Determines the ability to relocate fishing following environmental change*	Motorized boat ownership	Binomial: no=0; yes=1	0.1
	Non-motorized boat ownership	Binomial: no=0; yes=1	0.02
Occupational mobility *Allows for higher flexibility and secures food and income*	Normalized number of primary activities	Continuous: no activity=0; max=1	0.2
	Normalized number of secondary activities	Continuous: no activity=0;max=1	0.13
Material assets *Indicate the relative richness in the households within the district*	Type of house	Continuous: min=0; max=1	0.01
	Normalized number of domestic appliances	Continuous: no appliance=0; max=1	0.02
	Car ownership	Binomial: no=1; yes=1	0.03
Attachment *Indicates the willingness to move to other locations or change lifestyle on the long-term*	Place of birth	Binomial: French Polynesia=0; other=1	0.07
	Land and house ownership	Continuous: both=0; one=0.5; none=1	0.06
	Language spoken in household	Binomial: Polynesian language=0; Other=1	0.06
Education *Indicates the ability to take advantage of new information and employment opportunities*	Normalized level of formal education	Continuous: no formal education=0; higher studies=1	0.3

Appendix D (Chapter VI)

Assessing vulnerability in a multi-driver context

Vulnerability is generally framed as a set of external conditions (exposure) and internal properties (sensitivity and adaptive capacity) (Marshall et al. 2010). Exposure refers to the magnitude, frequency and/or duration to which the fishing organizations are subject to a particular driver of change. Sensitivity refers to their conditions mediating their short-term propensity to be influenced following the exposure, while adaptive capacity refers to the ability to implement effective and long-lasting responses to changes by minimizing, coping with, or recovering from the potential impact of a stressor. Here, we focused on vulnerability of fishing organizations to two major drivers of the Chilean artisanal fishing system: poaching and markets. Many studies have approached the measurement of vulnerability (i.e., the result of high exposure, high sensitivity and low adaptive capacity (Marshall et al. 2010)) as the combination of one specific and external component (exposure) and a suite of general intrinsic features (aggregated into sensitivity and adaptive capacity). Considering general aspects is important, but ignoring specific aspects may overlook the possibility that the properties leading to reduced or increased vulnerability differ depending on the driver. For instance, the conditions and factors that may be useful in confronting climate change are different than for economic shocks. Hence, a 'generic' view of vulnerability may be invalid. Hence, exposure is by essence a specific aspect of vulnerability while sensitivity, which in the context of this study refers to resource dependency (Marshall et al. 2007), is here considered as a general aspect of vulnerability. Finally, adaptive capacity is treated as the combination of general and specific aspects.

Each of the three vulnerability dimensions (i.e., exposure, sensitivity and adaptive capacity) is composed of either a specific aspect (exposure), a general aspect (sensitivity) or a combination of both (adaptive capacity) (SI Fig. S1). Each aspect is decomposed into 16 components in the context of poaching and markets (Table 1). Based on our survey, we created 20 indicators to quantify each vulnerability component (SI Table S1). Exposure to markets, which was not captured by our survey, was obtained from the undersecretary of fisheries (Table S1).

