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Poaching in marine protected areas: drivers of and responses to illegal fishing

Thesis submitted by

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This thesis is presented for the degree of Doctor of Philosophy in Marine Biology, within the College of Science and Engineering and the ARC Centre of Excellence for Coral Reef Studies, James Cook University, Townsville, Queensland Australia

For my grandfather, Arno Bergseth, who taught me what it was to be a gentleman and a scholar, and my father, Del Bergseth, who set a conservationist on his path with his first walk in the woods

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Statement of the Contribution of Others

This thesis includes collaborative work with my advisors Prof. Josh Cinner, Prof. Garry Russ, Dr. David Williamson, and other colleagues. During these collaborations, I was responsible for project design, data collection, analysis, interpretation and scientific writing of manuscripts for publication. My collaborators provided intellectual guidance, technical, editorial, and data collection assistance as well as financial support. Aaron MacNeil also provided statistical analysis as a collaborator in Chapter 5. James Aumend and others at the Great Barrier Reef Marine Park Authority's Field Management Compliance Unit provided feedback on study design and identified particular issues of importance for compliance management in the Marine Park. Special thanks to Dr. Adrian Arias for continued collaboration and discussion on many components of this thesis.

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Executive summary

Conservation is primarily reliant on people's compliance with rules and regulations. Yet, non-compliance is widespread throughout the world, and regularly negates expected conservation outcomes. The effects of non-compliance are especially apparent in protected areas, where even short bursts of poaching (defined in this thesis as fishing in no-take zones, or no-fishing marine reserves) can rapidly negate the effects of protection that often take decades to produce (e.g. Russ & Alcala 2010; Wittemyer *et al.* 2014). In addition, efforts to reduce poaching often fail due to a lack of capacity, resources, and/or general understanding of the drivers or influences on the behaviour (e.g. Gill *et al.* 2017). Thus, curtailing poaching depends on gathering detailed information to guide specifically targeted behavioural and management interventions. However, gathering information on poaching is inherently difficult because it is an illegal, clandestine, and often socially unacceptable activity.

This thesis therefore addresses this considerable knowledge gap by applying specialized techniques to measure, assess, and understand poaching. Many methods and approaches can be used to assess and measure poaching, but no single method is a panacea, because each is limited in the information it can provide and subject to different types of bias (Gavin *et al.* 2010; Arias 2015; Bergseth *et al.* 2015). Reliably assessing and measuring poaching therefore necessitates a triangulation approach that utilizes complimentary methods to provide a holistic picture of poaching. **Chapter 1** therefore provides a general introduction about the methods used to estimate compliance levels, as well as the different theoretical disciplines and approaches that can be used to understand poaching.

The focus of ***Chapter 2*** is to answer the first research question of my thesis: "*How can poaching be measured given its cryptic nature*?" In this chapter, I combine social surveys with a complementary field-based method to 'ground-truth' and assess the prevalence poaching by recreational fishers in the Great Barrier Reef Marine Park (GBRMP). I use three

specialized questioning techniques [Self-administered questioning (SAQ); the Randomised Response Technique (RRT; Warner 1965); the Unmatched Count Technique (UCT; Droitcour *et al.* 1991)] to estimate poaching rates. Concurrently, I develop and examine the potential of two theoretically grounded, proxy indicators of non-compliance (perceived prevalence of poaching and personally knowing a poacher). Both of these indicators yield higher estimates than those provided by specialised techniques, which suggests that specialised questioning techniques are still subject to underreporting. Furthermore, my findings indicate that the false consensus effect and social learning are causing poachers and their associates to overestimate the prevalence of poaching. I also quantify the accumulation rate of derelict fishing gears on fringing coral reefs inside and outside no-fishing reserves of two nearby inshore island groups. Surprisingly, I find no difference in the accumulation rate of derelict gears between areas open and closed to fishing. Overall, I demonstrate that a substantial amount of poaching is occurring in inshore island groups once thought to be among the best-enforced areas in the GBRMP, which has important management implications.

Understanding the drivers or influences on recreational fisher's poaching decisions is a critical component of designing management and behavioural interventions to curtail poaching. While numerous disciplines explore the complexities of human decision-making and resultant behaviours, most investigations of poaching suffer from 'disciplinary silo-thinking' which inhibits a comprehensive understanding of poaching behaviours (Von Essen *et al.* 2014). Thus, the focus of ***Chapter 3*** is to develop a multi-disciplinary approach to comprehensively examine the social dimensions of poaching. In this chapter, I integrate pertinent theories from a range of disciplines including sociology, criminology, wildlife biology, and psychology to develop 29 potential drivers of compliance behaviour. I then explore the prevalence and distribution of these potential drivers. I find that most fishers perceive high levels of legitimacy for management agencies and see poaching as personally and socially unacceptable. However, my research findings also suggest that two additional (mis)perceptions (pluralistic ignorance and a perceived lack of deterrence) are likely operative and at least partially responsible for the continuation of poaching in the GBRMP. Lastly, I reveal that fishers perceived two primary motivations to poach: better fishing in no-fishing reserves, and a low probability or risk of detection. These results suggest that extolling certain benefits of no-fishing reserves (i.e. they hold bigger and more fish) where enforcement capacity is low could actually lead to the perverse outcome of encouraging non-compliance. Based on these results, I highlight tools such as social norms approaches, strengthened coercive deterrence, and fear arousing communication that can be used to address these misperceptions and increase compliance.

In ***Chapter 4***, I ask: "*how are different poaching measure or proxies related to the potential drivers of compliance behaviour?*" Specifically, I collated and condensed 29 potential influences and drivers of poaching that were identified in Chapter 3, and empirically examined their influence on three indicators or proxies of poaching: the RRT, perceived levels of poaching, and personally knowing a poacher. The RRT model performed particularly poorly at identifying behavioural drivers of poaching, likely because the admitted level of poaching was lower than the intentionally introduced statistical noise (to obscure respondent's answers and ensure confidentiality). However, 66% of the drivers that were significant for one proxy indicator of poaching were also significant for the second proxy. When considered in light of the results from Chapter 2, this suggests that these proxies could be further integrated to estimate and understand poaching, especially in contexts where poaching is socially unacceptable and fairly rare.

Yet, many of the worlds marine protected areas suffer from a critical lack of capacity (Gill *et al.* 2017) and subsequent low levels of compliance, which negate expected outcomes.

Accordingly, natural resource management agencies are increasingly attempting to bolster compliance by engaging the latent surveillance and enforcement potential of resource users (GBRMPA 2016; Green 2016; Kohn 2016). However, little is known about the conditions or institutions that encourage this behaviour. In **Chapter 5** I fill this knowledge gap by collating and analysing more than 2000 fisher surveys from 55 MPAs in 7 countries to fill this knowledge gap by answering four research questions: 1) How many fishers have previously observed poaching?: 2) How do they respond to observed non-compliance?; 3) Why do fishers remain inactive?; and 4) What are the effects of institutions and conditions on fisher's responses to poaching? I found that nearly half of these fishers had previously observed poaching, but the most common response was inaction, typically due to conflict avoidance, a sense that it was not their responsibility or concern, and the perception that poaching was a survival strategy. Furthermore, I quantified how institutional design elements relate to fisher's responses to poaching, and highlight avenues to responsibly engage fishers while mitigating risk.

This body of work advances the current state of knowledge in compliance management, particularly in regard to the approaches necessary for fully measuring and understanding the prevalence and drivers of poaching. The findings provided by my multi-disciplinary approach advance our understanding and management of human behaviour in four concepts that are critical for effective conservation: 1) further consideration of how people process information; 2) re-conceptualizing how people behave; 3) developing communication strategies to bolster compliance; and 4) designing rules and interactions to shape behaviour. In addition, this thesis demonstrates the necessity of triangulating sources of compliance information, and delivers findings that have been directly adapted for communication and outreach strategies employed by one of the world's most iconic marine parks.

Chapters 2-6 were originally written and formatted for publication in peer-reviewed journals. They have since been edited and reformatted for submission as this thesis.

Publications associated with this thesis:

Bergseth, BJ, Williamson, DH, Sutton, S, Russ, GR, & Cinner, JE (2017). A social-ecological approach to assessing and managing poaching by recreational fishers. ***Frontiers in Ecology and the Environment***, 15, 67-73. *(Chapters 2 & 3)*

Bergseth, BJ & Roscher M (2018) Discerning the culture of compliance through recreational fisher's perceptions of poaching. ***Marine Policy***, 89, 132-141. *(Chapters 2 & 3)*

Bergseth, BJ, MacNeil, MA, & Cinner JE (in prep) Investigating behavioural drivers and potential proxy indicators of poaching by recreational fishers. Target journal: ***Journal of Environmental Psychology***. *(Chapter 4)*

Bergseth, BJ, Gurney, GG, Barnes, ML, Arias, A, & Cinner, JE (in 2nd round of reviews) Addressing poaching in marine protected areas through voluntary surveillance and enforcement. ***Nature Sustainability***. *(Chapter 5)*

Bergseth, BJ (2017) Effective marine protected areas require a sea change in compliance management. ***ICES Journal of Marine Science***, doi:10.1093/icesjms/fsx105. *(Chapter 6)*

Bergseth, BJ, Williamson, DH, Frisch, AJ, & Russ, GR (2016) Protected areas preserve natural behaviour of a targeted fish species on coral reefs. ***Biological Conservation***, 198, 202-209. *(Appendix 1)*

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CHAPTER 1: General Introduction

The problem of poaching in conservation

Conservation is in essence a human response that aims to mitigate the widespread degradation of nature that characterizes the modern world. However, many of the world's conservation efforts are hindered by pervasive non-compliance with conservation regulations. Here, I define compliance as adherence to rules and regulations, whereas non-compliance is defined as disobedience to, or violation of rules or regulations. The specific non-compliant behaviour examined in this thesis is poaching, which is defined in the Merriam-Webster dictionary as "trespassing for the purpose of stealing game, or to take game or fish illegally". Thus, I use the terms non-compliance and poaching interchangeably throughout the thesis, specifically in reference to non-compliance with no-fishing/no-take reserves, which are often zoned within larger marine protected areas (MPAs)

Poaching has serious ramifications. Even short bursts of poaching can have large deleterious effects on ecological outcomes that often take decades of conservation to produce (e.g. Russ & Alcala 2010; Wittemyer *et al.* 2014), and on which millions of resource-dependent stakeholders rely. In addition, free-riding (or individuals who gain benefits from poaching while everyone else complies) can undermine broader stakeholder support (Cardenas *et al.* 2000), and ultimately lead to a scenario where non-compliance becomes socially acceptable, thereby hindering management efforts (Hauck & Sweijd 1999). These consequences are often most apparent in the examples of large, charismatic megafauna such as elephants, rhinos, or sharks (e.g. Robbins *et al.* 2006; Vira & Ewing 2014; Büscher & Ramutsindela 2015), but the overall prevalence of poaching throughout the world is likely much higher than assumed, given that it is a clandestine and often highly profitable behaviour.

Problem Statement

Compliance is critical for effective conservation, yet poaching is widespread and regularly negates conservation efforts throughout the world. Confronting and curtailing poaching relies primarily on gathering reliable information on poaching activities, which can be used to design and guide targeted behavioural interventions. However, measuring, understanding, and curtailing poaching is inherently difficult given its illegal, clandestine, and often socially unacceptable behaviour.

Assessing and measuring poaching

A primary challenge in compliance management is detecting and estimating compliance levels. Many methods or approaches can be used to assess poaching, but each is subject to inherent strengths, limitations, and biases (e.g. Gavin *et al.* 2010; Arias 2015; Bergseth *et al.* 2015). For instance, direct observation methods can provide absolute estimates of poaching levels (e.g. Smallwood & Beckley 2012), but are markedly labour and cost-intensive. Biological surveys can be used to monitor target species populations and indicate potential impacts of poaching (e.g. Linares *et al.* 2012), but they cannot indicate who is poaching, or why (Bergseth *et al.* 2015). Similarly, recent studies demonstrate that fish change their behaviour (i.e. they become flightier, or fearful) as human fishing pressure increases, suggesting that fish behaviour could be used to indicate poaching in no-fishing reserves (Januchowski-Hartley *et al.* 2011; Bergseth *et al.* 2015). Alternatively, social surveys can estimate poaching levels and provide valuable information about drivers and stakeholders' perceptions of poaching. However, the validity of this information cannot be guaranteed because individuals are likely to provide untruthful answers when asked about self-involvement in an illegal behaviour (Tourangeau & Yan 2007).Yet, to date, a critical research gap is that few empirical studies have used a multi-methods approach to provide a holistic picture of the prevalence of poaching. This thesis addresses this gap by employing techniques from a range of both ecological and social science disciplines to detect and estimate poaching levels.

Numerous methods and approaches developed in social science can be used to estimate levels of illegal or undesirable behaviours. Here, I provide a general overview of some key methods of investigating poaching, which will be discussed in greater detail in the following chapters. Many of the specialised methods for estimating the prevalence of poaching often employ different administration techniques designed to keep an interviewer from knowing respondents' answers, thereby reducing question sensitivity and untruthful responses due to social desirability bias. For example, self-administered questioning eliminates the interviewer from the process and allows respondents to administer their own survey (Tourangeau & Yan 2007). Another approach is to use indirect questioning methods such as the Randomised Response Technique (RRT; Warner 1965) or the Unmatched Count Technique (UCT; Droitcour *et al.* 1991), which use statistical noise to obscure answers from interviewers, while allowing post-hoc calculations to determine the level of respondents who truthfully admitted to the behaviour. A second type of indirect questioning is to ask about perceptions of other people's behaviours. This approach reduces social desirability bias because it allows respondents to project their beliefs and behaviours through a mask of impersonality, thereby encouraging more truthful representations of behaviours and their prevalence in society (Fisher 1993).

Meanwhile, a lesser-utilized approach is to develop theoretically grounded proxy indicators or measures of poaching (but see St. John *et al.* 2012; Arias & Sutton 2013). For example, people who engage in an illegal behaviour often misperceive a 'false consensus' that others do as well, and therefore regularly overestimate the prevalence of that behaviour throughout society (Ross *et al.* 1977). If an established level of poaching exists for comparison, these overestimations can then potentially be used to indicate self-involvement in poaching (e.g. Petróczi *et al.* 2008; St. John *et al.* 2012). Furthermore, multiple theories in social psychology and criminology have repeatedly demonstrated how human behaviour is effectively learned from observing the actions (and their consequences) of our peers (e.g. Bandura & Walters 1977; Fischer 1995; Akers 2011). Thus, another potential indicator of poaching could be elicited by asking whether an individual knows a poacher, which would be considerably less confronting than directly asking someone if they poach.

Understanding poaching behaviour

In addition to understanding the prevalence of poaching, understanding why people poach is crucial for informing targeted behavioural interventions such as communication and outreach campaigns. Human behaviour is remarkably complex, and a wide variety of disciplines explore different aspects of human decision-making and subsequent behaviours. Thus, a large number of theories can be used to study and understand resource user's decisions to poach. However, many of these theories have been developed in one specific discipline, each of which has its own language, approach, and scale for examining behaviour. Accordingly, efforts to understand and curtail poaching often suffer from what has been called 'disciplinary silo thinking' and fail to capture all of the dimensions of poaching phenomena (Von Essen *et al.* 2014).This lack of multidisciplinary investigations of poaching is a second critical research gap addressed in this thesis.

This thesis integrates key concepts and theories from six social science fields. Becker's (1968) "Crime and Punishment: An Economic Approach" was the first theoretical framework developed to examine individual criminality (Kuperan & Sutinen 1998), and forms much of the foundation of modern criminology. Based on microeconomics, the general premise of rational choice theory is that people make 'rational', or purposeful decisions after weighing and comparing the costs and benefits of performing a behaviour. Thus, a classic criminological approach to compliance management is based on deterrence theory (e.g. Sherman 1993), which seeks to make the consequences greater than the benefits of breaking the law, or in this case, poaching. This is accomplished through the use of punitive or coercive measures, such as fines, gear confiscation, or, in extreme cases, imprisonment. Yet, numerous theories in social psychology and other disciplines demonstrate that human behaviour is not rational, and is instead heavily influenced by many more context specific characteristics, such as the actions of our peers, perceived social norms, and interactions with the governing authorities.

Second, the theory of planned behaviour (TPB; Ajzen 1991) is widely used in social psychology to understand and predict a wide range of human behaviours (Armitage & Conner 2001), including environmental behaviours such as compliance with conservation rules or regulations (e.g. Steinmetz *et al.* 2014). As an extension of the theory of reasoned action (Fishbein & Ajzen 1975; Ajzen & Fishbein 1980), the TPB is designed to capture people's deliberative decision-making processes. An important distinction here is that the TPB investigates behavioural drivers at the level of the individual (rather than the community) for a specific behaviour (e.g. poaching a specific animal in a specific context). Overall, the TPB postulates that people's behaviours are moderated by their intention (including motivations) towards the behaviour, which is itself shaped by three independent pre-determinants; attitudes towards the behaviour; subjective norms, or the social pressures associated with the behaviour; and perceived behavioural controls, or the perceived difficulty of performing the behaviour (Ajzen 1991). In addition, each of these three constructs are themselves shaped by antecedent, corresponding beliefs that reflect an individuals' underlying cognitive processes and structure (Armitage & Conner 2001).Yet, the general

TPB framework may overlook context-specific drivers of behaviour, and previous reviews have recommended expanding or tailoring the TPB to better fit the specific behaviour being investigated (Conner & Armitage 1998; Armitage & Conner 2001). Therefore, this thesis uses the TPB as a foundation on which to expand and incorporate numerous other variables and components from other disciplines and theories that are pertinent for investigating poaching in the context of MPAs.

A third disciplinary focus that is particularly relevant is centred on the effect of normative influences, which are increasingly recognized as important for a range of environmental behaviours, including poaching (e.g. Cialdini 2007; Thomas *et al.* 2016). The original composition of the TPB does acknowledge the importance of norms, but it often fails to capture all of the different normative pressures surrounding a behaviour (Armitage & Conner 2001). Detailed investigations of normative influences describe three distinct categories of norms that are likely to affect fisher's behaviour: 1) descriptive norms, or what other people do in a given situation; 2) injunctive norms, or the perceived moral rules of the group; and 3) personal norms, or a person's own values concerning a behaviour, such as morality or perceived responsibility (e.g. Conner & Armitage 1998; Cialdini 2007; White *et al.* 2009). Yet, the disposition and relationships between these norms are notably complex because they in turn reflect the dynamic and flexible nature of normative influences in societies (Cialdini *et al.* 1991). For instance, a fisher may believe that others regularly poach (descriptive social norm), while also believing that it is socially (injunctive social norm) and morally (personal norm) unacceptable. The importance of these normative influences also depends on the amount of attention an individual pays them, and whether they identify with the particular referent social group (Cialdini *et al.* 1991; Cialdini & Goldstein 2004; White *et al.* 2009). In addition, the strength or influence of these normative pressures also tends to change depending on the referent group's social distance; close peer groups like family and friends typically have stronger normative influence on an individual's behaviour than acquaintances, who in turn have more influence than a member of the general public (e.g. Yanovitzky *et al.* 2006). These influences are further complicated when studying a large population, because perceived norms and values may change between different social groups or subcultures that are likely to exist within the larger population.

Numerous other theories also exist to explain the complex interactions and influences of norms in society. For instance, social norms theory (Berkowitz 2005) posits that unhealthy behaviours (such as poaching) are reinforced by two social phenomena that cause people to overestimate the prevalence of that behaviour: pluralistic ignorance and false consensus. Pluralistic ignorance occurs when people incorrectly think others perform illegal or unhealthy behaviours more often than themselves, when they are actually similar. If left unchecked, these misperceptions may lead normally compliant individuals to adjust their behaviour to fit the misperceived norm, or to begin poaching because they believe others do as well (Prentice & Miller 1996; Schroeder & Prentice 1998). Furthermore, fishers who poach may also overestimate the prevalence of poaching by others because they perceive a 'false consensus' that other fishers also poach, when they are actually compliant (Berkowitz 2005). If operative, these two phenomena are likely working in tandem to reinforce and potentially encourage non-compliance in the fisher populations.

A fourth realm of relevant research is grounded mostly in social psychology and law, and examines how people's compliance behaviours are influenced and shaped by their perceptions of the legitimacy and trustworthiness of management and enforcement authorities (e.g. Kuperan & Sutinen 1998; Tyler 2006b, 2010; Levi *et al.* 2009). These perceptions often result from an individual's interactions with these agencies, thereby making identification with enforcement personnel and perceptions of both procedural and distributive justice important(e.g. Levi *et al.* 2009; Tyler 2010; Turner *et al.* 2016). If these preconditions are not met, resource users may decide to poach in acts of defiance, resistance, or rebellion against illegitimate or unjust management practices (e.g. Sherman 1993; Bell *et al*. 2007; Kahler & Gore 2012).

Similarly, a fifth relevant approach is to construct a typology of motivational categories for poaching. Although currently underutilized, this approach has been used to explain poaching in terrestrial and freshwater contexts (e.g. Muth & Bowe 1998; Kahler & Gore 2012; Von Essen *et al.* 2014). Yet, a critical shortcoming is that much of the literature discussing motivations to poach is speculative, rather than empirically grounded (Muth & Bowe 1998; Kahler & Gore 2012). In addition, these motivations to poach are often based on answers provided through interviews or surveys of previous violators or conservation officers (e.g. Bessey 1985; Eliason 2004). The advantage of this approach is that it provides direct answers from known poachers, or those that work to apprehend them. Yet, information gathered in this fashion should be interpreted with caution, because it is either provided from offenders themselves (and answers may therefore be untruthful or misleading), or by an enforcement officer who may unwittingly project their own beliefs and perception on poachers' behaviours. Furthermore, it may be difficult to gain access or permission to survey known offenders. Accordingly, an easy adaption of this typology approach is to indirectly examine the motivations to poach by eliciting people's perceptions of others behaviours, rather than their own (e.g. Kahler & Gore 2012).

Sixth, the commons literature describes the critical role that institutional design principles have in shaping resource user's behaviours, including compliance. In brief, institutions can be generally defined as the rules, both formal, and informal, that structure and influence people's behaviours and interactions (Ostrom 2005). Thus, institutional design principles should promote conditions that lead to cooperation and compliance in co-management contexts (Ostrom 1990). For instance, previous research has demonstrated how graduated sanctions (i.e. sanctions or penalties that increase in severity along with the frequency or severity of the offense) were positively related to fisher compliance in a range of management settings, including protected areas (Cinner *et al.* 2012). Similarly, participation in decision-making processes and local enforcement was an important determinant of resource user compliance in state-owned forest commons (Epstein 2017). However, many of the world's marine protected areas regularly fail to deliver their expected outcomes due to critical shortfalls in capacity (Gill *et al.* 2017). Natural resource management agencies are therefore increasingly seeking to bolster compliance by engaging latent surveillance by resource users (GBRMPA 2016; Green 2016; Kohn 2016), but little is known about the conditions and institutions that encourage or discourage this type of informal surveillance and enforcement. This constitutes the third critical research gap addressed in this thesis.