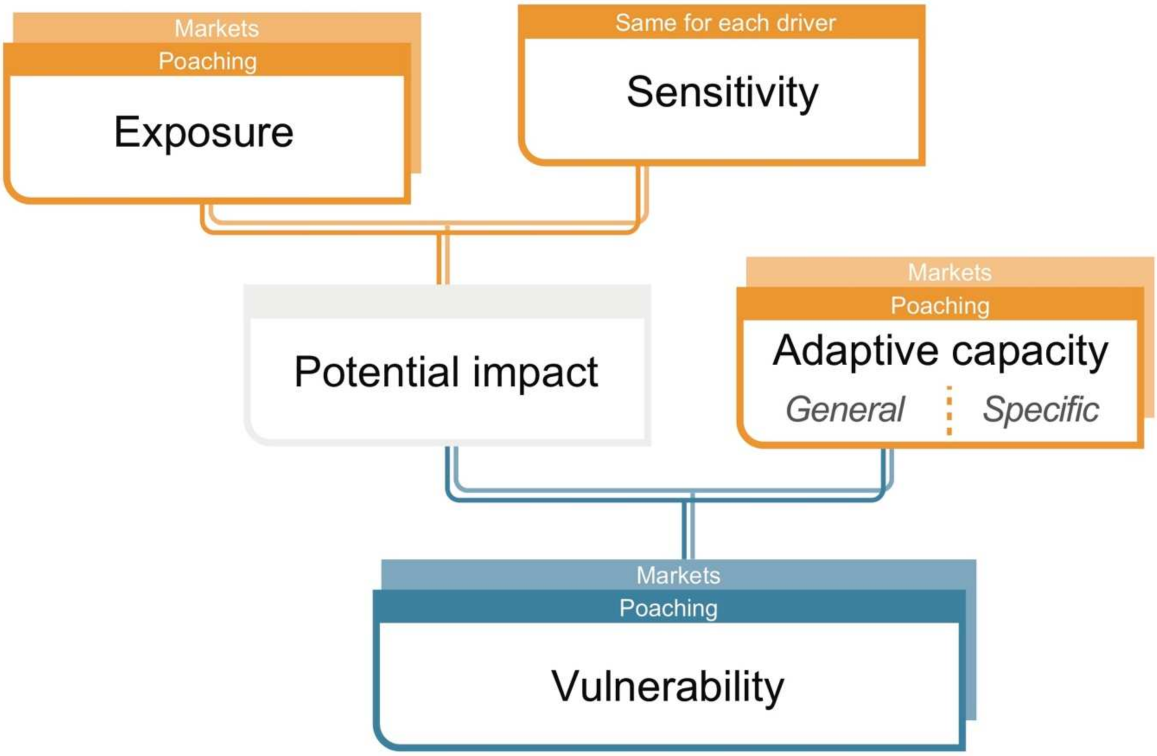


Figure S1: IPCC vulnerability framework adapted to multiple drivers. Note that exposure is specific to each driver while sensitivity remains general. Adaptive capacity has both general and specific aspects.

Table S1: Vulnerability dimensions, associated general and specific components and corresponding indicator.

Dimension / Aspect / 0. Component	Indicator
Exposure	
Specific (poaching)	
1. Level of poaching in TURF	1.1 Perceived level of poaching from syndicate's members
	1.2 Perceived level of poaching from outsiders
Specific (markets)	
2. Resource price volatility	2.1 Prices volatility index based on inter-annual variability
3. Resource price trend	3.1 Prices trends between 2005 and 2015 (slope of linear model)
Sensitivity	
General (resource dependency)	
4. Dependency on fishing	4.1 Proportion of households' members engaged in fishery versus non-fishery-related occupations
Adaptive capacity	
General	
5. Learning and knowledge	5.1 Level of formal education
	5.2 importance of TURF in ecological knowledge and awareness
6. Diversity and flexibility	6.1 Number of different livelihoods in a household
	6.2 Number of different types of marine resource targeted
7. Infrastructure	7.1 Log-distance to 17 infrastructure items
8. Material assets	8.1 Quality of household construction materials; number of household appliances
9. Social capital and trust	9.1 Participation in community events and trust in police, local officials, NGOs, fishery service and syndicate
10. Agency	10.1 Perceived leadership effectiveness
	10.2 Participation in decision-making
Specific (poaching)	
11. Conflict resolution mechanism	11.1 Use of graduated sanctions
12. Surveillance effectiveness	12.1 Perceived surveillance effectiveness of the best TURF
13. External support	13.1 Perceived level of external support for enforcing the TURF
14. Internal support	14.1 Perceived level of internal support for enforcing the TURF
Specific (markets)	
15. Relationship with the middleman	15.1 Level of trust in middleman
16. Gear diversity	16.1 Number of different gear utilized