Study objectives and thesis structure

The broad objective of this thesis is to contribute to the theory and application of measuring, understanding, and addressing non-compliance in conservation. I contribute to this theory with a detailed, in-depth investigation of poaching behaviours by fishers in MPAs. Earlier, I identified three critical knowledge gaps in the literature, which I address through three primary research questions (see also Fig. 1.1 below):

1) How can the prevalence of poaching be measured given its cryptic nature? (Chapter 2)
2) How are fisher's socioeconomic characteristics related to key indicators of poaching behaviour? (Chapters 3 & 4)
3) What influences people's self-reported voluntary surveillance and enforcement behaviours? (Chapter 5)

I address these questions through theory-driven research that focuses largely on the context of poaching by recreational fishers. Recreational fishers are exciting study subjects for this type of investigation, because their poaching decisions are likely more complex than the desire for financial gain or livelihood concerns. Throughout my thesis, I use theories and approaches developed in a wide range of scientific disciplines including ecology, social psychology, and criminology to quantify and empirically explore the complexities that characterize human behaviour and (non)compliance. After applying these theories, I discuss my findings in the context of compliance management, thereby using cutting edge social science theories and techniques to address critical research priorities identified by compliance managers in one of the world's most iconic marine parks.

An additional aim of this thesis was to develop a metric of fish behaviour (flight-initiation distance; FID, which measures how close a diver can approach a fish before it flees) for use as a proxy indicator of poaching in no-fishing reserves. In short, previous research has shown that fish become more flighty when they are exposed to fishing (Johannes 1981), and in some instances, even if they are inside marine reserves while the surrounding seascape is heavily fished (Feary *et al.* 2011; Januchowski-Hartley *et al.* 2011; Januchowski-Hartley *et al.* 2015). I initially aimed to determine whether FID could be used to indicate instances of poaching inside a no-fishing marine reserve. This required two research components or stages. The first stage of research was to examine whether FID of targeted fish species was markedly different in no-fishing reserves compared to areas open to fishing. The second stage was designed as a removal experiment, where I planned to catch and remove fish from a no-fishing reserve (i.e. simulated poaching event), and subsequently document the magnitude and rapidity of changes in FID for remaining fish populations, as well as the time period required for FID to return to pre-poaching levels. These are three critical pieces of information that must be established before FID can be employed with any sort of confidence to indicate poaching. However, I was unable to secure research permits for this second stage, and had to modify my thesis structure accordingly. Therefore, the initial stage of research is included as Appendix 1.

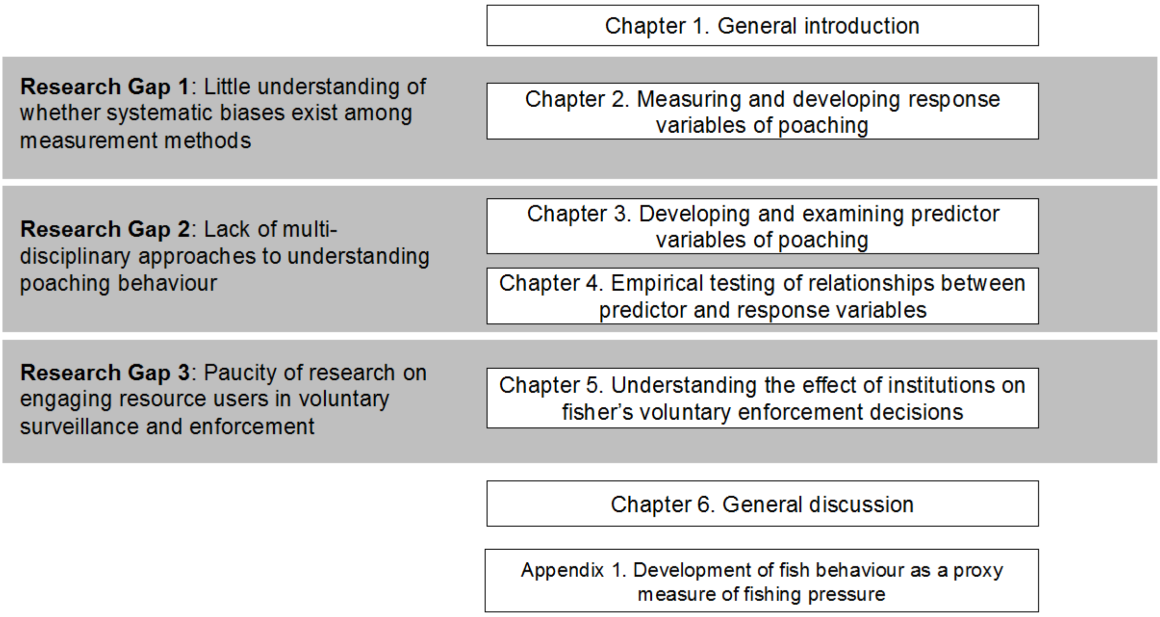

Fig. 1.1. Chapter structure for this thesis including indications of research gaps addressed.

Study system

The majority of this research (Chapters 2-4) is situated in Australia's Great Barrier Reef Marine Park (GBRMP), which is a large multi-use marine park that extends more than 2000 km along Australia's eastern coast and encompasses ~345,000 km^2. It is widely considered one of the best-managed coral reef ecosystems in the world. Activities such as recreational fishing are regulated by a multiple-use spatial zoning system that confines different activity types to specific locations (e.g. areas open to fishing and no-fishing reserves), and additional fisheries regulations such as catch and size limits. Studying poaching by recreational fishers in the GBRMP is an ambitious task because it has a large, diverse, and widely distributed recreational fisher population – roughly 171,000 recreational fishers live adjacent to the GBRMP and use numerous fishing gears, including line and spearfishing (Fig. 1.2a, b; Webley *et al.* 2015). The GBRMP has a comprehensive risk-based compliance management system to direct and coordinate multi-agency enforcement vessels and patrols (Fig. 1.2c), yet the sheer size of the park makes compliance monitoring an ever-present challenge. Furthermore, recent enforcement reports indicate that poaching infractions by recreational fishers may be increasing, for reasons unknown (GBRMPA 2015). It is therefore an ideal, and challenging study system for this investigation.

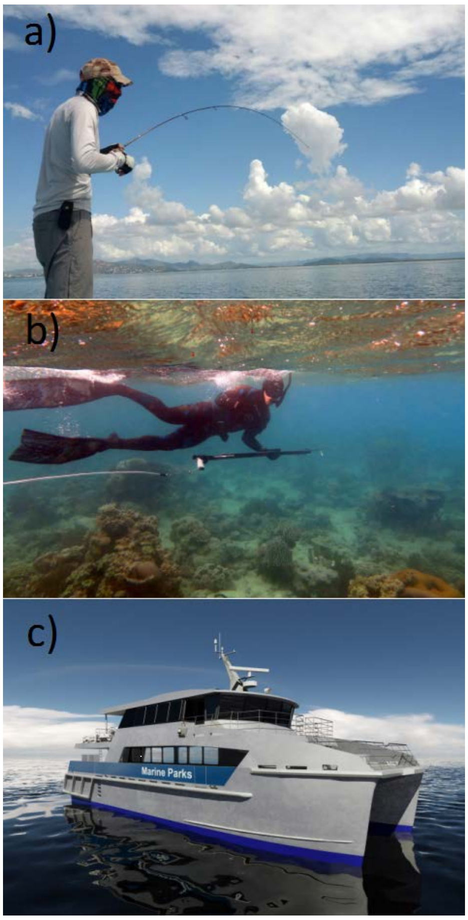

Fig. 1.2. Recreational fishing is a popular pastime for people living adjacent to the 2000 km long Great Barrier Reef Marine Park. a) Line fishing is the predominant type of recreational fishing, but b) spearfishing is rising in popularity. c) The Great Barrier Reef Marine Park Authority is responsible for coordinating enforcement patrols with Queensland's Department of National Parks, Recreation, Sport and Racing, and the Queensland Police. Photos courtesy of (a) Adrian Arias and (c) Incat Crowther.

In addition, addressing the third knowledge gap (i.e. paucity of information on engaging resource users in voluntary surveillance and enforcement) necessitated the expansion of this thesis' study scope beyond the GBRMP. I therefore collate and analyse data from six additional countries (Costa Rica, Indonesia, Papua New Guinea, Kenya, Tanzania, & Madagascar) to answer the third primary research question (i.e. What influences people's self-reported voluntary surveillance and enforcement behaviours?). More information on this study system and context is available in Chapter 5.

CHAPTER 2: A social-ecological approach to assessing and managing poaching by recreational fishers

2.1 Synopsis

Detecting and measuring the prevalence of poaching is a critical component of compliance management. Yet, estimating the level of an illegal behaviour is inherently difficult, and every measurement method has biases and limitations. However, most investigations of compliance use only a single method to estimate poaching, rather than developing specialized multi-disciplinary approaches to assess poaching. Here, I use a social-ecological approach to estimated poaching by recreational fishers in no-fishing reserves of the GBRMP, Australia. This multi-disciplinary investigation incorporated social surveys and quantified derelict (lost/discarded) fishing gear to reveal three key findings: 1) between 3-7% of fishers admitted to poaching within the last year using specifically designed estimation techniques (i.e. self-administered questioning, randomised response technique, and unmatched count technique); 2) poaching activities were often concentrated at certain times (holidays) and in specific places (poaching hotspots); and 3) personally knowing a poacher and perceived levels of poaching may be useful proxy indicators of poaching. My combined social-ecological approach revealed that even in an iconic marine park such as the GBRMP, poaching is higher than previously assumed, which has implications for effective management.

2.2 Introduction

Detecting and assessing compliance levels is a critical component of compliance management. Numerous methods can be used to estimate poaching levels in MPAs, but each is subject to its own biases, limitations, and applicability (Bergseth *et al.* 2015). For instance, indirect observations such as quantifying derelict fishing gears can provide indications of fishing pressure inside no-fishing reserves compared to fishing areas, but cannot indicate who is poaching, or why (Arias 2015; Bergseth *et al.* 2015). Conversely, social surveys can provide estimates of the prevalence of non-compliance in a population of fishers, but it is often difficult to ascertain the biological impacts of this poaching without labour-intensive models (e.g. Little *et al.* 2005). Yet, traditional examinations of compliance are typically limited to a single disciplinary approach that provides only part of the story (e.g., quantifying fish populations and making assumptions about poaching prevalence, or using social surveys to ask about poaching, without any field-based "ground-truthing"). Although multi-disciplinary approaches are demonstrably beneficial for fully understanding compliance, relatively few studies to date have incorporated a multi-disciplinary approach to estimate compliance levels.

Social surveys are increasingly employed to examine non-compliance in MPAs, but estimates of non-compliance are often subject to underreporting due to social desirability bias (Tourangeau & Yan 2007). However, specialized techniques developed in the social sciences can be used to reduce untruthful answers by eliminating the social pressures that people perceive when being directly questioning by an interviewer. Two methods in particular have been increasingly implemented to investigate poaching in conservation; the randomised response technique (RRT; Warner 1965) and the unmatched count technique (UCT; Droitcour *et al.* 1991). The "forced answer" RRT aims to reduce bias by using a randomising device (e.g. dice) to obscure the respondent's answer from the researcher so that subjects are more likely to answer truthfully. For example, instructions may ask respondents to hide the dice from the researcher and answer yes if they rolled a 1, no if they rolled a 12, and answer truthfully if they rolled anything else. In this case, only the respondent knows if they are instructed to truthfully answer about poaching, or are being "forced" by the device to automatically answer yes or no, thereby ensuring confidentiality. However, the probabilities of being forced to answer “yes”, “no” or truthfully are known, which allows post-hoc determination of the percentage of respondents who admitted to poaching (Warner 1965). The UCT also protects respondents’ confidentiality by asking respondents to indicate how many, but not which, behaviours they have engaged in from a list (Droitcour *et al.* 1991). Respondents get either a control or treatment list; the control list asks respondents about legal behaviours, while the treatment asks about the same legal behaviours, plus poaching. Comparing the mean number of behaviours reported by each population (control and treatment) then indicates the percentage of the population reporting the sensitive behaviour (Dalton *et al.* 1994)

Aside from the increasingly implemented RRT and UCT, numerous other techniques can be used to reduce untruthful answers when asking about poaching. For instance, self-administered questioning (SAQ), removes the interviewer completely from the process, thereby reducing social desirability bias (Tourangeau & Yan 2007). Alternatively, researchers can seek to understand poaching by eliciting respondents’ perceptions about other people’s behaviours. This type of indirect questioning allows respondents to project their own beliefs, attitudes, and behaviours through a mask of impersonality, thereby reducing social desirability bias (Fisher 1993).

Eliciting perceived levels of poaching reduces bias because it does not directly ask respondents if they poach, and can produce estimates similar to those provided by more robust methods such as the RRT (Arias & Sutton 2013; Cross *et al.* 2013). In addition, people that engage in illegal or undesirable activities are more prone to misperceive a ‘false consensus’ that others do the same, and therefore overestimate the overall prevalence of that activity (Ross *et al.* 1977; Mullen & Hu 1988). This ‘false consensus effect’ has been documented in a range of contexts, including carnivore killing (St. John *et al.* 2012), illegal hunting (St. John *et al.* 2015) and fishing (Hatcher *et al.* 2000; Arias & Sutton 2013; Bova *et* *al.* 2017). Furthermore, these overestimations have also been used to indicate self-involvement in illegal behaviours (e.g. Petróczi *et al.* 2008; St. John *et al.* 2012), and may therefore be useful in indicating whether a fisher poaches. Similarly, multiple theories in social science demonstrate that individuals are heavily influenced by the behaviours of their peers. For instance, subculture theory suggests that the recreational fishing population is likely composed of numerous social groups or subcultures that have their own set of values or norms regarding poaching (Fischer 1995). Likewise, social structure social learning theory (SSSL) describes how people learn from observing the behaviours (and consequences) of the people and peer groups they interact with (Akers 2011). In this context, subculture and SSSL theory would predict that fishers would learn, and look more favourably on poaching if they interact with, and observe the actions of poachers who successfully catch more fish, especially if they avoid detection or punishment. Thus, an additional potential proxy measure or indicator of poaching is whether a fisher personally knows someone who poaches. Yet, very few studies have yet explored the potential of using these measures as potential proxy indicators of poaching (but see St. John *et al.* 2012; Arias & Sutton 2013).

The Great Barrier Reef Marine Park (GBRMP; Fig. 2.1) is widely considered one of the best-managed marine ecosystems in the world, but considerable amounts of poaching by fishers still occur (Arias & Sutton 2013; Williamson *et al.* 2014; GBRMPA 2015). Research undertaken in 2009 indicated that the level of poaching by recreational fishers is ~10% (Arias & Sutton 2013), but information from the Great Barrier Reef Marine Park Authority (GBRMPA) indicates that this level may be rising (GBRMPA 2015), for reasons yet unknown. Here, I use the iconic GBRMP as a lens through which I investigate two questions that are critical for effective marine conservation: 1) What is the prevalence of poaching?; and 2) How do potential proxy indicators or measures of poaching compare to established measurement methods?

2.3 Methods

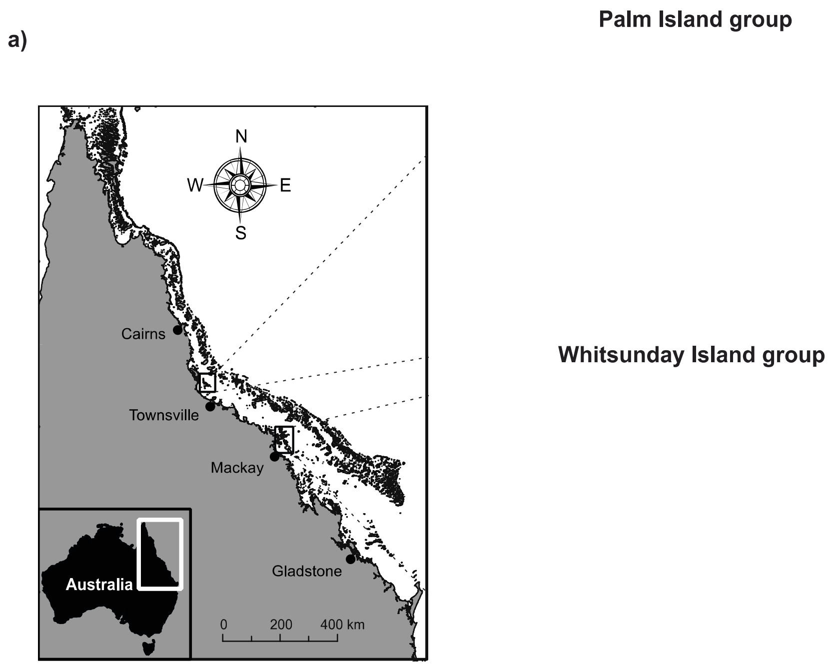

Fig. 2.1. a) Study sites for underwater visual censuses of discarded fishing lines in the Great Barrier Reef Marine Park (GBRMP), Australia: b) Palm Island group and c) Whitsundays Island group. Blue and yellow = fished zones, green = no-fishing zones. Yellow zones are areas where fishers are limited to a single hook and line, and cannot spearfish. Red circles indicate poaching hotspots as identified in: d) Density of re-accumulated fishing line by site (per 1000m^2). Horizontal dotted lines show mean (±SE) re-accumulated line densities for no-fishing (4.28 ± 0.88) and fished sites (4.84 ± 0.75). Poaching hotspots are red, and defined as sites with re-accumulated line densities higher than the mean density. Site numbers correspond to site locations illustrated in b) and c). * denotes sites open to fishing by traditional owners under Native Title Act (1993)

Study sites

This study examined poaching in the Townsville-Whitsunday region of the GBRMP, which had the highest number of detected poaching infractions between 2014-2015 (GBRMPA 2015). Social surveys were conducted in Townsville, Australia, while derelict fishing gears were quantified on fringing coral reefs of two inshore island groups located in the Townsville-Whitsunday region (the Palm and Whitsunday Island groups; Fig. 2.1, Table 2.1). Townsville is located 40 NM from the Palm Island group, and was chosen as the main study site for social surveys because it is the major urban centre in northern Queensland, which has a large population of recreational fishers (36,000; Webley *et al.* 2015). Derelict fishing gears were quantified on coral reefs of the Palm and Whitsunday Island groups, which both receive high levels of tourism use and recreational fishing effort (McCook *et al.* 2010; Webley *et al.* 2015). Traditional owners are legally allowed to fish in no-fishing reserves of the Palm Island group under the Native Title Act (1993), with implications that are further examined in the discussion. Aside from no-fishing reserves (green zones) and areas open to fishing (blue zones), both island groups also contain limited fishing areas where fishers are limited to a single hook and line (yellow zones; Fig. 2.1).

Derelict fishing gear surveys

Underwater visual census (UVC) was used to quantify the total number of derelict fishing lines at 14 monitoring sites in the Palm Islands and 16 sites in the Whitsunday Islands (30 sites in total) during 2012 and 2014 (Fig. 2.1). Each fished site (15) was paired with a no-fishing site (15) that was similar in terms of reef structural morphology, usage patterns, and access to boaters. Site selection was based on a larger collection of pre-existing long term-monitoring sites (see Williamson *et al.* 2014). In most cases, we chose sheltered site locations that are typically accessible to fishers in calm to moderate conditions (<15 knot winds).

Immediately following the 2012 UVC, teams of 4 – 6 divers removed all derelict fishing gear from the 30 monitoring sites between depths of 4 m and 12 m (16 – 24 diver-hours spent cleaning each site). Transect tapes were used to measure the length and width of survey areas, ensuring that very little, if any gear remained after the initial clean-up. All sites were resurveyed in 2014 to quantify the amount of gear that had re-accumulated since 2012 (Table 2.1). Divers swam the entire length of each sighted line during UVCs and cleaning to ensure that each line was only recorded once, regardless of its length. Differences in mean line accumulation per site between fished and no-fishing areas were tested using an independent samples t-test.

Social surveys

A total of 682 social surveys were conducted at Townsville boat ramps from April to September 2015. All surveys were collected on iPads using iSurvey software. All respondents were approached, given a brief explanation of the survey (i.e. 'I am asking people about green zones (GBRMP no-fishing zones), the way they are managed, and whether you think people follow the rules'), and invited to participate. To maximize truthful responses, surveyors emphasized that it was an anonymous survey designed and conducted by student researchers that had no affiliation with local law enforcement or management authorities. In addition, all respondents were assured that surveyors were not collecting any identifying information (e.g. registration numbers, etc.) that could later be used to identify them. I also used familiar and forgiving wording (Tourangeau & Yan 2007) to describe the survey aims, thereby reducing potential response bias due to the sensitivity of the topic. This included using descriptions like 'fishing in a green zone' and avoiding stigmatizing terms such as 'fishing in a no-take/no-fishing zone', 'compliance' or 'poaching'.

Table 2.1. Study site information for underwater visual census of discarded fishing gear. [a] Accumulated line values were corrected to account for differences in total area surveyed between years.

Region	Island	Site	Zone	Line removed/1000m^2 (2012)	Accumulated line/1000m^2 (2014)[a]	2012 survey area (m^2)	2014 survey area (m^2)
Palm Islands	Pelorus	1	Fished	13.71	5.63	6200	6400
Palm Islands	Pelorus	2	Fished	20.16	9.38	6400	6400
Palm Islands	Pelorus	3	Fished	20.20	3.37	5000	7425
Palm Islands	Pelorus	4	Fished	12.86	5.83	7000	7200
Palm Islands	Pelorus	5	Fished	22.75	7.78	8000	7200
Palm Islands	Orpheus	6	No-take	20.76	6.50	6600	8000
Palm Islands	Orpheus	7	No-take	1.50	1.00	6000	6000
Palm Islands	Orpheus	8	No-take	4.87	2.11	7600	7600
Palm Islands	Orpheus	9	No-take	4.57	3.14	7000	7000
Palm Islands	Orpheus	10	No-take	3.43	1.41	7000	6400
Palm Islands	Curacoa	11	Fished	43.94	10.44	6600	6800
Palm Islands	Curacoa	12	No-take	33.64	8.86	4400	4400
Palm Islands	Curacoa	13	No-take	47.25	11.75	4000	4000
Palm Islands	Curacoa	14	Fished	9.00	2.63	8000	8000
Whitsunday Islands	Hayman	15	No-take	12.73	6.85	4400	7450
Whitsunday Islands	Hayman	16	Fished	12.32	1.25	5600	4000
Whitsunday Islands	Hook	17	Fished	17.96	2.63	5400	6075
Whitsunday Islands	Hook	18	Fished	13.64	4.83	6600	7450
Whitsunday Islands	Hook	19	Fished	32.50	4.26	4400	7050
Whitsunday Islands	Black	20	No-take	22.06	3.08	3400	7150
Whitsunday Islands	Black	21	No-take	8.24	3.12	5950	3850
Whitsunday Islands	Langford	22	No-take	10.52	8.58	5800	6170
Whitsunday Islands	Hook	23	No-take	5.79	0.88	7600	8000
Whitsunday Islands	Hook	24	Fished	14.55	5.75	5600	4000
Whitsunday Islands	Whitsunday	25	Fished	4.00	6.00	4000	7000
Whitsunday Islands	Dumbbell	26	No-take	3.57	3.88	7000	8000
Whitsunday Islands	Dumbbell	27	No-take	3.18	0.50	6600	8000
Whitsunday Islands	Deloraine	28	Fished	5.29	0.86	4725	7000
Whitsunday Islands	Whitsunday	29	Fished	3.26	0.63	4600	8000
Whitsunday Islands	Esk	30	No-take	5.00	2.63	3600	8000

Estimating poaching prevalence with specialized questioning techniques

To examine levels of poaching, I used three specialized techniques specifically designed to reduce bias and underreporting and two potential proxy measures of poaching: 1) self-administered questioning (SAQ; Have you knowingly fished in a green zone within the last 12 months?); 2) the Randomised Response Technique (RRT; Have you knowingly fished in a green zone within the last 12 months?); 3) the Unmatched Count Technique (UCT); 4) perceived level of poaching (PLP; In your opinion, what proportion of recreational fishers do you think have fished in a green zone in the last 12 months [0-100%]?); and 5) personally knowing a poacher (PKP; Do you personally know someone who fished in a green zone in the last year?; Table 2.2). All randomly selected respondents answered SAQ, PLP, and PKP questions, and either (i) RRT, (ii) UCT treatment, or (iii) UCT control list (i.e., surveyors switched between techniques (i), (ii), and (iii) with each subsequent respondent during a period where I trialled the RRT and UCT methods [April-December 2015]. Evaluation and comparison of these two methods during the trial period indicated that the RRT was more robust, and enabled further analysis of behavioural drivers of poaching (see Chapter 4), so I retained the RRT and dropped the UCT method after the trial; Table 2.2).