Correlations among vulnerability indicators

According to the IPCC framework of vulnerability, exposure, sensitivity and adaptive capacity are should be independent. This is the case here for both stressors (no shared indicators and no statistical correlation). As expected, aspects of vulnerability sharing indicators were correlated in most cases (15 out of 18 pairwise comparisons). Despite shared indicators, some were nevertheless not correlated (3 out of 18 cases). For instance, no significant correlation was detected between vulnerability to poaching and vulnerability to markets, despite the fact that they have ten indicators in common. This can be due to the combined action of negative relationship with other components (e.g., V_mk and E.po) and the addition of specific aspects of vulnerability. Among the unexpected correlations (i.e., significant correlation between aspects of vulnerability that have no indicator in common), exposure was positively correlated with specific adaptive capacity (and consequently total adaptive capacity) in the context of market forces.

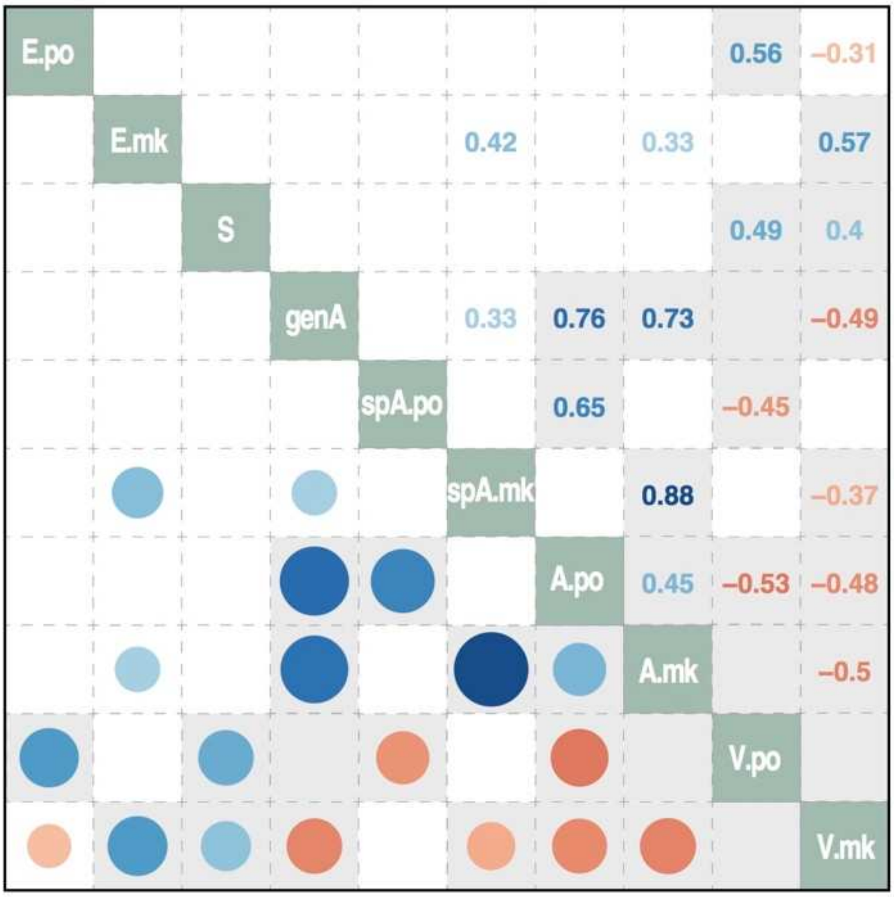


Figure S2: Spearman rank correlations amongst vulnerability aspects. Circle size and color indicate the correlative strength and direction, respectively (blue, positive; red, negative). Only values for significant correlations ($P < 0.05$) are displayed.). E.po and E.mk, exposure to poaching and markets. S, sensitivity. GenA, general adaptive capacity. spA.po, spA.mk, specific adaptive capacity to poaching and markets. A.po and A.mk, adaptive capacity to poaching and markets. V.po and V.mk, vulnerability to poaching and market. Aspects sharing at least one indicator are indicated by grey boxes and denote where correlations may occur expected (e.g., adaptive capacity to poaching is calculated by combing general and specific adaptive capacity, and is therefore expected to be correlated with those two components).