SAQ was accomplished by handing an iPad to the respondent, who was asked to truthfully answer whether they had fished in a no-fishing reserve, knowing that the surveyor would not see their response. Fishers also estimated the level of poaching (PLP) to the nearest 1%. Perceived levels of poaching were used because previous research indicates that this method reduces question sensitivity, and provided estimates that align with those provided by more robust techniques (i.e. RRT; Arias & Sutton 2013; Cross *et al.* 2013). Although yet to be explicitly tested, personally knowing a poacher is theoretically plausible for use as a proxy indicator of compliance, and also removes social desirability bias compared to more direct questioning techniques.

The specific randomised response technique employed in this study was the "forced choice" design (Warner 1965). This design uses a randomising device (Fig. 2.2) to "force" respondents to answer in one of three ways (Fig. 2.3). Depending on the colour of the bead shown by the randomising device, respondents were directed to: 1) automatically answer "yes" (5%); 2) automatically answer "no" (10%); or 3) truthfully answer the question (85%): "Did you knowingly fish in a no-take zone in the last 12 months?" (Fig 2.2). As in Horvitz *et al.* (1976), the following equations were used to estimate the level of non-compliance:

(1) To calculate the proportion of the population (π_y) with the non-sensitive attribute (π_y):

$$\pi_y = \frac{P_2}{(P_2 + P_3)} = \frac{P_2}{1 - P_1}$$

(2) To calculate the proportion of the population with the sensitive attribute (π_S) when (π_y) is known. P is the probability of choosing the sensitive attribute ($P=P_1$), while λ is the observed P of "yes" in the RRT section:

$$(\pi_S|\pi_y) = \frac{\lambda - (1 - P)\pi_y}{P}$$

(3) To calculate the variance, with λ being the probability of a "yes" response ($\lambda = P\pi_S + (1 - P)\pi_y$):

$$var(\pi_S|\pi_y) = \frac{\lambda(1 - \lambda)}{nP^2}$$

Fig. 2.2. Randomising device used with randomised response technique. The device has a small viewing window that enables respondents to view the colour of the bead without the surveyor's knowledge. The colour of the bead was used to direct the respondent how to answer (i.e., forced yes (5%), forced no (10%), or truthfully (85%)). See Figure 2.3 for decision tree.

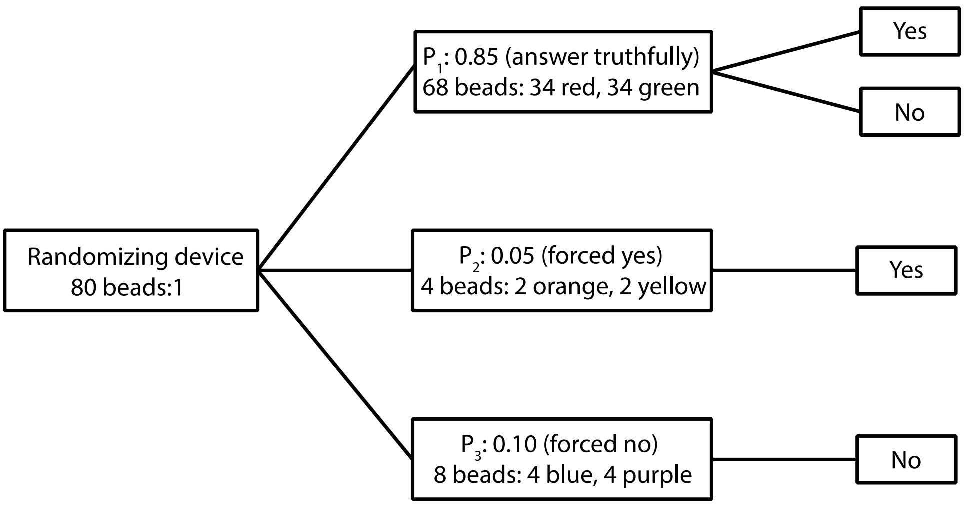

Fig. 2.3. Randomised response technique decision tree (Adapted from Arias & Sutton 2013).

The UCT was first introduced by Miller (1984) and applied by Dalton *et al.* (1994) . In this method, a respondent is randomly and unknowingly assigned to either a "control" or a "treatment" group. If assigned to the control group, the respondent was asked to identify how many, *but not which*, of four legal (control) activities they have done in the last year (Table 2.2). The list for the treatment group consisted of the same four legal activities as the control group, with the addition of a fifth, illegal activity (fishing inside a no-take zone in the last year; Table 2.2). An estimate of poaching levels can then be achieved by subtracting the mean number of the control group from the mean number of the treatment group (e.g. treatment mean of 3.5 minus control mean of 3.3 indicates a poaching level of .20, or 20%). This method thus allows researchers to compare the mean number of activities for the control and the treatment group, and enables an estimation of non-compliance levels (people who fished inside a no-take zone in the last year).

Analysis of proxy indicators

I used Welch Two Sample t-tests to determine whether false consensus and subculture/social structure social learning were affecting fisher's perceptions of poaching. To determine if the false consensus effect was operative, I compared the perceived levels of non-compliance (0-100%) of fishers who admitted to poaching (n = 21) via a self-administered question (i.e. "Have you fished in a green zone in the last 12 months?") with fishers who did not admit to poaching. To identify whether subculture/social structure social learning was operative, I compared the perceived levels of non-compliance (0-100%) for fishers who knew poachers (n = 86) and fishers who did not (n = 595). Because I had unequal sample sizes for both t-tests, I also calculated Hedges' *g* values, which provide a measure of effect size that is weighted according to the sample size for each group (Lakens 2013), using the *lsr* package in R (Team 2016).

Table 2.2. Techniques used to estimate poaching levels in the Great Barrier Reef Marine Park. Respondents always received techniques 1-3 and either technique 4 or 5 (a or b) during an initial methodological trial period (April –December 2015). During this time, surveyors switched between methods 4, 5a, and 5b with each subsequent survey. Differences in sample sizes therefore reflect the discontinuation of the Unmatched Count Technique after the initial pilot study indicated that this was the least robust method for estimating and understanding compliance.

Technique	Brief description	Metric or output	Result
1) Self-administered questioning	Respondents answer using iPad	"Yes" or "no" response	21 out of 681 fishers admit to poaching = 3.1%
2) Perceived level of poaching	Respondents indicate what percentage of population they believe has fished in a no-fishing reserve in the last year	0 – 100%	682 responses; mean perceived level of poaching= 9.7%
3) Personally know a poacher	Whether respondents knew someone who fished in a no-fishing reserve in the last year	0 – 100%	682 responses; 12.6% report knowing a poacher
4) Randomised Response Technique	Respondents' answer determined/forced by randomising device (% chance of answer type) - Forced "yes" (5%) - Forced "no" (10%) - Truthful "yes" or "no" (85%)	Yes or no response	504 responses; estimated level of poaching = 6.9% (based on formulas provided by Horvitz et al. 1976)
5) Unmatched Count Technique	Respondents identify how many but not which activities they've done in the last year	0-100%	2.578 (treatment mean) – 2.531 (control mean) = .046; 4.7% estimated level of poaching
a) Control – 4 legal behaviours	- Trolling for pelagic species - Spearfishing - Line fishing - Fishing near a no-fishing boundary (outside of)	___ out of 4 possible activities	76 responses; mean = 2.531
b) Treatment – 4 legal behaviours + 1 sensitive behaviour (S)	- Trolling for pelagic species - Spearfishing - Line fishing - Fishing near a no-fishing boundary (outside of) - Fishing in a no-fishing zone (S)	___ out of 5 possible activities	79 responses; mean = 2.578

2.4 Results

Quantification of discarded fishing gear

Derelict fishing gear re-accumulated to varying degrees at all sites in the Palm and Whitsunday Island groups between 2012 and 2014. No differences were detected in the mean density of re-accumulated gear between fished sites and sites in no-fishing reserves (t_2 = 1.085, *df* = 26.275, *P* = 0.2878) (Fig. 2.4a). In the Palm Island group, re-accumulation of fishing gear was concentrated at several sites (i.e. poaching hotspots) within reserves (Sites 6, 12, & 13; Fig. 2.1b, d). Gear re-accumulation was more evenly distributed within and among reserve sites in the Whitsunday Islands, (Table 2.1), but was particularly high at two sites (Sites 15 & 22: Fig. 2.1c, d). With the exception of 2 crab pots and 1 fish trap, all of the gear recorded was monofilament or braided nylon line, most commonly with lead sinkers and hooks attached.

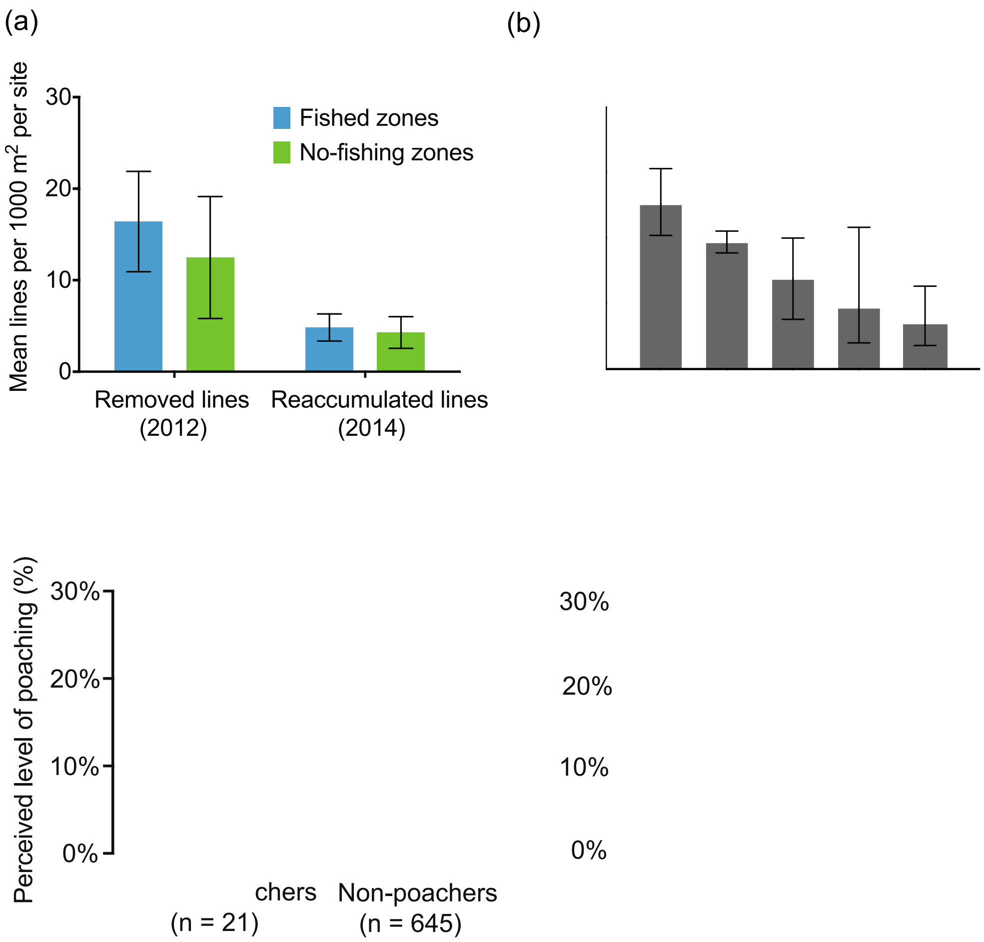

Fig. 2.4. a) Mean (±95% CI) number of fishing lines removed (2012) and re-accumulated (2014) per site for fished (blue bars) and no-fishing (green bars) zones. b) Prevalence of poaching estimates (mean ±95% CI) provided by two potential proxy techniques, Personally knowing a fisher (PKP), Perceived level of poaching (PLP), and three specialized techniques, the Randomised Response Technique (RRT), the Unmatched Count Technique (UCT), and Self-administered questioning (SAQ). c) Perceived non-compliance rates of admitted poachers (n = 21) and non-poachers (n = 645), and d) fishers who knew poachers (n = 86) and those who did not (n = 595). Mean values displayed with 95% confidence intervals.

Estimates of poaching

The estimated prevalence of poaching varied according to the method employed (Fig. 2.4b). The two proxy indicators returned the highest estimates: nearly 13% (12.6 ± 0.01 SE, n = 680 respondents) of fishers personally knew a poacher, while the perceived mean level of poaching was 9.7% (± 0.42 SE; n = 682 respondents, median = 5%). The range of perceived levels of poaching was quite variable, as people thought that anywhere between 0 to 70% of the recreational fisher population had poached within the last year. The specialised techniques returned lower estimates of poaching. The RRT (n = 504 respondents) provided an overall estimate of 6.9%, and detected a considerable pulse in poaching activity over the 2015 Easter weekend (43.6% non-compliance over Easter, compared to 6.9% overall). The UCT yielded a non-compliance estimate of 4.7% (± 0.17 SE; n = 155 total respondents – 76 control, 79 treatment) while self-administered questioning provided an estimate of poaching levels at 3.1% (± .01 SE; n = 681 respondents). Fishers who admitted to poaching via self admitted questioning had larger estimates of non-compliance compared to fishers who did not admit to poaching (15.7% vs. 9.5%; Fig. 2c). This relationship was not statistically significant (t = -1.48, *df* = 20.41, p = 0.154), but suffers from an extremely small sample size. The effect size (*g* = 0.56) however, indicates a medium relationship. In addition, fishers who knew poachers had significantly higher estimates of non-compliance (t = -3.28, *df* = 99.25, p = 0.001, *g* = 0.47) compared to fishers who did not admit to knowing a poacher (14.2% vs. 9%; Fig. 2.4d).

2.5 Discussion

Poaching is a global issue of critical importance for global conservation efforts, which I demonstrate in one of the world's most iconic world heritage-listed marine parks. Inshore coral reef reserves have long been considered some of the best-protected areas within the GBRMP (Davis *et al.* 2004; McCook *et al.* 2010), yet I document surprisingly high levels of poaching in these areas. Interestingly, both of the proxy indicators of poaching provided higher estimates (10-13%) than did the specialised questioning techniques (3-7%). This encourages further examination of these proxy indicators, and suggests that underreporting of self-involvement (which is common in social surveys; Tourangeau & Yan 2007; Nuno & St. John 2015) may still occur even with the use of specialized questioning techniques. Indeed, previous research has highlighted that underreporting of sensitive behaviours can occur even with use of specialized techniques such as the RRT (Landsheer *et al.* 1999; Lensvelt-Mulders & Boeije 2007; St. John *et al.* 2012). The mean level of perceived non-compliance (9.7%) in this study was identical to that estimated by Arias and Sutton (2013), but I found nearly two-fold more fishers that reported knowing someone who had poached compared to Arias and Sutton (2013). The latter result (increase in fishers who knew poachers) may reflect the rising level of non-compliance reported by enforcement patrols (e.g. GBRMPA 2015). If so, this may also suggest that a lag effect exists in people's perceptions – fishers still perceive the same levels of non-compliance, even if they are potentially increasing.

Nonetheless, if the potentially conservative estimates (3-7%) returned by specialised questioning techniques are representative of the entire recreational fisher population (~171,000 recreational fishers live adjacent to the GBRMP; Webley *et al.* 2015), this could suggest that anywhere between 5,000 and 12,000 fishers may have poached in the GBRMP in the last year. While I did not quantify days of poaching per person, the re-accumulation rates of derelict fishing gear indicate that no-fishing reserves in the Palm and Whitsunday Islands may be subject to nearly as much fishing effort as areas that are legally open to fishing. Most of the gear recovered was firmly entangled (or encrusted) on the reef, and typically had lead weights and hooks still attached to the line. For this reason, I am confident that the line was not wind-blown or current-driven, and was instead snagged on the reef and discarded by fishers.

My social-ecological approach helped reveal three critical pieces of information for compliance management: 1) the magnitude of poaching in the GBRMP; 2) the potential applicability of using proxy indicators to examine poaching behaviours; and 3) two mechanisms (false consensus effect and social learning or socialisation) that are likely maintaining and encouraging continued poaching. First, understanding the magnitude of poaching is a necessary component of optimising enforcement in compliance management. Optimising enforcement is crucial because resources for enforcement are always limited; enforcement patrols are financially expensive, logistically difficult, and typically produce low detection rates (Arias *et al.* 2015). Therefore, considerable resources can be saved by targeting patrols to locations and periods where poaching is more likely. Here, the combination of derelict fishing gear surveys and social surveys produced both spatial and temporal information on poaching activities. For instance, my social surveys detected a large pulse in poaching activity (43% of respondents admitted to poaching via RRT) during a holiday weekend with good weather (2015 Easter weekend), which suggests that these are high-risk periods that necessitate greater enforcement presence. Furthermore, most of the poaching hotspots identified through derelict fishing gear occurred near the edges of reserves, or surprisingly, in high usage areas where we expect some degree of passive surveillance to be afforded (e.g., near resorts, research stations, or public mooring areas). This may suggest that a component of poaching activities occur at night, a finding that concurs with a recent assessment by the GBRMPA (GBRMPA 2015). Thus, knowledge of where and when poaching occurs can be used to target enforcement patrols, which should result in higher detection rates, and higher cost-effectiveness.

I also found support for further development and testing of proxy indicators of poaching. The considerable difference in perceived poaching levels between compliers and admitted non-compliers does suggest that the false consensus effect (Ross *et al.* 1977) is operative amongst poachers in the recreational fisher population. While I did not see a statistically significant difference in perceived non-compliance between admitted poachers and non-poachers, this was likely have due to the small sample size of admitted poachers (n = 21). In this case, the Hedges *g* effect size (which accounts for differences in sample sizes) indicates a notable difference in perceived levels of poaching between self-admitted poachers and reported compliers. This parallels previous research describing how non-compliant fishers provide higher estimates of non-compliance than compliant fishers (Hatcher *et al.* 2000; Arias & Sutton 2013; Bova *et al.* 2017). Thus, it appears that the same misperceptions of false consensus that allow poachers to justify and reinforce their noncompliance (e.g. Dawes *et al.* 1977; Ross *et al.* 1977; Berkowitz 2005), may also be used to indicate their self-involvement in poaching (e.g. Petróczi *et al.* 2008; St. John *et al.* 2012).

In addition, fishers who knew poachers had significantly higher estimates of poaching than those who did not. If considered in the context of SSSL (Akers 2011), this may suggest that these fishers could be 'learning' that poaching is more prevalent or acceptable compared to fishers who do not associate with poachers. Alternatively, subculture theory would suggest that poachers are more likely to associate with other fishers who also poach. This latter explanation seems to be at least partially supported by previous research in the GBRMP that demonstrated how recreational fishers in the GBRMP who personally knew poachers were more likely to themselves poach (Arias & Sutton 2013). SSSL also posits that these behaviours are discouraged or reinforced by the observed consequences for those who perform the behaviour (Nicholson & Higgins 2017). Therefore, fishers who know poachers may be more prone to poach if they see others doing it, benefiting from it (e.g. catching more fish), and avoiding punishment (e.g. not getting caught or fined). This is likely occurring here, considering the size of the marine park (~345,000 km^2), and the demonstrably low rates of detection and prosecution of infringements inherent in marine fisheries (Sutinen *et al.* 1990; Kuperan & Sutinen 1998; Arias *et al.* 2015). Ultimately, this lends further support to the idea that poaching subcultures (e.g. Fischer 1995) may exist within the recreational fishing population of the GBRMP, and may be reinforced by social learning within these subgroups. If left unchecked, these poaching subcultures could lead to a negative cascading effect that encourages further non-compliance. To combat this cascade, compliance management could therefore focus enforcement actions and harsh punishments on those who knowingly and repeatedly break the rules (Arias *et al.* 2015), which I discuss below.

A range of tools can be used to increase the harshness of penalties on intentional poachers, whether through fines or the threat of fishing gear confiscation (Becker 1968; Grasmick & Bryjak 1980; Garoupa 1997). Along these lines, the GBRMPA recently increased the fine for poaching in a no-fishing zone from $1800 AUD to $2100 AUD (GBRMPA 2017c). If infringement notices are ineffective in deterring offenders, recidivists can also be prosecuted in Queensland State Courts, where both a criminal conviction and court-imposed fine could result. In addition, the GBRMPA has a variety of further enforcement actions that can be employed to deal with recidivists or problematic individuals, including the power to issue written directions that prohibit or impose conditions on a person's entry to and use of the GBRMP for a period of time (J. Aumend, GBRMPA Field Compliance Management Unit, personal communication). However, increasing the costs and harshness of penalties does run the risk of alienating fishers if they are considered too harsh, unjust, or illegitimate. For instance, previous studies have described how poaching can occur as acts of defiance, resistance, or rebellion against illegitimate or unjust management practices (Bell *et al.* 2007; Filteau 2012; Von Essen *et al.* 2014). Yet, this is unlikely in context of the GBRMPA, which utilises a system of graduated sanctions (the harshness of penalties increases with the extent or severity of infringements), and enjoys high levels of support from fishers (Arias & Sutton 2013).

As mentioned previously, the methods used in this study do have limitations. For instance, derelict gear surveys cannot provide the identity of fishers who discarded the gear, so I was unable to partition gear left by poachers from gear left by traditional owners legally allowed to fish in reserves of the Palm Island group. Thus, sites 12 and 13 (Fig. 2.2) could be traditional owner fishing hotspots rather than poaching hotspots. Regardless, these no-fishing reserves are receiving as much, or more, fishing effort than officially designated fishing areas. Derelict gear surveys are also unlikely to detect poaching by spearfishers because gear is rarely discarded, so other measures like direct observation should be used to assess poaching by spearfishers (Bergseth *et al.* 2015). Similarly, the potential proxy indicators introduced in this study are still in the early stages of development, and require considerably more ground-truthing and verification before they can be used with any level of confidence to unequivocally indicate poaching behaviour. For instance, the estimate derived from the number of fishers who personally knew a poacher is largely dependent on the characteristics of the social network and interconnectedness of each fishers (or poacher) within that network. Thus, it is conceivable that fishers who frequent the same access point may know the same poacher, which could inflate this estimate. Social network analysis can examine the connections and links between different fishers and would therefore be well placed to elucidate this uncertainty.

Reducing poaching and non-compliance is one of the largest challenges to ensuring the efficacy of the world's protected areas (McCauley *et al.* 2016). However, these efforts rely on a solid understanding of human behaviour, reliable measures of clandestine illegal activities, and flexible, adaptive management frameworks that can deal with the uncertainties inherent in these contexts. Thus, future research should prioritize investigations of the cognitive and social aspects of individual behavioural decisions, further development of measures and proxies that can be used to estimate compliance, and the frameworks that regulate human behaviour in conservation.

CHAPTER 3: Discerning the culture of compliance through recreational fisher’ perceptions of poaching

3.1 Synopsis

Curtailing poaching is a vital component of compliance management. However, initiatives aiming to increase compliance with protected areas often fail due to a paucity of detailed information needed to guide targeted management and behavioural interventions. In this regard, effective compliance management requires understanding why resource users are breaking the rules, why these behaviours continue to occur, and how to effectively confront non-compliance. Here, I interviewed 682 recreational fishers of the Great Barrier Reef Marine Park (GBRMP) to examine the social dimensions of compliance management. Specifically, I elicited fisher’s perceptions of management and why others may be motivated to poach, as well as the beliefs, attitudes, normative influences, consumptive orientation, and the perceived behavioural controls that may influence fishers’ poaching behaviours. Encouragingly, most fishers had high perceptions of the legitimacy of management agencies and thought poaching was socially and personally unacceptable. However, my findings suggest that three mechanisms or (mis)perceptions are likely operative and encouraging continued non-compliance by fishers. These include pluralistic ignorance (i.e. compliant individuals overestimating the prevalence of poaching), the perception of better catches in no-fishing reserves, and a perceived lack of deterrence. In summary, these results suggest that extolling certain ecological benefits of marine reserves where enforcement capacity is low could lead to the perverse outcome of encouraging non-compliance, especially when compliant users overestimate the prevalence of poaching. Numerous tools can be used to address and correct these perceptions, including social norms and influence approaches, fear-arousing communications, and social outreach. If properly implemented, these tools and approaches should not only increase compliance but also reduce support (whether active or passive) for a culture of non-compliance.

3.2 Introduction

Effective marine conservation necessitates both maintaining high levels of fisher compliance and reducing non-compliance. Reducing non-compliance in turn requires understanding why fishers are poaching, identifying the underlying mechanisms that may be encouraging these behaviours, and the strategies that can be used to encourage compliance and discourage poaching. Yet, asking people if and why they poach is unlikely to yield truthful responses in instances where poaching is socially undesirable. Yet, anonymous social surveys that are grounded in relevant theories of behaviour can be used to identify the potential mechanisms, perceptions, or beliefs that may be supporting or encouraging continued non-compliance (e.g. Hardeman *et al.* 2002; Berkowitz 2005). This information can then guide targeted behavioural interventions that aim to reduce poaching, such as persuasive communication and social norms marketing or influence approaches (e.g. Ham *et al.* 2009; Abrahamse & Steg 2013; Bova *et al.* 2017).

Numerous theories of human behaviour can be used to examine poaching by recreational fishers. Most applicably, the theory of planned behaviour is regularly used in studies where the ultimate objective is influencing human behaviour (St. John *et al.* 2010). The TPB is often successful in attaining desired shifts in behaviour (Hardeman *et al.* 2002), and is increasingly recommended and applied to study poaching behaviours (e.g. Steinmetz *et al.* 2014; Arias 2015; Thomas *et al.* 2016). However, reviews of the TPB do recommend modifying or extending the TPB framework to better fit the specific behaviour and context in which it is being investigated (Conner & Armitage 1998; Armitage & Conner 2001). For instance, comprehensive investigations of normative influences typically recognise three distinct types of norms. These include descriptive social norms (i.e. the perception of whether others poach), injunctive social norms, (i.e. the perceived social acceptability of poaching), and personal or moral norms (e.g. White *et al.* 2009). The influence of these social norms also depends on an individual's identification (or lack thereof) with referent social groups, and the social distance between the social group and the individual. As such, the normative influences of family members, close friends or other 'ingroup' members are typically stronger than the influence of 'outgroup' individuals, or members of the general public (Yanovitzky *et al.* 2006). In addition, normative misperceptions tend to increase with social distance due to a phenomenon called pluralistic ignorance (e.g. Prentice & Miller 1996). These misperceptions (i.e. compliant individuals wrongly thinking that others poach, or believe poaching is socially acceptable) are particularly dangerous, because they could sway previously compliant fishers to change their behaviour and begin poaching to fit the misperceived norm if left uncorrected (Berkowitz 2005).

Another relevant disciplinary focus in social psychology and law highlights how compliance (i.e. behavioural legitimacy) is shaped by people's perceived obligation to follow the rules. This sense of obligation to obey is called value-based legitimacy, and reflects people's recognition of a governing body's right to rule (Tyler 2006a). This legitimacy is itself derived from the structure and processes employed by the authorities to shape people's behaviours (Levi *et al.* 2009). As such, legitimacy is often shaped by other perceptions of management, including those of justice, trust, and identification with management or enforcement personnel (Tyler 2010). If these preconditions are not met, resource users may consider management illegitimate, untrustworthy, or unjust, and decide to poach in acts of defiance, rebellion, or protest (Filteau 2012; Kahler & Gore 2012; Von Essen *et al.* 2014). Indeed, examinations of previously apprehended wildlife poachers have confirmed that poachers are subject to these motivations to poach, and numerous other reasons, including opportunism, livelihood benefits, or even in acts of 'gamesmanship' where they attempt to outwit enforcement (Muth & Bowe 1998; Eliason 2004). In addition, fishers with higher consumptive orientation (i.e. those who place a higher value on catch-related aspects of fishing; Sutton 2003) or specialization may also be more inclined to poach than less specialized fishers or fishers whose satisfaction is not dependent on catching and retaining fish (e.g. Magee *et al.* 2018). In total, people's decisions to poach are complex, likely context-dependent, and are unlikely to be fully understood using a single disciplinary approach. A critical research gap is that, to date, most investigations of why people poach suffer from 'disciplinary silo-thinking' that fails to provide a holistic picture of the many drivers or conditions that might lead someone to poach (Von Essen *et al.* 2014). Here, I address this critical research gap using a detailed, multi-disciplinary investigation to explore the social components and perceptions of poaching, and identify potential mechanisms or phenomena that may encourage or maintain poaching behaviours in fishers on the Great Barrier Reef.

This study examines recreational fishers' perceptions of poaching in the Great Barrier Reef Marine Park, Australia. Specifically, I ask: 1) What are recreational fishers' perceptions of poaching?; and 2) How can these perceptions be used to understand the culture of compliance? I answer these two critical research questions using 682 recreational fisher surveys to assess fishers' i) beliefs, ii) attitudes, iii) normative influences, iv) consumptive orientation, and v) the perceived behavioural controls that may influence fisher's decisions to poach. Additionally, I examine vi) key sociodemographic information about fishers, such as their age, gender, household income, and vii) why others might be motivated to poach. In the following chapter, I use these seven components to predict poaching behaviour based on the three indicators of poaching from Chapter 2.

3.3 Methods

Survey design

As in explanations of the survey aims (Chapter 2), I used the same types of forgiving wording throughout all aspects of question design. Further care was taken to reduce bias by placing potentially sensitive questions (e.g. self-administered question about whether they had personally poached from Chapter 2) towards the end of the survey. The TPB (Ajzen 1991) was used as a foundation on which to build an expanded investigation of poaching behaviours. As in TPB investigations, the design of the quantitative survey was guided by preliminary pilot testing of open-ended qualitative research questions to identify all salient perceptions and reasons to poach, including fisher's attitudes, who might care if they poached (normative groups), what would keep them from poaching (perceived behavioural controls), and the potential outcomes of poaching (outcome beliefs). The TPB framework was further expanded to ask about all three types of normative influences (descriptive, injunctive, and personal/moral) as described by extensive investigations (e.g. Cialdini *et al.* 1991; White *et al.* 2009). Tables 3.1 & 3.2 describe the TPB-related indicators collected in this research, as well as the statements used to assess fisher's consumptive orientations.

Additional information that may be useful for understanding compliance was also elicited, such as fisher avidity, the importance of fishing, income, sex, whether they fished alone or with others, if they had previously engaged in management, and whether they had been inspected by enforcement (Table 3.3). Most questions used 7-point Likert scales to quantify people's perceptions [i.e. 1 = strongly disagree, or very unlikely (outcome beliefs), 4 = neutral, 7 = strongly agree or very likely; Tables 3.1 - 3.3]. Fishers were asked to indicate how many days they had line-fished and spearfished (the two most common types of recreational fishing on the GBR) within the last 12 months. To enable comparisons of these two fisher groups, I categorized respondents as either line fishers or spearfishers based on which type of fishing comprised the majority of their time (i.e., >50% of days fished).

Table 3.1. Descriptions of recreational fisher's compliance perceptions, including attitudes towards management, norms, and perceived behavioural controls.

Variable	Description	Data type
Attitudes towards management		
Value-based legitimacy	Respondent's sense of obligation to obey Marine Park zoning regulations	Ordinal (Likert Scale 1-7)
Motive-based trust	Respondent's trust (or not) that Marine Parks personnel will do their job effectively and in the public good	Ordinal (Likert Scale 1-7)
Identification	Whether Marine Parks personnel share the same background, morals, and goals	Ordinal (Likert Scale 1-7)
Procedural justice (fairness)	Whether Marine Parks personnel use fair processes and make fair decision when dealing with fishers	Ordinal (Likert Scale 1-7)
Procedural justice (respect)	Whether Marine Parks personnel are approachable and respectful when dealing with fishers	Ordinal (Likert Scale 1-7)
Distributive justice	Whether the current zoning plan allows everyone a "fair or equal share" of benefits and resources on the Great Barrier Reef	Ordinal (Likert Scale 1-7)
Norms		
Personal norm	Personal or moral acceptability of poaching	Ordinal (Likert Scale 1-7)
Injunctive norms	Perceived social acceptability of poaching for three social groups: 1) friends and family; 2) fishers they know; and 3) fishers they do not know	Ordinal (Likert Scale 1-7)
Motivation to comply (with social pressure)	Whether respondents cared if social groups approved of poaching: 1) family and friends; 2) fishers they know; and 3) fishers they do not know	Ordinal (Likert Scale 1-7)
Descriptive norms	Perceived prevalence of poaching for: 1) fishers they know; and 2) fishers they do not know	Ordinal (Likert Scale 1-7)
Perceived behavioural controls		
Access to zoning regulations	I know where to get information on no-take boundaries	Ordinal (Likert Scale 1-7)
Knowledge of no-take boundaries	I am aware of the no-take boundaries where I fish	Ordinal (Likert Scale 1-7)
Fines	The risk of getting fined would prevent me from fishing in a no-take zone	Ordinal (Likert Scale 1-7)
Confiscation of fishing gear	The risk of getting fishing gear confiscated would prevent me from fishing in a no-take zone	Ordinal (Likert Scale 1-7)
Social shame	The risk of social shame or disapproval would prevent me from fishing in a no-take zone	Ordinal (Likert Scale 1-7)

Table 3.2. Descriptions of recreational fisher's consumptive orientation and outcome beliefs.

Variable	Description	Data type
Consumptive Orientation		
Larger fish	- I fish where there is a chance for a big fish - I am happiest with catch a sport fish - Rather catch a big fish instead of many fish - The bigger the fish I catch the happier I am	Ordinal (Likert Scale 1-7)
More fish	- The more fish I catch the happier I am - It doesn't matter how many fish are caught - A successful trip is when many fish are caught - I am not happy unless I catch many fish	Ordinal (Likert Scale 1-7)
Outcome Beliefs		
More fish	Fishing in a no-fishing zone would result in catching more fish	Ordinal (Likert Scale 1-7)
Larger fish	Fishing in a no-fishing zone would result in catching larger fish	Ordinal (Likert Scale 1-7)
Higher quality or rare fish	Fishing in a no-fishing zone would result in catching a higher quality/rarer fish	Ordinal (Likert Scale 1-7)
Fines	Fishing in a no-fishing zone would result in getting fined	Ordinal (Likert Scale 1-7)
Confiscation of fishing gear	Fishing in a no-fishing zone would result in the confiscation of my boat or fishing equipment	Ordinal (Likert Scale 1-7)
Social shame	Fishing in a no-fishing zone would result in social shame or disapproval due to getting caught	Ordinal (Likert Scale 1-7)
Removal of breeding stock	Fishing in a no-fishing zone would result in the removal of important fish from the breeding stock	Ordinal (Likert Scale 1-7)
Other damage to the environment	Fishing in a no-fishing zone would result in other damage to the environment (e.g. damaging corals due to anchors, losing gear, etc.)	Ordinal (Likert Scale 1-7)

Table 3.3. Descriptions of sociodemographic variables collected.

Variable	Description	Data type
Fisher avidity	Days fished in last year	Continuous
Importance of fishing	Relative importance of fishing compared to other recreational activities in the GBRMP	Ordinal
Education	Highest level of finished education	Categorical
Income	Combined total household income (pre-tax)	Ordinal
Sex	Respondent's sexual identification	Categorical
Fish with others	Whether a respondent fished with others	Binomial
Previously inspected	Whether a respondent had previously been inspected by Marine Parks personnel	Binomial
Engagement	Whether a respondent had previously engaged in the fisheries management processes	Binomial

Perceived motivations to poach

This survey also asked about a total of nine perceived motivations to poach, which were determined by preliminary pilot testing of open-ended qualitative research (i.e., "why do you think people would fish in a no-fishing reserve?"). Respondents were asked to rate their level of agreement or disagreement with each motivation to poach on a 7-point Likert scale (1 = strongly disagree, 4 = neutral, 7 = strongly agree). The motivations to poach in a no-fishing reserve included in this study were: 1) because the fishing is better; 2) don't think they'll get caught; 3) don't care about conservation; 4) disagreement with no-fishing reserves; 5) accidental; 6) fishers have the right to fish where they want; 7) to see if they can get away with it; 8) somebody told them they couldn't; and 9) an act of protest or rebellion. I used a multivariate analysis of variance (MANOVA) in R (Team 2016) to test for differences in perceived motivations to poach between spearfishers and line fishers.

3.4 Results

Descriptive analysis

The average fisher in the GBRMP is male, spends ~ 34 days/year fishing, makes $90-135,000 AUD, and has a tertiary education. Most (73%) fishers said fishing was the most important activity that they undertook in the GBRMP, and most (78%) had previously been inspected by marine parks personnel. The majority (90%) of fishers said they typically fished with family or friends, whereas 10% of fishers typically fished alone. Only 25% of fishers had previously engaged in participatory management processes (i.e. contacted a government representative, made a submission to a government agency, or attended a public meeting about a fisheries-related topic).

Attitudes towards management

Fishers perceived very high levels of legitimacy: 98% agreed that they were obliged to obey zoning regulations (Fig. 3.1, Table 3.4). Most fishers also believed that the processes and decisions made by GBRMPA personnel were procedurally just (Fig. 3.1, Table 3.4). Similarly, most fishers also believed that the GBRMP allowed for distributive justice, but 16% did not think the current zoning plan was fair or just (Fig. 3.1, Table 3.4). Most fishers (85%) also trusted marine parks personnel and believed they did their job effectively and in the public good, but 10% of fishers did report some level of distrust (Table 3.4).

Table 3.4. Recreational fisher's attitudes towards management authorities in the Great Barrier Reef Marine Park. All responses were assessed on a 7-point Likert scale that ranged from strongly disagree (1), moderately disagree (2 & 3), neutral (4), moderately agree (5 & 6) and strongly agree (7). Bolded values show largest percentage of answers for each question.

Attitudes towards management	Strongly Disagree	Moderately Disagree	Neutral	Moderately Agree	Strongly Agree
Value-based legitimacy: I am obliged to obey the Marine Park zoning regulations.	0%	<1%	<1%	13%	**85%**
Motive-based trust: I trust that Marine Parks personnel will do their job effectively and in the public good.	3%	7%	5%	36%	**49%**
Identification: Marine Parks and Fisheries officers share the same background, morals, values and goals that I do.	4%	6%	14%	**40%**	36%
Procedural justice (fairness): Marine Parks personnel use fair processes and make fair decisions when dealing with fishers.	4%	4%	16%	37%	**38%**
Procedural justice (respect): Marine Parks personnel are approachable and respectful when dealing with fishers.	2%	3%	15%	37%	**43%**
Distributive justice: The current zoning plan (fishing zones, no-take zones, etc.) allows everyone a "fair or equal share" of the benefits and resources available on the Great Barrier Reef.	7%	9%	7%	35%	**42%**

Fig. 3.1. Recreational fisher's poaching-related attitudes towards management, norms, and perceived behavioural controls, illustrated as overall median scores. All responses were assessed on a 7-point Likert scale that ranged from strongly disagree (1), moderately disagree (2 & 3), neutral (4), moderately agree (5 & 6) and strongly agree (7).

Norms

A large majority of fishers (97%) said that it was personally unacceptable to fish in a no-take zone (Fig. 3.1, Table 3.5). Most fishers also thought that others would not approve of poaching (injunctive norms), but this decreased as social distance increased (i.e. most thought that friends and family would not approve, but were unsure about fishers they did not know; Fig. 3.1, Table 3.5). Similarly, respondents believed that fishers they knew were less likely to poach than fishers they did not know: 57% of fishers believed that fishers they did not know had poached in the last 12 months (descriptive norms; Table 3.5). Fishers also reported reasonable motivations to comply with all reference groups (friends and family, fishers they knew, and fishers they did not know; Table 3.5). However, a moderate level (16-21%) of fishers reported not caring about whether others would approve of them poaching (Table 3.5).

Perceived behavioural controls

Most fishers agreed that the risk of being fined, having gear confiscated, or social shame would prevent them from poaching (Fig. 3.2, Table 3.6). However, moderate levels of fishers indicated that the risk of being fined, gear confiscation, or social shame would not prevent them from fishing in a no-take zone (Table 3.6). Most notably, 25% of fishers did not agree that social shame or disapproval would prevent them from poaching (Table 3.6). The large majority of fishers also reported high levels of awareness of the no-take boundaries where they fished, and the ability to get information on zoning regulations (Fig. 3.2, Table 3.6).

Table 3.5. Normative influences and recreational fishers' motivations to comply with social groups. All responses were assessed on a 7-point Likert scale that ranged from strongly disagree (1), moderately disagree (2 & 3), neutral (4), moderately agree (5 & 6) and strongly agree (7). Bolded values show largest percentage of answers for each question. Note that

each injunctive norm question is also paired with a motivation to comply question for each social group (3).

Norms	Strongly Disagree	Moderately Disagree	Neutral	Moderately Agree	Strongly Agree
Personal norm: It is acceptable to me if I fish in a no-take zone.	**89%**	8%	1%	1%	<1%
Injunctive norm (1): My **friends** and **family** would **approve** of me fishing in no-take zones.	**77**%	17%	2%	2%	2%
Motivation to comply: I care whether my **friends** and **family** would **approve** of me fishing in no-take zones.	10%	6%	6%	29%	**46%**
Injunctive norm (2): **Fishers I know** would **approve** of me fishing in no-take zones.	**72%**	18%	4%	4%	2%
Motivation to comply: I care whether **fishers I know** would **approve** of me fishing in no-take zones.	10%	8%	7%	29%	**45%**
Injunctive norm (3): **Fishers I do not know** would **approve** of me fishing in no-take zones.	**65%**	23%	5%	6%	<1%
Motivation to comply: I care whether **fishers I do not know** would **approve** of me fishing in no-take zones.	13%	9%	8%	31%	**37%**
Descriptive norm: Fishers I know have fished in a no-take zone in the last 12 months.	**75%**	8%	3%	5%	8%
Descriptive norm: Fishers I do not know have fished in a no-take zone in the last 12 months.	17%	10%	15%	**30%**	27%

Table 3.6. Recreational fisher's perceived behavioural controls of poaching in no-take zones of the Great Barrier Reef Marine Park. All responses were assessed on a 7-point Likert scale that ranged from strongly disagree (1), moderately disagree (2 & 3), neutral (4), moderately agree (5 & 6) and strongly agree (7). Bolded values show largest percentage of answers for each question.

Perceived behavioural controls	Strongly Disagree	Moderately Disagree	Neutral	Moderately Agree	Strongly Agree
Access to information: I know how to get information on no-take boundaries.	<1%	<1%	<1%	8%	**91%**
Awareness of no-take boundaries: I am educated about, or aware of, the no-take zone boundaries where I fish.	<1%	<1%	<1%	10%	**88%**
Fining: The risk of being fined would prevent me from fishing in a no-take zone.	10%	6%	3%	23%	**58%**
Gear confiscation: The risk of getting my boat, fishing equipment, or other property confiscated would prevent me from fishing in a no-take zone.	13%	8%	2%	18%	**59%**
Social shame: The social shame or disgrace of being caught would prevent me from fishing in a no-take zone.	16%	9%	5%	30%	**40%**

Consumptive orientation and outcome beliefs

Most fishers reported consumptive orientations that leaned towards catching larger fish rather than catching many fish (Fig. 3.2, Table 3.7). A large majority of fishers believed that fishing in a no-take zone would result in getting fined (93%), the confiscation of property such as fishing equipment (81%), or social shame and disapproval due to being caught (73%); Fig. 3.2, Table 3.7). Similarly, most fishers (83%) agreed that poaching would remove important fish from the breeding stock and other damage the environment (77%). Overall, fishers also agreed that poaching would likely result in catching more, bigger, or more rare fish, but these beliefs were more variable (Fig. 3.2, Table 3.7).

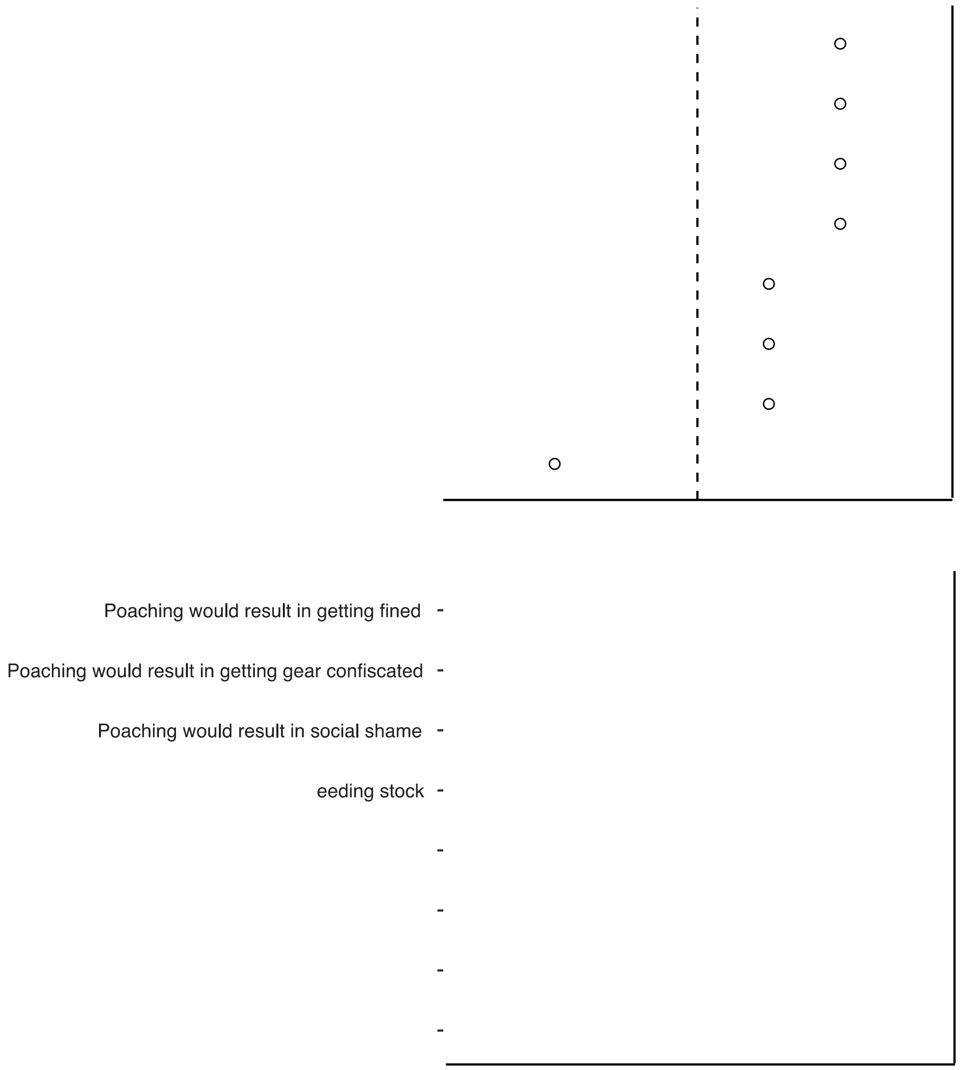

Fig. 3.2. Recreational fisher's consumptive orientation and outcome beliefs about poaching in no-take zones of the Great Barrier Reef Marine Park, illustrated as overall median scores. All responses were assessed on a 7-point Likert scale that ranged from strongly disagree (1), moderately disagree (2 & 3), neutral (4), moderately agree (5 & 6) and strongly agree (7).