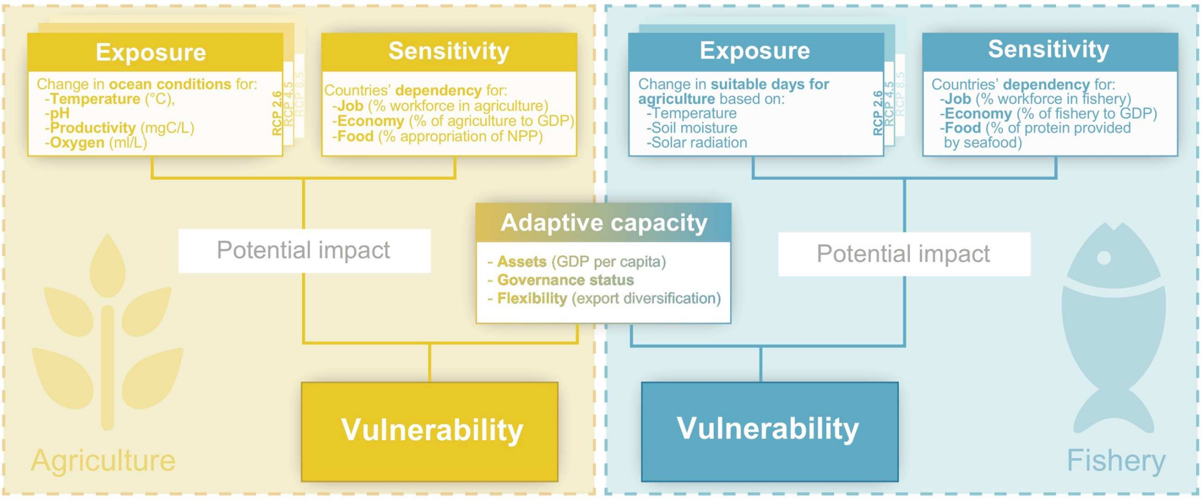
These results show that despite shared components/aspects (sensitivity and general adaptive capacity), very low (not statistically significant) correlation between different aggregated vulnerabilities was measured (Fig S2). Some study sites were highly vulnerable to one particular driver and weakly vulnerable to another (Fig. 3B in main text). This (1) emphasizes the role of specific components of vulnerability (exposure, but also specific adaptive capacity) in determining vulnerability and (2) highlights how prioritization arising from vulnerability analyses may be biased if important drivers are missing, thus stressing the need to perform more representative vulnerability assessment to shift from a problem-centered to a community-centered approach to vulnerability.

Appendix E (Chapter VII)

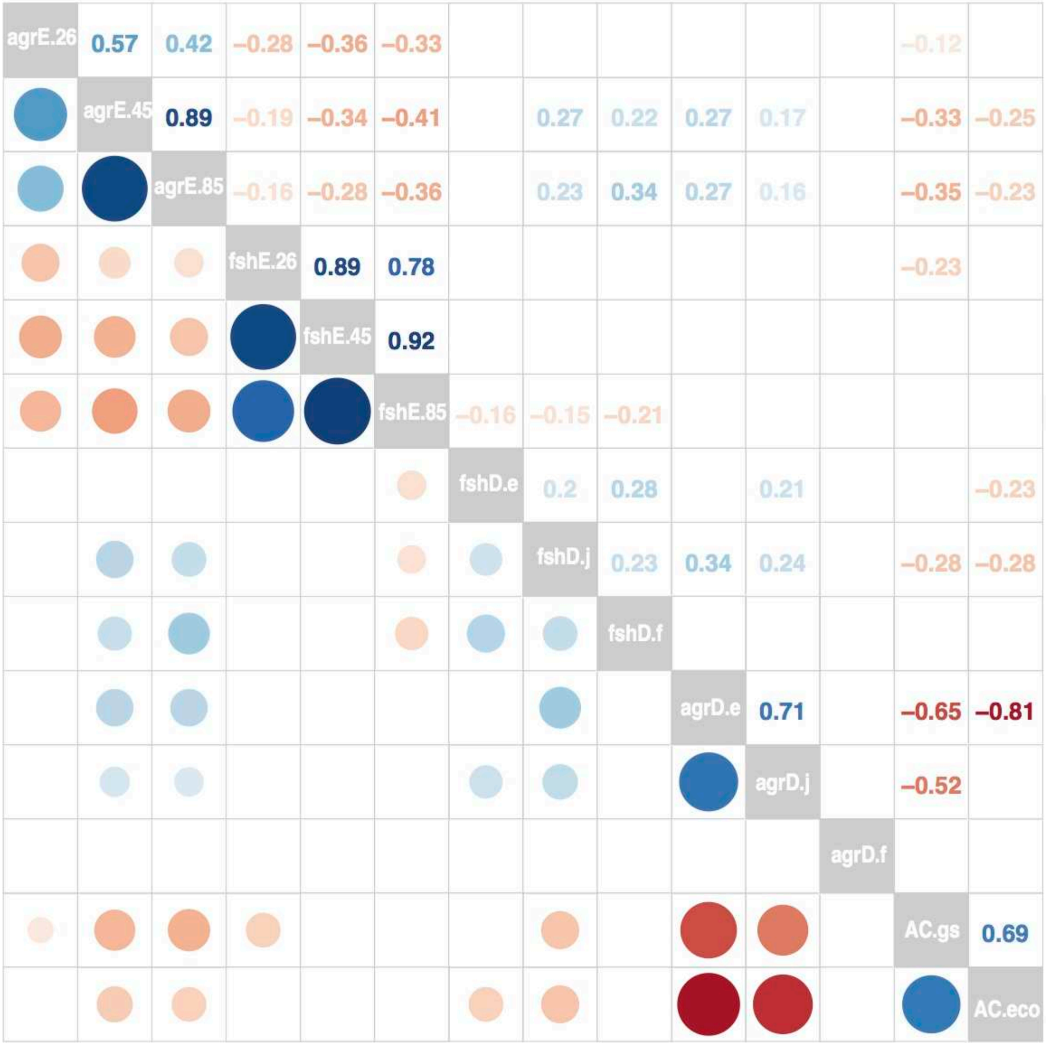
Extended Data Table 1: Indicators used to assess the vulnerability of countries to the impacts of climate change on agriculture and marine fisheries. Data sources are provided along with the coefficient of determinant in predicting overall country vulnerability to risk associated with impacts of climate change on agriculture and marine fisheries.

AGRICULTURE	Source
Exposure	
Suitable days for agriculture	
Change in the number of SDFA on agricultural lands	(Mora et al. 2015)
Sensitivity	
Dependency on agriculture	
Food dependency (% appropriation of primary production)	(Imhoff et al. 2004)
Job dependency (% workforce employed by agriculture)	(CIA 2015)
Revenue dependency (% agriculture value to GDP)	(FAO 2014b)
Adaptive Capacity*	
Governance status	
Composite governance index	(The World Bank Group 2015)
Assets	
GDP per capita PPP	(CIA 2015)
Flexibility	
Exportation diversification	(IMF 2014)
FISHERY	**Source**
Exposure	
Change in ocean condition	
Cumulative average of absolute change in temperature, pH, oxygen and productivity within the EEZ	(Mora et al. 2013)
Exposure	
Food dependency (% animal protein consumption supplied by seafood)	(FAO 2014b)
Employment dependency (% workforce employed by marine fishing)	(Teh & Sumaila 2013)
Revenue dependency (% landed seafood value to GDP)	(Pauly 2007)
Adaptive Capacity*	
Governance status	
Composite governance index	(The World Bank Group 2015)
Economy	
GDP per capita PPP	(CIA 2015)
Flexibility	
Exportation diversification	(IMF 2014)

* For both sectors, we used the same indicators to measure “adaptive capacity”, assuming that countries’ ability to implement effective and long term responses to changes were identical for agriculture and fishery.