Table 3.7. Recreational fisher's consumptive orientation and outcome beliefs about poaching in no-take zones of the Great Barrier Reef Marine Park. All responses were assessed on a 7-point Likert scale that ranged from strongly disagree (1), moderately disagree (2 & 3), neutral (4), moderately agree (5 & 6) and strongly agree (7). Bolded values show largest percentage of answers for each question.

Outcome beliefs	Strongly Disagree	Moderately Disagree	Neutral	Moderately Agree	Strongly Agree
More fish: Fishing in a no-take zone would result in catching more fish.	8%	11%	10%	34%	**37%**
Bigger fish: Fishing in a no-take zone would result in catching bigger fish.	7%	14%	12%	**36%**	31%
Rare fish: Fishing in a no-take zone would result in catching higher quality/rarer fish.	10%	13%	14%	**36%**	27%
Removal of breeding stock: Fishing in a no-take zone would result in the removal of important fish from the breeding stock.	4%	7%	6%	29%	**54%**
Damage to the environment: Fishing in a no-take zone would result in other damage to the environment (e.g. anchor damages, losing gear).	8%	13%	6%	30%	**43%**
Being fined: Fishing in a no-take zone would result in getting fined.	1%	5%	1%	14%	**79%**
Confiscation of property: Fishing in a no-take zone would result in the confiscation of my boat or fishing equipment.	4%	7%	8%	26%	**55%**
Social shame: Fishing in a no-take zone would result in social shame or disapproval due to getting caught.	4%	7%	6%	29%	**54%**

Perceived motivations to poach

In general, comparisons of the perceived motivations to poach did not differ between spearfishers and line fishers (MANOVA: Pillai's = 0.013, *df*=1, 671, *P*=0.431), though spearfishers (n = 27) ranked poaching as acts of defiance or rebellion/protest as lower than line fishers (n = 655). Fishers' primary perceived motivation to poach was the perception of higher catches in reserves, along with a low risk of detection, followed by poachers not caring about conservation, and disagreeing with reserves (Fig. 3.3). Respondents slightly agreed that people might fish in reserves because it was an accident, they believed they had the right to fish where they want or to see if they could get away with poaching, and did not agree that people poached as an act of defiance, or as an act of protest or rebellion against management agencies.

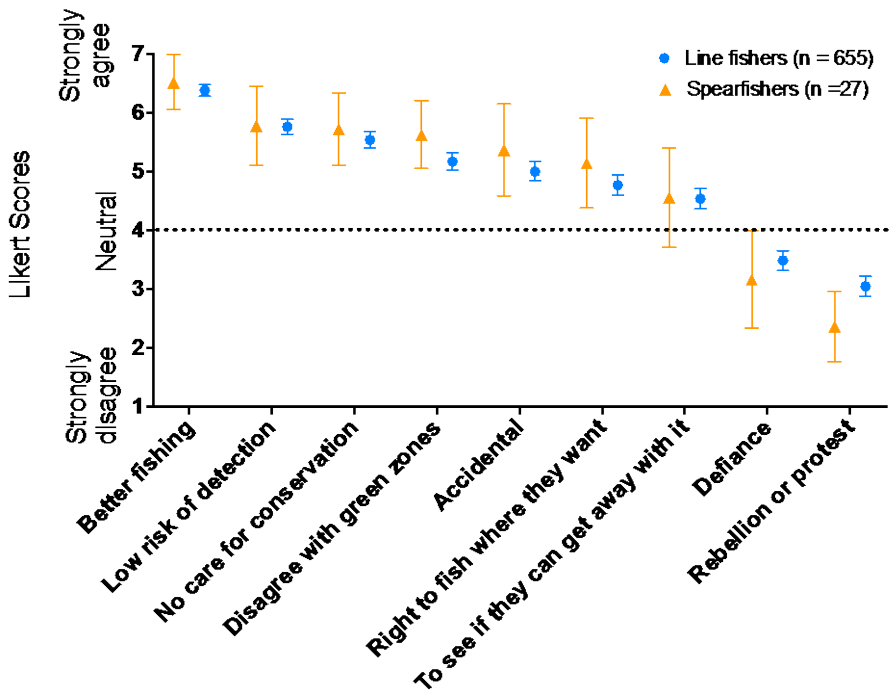

Fig. 3.3. Perceived motivations for fishers to poach, ranked in order from highest to lowest (left to right). Rankings are based on 7-point Likert scales of agreement or disagreement with each perceived motivation to poach. Mean scores (±95% CI) for line fishers are blue, and orange for spearfishers. The dotted horizontal line indicates a neutral score (4).

3.5 Discussion

Most recreational fishers in the GBRMP viewed poaching as both personally and socially unacceptable. Furthermore, most fishers see the GBRMPA as legitimate, just, and trustworthy; all of which are pre-conditions that encourage voluntary compliance (Tyler 2010). Encouragingly, compliance by recreational fishers in the GBRMP is quite high, but a considerable amount of poaching is still occurring (Chapter 2; Arias & Sutton 2013; GBRMPA 2015, 2017a). Although some level of non-compliance is inherent in fisheries management, it is desirable to discourage and reduce non-compliance as much as possible. Further reducing this non-compliance relies on understanding why some fishers continue to poach, and identifying pertinent aspects of compliance management that can be used to encourage compliance. Encouraging compliance necessitates approaches that use traditional, coercive deterrence levers, and complementary normative-based levers, both of which are necessary for high compliance (Arias 2015). In addition to the false consensus effect and social learning introduced in Chapter 2, this data suggests two other (mis)perceptions or mechanisms are likely operative and encouraging poaching behaviour in the recreational fishing population of the GBRMP. One of these is related to social norms (pluralistic ignorance), while the second was a perceived lack of deterrence. I unpack these in depth below, and highlight approaches and strategies that can be used to change these perceptions and further encourage compliance.

Pluralistic ignorance occurs when compliant individuals incorrectly perceive the attitudes and behaviours of others as different from their own (i.e. overestimating the acceptability and prevalence of poaching), when they are in fact similar (Schroeder & Prentice 1998; Berkowitz 2005). Research suggests that this has previously occurred in fisheries compliance contexts (e.g. Hatcher *et al.* 2000; Bova *et al.* 2017), and my data suggest that this phenomenon is also operative in the GBRMP recreational fisher population.

Specifically, respondents thought that the general fishing public, or fishers they did not know, poached more than themselves or fishers that they knew. Similarly, fishers also thought that the social acceptability of poaching increased with social distance; fishers felt most strongly that their friends and family would not approve of poaching, less so about fishers they knew, and least strongly that fishers they did not know would consider poaching as socially unacceptable. If these misperceptions are not corrected, they might sway previously compliant fishers to adjust their own behaviour to fit the misperceived norm of poaching (e.g. Prentice & Miller 1996; Hatcher *et al.* 2000; Bova *et al.* 2017).

Preventing normative-based backslides, or potential increases in poaching, will rely in part on providing tailored normative information to correct fisher's misperceptions (e.g. Steg & Vlek 2009). Social norms approaches or marketing that provide accurate normative feedback have seen considerable success in reducing a variety of problem behaviours (e.g. Fabiano *et al.* 2003; Haines & Barker 2003; Hancock & Henry 2003), and have been suggested for fisheries compliance (Arias & Sutton 2013; Thomas *et al.* 2016; Bova *et al.* 2017). As noted earlier, fishers in the GBRMP misperceived the prevalence of poaching by others. Although often underappreciated, this descriptive social norm is demonstrably powerful in shaping compliance behaviours (Cialdini 2007). A pertinent strategy to reduce poaching could therefore emphasize that nearly all fishers do comply, and highlight that most fishers think poaching is both morally and socially unacceptable. Communicating high compliance levels should correct the misperceptions of problematic poachers who use it to rationalize their own behaviours via the false consensus effect, and further reduce the probability that compliant fishermen act as 'carriers of the misperception' who inadvertently contribute to a culture that allows poaching to continue (Perkins 1997; Berkowitz 2005). In addition, highlighting the fact that most fishers find poaching to be socially and morally unacceptable would lever the considerable power of shame and collective moral judgements to encourage fisher's compliance (Cialdini & Goldstein 2004; Gezelius 2004; Cepić & Nunan 2017). Multiple avenues exist to deliver and diffuse this normative feedback among recreational fishers. These include communication strategies such as press releases, articles in fishing periodicals, and empowering block leaders, role models, or key players in social networks (Sandström *et al.* 2014; Thomas *et al.* 2016; Mbaru & Barnes 2017). In the context of recreational fishing in the GBRMP, recognized leaders or role models such as television fishing show hosts or fishing club members may be well placed to shape and influence normative perceptions in the recreational fishing community. In a recent social outreach intervention, practitioners were able to substantially reduce poaching by targeting outreach efforts at local leaders, adults, and school children (Steinmetz *et al.* 2014). Although focused on terrestrial poaching in a developing country, approaches from Steinmetz *et al.* (2014) could potentially be applied to address poaching by recreational fishers in the GBRMP. These include: offering opportunities for action, increasing fisher's perceptions of their ability to report or reduce poaching, and further increasing the social pressure to comply. Most notably, social outreach that targets children may be effective in delivering long-term compliance, because the values gained during these younger, impressionable years often last into adulthood (Robertson *et al.* 2012).

Yet, it is important to note that a substantial proportion of fishers (16-22%) indicated that they did not care whether others would approve or disapprove of them poaching. Similarly, most fishers thought they would face social shame or disapproval if they were caught poaching, but again, nearly a quarter of all fishers said this would not prevent them from poaching. This result supports previous findings that social disapproval may be a tertiary, rather than a primary driver of compliance in the GBRMP (Arias & Sutton 2013). Thus, maintaining high levels of compliance will rely on both social norms approaches and complementary coercive deterrence measures. For example, most recreational fishers believed that poaching in the GBRMP could result in getting a fine (93%) or having gear confiscated (81%). Conversely, a substantial portion of fishers indicated that the risk of being fined or having gear confiscated would not prevent them from poaching (16% and 21%, respectively). Ideally, this suggests that these fishers aren't deterred from poaching based on the risk of punishment, but instead choose to comply due to personal values and morals (e.g. Kuperan & Sutinen 1998; Sutinen & Kuperan 1999; Gezelius 2004). However, a more prudent interpretation is that fishers either don't see the threat of fining or confiscation as severe enough to deter them from poaching, or alternatively, do not think they will be caught in the act (as would be suggested by Grasmick & Bryjak 1980; Grasmick & Green 1980). Although I did not explicitly ask fishers whether they perceived a high likelihood of detection, the latter explanation is most supported by my findings. Specifically, fishers believed that poaching in the GBRMP is motivated by opportunity—fishers poach because: 1) they know they are likely to attain higher catches; and 2) they understand there is a low likelihood of getting caught. These findings have multiple implications for compliance management in the GBRMP, which I discuss below.

The perception of better catches suggests that fishers accept the rationale for no-fishing reserves (i.e., they produce more, and larger fish). Therefore, anti-poaching communication strategies based solely on the information deficit model (i.e., that fishers poach because they lack information that reserves produce more, and larger fish) are likely to be ineffective, or even counter-productive in reducing poaching. Providing information about the potential for reserves to support fisheries outside of reserve boundaries may be an alternative strategy, but this relies on poachers using long-term planning horizons, or deciding not to poach because they care about the long-term efficacy of the GBRMP. Conversely, my findings suggest that fishers poach opportunistically, based on short-term planning horizons. Curtailing poaching is therefore likely to require changing fisher's perceptions of the likelihood of being caught.

One way to increase fisher's perceptions of risk is to increase the *actual* probability of being detected while poaching (e.g. Hatcher *et al.* 2000), whether through increases in patrol effort, patrol efficiency (e.g. risk-based patrol planning), or voluntary enforcement actions by other fishers (i.e. confronting or reporting poachers). It is worth noting that the GBRMP Field Compliance Unit actively pursues all of these strategies (GBRMPA 2016, 2017d). However, these options are often untenable elsewhere, because many of the world's MPAs often suffer from a lack of critical management and enforcement capacity (Gill *et al.* 2017). It is therefore improbable that most MPAs could secure the necessary components to increase the probability of detecting poaching. Most notably, these components include increased funding and enforcement capacity, detailed information on non-compliant activities, or an understanding of how to responsibly encourage voluntary reporting/enforcement actions that may put fishers at risk of costly confrontations, retribution, or revenge by poachers (see Chapter 5). The need for an alternative approach is further emphasized by the fact that while undoubtedly important, instrumental and coercive deterrence cannot solely guarantee high compliance—there will always be individuals (chronic offenders) who choose to break the law in a premeditated fashion, regardless of the potential penalty (Kuperan & Sutinen 1998; Sutinen & Kuperan 1999).

Another way to deter poaching is to increase the *perceived* probability of detection (Hatcher *et al.* 2000)through fear appeals, or fear-arousing communications. Fear-arousing communications have previously resulted in significant reductions of people's problem behaviours (Sutton & Hallett 1988; Witte & Allen 2000), and may be particularly useful because recreational fishers in the GBRMP do perceive a low probability of detection as a primary motivation to poach. Furthermore, fishers do not know the actual probability of detection, so their behaviour is instead influenced by the perceived threat of enforcement (Grasmick & Bryjak 1980; Grasmick & Green 1980). Increasing the perceived probability of detection could therefore be accomplished by emphasizing the capabilities of enforcement agencies, whether by publicizing technological advancements used to catch poachers (e.g. night vision, radar, drones), or instances of successful apprehension and punitive actions (e.g. fines, gear confiscation, etc.). This approach is also likely to be cost-effective, because messages can be tailored and delivered to target audiences with low-cost press releases and social media platforms including fishing forums and blog posts. In response to increasing numbers of reported poaching offenses by recreational fishers and "apparent growing complacency and negligence towards zoning compliance" (GBRMPA 2015, 2017a), the GBRMPA has employed a number of steps (including those described above) to address the ongoing issue of poaching by recreational fishers (e.g. GBRMPA 2017c, a, d, b). Therefore, continued research on this topic would be well placed to document the effects of these efforts on fisher's perceptions, and any resultant changes in compliance behaviours.

An important caveat that must be considered is that I asked fishers about the perceived motivations for others to poach, rather than themselves. The information elicited in this manner may be different from that derived from directly asking poachers about why they themselves poach. However, asking fishers directly why they poach is likely to result in untruthful responses, or denial of poaching in the first place. In addition, previous research (e.g. Fisher 1993) has demonstrated that asking people about others' motivations allows respondents to project their own beliefs and perceptions onto others through a mask of impersonality, thereby encouraging more truthful representations of behaviours, beliefs, and motivations.

When considered in combination with Chapter 2, this investigation illustrates that numerous (mis)perceptions regarding poaching exist and may be responsible for the overestimation and continuation of poaching by recreational fishers in the GBRMP. Encouragingly, these perceptions should be pliable to targeted behavioural interventions such as social outreach, fear arousing communication, and social norms marketing and influence approaches. Depending on the approach, further reducing non-compliance and poaching can likely be accomplished by emphasizing three specific messages: 1) nearly every recreational fisher thinks that poaching is socially and morally unacceptable; 2) almost all recreational fishers follow the rules, and those who do not are in the minority; and 3) the likelihood of being detected while poaching is high, as are the consequences. If successfully delivered, these normative and instrumental-based messages should correct misperceptions of poaching held by both poachers and compliant fishers, and further encourage a culture of compliance.

CHAPTER 4: Investigating behavioural drivers and potential proxy indicators of poaching by recreational fishers

4.1 Synopsis

Here, I interview 682 recreational fishers in the Great Barrier Reef Marine Park, Australia to investigate relationships between potential behavioural drivers (Chapter 3) and proxy indicators of poaching (Chapter 2). Specifically, I collated and condensed the 29 potential drivers of poaching identified in Chapter 3, and examined their ability to predict three proxy indicators of poaching. The Random Response Technique (RRT) model was not supported by the potential behavioural drivers of poaching, which was likely because the level of intentionally introduced statistical noise (to obscure a respondent's answer and ensure confidentiality) was higher than the actual level of estimated poaching (7%). The other proxy indicators may provide useful information for community-level investigations of poaching, but I found limited evidence that they can be used to unequivocally identify individuals who poach. This multi-method approach provided robust estimates of likely poaching rates of recreational fishers, but highlights the difficulties of empirically studying and identifying what drives an individual's poaching behaviour when poaching rates are relatively low.

4.2 Introduction

Efforts to curtail poaching in MPAs often fail due to incomplete understanding of what influences or drives fishers' poaching behaviours. Accordingly, the effectiveness of efforts to reduce poaching therefore hinges on the ability to ascertain what differentiates poachers from compliant individuals, and reducing the behavioural drivers of poaching. Reducing or eliminating the behavioural drivers of poaching can be accomplished in many ways, whether through ‘structural fixes’ such as changes in management policies, laws, or institutional design principles (Heberlein 2012; Arias 2015), or through ‘cognitive fixes’ that seek to change human behaviour with tools like communication, outreach, and social influence strategies (Ham *et al.* 2009; Heberlein 2012; Abrahamse & Steg 2013). Yet, these fixes or interventions strategies are primarily dependent on the disciplinary approach and scale at which the initial investigations of behavioural drivers occur.

Depending on the disciplinary lens used to examine compliance, behavioural drivers of poaching can comprise a diverse suite of attributes or conditions. At the macro-scale, numerous institutional design conditions are demonstrably important drivers of compliance levels. MPA size has previously been linked to differences in compliance levels, but the direction of this relationship is inconsistent, and seems to be heavily influenced by context. For instance, compliance levels were higher in small MPAs in Costa Rica (Arias *et al.* 2015), but higher in large conservation areas in Canada (Haggarty *et al.* 2016). Previous research has also illustrated how fisher compliance is higher when graduated sanctions are present (i.e. penalties that increase with the severity or frequency of infringements; Cinner *et al.* 2012). A further condition that is demonstrably important for compliance is the inclusion of resource users in resource management, whether through formal consultation, participatory decision-making processes, rule making, or monitoring and enforcement (Hatcher *et al.* 2000; Pollnac *et al.* 2010; Arias *et al.* 2015; Epstein 2017).

At the micro-level, research has identified a considerable number of traits or individual-level characteristics that drive fisher compliance behaviours. Fishers’ perceptions of management such as legitimacy, effectiveness, and fairness are especially important determinants of compliance (Kuperan & Sutinen 1998; Nielsen & Mathiesen 2003; Arias *et al.* 2015), and numerous studies have illustrated how poaching can occur as forms of protest, rebellion, or defiance against management that is considered illegitimate or unjust (Bell *et al.*

2007; Kahler & Gore 2012; Von Essen *et al.* 2014). Fishers' perceptions of risk, deterrence and opportunity are also key drivers of compliance—poaching generally tends to increase when fishers perceive more opportunity and lower probabilities of being detected or prosecuted (Kuperan & Sutinen 1998; Sutinen & Kuperan 1999; Hatcher *et al.* 2000). Normative influences often play a vital role in shaping environmental compliance behaviours (Cialdini *et al.* 1991; Cialdini 2007; White *et al.* 2009), and are increasingly recognized as important determinants of fishers' compliance behaviours (Hatcher *et al.* 2000; Gezelius 2004; Thomas *et al.* 2016; Bova *et al.* 2017). In addition, a lack of knowledge or awareness of MPA boundaries may also drive 'accidental' non-compliance, as was demonstrated in Canada (Lancaster *et al.* 2015), but fishers in the GBRMP reported very high levels of awareness of information source and boundaries of no-fishing reserves (Chapter 3).

Although the above studies have examined how different conditions are related to compliance, or perceptions of compliance, most investigations of poaching drivers suffer from several key shortcomings. First, is that many of these investigations are compartmentalized, or grounded in a single disciplinary realm or approach (e.g. institutional approach), and are therefore likely to overlook key drivers identified in other research disciplines (e.g. normative approach). A second shortcoming concerns the response variable used to measure or indicate compliance. For instance, many studies focus solely on fishers' perceived levels of compliance (e.g. the magnitude or frequency that other people poach) as the response variable (e.g. Cinner *et al.* 2012; Arias & Sutton 2013; Lancaster *et al.* 2015). As noted in the previous chapter, this assumes that people's perceptions are reliable indications of actual compliance levels, which has a degree of support (Chapter 2). However, this approach does not attempt to identify actual poachers, and instead identifies the drivers related to general levels of compliance. In addition, previous attempts at identifying the correlates to poaching using the RRT as a response variable have failed to account for the introduced variance in the model (Arias & Sutton 2013). Furthermore, many of the other studies that examine drivers of individual fishers' poaching behaviours use direct questioning to ask respondents whether they have poached (e.g. Thomas *et al.* 2016; Bova *et al.* 2017). This approach makes a substantial assumption that respondents will admit truthfully to an illegal activity, which may be the case if poaching is pervasive and not seen as socially undesirable. However, I demonstrated that this type of self-administered questioning was still subject to considerable underreporting in the GBRMP (Chapter 2), to the point where it was not feasible to try to model drivers for these self-admitted poachers (n=21).

This chapter ties together the two previous data chapters by investigating how a broad suite or potential drivers or predictors of poaching behaviour (identified in Chapter 3) are related to the three key methods for estimating and indicating poaching (RRT, perceived levels of poaching, and personally knowing a poacher; Table 4.1) that were identified in Chapter 2. I pay particular attention to the degree to which potential drivers are consistent across the different poaching estimate indicators. This investigation makes a unique contribution because few studies have yet assessed the feasibility of using these proxy indicators to profile fishers who poach (but see St. John *et al.* 2012), or compared these approaches to determine whether these theoretically grounded proxies align or triangulate.

Table 4.1. Indicators or measures of poaching developed in Chapter 2.