Extended Data Figure 1: Overview of indicators used to quantify each dimension of vulnerability. Exposure is sector-specific and changes with emission scenarios, dependency is only sector-specific and adaptive capacity is the same across sectors and scenarios



Extended Data Figure 2: Spearman rank correlations amongst vulnerability indicators. Circle size and color indicate the correlative strength and direction, respectively (blue, positive; red, negative). Variables ordered by dimensions (that is, exposure, dependency, and adaptability) and not hierarchical clusters, displaying values for significant correlations only ($P < 0.05$). agrE.26, agrE.45 and agrE.85, agriculture exposure according to RCP 2.6, RCP 4.5 and RCP 8.5 emission scenarios, respectively; fshE.26, fshE.45 and fshE.85, fisheries exposure according to RCP 2.6, RCP 4.5 and RCP 8.5 emission scenarios, respectively; agrD.e, agrD.j and agrD.f, dependency on agriculture for economy, job and food, respectively; fshD.e, fshD.j and fshD.f, dependency on marine fisheries for economy, job and food, respectively; AC.gs, governance status; AC.eco, log[GDP per capita].

Abstract

Contemporary sustainability science and practice must embrace the complexity of social-ecological systems and capitalize on the lessons learned from the recent theoretical and applied advances made in various disciplines. This can be accomplished in particular by incorporating this extensive knowledge into management and decision making through integrative and operational frameworks. Based on contrasting but complementary case studies (coral reef fishery in Moorea, French Polynesia; artisanal benthic fishery in Chile and global food systems), and drawing from the recent development in social-ecological science, we extended the use of the social-ecological vulnerability framework by (1) mapping human-nature dependencies in the context of resource-user interactions, (2) integrating the temporal dimension, (3) accounting for multiple drivers of change and (4) their impact on diverse entities of the system considered. Specifically, using the Progressive-Change BACIPS approach in Moorea (Chapter I), we show that the current marine spatial planning has unlikely contributed to improve ecological outcomes, thus casting doubts on its capacity to meet its conservation and fishery management objectives. In Chapter II we showed how the use of mixed methods combining different data sources allowed us to map fishing effort in the challenging context of Moorea where fishing is diffuse among inhabitants and along the coast. Using this information within a vulnerability framework (Chapter III) we mapped resource-user interdependencies, to highlight focal areas for management interventions in Moorea. Then we developed a framework combining spatial social and ecological vulnerabilities to recommend interventions portfolio in those focal areas, specifically targeting each social and ecological conditions within each management spatial unit. In Chapter V we showed how vulnerability assessments in two different times can be used to capture combined changes in social-ecological systems in response to direct and indirect drivers. This approach precluded us to identify the specific response of the social-ecological system to each of the drivers, therefore, in Chapter VI, we developed a framework to disentangle the effect of direct and indirect drivers on the vulnerability of fishing communities involved in the management of common-pool resources through TURFs. Finally, we assessed in Chapter VII how countries are vulnerable to the effects of climate change on agriculture and fisheries globally to show how vulnerability assessments can be suited in a cross-sectorial management context. This interdisciplinary work provided the foundation to represent key linkages in social-ecological systems, understand the underlying sources of unsustainability, and address these through a set of targeted and context-grounded management interventions and policy actions. This thesis provides a new perspective on human-nature linkages and has a number practical implications for managers, conservation planners, and policy-makers that seek to incorporate a social-ecological perspective to tackle sustainability issues from local to global scales.

Key-words: interdisciplinarity – operationalization – science-policy interface – social-ecological systems - vulnerability