Poaching indicator	Description
Randomised Response Technique (RRT)	Respondents who truthfully admitted to poaching via the RRT
Perceived levels of poaching	Respondents' perception of the level of poaching in fisher population (0-100%)
Personally knowing a poacher	Whether a respondent admitted to personally knowing a poacher

4.3 Methods

Predictor variables

Whenever possible, I reduced the number of predictor variables to be used in final models by producing theoretically grounded latent traits. However, seven indicators were used directly from Chapter 3 because they were not applicable for reduction into latent traits. These included fisher avidity, importance of fishing, whether respondents fished with others, engagement, previous experience of being inspected, descriptive social norm, and the personal norm (Table 4.2). I then reduced 17 other indicators into 10 latent traits. Indicators were selected and grouped for latent trait production based on theoretical discourse and disciplinary approaches. Based on the theory of planned behaviour, I made three latent traits to capture fisher's beliefs: the first trait captured fisher's beliefs about reserves (i.e. whether they worked), potential punitive consequences of poaching, and whether they believed poaching would have environmental consequences. I also constructed one latent variable to capture the injunctive social pressures (i.e. what others think about poaching and whether each fisher cared what others thought). I constructed one latent trait to capture fisher's attitudes towards the management process, and another latent trait that encapsulated deterrence based behavioural controls. Aside from the theory of planned behaviour, I constructed two latent traits to capture fisher's perceived motivations for others to poach. The first latent trait was designed to capture opportunistic drivers of poaching, whereas the second was designed to capture rebellion-based drivers for poaching. Lastly, I constructed two latent traits to capture fisher's consumptive orientations towards both big fish and more fish (Table 4.2).

Latent variable construction

I constructed latent variables using exploratory factor analysis and data extraction via principal component analysis. Data were considered suitable for latent variable construction when all of the following conditions were met: 1) correlation coefficients scores were above 0.3; 2) KMO measure of sampling adequacy scores were above 0.5; and 3) when the Bartlett's test of sphericity was significant ($p<0.05$). I also used Kaiser's criteria (Eigenvalue > 1 rule; Kaiser 1960) and the Scree test (Cattell 1966) to guide factor extraction. Factor loading scores for each latent variable are reported in Table 4.3.

Table 4.2. Description of predictor variables used in models.

Variable	**Description**	**Type**
Fisher avidity	Days fished per year	Continuous
Importance of fishing	Relative importance of fishing as a recreational activity in the Great Barrier Reef Marine Park	Ordinal
Fish with others	Whether a respondent fished alone or with others	Dichotomous (yes or no)
Previous experience of being inspected	Whether a respondent had previously been inspected by Marie Parks officers	Dichotomous (yes or no)
Engagement	Whether a respondent had ever contacted a government representative, made a submission to a government agency, or attended a public meeting on a fisheries related issue.	Dichotomous (yes or no)
Reserve beliefs	Respondents' beliefs about whether poaching would result in catching larger, more, or rarer fish	Latent variable
Consequence beliefs	Respondents' beliefs about whether poaching would result in getting gear confiscated, fined, or social shame	Latent variable
Beliefs about environmental consequences of poaching	Respondents' beliefs about whether poaching would result in environmental damage to the environment (e.g. anchor damage, removal of breeding stock, etc.)	Latent variable

Process	Respondents' attitudes about management processes, including trust, identification, procedural justice, distributive justice, and legitimacy	Latent variable
Rebellion-based perceived motivations to poach	Respondents' perceived motivations for others to poach based on disobedience (disagreement with no-take zones, protest, defiance, right to fish wherever they want, "catch me if you can" attitudes)	Latent variable
Opportunistic-based perceived motivations to poach	Respondents' perceived motivations for others to poach based on opportunity (better catches, do not care about conservation, low risk of being caught)	Latent variable
Consumptive orientation – big fish	Degree to which respondents value catching big fish while fishing	Latent variable
Consumptive orientation – more fish	Degree to which respondents value catching more fish while fishing	Latent variable
Deterrence based perceived behavioural control	Degree to which respondents would be deterred by the threat of fines, gear confiscation, or social shame	Latent variable
Injunctive social norm	Respondents' perceptions about the social acceptability of poaching, and the degree to which they care about others' approval or disapproval regarding poaching (high scores indicate belief that poaching is socially acceptable, and do care what others think about their decision to poach)	Latent variable
Descriptive social norm (general fishing public)	The degree to which a respondent believes that others have poached in the last 12 months (high scores indicate belief that others have poached)	Ordinal
Personal (injunctive) norm	The degree to which a respondent believes it is acceptable to poach (high scores indicate poaching is personally acceptable)	Ordinal

Table 4.3. Factor loading scores for latent variable production.

Item	Factor loading scores
Reserve beliefs	
Reserves produce more fish	.890
Reserves produce larger fish	.864
Reserves produce rarer fish	.801
Consequence beliefs	
Poaching would result in gear confiscation	.855
Poaching would result in a fine	.822
Poaching would result in social shame	.560
Beliefs about environmental consequences of poaching	
Poaching would result in environmental damage	.829
Poaching would remove important breeding stock	.821
Poaching would result in environmental damage (negative flip of forgivingly worded statement)	.749
Process	
Motive-based trust	.795
Procedural Justice (fairness)	.768
Procedural Justice (respect)	.716
Identification with authorities	.699
Distributive justice	.616
Value-based legitimacy	.420
Perceived motivations to poach (rebellion)	
Act of defiance	.707
Disagreement with no-fishing reserves	.659
Act of protest	.638
Right to fish where they want	.623
For fun, or to see if they can get away with it	.618
Perceived motivations to poach (opportunistic)	
Low perception of being caught	.776
Do not care about conservation	.748
Perception of better catches in no-fishing reserves	.641
Consumptive orientation for big fish	
Rather catch 1 or 2 big fish than 10 smaller fish	.753
Bigger the fish = better the trip	.733
Happiest catching challenging sport fish	.671
Fish where there is a chance for big fish	.534
Consumptive orientation for more fish	
Happiness is dependent on catching many fish	.835
Successful fishing trip is catching many fish	.819
Not happy with fishing unless catching a lot	.608
Perceived behavioural control (deterrence)	
Risk of fine is a deterrent to poaching	.919
Gear confiscation is a deterrent to poaching	.917
Social shame is a deterrent to poaching	.839
Injunctive social norm	
Injunctive social norm – known fishers	.855
Injunctive social norm – friends and family	.801
Injunctive social norm – general fishing public	.718

Data analysis

I used three different models to examine how the 17 potential poaching predictors (Table 4.2) are related to the RRT, perceived rates of non-compliance, and knowing a poacher (Table 4.4).

Table 4.4. Description of response variables used in models.

Response variable	Description	Type	Sample size
Model 1	Whether a fisher chose to knowingly fish in a no-take zone in the last 12 months, as provided by the Randomised Response Technique (RRT)	Dichotomous (yes or no)	504
Model 2	Fishers' perceived level of poaching in the last 12 months	Continuous (0-100%)	681
Model 3	Whether or not a fisher reported knowing someone who had poached in the last 12 months	Dichotomous (yes or no)	681

For Model 1, I worked with a statistician (Dr. Aaron MacNeil) to develop a Bayesian approach to identify relationships between the 17 predictor variables and admitting to poaching via the RRT. Recall from Chapter 2 that the RRT uses a randomising device (e.g. dice) to conceal respondent's answers and provide confidentiality, thereby encouraging truthful answers. I used a randomising device design adapted from Arias and Sutton (2013) that ensured only the respondent knew whether they were answering truthfully [85%], or being instructed (i.e. forced) to automatically answer 'yes' [5%], or 'no' [10%] (see Chapter 2). Knowing these fixed probabilities allowed me to estimate the amount of truthful yes answers post-hoc. While useful for reducing potential bias due to the sensitivity of poaching, this design adds uncertainty (reduces information) in the resulting data that can be difficult to analyse. Because there were so few recorded poaching events in the RRT data (11%, including forced answers), I adopted the recommended Bayesian approach for rare events, using a logistic Bernoulli model with ~Cauchy (0, 2.5) priors for the covariates and a

~Cauchy (0,10) intercept (Gelman *et al.* 2008). I used a Bernoulli distribution in this case because the dependent variable (admitted poaching via RRT) was effectively binomial, but I had to account for the intentionally introduced variance of forced responses (i.e. predict the probability of particular outcomes rather than the outcomes themselves).

In Model 2, I used a general linear model (package *MASS*) to examine the relationship between potential drivers of (non)compliance and perceived levels of poaching. For Model 3, I used a rare events logistic regression model (relogit; package *zelig*) to examine relationships between our theoretically grounded covariates, or drivers of (non)compliance, and personally knowing a poacher (Table 4.1, Table 4.4). In this model, the response variable was binary (either knowing a poacher or not), but rare, thus a modified logistic regression was appropriate. The relogit uses a standard logistic regression that corrects for the bias encountered when a sample size is small, or the actual event (knowing a poacher) is rare, and was therefore appropriate here (13% of fishers reported knowing a poacher).

All predictor variables were standardized, and when applicable, log (x+1) transformed (i.e. fisher avidity, perceived non-compliance) before analysis. All models were checked for overdispersion; the data were not overdispersed relative to binomial (knowing a poacher) or Gaussian distributions (perceived non-compliance). Models were checked for multicollinearity using variance inflation factors as in Pan & Jackson (2008). Conventional statistical models and graphical outputs were performed in R (version 3.3.2; Team 2016), while the Bayesian analysis was performed using the Bayesian Python package PyMC3 (Salvatier *et al.* 2016). Model outputs were plotted using the packages *SJplot* and *bayesplot* packages.

Model selection

I verified model fit through comparison with null models (the two mixed models contained the model structure but no explanatory variables). My full models performed better than the respective null models for models 2 and 3, but not for model 1 (Table 4.5). Each explanatory variable was selected for inclusion because it was theoretically grounded and could therefore sensibly be considered as having an effect, even if small or uncertain. For this reason, I chose not to remove 'non-significant' explanatory variables from the model because doing so would be the equivalent of fixing parameter estimates at exactly zero, which is a highly-subjective modelling decision after explanatory variables have been identified as potentially important (Gelman & Hill 2007).

Table 4.5. Model fits and selection criteria, including number of replicates (n), number of parameters (K), Aikake Information Criterion (AIC), and Deviation Information Criterion (DIC).

Model	**n**	**K**	**AIC**	**DIC**
Model 1 Null	504	0	-	351
Model 1 Full	504	17	-	451
Model 2 Null	671	0	977	-
Model 2 Full	671	17	923	-
Model 3 Null	670	0	512	-
Model 3 Full	653	17	466	-

4.4 Results

There was no strong evidence for the predictive model based on the RRT because of the considerable variation in the predictor variables or parameter coefficients (Fig. 4.1a). Quite simply, the imprecision (wide error bars) resulting from the introduced error meant that even covariates with a reasonably strong effect (e.g. process and injunctive social norms) still overlapped zero. However, I did find a number of variables that were able to differentiate and predict compliance versus poaching for the other proxy indicators: three for knowing a poacher and eight for high estimates of non-compliance. Importantly, two of the three covariates (fisher avidity and the descriptive social norm, or the belief that other fishers had poached in the last year; Fig. 4.1b, c) that were able to differentiate between poaching and compliance for the proxy of knowing a poacher were also related to high estimates of non-compliance. High estimates of perceived poaching were also positively related to the importance of fishing, environmental beliefs, injunctive social norms, personal norms, and deterrence as a perceived behavioural control, whereas reserve beliefs were negatively related (Fig. 4.1b). In addition to positive relationships with fisher avidity and the descriptive social norm, knowing a poacher was also positively related to fishing with others (Fig. 4.1c).

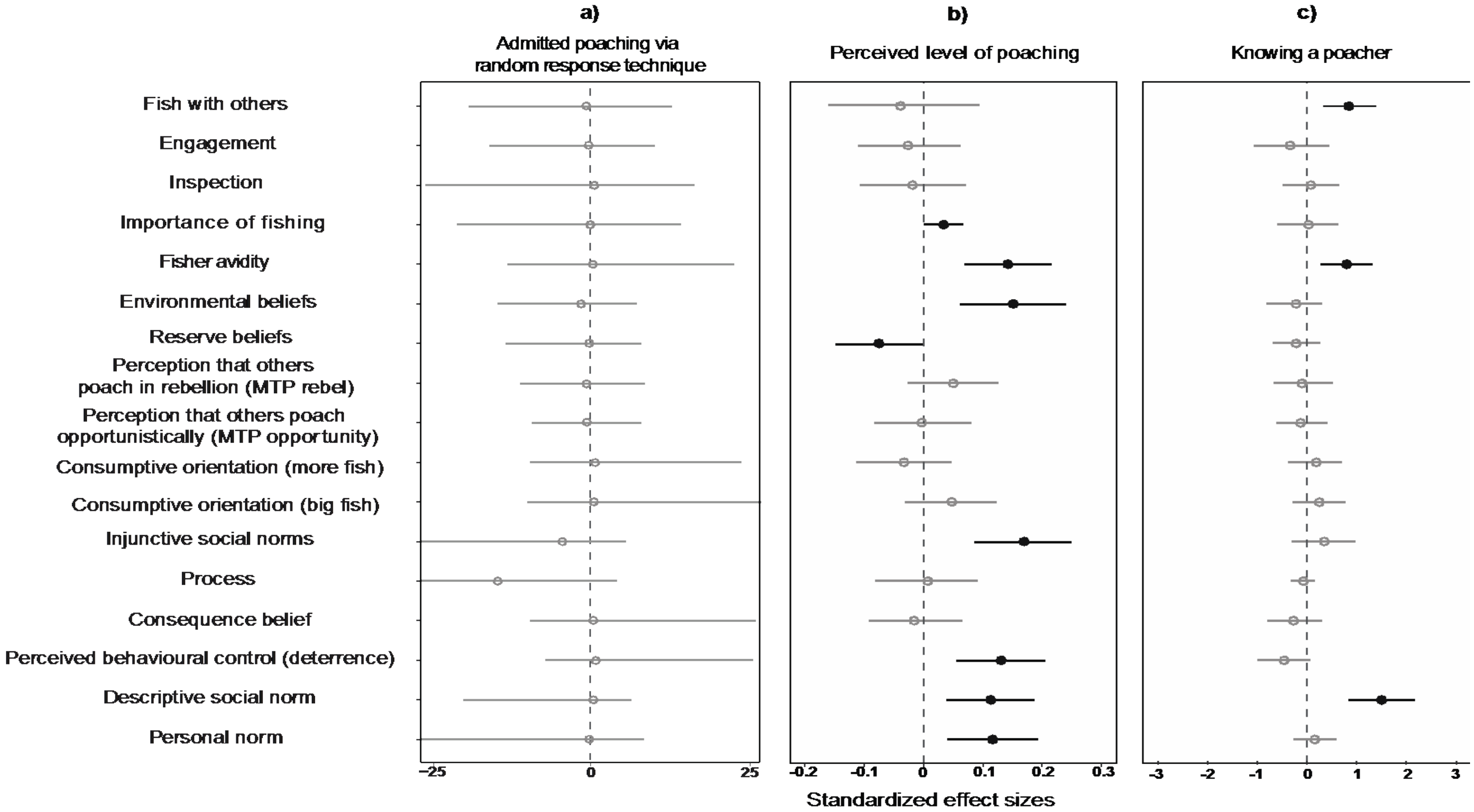

Fig. 4.1. Comparison of predictive, and potential proxy indicators models of poaching. Scores to the right of the dotted line indicate (potential) poaching, whereas scores to the left of the dotted lines correspond with compliance. Parameter estimates are standardized effect sizes and 95% confidence intervals for the 13 behavioural drivers or covariates (*y* axis). Responses included: admitted poaching via the randomised response technique (a); perceived levels of poaching (b); and personally knowing a poacher (c). Precise estimates are indicated with black, filled circles and confidence intervals, while imprecise estimates are indicated with grey, open circles and confidence intervals. Effect sizes were standardized within each response, relative to the mean divided by two times their standard deviation.

4.5 Discussion

Predicting and identifying behavioural drivers of poaching is inherently difficult. Many studies have made such attempts, but often in areas where illegal behaviours are deemed as more socially acceptable and are fairly prevalent (e.g. St. John *et al.* 2012). This is not the case in the Great Barrier Reef Marine Park, where poaching is relatively uncommon (Chapter 2). However, I found that two of the three potential indicators of poaching behaviours could be predicted by theoretically derived behavioural drivers. These two models (perceived level of poaching and personally knowing a poacher) had correspondence on two covariates: descriptive social norms and avidity.

The descriptive social norm, or belief that other fishers had poached in the last year, was significantly related to both of our theoretically grounded poaching proxies. Although once described as ‘underappreciated sources of social control’ (Cialdini 2007), recent examinations of recreational fisher compliance have emphasized the critical role that descriptive social norms have in shaping fisher’s compliance decisions. In New Zealand, descriptive social norms proved to be primary pre-determinants of recreational fisher’s compliance with both size and daily limits (Thomas *et al.* 2016). In addition, both my research (Chapter 2) and studies in other regions of the world find similar results, and demonstrate how poachers had substantially higher estimates of perceived non-compliance (or descriptive social norms) than compliant fishers (Bova *et al.* 2017). The importance of descriptive social norms has also been well documented for a wide variety of other social behaviours (e.g. Festinger 1954; Milgram *et al.* 1969; Berkowitz 2005; Schultz *et al.* 2007). Overall, the positive relationship of the descriptive social norm with both proxy measures appears to support the applicability of both proxy indicators of poaching. For instance, social structure social learning theory (Akers 2011) would predict that fishers learn from observing the behaviours of others, and this result suggests that fishers either know, or believe that others poach. Some extent of social learning in regards to poaching is therefore likely to exist, considering that 57% of fishers believed that other fishers had poached in the last year (Chapter 3), and 27% of these fishers had previously observed other fishers poaching (Chapter 5). Overall, the positive relationship between perceived non-compliance and the descriptive social norm is relatively intuitive because they are two different ways of measuring fisher's perceptions of other people's behaviours. Yet, this observed consistency serves to support suggestions that the false consensus effect may be operational among poachers, and lends some credence to the potential use of high levels of perceived non-compliance as a proxy indicator of poaching. This is further supported by other studies of recreational fisher compliance suggesting that the false consensus effect is operational amongst poachers (Chapter 2; Arias & Sutton 2013; Bova *et al.* 2017). Yet, I also found numerous counter intuitive relationships between perceived non-compliance and my predictor variables, which indicates that fishers who overestimate the prevalence of poaching should not unequivocally be considered poachers. I explain these counter intuitive relationships later in this discussion.

Avid fishers were more likely to know poachers and perceive higher levels of non-compliance. When viewed in light of the false consensus effect and social structure social learning and/or subculture theory, this result may suggest that avid fishers are more likely to themselves poach. The rationale for this relationship comes from the closely related concept of specialization. Studies on fisher specialization consistently demonstrate that highly specialized (or avid) fishers place higher importance on non-catch related motivations to fish (Magee *et al.* 2018). Yet, specialized fishers have also been described as being more supportive of fisheries management measures such as catch limits compared to MPAs, even when MPAs do not affect their ability to pursue non-catch related aspects of fishing (Salz & Loomis 2005; Voyer *et al.* 2014; Martin *et al.* 2016). Recent research has further explored the interactions between fisher specialization, avidity, and motivations to fish, and suggests that specific categories of fishers such as ‘hunter-gatherers’ are more motivated by catch-related aspects, and may therefore have a greater tendency towards non-compliance (Magee *et al.* 2018). However, an alternative (and perhaps more plausible) explanation would be that people who fish more often are more likely to know a larger number of fishers overall (including poachers), and are also more likely to see or hear of poaching because they spend more time on the water. The latter explanation would align with research that suggests avid recreational fishers are more interested in engaging in management processes (Li *et al.* 2010), and are more likely to have conservation-oriented normative values (social and moral) that influence their fishing decisions (Buchanan 1985; Ditton *et al.* 1992; Gigliotti & Peyton 1993). The significant relationship between avidity and both of my potential proxy indicators of poaching emphasizes that further research in this realm is warranted.

The only other predictor variable that was able to differentiate between compliance and poaching for the proxy based on knowing a poacher was whether or not an individual fished with others, which is a relatively straightforward result—people who fish with others are more likely to have a larger social network, and therefore have a higher probability of knowing a poacher. However, perceived non-compliance was related to numerous other potential drivers. Some of these relationships are intuitive and seem to support its application as a poaching proxy, but other relationships are counter-intuitive and therefore obfuscate the applicability of this proxy’s ability to profile potential poachers. For instance, the positive relationships of personal and injunctive norms with perceived non-compliance lend credibility to this proxy. Both of these norms are demonstrably important in shaping many different types of social behaviours (Berkowitz 2005; Schultz *et al.* 2007; White *et al.* 2009), including recreational fisher compliance (Chapter 3; Thomas *et al.* 2016; Bova *et al.* 2017). It is therefore conceivable that fishers who believe poaching is personally and socially acceptable would be more likely to poach.

However, perceived non-compliance was related to several variables in ways that are counter-intuitive for a proxy indicator of individual poaching behaviour. For instance, perceived non-compliance was negatively related to reserve beliefs (that is the beliefs that poaching in a reserve would result in catching larger, more, or rare fish compared to areas open to fishing). My findings in Chapter 3 suggest that poaching by recreational fishers may be opportunistic in nature, with a primary motivation being the perception of better catches in no-take zones, so one would expect to see a positive, rather than a negative relationship between these variables. I also discovered two other counter-intuitive relationships: both deterrence-based behavioural controls and beliefs that poaching would have negative consequences for the environment were positively related to perceived levels of poaching. Theoretically, it would be expected that individuals who poach would be more likely to refute, rather than acknowledge the negative impacts of poaching on the environment. Similarly, fishers who view deterrence (in the form of fines, gear confiscation, or social shame) as behavioural controls would be expected to be more compliant, rather than more prone to poach. It must be noted that the approach used here to investigate relationships between theoretically derived drivers of poaching and key proxies of poaching is correlative rather than causal. Therefore, a mechanistic understanding of these relationships is simply not possible, but warrants further investigation.

Of the three models developed to predict compliance, there was no strong evidence for the RRT model given the data; the coefficients or potential drivers had very large amounts of variance and were unable to differentiate between poaching and compliance. This is likely because the amount of intentionally introduced statistical noise (15%; to obscure respondent's answer from interviewers and thereby protect confidentiality) was higher than the actual estimated level of poaching. This highlights a considerable drawback of the RRT, especially because I employed a very powerful design (85% chance of being directed to answer truthfully; see Chapter 2, Fig. 2.3). In addition, an extensive meta-analysis (Lensvelt-Mulders *et al.* 2005) compared estimates provided by RRT studies to known population values, and illustrated how wrong (or potentially untruthful answers) regularly occurred in these studies and ranged anywhere between 11-54%. When considered in conjunction with the fact that both of the other proxy indicators yielded higher estimates of non-compliance (Chapter 2), it is not implausible that untruthful answers by fishers obstructed this analysis and obscured potentially significant relationships.

In all, this investigation demonstrates the considerable difficulties faced when attempting to examine the behavioural drivers of illegal and socially unacceptable activity. Firstly, the RRT returned a very low estimated level of poaching by recreational fishers in the GBRMP (~7%; Chapter 2). This low level made it very problematic to gain sufficient sample sizes of poachers to run individual-level quantitative models to examine the conditions that could predict whether or not someone was likely to poach, because the introduced level of error (i.e. 15% forced responses) was higher than the admitted level of poaching. Secondly, a large majority of recreational fishers also believed that poaching was both personally and social unacceptable (Chapter 3). Both of these outcomes are positive reflections of the competent compliance management of the GBRMPA, which also enjoys very high levels of perceived legitimacy (Chapter 3). However, the interaction of these two conditions makes a very challenging and problematic setting in which to investigate and quantify individual level behavioural drivers of poaching. In this context, nearly all of the fishers surveyed here were compliant or they answered untruthfully due to the socially unacceptable nature of poaching.

Given these circumstances, an alternative approach to understanding poaching behaviour would have been to utilize a separate survey to elicit information from previously apprehended offenders. In essence, this would have guaranteed that the people identified as poachers were actual poachers (or at least had been caught poaching) and provided a larger sample of known poachers to interview about their motivations and characteristics. I made repeated collaborative efforts with GBRMPA's field compliance management unit to survey fishers who had been issued citation notices, but was unable to gain permission due to privacy protection laws. However, even this approach would have had drawbacks, since it would have only surveyed known poachers, and could not have compared them to non-poachers. An important caveat that must be acknowledged in this study is that I did ask respondents about what would prevent them from poaching (i.e. fines, gear confiscation, shame), but I did not elicit respondent's perceptions of the likelihood of being caught while poaching. It is possible that respondents then acknowledge the potential consequences of being caught, while simultaneously perceiving a very low risk of actually being caught in the act. In this case, the perceived behavioural controls that I measured may have failed to fully encompass all of the different aspects that deter or encourage poaching by fishers.

Another caveat that should be considered here is that I tested the effects of individual level attributes (i.e. a respondent's beliefs, attitudes, etc.) on two proxies that may not completely capture an individual's choice to poach. However, both of these measures have been previously suggested as useful proxies at both the community level (e.g. determining the level of poaching in a population of fishers), and as potential indicators of whether an individual engages in illegal activity (e.g. Petróczi *et al.* 2008; Arias & Sutton 2013; Cross *et al.* 2013). Thus, the main aim of this chapter was to push and develop the limits of knowledge and theory to further contribute to the discussion of whether these measures can be deployed as proxy indicators of an individual's behaviour.

Overall, I illustrate how proxy indicators may be useful to examine poaching at the system, or community level, but are less effective for identifying individuals who personally engage in poaching. While previous research indicates that the false consensus effect is likely operative amongst poachers, my results show limited evidence that overestimations of non-compliance can be used to unequivocally identify an individual as non-compliant. Knowing a poacher may also increase the probability that a fisher themselves decides to poach, but this is at best only an indicator that an individual is at higher risk of poaching, rather than an absolute identifier. Lastly, the RRT is useful in providing estimates of the overall level of poaching, but the ability of this technique to identify behavioural drivers of poaching is directly related to the level of non-compliance. If the level of introduced error is higher than the level of non-compliance, which occurred here, it may be very difficult to use the RRT to go beyond estimating levels of poaching.

CHAPTER 5: Addressing poaching in marine protected areas through voluntary surveillance and enforcement

5.1 Synopsis

Poaching renders many of the world's marine protected areas (MPAs) ineffective. Because enforcement capacity is often limited, managers are attempting to bolster compliance by engaging the latent surveillance potential of fishers. Yet, little is known about how fishers respond when they witness poaching. Here, I collated 2111 surveys of fishers living adjacent to 55 MPAs in seven countries and found that 48% had previously observed poaching. However, the most common response was inaction, with the primary reasons being: 1) conflict avoidance; 2) a sense that it was not their responsibility or jurisdiction; and 3) the perception that poaching was a survival strategy. I also quantified how different characteristics or conditions such as institutional design elements related to fishers' responses to poaching, and highlight avenues to engage fishers while mitigating risks. These include emphasising how poaching personally affects each fisher, promoting stewardship and personal responsibility norms, and poverty alleviation to reduce the need for fishers to poach for survival.

5.2 Introduction

Marine protected areas (MPAs) are increasingly implemented to improve or maintain ecological conditions in marine ecosystems. The efficacy of these areas hinges largely on user compliance with MPA regulations, but ensuring this is a persistent problem. A myriad of non-compliant activities (e.g. pollution, illegal development, etc.) threaten MPA effectiveness worldwide. Of these, poaching (i.e. fishing in no-fishing zones of MPAs) is particularly prevalent, and regularly renders many of the world's MPAs ineffective (Kelleher *et al.* 1995; Mora *et al.* 2006; Rife *et al.* 2013). This often occurs because of limitations in enforcement (e.g. McClanahan 1999; Lundquist & Granek 2005; Byers & Noonburg 2007), which typically arise from a lack of resources and capacity (e.g. Gill *et al.* 2017). Given that resources for enforcement are likely to remain limited in the future, it is critical to understand how compliance can be improved with minimal additional resources.

Natural resource management (NRM) agencies are increasingly attempting to bolster compliance by engaging the latent surveillance and enforcement capacity of hunters and fishers (e.g. GBRMPA 2016; Green 2016; Kohn 2016). Here, I use the terms 'surveillance' and 'enforcement' specifically in reference to the voluntary actions that fishers take after observing poaching, or fishing in no-fishing zones of MPAs. Engaging fishers could have serious repercussions, most notably increased conflict and potential retaliation by poachers. This necessitates mitigating conflict risks while understanding individual's decisions to contribute to NRM, often in changing governance contexts. Fortunately, this is in part an exercise in community engagement in NRM—a topic that has received considerable attention (e.g. Cox *et al.* 2010; Gutierrez *et al.* 2011; Alexander *et al.* 2015).

Numerous conditions or institutional design elements can facilitate community engagement in NRM. These include, but are not limited to, rule agreement (DeCaro *et al.* 2015), identifying change agents (Gutierrez *et al.* 2011), capacity building (McConney & Pena 2012), participation in decision-making (Gurney *et al.* 2016; Epstein 2017), devolution of powers (Pomeroy & Berkes 1997), MPA size, design, or age (McClanahan *et al.* 2009), graduated sanctions (Cinner *et al.* 2012), perceived legitimacy (Levi *et al.* 2009), and the perceived effectiveness of management (McClanahan 1999; Gurney *et al.* 2014; Davis *et al.* 2015). While not explicit examinations of the relationships between these conditions and voluntary surveillance by stakeholders, these studies do provide a solid foundation on which to base further research.

This study examines voluntary surveillance and self-enforcement by 2111 fishers in 55 MPAs spanning seven countries (Fig. 1). Specifically, I ask: 1) how many fishers have witnessed poaching?; 2) what do fishers do when they observe poaching?; 3) why don't fishers confront or report poachers?; and 4) how do certain conditions or institutional design elements relate to fishers' actions after observing poaching? In contrast to the majority of research examining surveillance and enforcement in NRM, fishers interviewed here were not formally performing surveillance or enforcement patrols. Instead, this study examines voluntary surveillance and/or engagement, which occur largely outside the realm of formal enforcement, with the exception of customary marine tenure contexts, which I discuss further below.

5.3 Methods

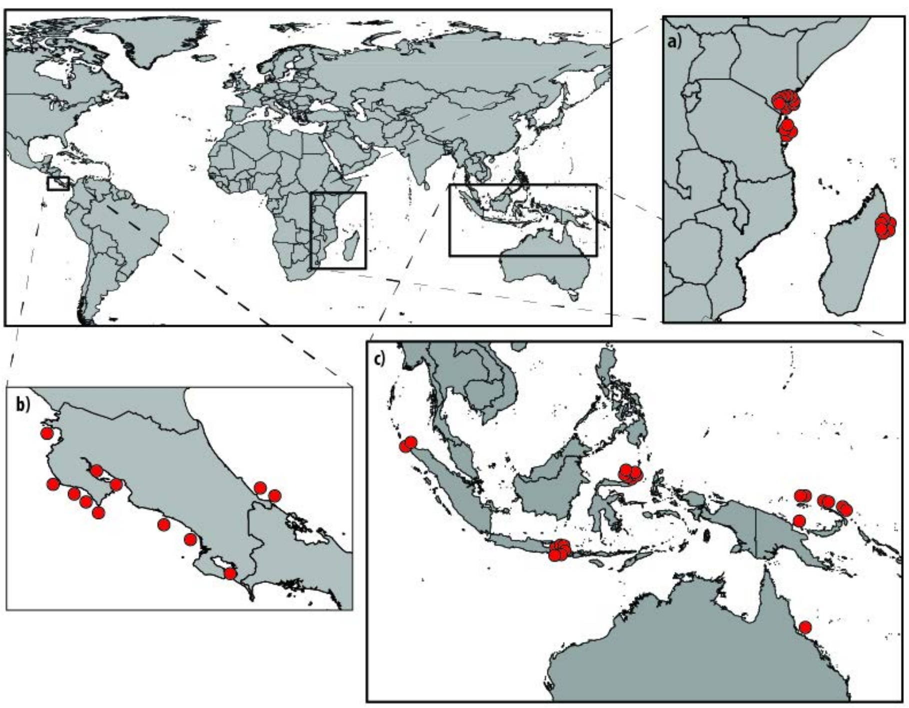

Fig. 5.1. Map of 55 study sites across the world in a) Kenya, Tanzania, and Madagascar; Western Indian Ocean (WIO), b) Costa Rica, and c) Indonesia, Papua New Guinea, and Great Barrier Reef Marine Park; Australia. Red dots indicate approximate locations of study sites (MPAs).

Sampling

I surveyed a total of 2111 fishers from communities adjacent to, or nearby 55 MPAs in 7 countries (Fig. 5.1). MPAs investigated in this study encompassed a range of spatial closures, from small traditional closures such as "tambu" areas in Papua New Guinea, to large, no-take closures in areas such as Australia's Great Barrier Reef Marine Park. Overall, a purposive sampling design was used to allow the targeting of active fishers. Respondents were therefore approached at boat ramps or landing sites (Australia, Costa Rica, Kenya), systematically surveyed from households (e.g. every third household in the village; Indonesia, Madagascar, Papua New Guinea), or identified from lists of fishers provided by local leaders (Costa Rica, Tanzania). Respondents' responses were coded, such that they could cite more than a single action and/or reason for the action, upon observing poaching (e.g. confront and report), so actions were not always mutually exclusive. Reasons for responses to observed poaching were recorded whenever offered. With the exception of reasons for inaction, these were too rare to enable further quantitative analysis. Thus, I offer these other salient reasons as qualitative evidence or quotes in the discussion to further support my findings and suggestions on how to responsibly engage fishers in surveillance and enforcement.

Conditions, institutional design elements and data integration

Data were integrated from five different datasets, including this thesis, Cinner *et al.* (2012), Gurney *et al.* (2016), Arias *et al.* (2015) and one unpublished dataset. These data were collected for different purposes, but all originate from the same research lab and consequently are highly comparable. In all, I asked about four conditions or institutional design elements that are likely to affect voluntary enforcement: rule agreement, graduated sanctions, participation in decision-making, and MPA size (Tables 5.1, 5.2). Participation in decision-making was ascertained by asking if resource users had previously held formal leadership roles, attended committee meetings, or contacted/made submissions to governance or management representatives. Rule agreement was determined by asking resource users if they agreed with the protected area regulation. Graduated sanctions are defined here as sanctions, or punishments that increased along with the occurrence, or severity of offences, as in Cinner *et al.* (2012).

Table 5.1. Descriptions of variables explored using binomial regression models.

Explanatory variables	**Description**	**Type**
Rule agreement	Whether respondents agreed with protected area regulation	Binary
Graduated sanctions	Presence or absence of sanctions	Binary
Participation in decision-making	Whether respondents had previously participated in decision-making processes	Binary
Marine protected area size	Small (0-100 km^2), or large (<100km^2)	Binary
Random effects		
Country	Costa Rica, Kenya, Tanzania, Madagascar, Indonesia, Papua New Guinea, Australia	Multinomial
Site	55 sites	Multinomial

Table 5.2. Response and explanatory variables used for binomial regression models. In some instances, a question about institutional conditions was not covered in a particular data set, so sample sizes differ among analyses.

Response variables	**Sample size**
Join poachers	98
Do nothing (inaction)	431
Report poachers	284
Confront poachers	259
Explanatory variables	
Rule agreement	488
Graduated sanctions	979
Participation in decision-making	994
MPA size	1020

Analysis

To assess how conditions and institutional design elements were related to fishers' responses to observed poaching, I used generalized linear mixed models with a binomial distribution. I modelled each type of behaviour separately because the response variables were not always mutually exclusive—for example, fishers could both report and confront poachers, so I ran a separate regression model for each action. I set country and site *a priori* as random factors to account for non-independence of data arising from repeated sampling within each site within each country and checked models for multicollinearity using variance inflation factors, as in Pan & Jackson (2008). All models were checked for overdispersion; the data were not overdispersed relative to a binomial distribution for all four models. I compared full models to a null model, which contained the model structure (i.e. random effects) but no explanatory variables. In all cases the full model performed better than its respective null model (Table 5.3). I did not remove 'non-significant 'explanatory variables from the model because each explanatory variable was carefully considered for inclusion and could therefore reasonably be considered as having an effect, even if small or uncertain; removing factors from the model is equivalent to fixing parameter estimates at exactly zero; a highly-subjective modelling decision after explanatory variables have already been selected as potentially important (Gelman & Hill 2007). All analyses were undertaken in R (version 3.3.2), using the packages *lme4*, *arm,* and *car*.

Table 5.3. Model fit scores for binomial models.

Model	**AIC**
Join Null	162
Join Full	153
Do Nothing Null	491
Do Nothing Full	490
Confront Null	519
Confront Full	508
Report Null	539
Report Full	526

5.4 Results & Discussion

Interestingly, nearly half of all the fishers interviewed here (48%; 1020 of 2111) had previously observed poaching, yet the most common response was to do nothing after observing non-compliance (Fig. 5.2). Besides inaction, fishers described three other responses to observed poaching: reporting, confronting, or joining poachers, at frequencies that varied by region (Fig. 5.2). Inaction was particularly prevalent in the Great Barrier Reef Marine Park (GBRMP; Australia) and Costa Rica, and least common in Papua New Guinea and Indonesia, where fishers regularly confronted or reported poaching (Fig. 5.2). Overall, fishers reported poachers slightly more than they confronted them (28% and 25% relative frequencies, respectively; Fig. 5.2). Joining poachers was relatively rare (10% relative frequency), except for in the Western Indian Ocean, where almost 20% of fishers who sighted poaching stated that they had joined in the act (Fig. 5.2). The considerable number of fishers who remained inactive after observing poaching represent substantial human capital that might be engaged to fill critical gaps in enforcement capacity existent in many of the world's MPAs (e.g. Gill *et al.* 2017). Indeed, numerous NRM agencies are continuing to develop programs that aim to harness the latent surveillance and enforcement capacity of resource users (e.g. GBRMPA 2016; Green 2016; Kohn 2016). Yet, the success of these efforts will rely on understanding why some individuals remain inactive, why others choose to voluntarily engage, and whether different conditions or institutional design elements can be utilized to encourage engagement in this context. Most importantly, these engagement efforts must also mitigate the risks inherent in engaging resource users in surveillance and enforcement capacities, which I discuss further below.

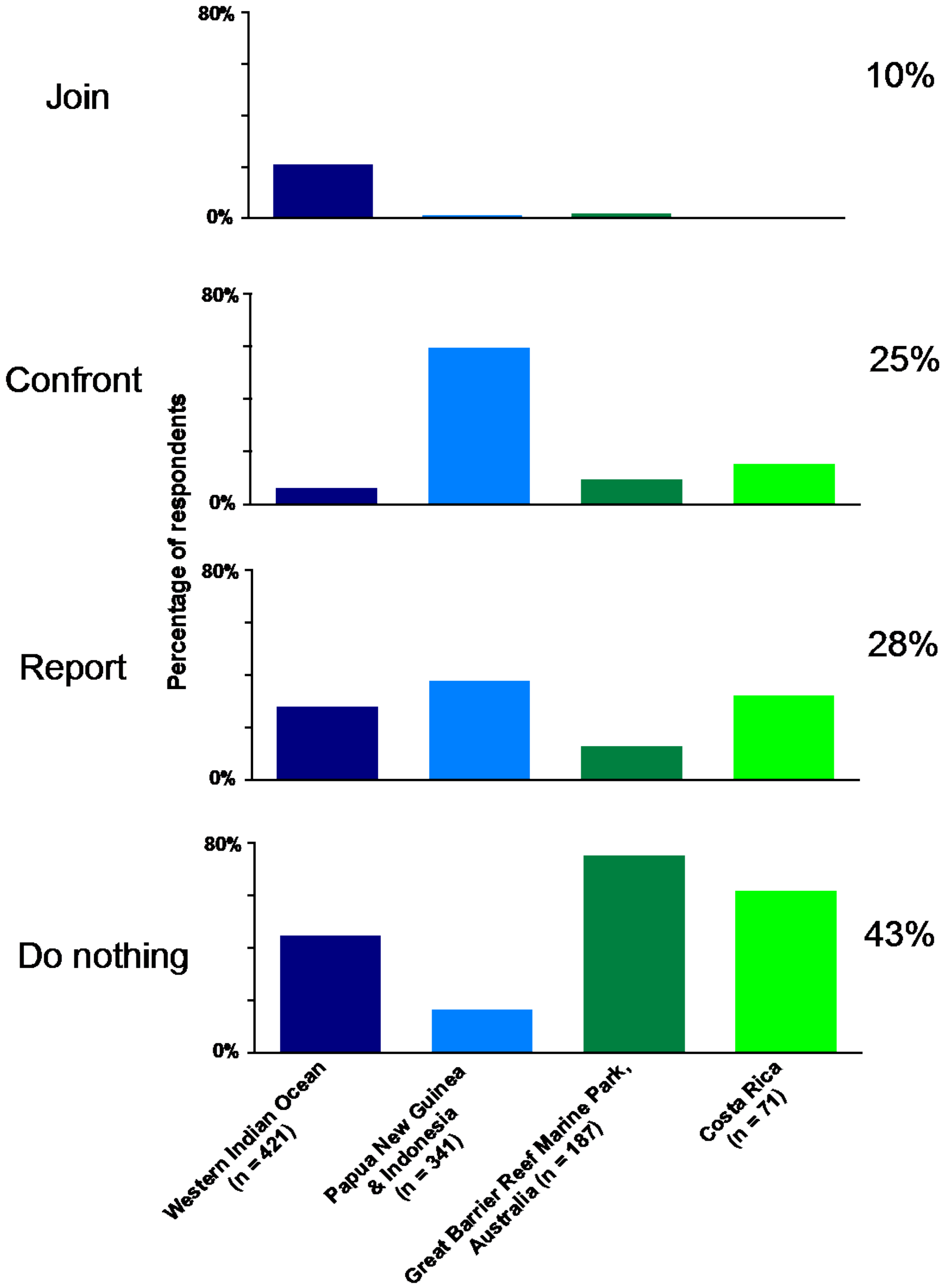

Fig. 5.2. Fishers' responses to observed poaching, displayed as relative frequencies for each world region (% – left y axis), and the relative frequency of each action for the entire dataset (% on right). Actions were not always mutually exclusive (e.g. a fisher could both confront and report poachers), so frequencies may exceed 100%. Countries included in Western Indian Ocean were Kenya, Tanzania, and Madagascar.

Reasons for inaction

Fishers gave numerous reasons for inaction, but the three most cited were: 1) to avoid conflict; 2) because it was not their jurisdiction or responsibility (i.e. poaching occurred in an area they did not have rights over, or it wasn't their concern); and 3) the perception that poaching was a survival or livelihood strategy. Overall, the predominant reason for inaction was conflict avoidance (with the exception of the GBRMP; Fig. 5.3), which suggests that many fishers view voluntary surveillance and enforcement as a costly behaviour. Studies of cooperation in resource management settings have demonstrated the critical role of costly norm enforcement in stabilizing large-scale cooperation (e.g. Henrich *et al.* 2006; Rustagi *et al.* 2010), but "costly" in these investigations often refers to the personal costs of involvement in formal monitoring or enforcement patrols, such as time and money. Yet, my findings suggest that fishers also see voluntary enforcement as costly in terms of the conflict with, and potential retaliation by poachers. While I discuss this further below, this emphasizes the necessity of including complementary risk mitigation strategies with any efforts to engage fishers in voluntary surveillance and enforcement. Lastly, the marked absence of conflict avoidance as a reason for inaction in the GBRMP may also reflect pronounced cultural differences between a relatively individualistic society (Australia) that emphasizes a "dominant" conflict communication style, and collectivistic societies (the six other countries studied here) that tend to avoid direct conflict as a way of saving face (Hofstede 1980; Ting-Toomey 1991; Oyserman *et al.* 2002).

The second most frequently cited reason for inaction was a sense that it was not a fisher's jurisdiction or responsibility (Fig. 5.3). Interestingly, this response was particularly prevalent in Papua New Guinea, Indonesia, and the GBRMP, which have contrasting systems of ownership rights over marine resources. The study sites in Papua New Guinea and Indonesia exercise forms of customary marine tenure, which allow local fishers to legally exclude fishers from other areas from accessing fishing grounds. This type of territorial ownership of marine resource is widely promoted as a key foundation for sustainable governance (e.g. Afflerbach *et al.* 2014), yet my results suggest a considerable drawback of this type of local management—fishers typically have enforcement rights only in their clan or community's areas and have disincentives to be involved in enforcement activities elsewhere. In Papua New Guinea, respondents regularly remarked that reporting infringements in marine areas owned by other clans would be perceived as interfering in the other clan's business and was strongly frowned upon. In the GBRMP, this reasoning for inaction may reflect a high degree of individualism and apathy toward reporting, or fisher dependence on formal enforcement practices and mechanisms, which has been demonstrated in numerous social dilemma experiments (e.g. Mulder *et al.* 2006; Chen *et al.* 2009; DeCaro *et al.* 2015). Finally, fishers in developing countries often chose not to report or confront poachers because they believed poaching was a survival or livelihood strategy (Fig. 5.3). While addressing this particular issue is beyond the scope of this study, this result does highlight the potential for poverty reduction strategies to reduce poaching in MPAs located in developing countries (Arias *et al.* 2015).

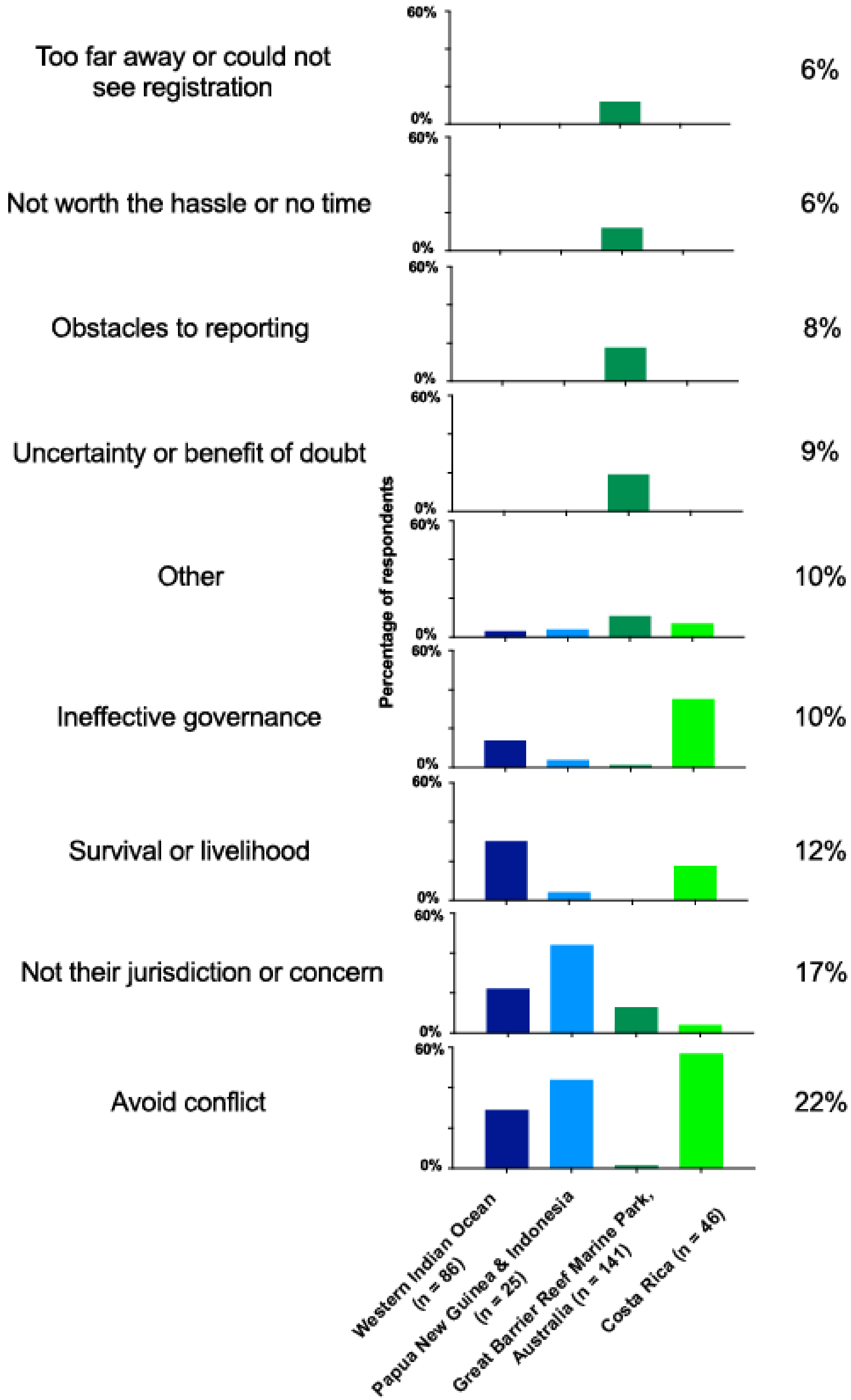

Fig. 5.3. Fishers' reasons for inaction, displayed as relative frequencies for each world region (% – left y axis), and the relative frequency of each action for the entire dataset (% on right). Countries included in Western Indian Ocean were Kenya, Tanzania, and Madagascar.

Reasons for action

Engaging inactive fishers will rely in part on identifying and levering the salient reasons cited by fishers who took voluntary action after observing poaching. Although the cultures in six of the seven countries in this study can be considered “collectivistic” (Oyserman *et al.* 2002), cultures are notably dynamic and cannot solely explain individual-level behaviour (Hofstede 1980; Oyserman *et al.* 2002). This is further highlighted by my results, because many of the reasons for positive action (i.e. reporting or confronting poachers) offered in these collectivistic societies actually had individualistic underpinnings (e.g. “illegal fishing affects me”, or “I’m personally invested in the MPA”). Thus, emphasizing how poaching personally affects each fisher may be useful for engaging fishers that typically avoid conflict, or don’t view enforcement as their job or responsibility.

A second approach to activating voluntary surveillance and enforcement by fishers is levering beliefs about conservation, stewardship, and moral norms. These were offered as reasons for action in most countries, which suggests that these constructs are already operational and present in some capacity. Cultivating these norms and beliefs may rely on identifying and empowering ‘key players’, ‘change agents’, and ‘block leaders’, or resource users well placed to work within their social network to promote concepts of stewardship and moral responsibility towards protected areas (e.g. Abrahamse & Steg 2013; Alexander *et al.* 2015; Mbaru & Barnes 2017). If effectively fostered, these internalized stewardship beliefs and feelings of morality responsibility should lead to strong commitments by fishers to voluntarily enforce, which are necessary for long-lasting behavioural change (Matthies *et al.* 2006).

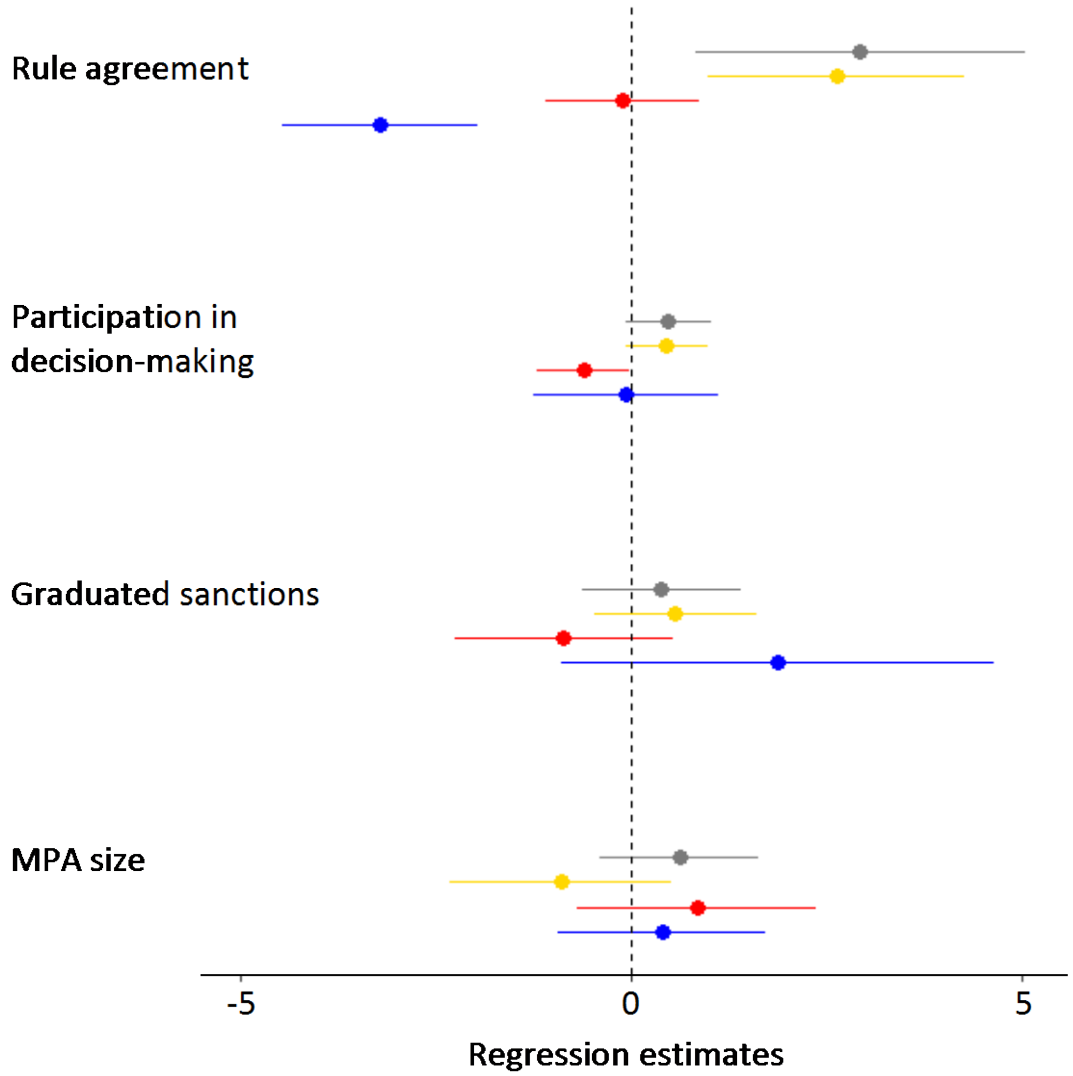

Fig. 5.4. Relationships between institutional design elements and fishers' voluntary enforcement actions, displayed as regression estimates with 95% confidence intervals. Four possible actions include reporting (grey), confronting (yellow), doing nothing (red), or joining (blue) poachers. Note that the reference level for marine protected area (MPA) size is 'small'.

Effect of conditions and institutions on surveillance and self-enforcement

Overall, rule agreement and participation in decision-making were positively related to reporting and confronting poachers (Fig. 5.4, Table 5.4). In addition, rule agreement was negatively related to joining poachers, while participation in decision-making was negatively related to inaction (Fig. 5.4, Table 5.4). However, reporting and confronting poachers were only significantly related to rule agreement and not to participation in decision-making, which suggests that the outcome (rule agreement) was more important than the procedure (participation). This is surprising, because extensive research on the topic of justice has demonstrated that procedural justice (i.e. whether people were allowed to participate) is typically more influential than distributive justice (i.e. perceived fairness of the rule, and resultant agreement with outcome) (e.g. Lind & Tyler 1988; Tyler 2000; Cohen-Charash & Spector 2001). Yet, research on the role of justice in the workplace suggests that distributive justice concerns can outweigh procedural justice if an organization or workplace is not in harmony, or if productivity and efficiency are the focus (Barrett-Howard & Tyler 1986; Lind & Tyler 1988). When viewed through this lens, my findings suggest that the competing self-interests of fishers could actually lead toward a culture of disharmony, and further emphasize the importance of equitable management practices (e.g. Cinner *et al.* 2014; Gurney *et al.* 2015; Gill *et al.* 2017), especially for encouraging voluntary surveillance and enforcement behaviour. Lastly, the significant, negative relationship between participation in decision-making and inaction also emphasizes that perceptions of procedural justice are still likely to play a role in stakeholder engagement, even if I did not find a statistically significant relationship with reporting or confronting poachers in this study (Fig. 5.4, Table 5.4).

Table 5.4. Model coefficients, standard errors, confidence intervals, z values, and p values for the relationships of institutional design parameters with fishers' actions after observing non-compliance, or poaching in no fishing zones of marine protected areas. P values in bold are significant at the 0.05 level.

Response variable	Explanatory variable	Estimate	Std. Error	Lower 95% CI	Upper 95% CI	z value	Pr(>|z|)
Report	Graduated sanctions	0.3875	0.5150	-0.6218	1.3969	0.7525	0.4517
	Participation in decision-making	0.4791	0.2755	-0.0608	1.0190	1.7393	0.0820
	MPA size	0.6191	0.5155	-0.3912	1.6294	-1.2010	0.2297
	Rule agreement	2.9341	1.0778	0.8217	5.0465	2.7225	**0.0065**
Confront	Graduated sanctions	0.5622	0.5294	-0.4755	1.5999	1.0619	0.2883
	Participation in decision-making	0.4566	0.2694	-0.0714	0.9847	1.6951	0.0901
	MPA size	-0.8994	0.7185	-2.3076	0.5088	1.2519	0.2106
	Rule agreement	2.6246	0.8404	0.9773	4.2718	3.1228	**0.0018**
Inaction	Graduated sanctions	-0.8601	0.7107	-2.2532	0.5329	-1.2102	0.2262
	Participation in decision-making	-0.6095	0.2990	-1.1954	-0.0235	-2.0386	**0.0415**
	MPA size	0.8430	0.7773	-0.6804	2.3665	-1.0846	0.2781
	Rule agreement	-0.1027	0.5010	-1.0847	0.8792	-0.2051	0.8375
Join	Graduated sanctions	1.8834	1.4128	-0.8857	4.6526	1.3331	0.1825
	Participation in decision-making	-0.0648	0.6017	-1.2442	1.1145	-0.1078	0.9142
	MPA size	0.4002	0.6785	-0.9296	1.7300	-0.5899	0.5553
	Rule agreement	-3.2083	0.6368	-4.4565	-1.9602	-5.0380	**0.0000**

My models also indicated that graduated sanctions had a positive but highly uncertain relationship (i.e. wide confidence intervals that overlapped zero) to joining poachers (Fig. 5.4, Table 5.4)—a relationship driven almost completely by respondents from villages surrounding two MPAs in Kenya, which had graduated sanctions. This could be interpreted as support for the "crowding out" effect (e.g. Vollan 2008), where external sanctions effectively crowd out fishers' intrinsic motivation to follow the rules or become active in enforcing the rules themselves. Yet, this is unlikely given that the MPAs and graduated sanctions in question were locally developed and managed. Furthermore, these same MPAs show very good trends of fish stock recovery (Cinner & McClanahan 2015), and less than 10% of all fishers surveyed in these communities told interviewers that they had seen poaching, so this may instead reflect a small subculture of non-compliers. In all, this counter-intuitive result highlights the importance of considering contextual information and detail that could be lost during large-scale comparative studies. This point is further emphasised by the significant, negative relationship between participation in decision-making and inaction, but a lack of a significant, positive relationship with either reporting or confronting poachers. Lastly, it is important to acknowledge that there are likely a multitude of other conditions and institutions (and interactions between these) that affect surveillance and enforcement (e.g. Cox *et al.* 2010; Warner & Pomeroy 2012; Turner *et al.* 2016), but measuring these concepts was beyond the scope of this study.

Mitigating risks

Efforts to encourage voluntary surveillance and enforcement may expose resource users to increased risk of conflict, violent reprisals, and in extreme cases, death (e.g. McSkimming & Berg 2008; Witness 2017). Accordingly, I do not advocate confrontations between non-trained and legally powerless fishers with potentially dangerous offenders, especially when enforcement authorities exist. Fortunately, institutions can be used to reduce risks by providing appropriate avenues, incentives, and protection to aid those who report non-compliance to the authorities (i.e. whistleblowers). For instance, community crime prevention initiatives such as Crime Stoppers (Lippert 2002) or anti-poaching hotlines (e.g. McSkimming & Berg 2008; Green 2016) encourage people to anonymously report crimes to enforcement authorities. These programs can increase group cohesion, enhance relations between people and enforcement authorities, and perhaps most importantly, act as deterrents for future crimes (Bursik & Grasmick 1993; Lippert 2002). Yet, the anonymity granted in these programs can be abused—poachers in Costa Rica admitted to reporting false infringements in other regions of MPAs to misdirect enforcement patrols from their own poaching activities elsewhere (A. Arias, personal communication). However, offering financial incentives can discourage false or misleading reports; some laws in the United States of America (e.g. Lacey Act and MARPOL Protocol; Kohn 2016) (e.g. Lacey Act and MARPOL Protocol; Kohn 2016) have provisions that allow financial rewards for whistleblowers contingent upon successful prosecution. While not a panacea for reducing crime, whistleblowers are the single most important source of fraud detection worldwide (ACFE 2014), and could therefore be particularly advantageous in combatting poaching if responsibly utilized (Kohn 2016).

An important caveat that must be acknowledged is that social desirability bias may have influenced respondent's answers about this potentially sensitive topic. While I cannot completely account for all of the biases inherent in this type of an investigation, I did use site and country as random effects in my statistical models to account for potential site or country-level biases. It is also possible that the type of fishing (i.e. recreational vs subsistence fishers) may affect fisher's response to poaching, and should therefore be considered and investigated in future research.

Here, I reveal a considerable portion of inactive human capital that could be harnessed to address the widespread lack of enforcement capacity currently hindering many of the world's MPAs (Gill *et al.* 2017). Effectively and responsibly harnessing this latent capital will depend on mitigating risks while removing barriers to inactivity, and fostering the norms, attitudes, beliefs, and institutional design elements that promote voluntary action. Numerous avenues could be used to engage fishers in this regard. These include the emphasis of how poaching personally affects each fisher, the promotion of stewardship norms and moral responsibility, and poverty alleviation programs to reduce the necessity to poach for survival. My results also suggest that the competing self-interests of fishers could lead to a culture of disharmony that discourages cooperation, underscoring the importance of equitable management outcomes for encouraging voluntary enforcement. NRM agencies throughout the world continue to develop programs to bolster compliance through voluntary surveillance and enforcement—further research and understanding of this topic, with special emphasis on risk mitigation, could therefore play a critical role in informing and affecting the outcomes of these efforts.

CHAPTER 6: Discussion and conclusions

6.1 General discussion

This thesis contributes knowledge and further understanding on resource user compliance, which underpins and often determines the success of conservation efforts. Indeed, ‘conservation means behaviour’ and more specifically, changing or influencing human behaviour to attain desired conservation outcomes (Schultz 2011). A lack of compliance effectively impairs these conservation efforts and results in ‘paper parks’ or other measures that serve to cast doubt on whether conservation can be effective. As mentioned previously, assessing, understanding, and curtailing poaching behaviours requires multi-disciplinary approaches that recognize and address the complex nature of human behavioural decisions in social-ecological systems. Prior to this thesis, multi-disciplinary, quantitative investigations of poaching were rarely attempted. Furthermore, very few studies triangulated compliance information. Here, I demonstrated how such investigations can provide larger, holistic pictures of poaching phenomena, including causes, levels, proxy indicators, and potential methods to curtail poaching. This approach does require substantial field time, both ecological and social, as well as regular collaborative efforts with enforcement authorities. However, this ensured that I could partner cutting edge social science theories and techniques with compliance management priorities. This yielded a substantial increase in the understanding of theoretical underpinnings of human behaviour in the context of compliance with conservation regulations. In addition, this thesis provided research results that were quickly assimilated and applied by compliance managers in one of the world’s most iconic marine parks, which I detail further below.

6.2 Key findings and contributions

One of the major contributions of this thesis is the approach I developed to detect and estimate poaching levels (Chapter 2). Using six methods (5 social, 1 ecological,) to estimate poaching levels addressed a primary difficulty in the study and management of compliance: determining the level of non-compliance, and assessing the reliability of the provided estimates. A key finding here was that even the specialized techniques designed to increase truthful answers may be subject to underreporting based on fishers' own estimates of the prevalence of poaching. Thus, these methods may be less reliable in systems where poaching is considered socially unacceptable or a taboo subject, because respondents may be likely to lie regardless of the technique employed. In these cases, examining proxy indicators of poaching, or asking about other people's poaching behaviours may yield better estimates because it reduces the social desirability bias while allowing people to project their own behaviours and beliefs through a mask of impersonality. However, I also demonstrate the current lack of understanding that impedes the application of these proxy indicators of poaching to go beyond estimating poaching levels and actually identify poachers (Chapter 4). More research on this topic is therefore warranted.

In addition, I illustrated the benefits of 'ground-truthing' estimates provided by social techniques with field-based ecological measurements. For instance, the range of non-compliance estimated by social techniques (3-13%; Chapter 2) would likely be considered quite low when compared to other MPAs in the world. Yet, my approach of quantifying derelict fishing gears via underwater visual censuses on inshore islands groups of the GBRMP revealed what these levels might translate to in the ecosystem; no-fishing reserves were receiving nearly the same amount of fishing pressure relative to areas open to fishing. This was surprising, because these inshore island groups were once considered to be among some of the best-enforced areas in the GBRMP. However, none of these approaches could answer the critical question of who, or why fishers were choosing to poach, further highlighting the need to complement these methods with quantitative examinations of the social dimensions of poaching.

My multi-disciplinary examination of the social dimensions of fishers' compliance decisions yielded findings that enable me to recommend five avenues to further advance our understanding and management of compliance in the quest for effective conservation outcomes: 1) further consideration of how people process information; 2) re-conceptualizing how people behave; 3) developing communication strategies to bolster compliance; 4) designing rules and interactions to shape behaviour; and 5) developing a holistic understanding of poaching.

1) How people process information

A common misconception in science is that people remain sceptical or fail to embrace scientific findings because they lack adequate information and understanding about the topic, which can be remedied by providing them with more information. Although this "information deficit hypothesis" has been discredited by an expanse of literature (e.g. Kahan *et al.* 2012), conservationists often assume that raising awareness will change people's behaviour (Schultz 2011; Heberlein 2012). For instance, many communication projects assume that fishers would comply if they had more information about benefits of no-fishing reserves (i.e. they produce more, and bigger fish, which may be exported). However, my findings demonstrate that fishers in the GBRMP were already aware of the benefits of reserves, and cited better catches as the primary motivation to poach (Chapter 3). When combined with the low perception of detection while poaching (Chapter 3), my findings suggest that extolling the benefits of no-fishing reserves could result in the perverse outcome of encouraging fishers to poach, especially when reserves lack enforcement capacity, which is demonstrably widespread (Gill *et al.* 2017). However, the GBRMP does not lack enforcement capacity, and has an advanced compliance monitoring program that uses risk–based planning to guide aircraft-, vessel-, and land-based enforcement patrols. This serves to further highlight the fact that enforcement alone is not enough to ensure high compliance, thereby necessitating further understanding of human behaviour.

2) Re-conceptualizing how people behave

Traditional models of human behaviour are often based on the economic premise of human beings as rational actors who make decisions based on the costs and benefits associated with the behaviour (e.g. Becker 1968). However, extensive research from social science disciplines illustrates that human behaviour is not always rational, and is heavily influenced by the norms surrounding them (Ostrom 1998; Cialdini 2003; Keizer & Schultz 2011). This notion was recently validated in a recreational fisheries context, where social norms (e.g. the social acceptability of poaching, and whether others poached), had the greatest influence on fishers' compliance behaviour (Thomas *et al.* 2016).

In addition, my findings suggest that three normative (mis)perceptions or mechanisms are also operative and working to encourage and reinforce a culture of non-compliance in the GBRMP recreational fisher population. One such misperception is that poachers likely conceive a 'false consensus' that others also poach, which allows them to justify their own continued non-compliance (Chapter 2). Furthermore, fishers who know poachers have significantly higher estimates of the level of poaching, which implies that fishers who know poachers believe that poaching is more common than fishers who do not associate with poachers (Chapter 2). These misperceptions are also likely working in tandem with another phenomenon called pluralistic ignorance, where compliant fishers wrongly assume that poaching is more common than it actually is (Chapter 3). Failing to correct these misperceptions could thus result in a cascading effect where previously compliant fishers change their behaviour to fit the misperceived norm of poaching (Prentice & Miller 1996).

3) Developing communication strategies to bolster compliance

In this thesis, I revealed numerous misperceptions, mechanisms, and potential drivers of poaching behaviours that should be malleable to targeted management and behavioural interventions such as policy changes and persuasive communication. Most of these misperceptions centred on the behaviour of others, so normative-based communications may be particularly effective in reducing poaching behaviours. Encouragingly, I found that most fishers comply with no-fishing reserves and believe poaching is socially unacceptable (Chapter 3). Reinforcing this first message may be particularly useful because it levers the descriptive norm (i.e. almost everyone else complies), which is a demonstrably powerful tool for social control (Cialdini 2007). If widely publicized, this message should correct the 'false consensus' among poachers that allows them to justify and continue poaching, as well as the pluralistic ignorance that causes compliant fishers to misperceive and overestimate the prevalence of poaching. Emphasizing the fact that almost all fishers believed poaching was socially unacceptable should also increase the power of shame to deter poachers, given that a moderate level of fishers said that social shame would not keep them from poaching (Chapter 3).

As mentioned previously, fishers did recognize the low probability of detection while poaching (Chapter 3), which is common in most marine fisheries (Kuperan & Sutinen 1998). This can be addressed in two ways. The first solution is increasing the actual probability of being detected, but this is often resource intensive, and the majority of the world's MPAs already lack critical staffing and enforcement capacity (Gill *et al.* 2017). A second, more suitable solution is to increase the perceived likelihood of detection with targeted communication strategies, because fishers do not know the actual probability of detection, so their behaviour is instead shaped by the perceived threat of enforcement (Grasmick & Bryjak 1980; Grasmick & Green 1980). This is relatively cost-effective compared to increasing the actual probability of detection, and can be accomplished by publicizing instances of successful apprehension and prosecution, the technologies being used to detect poaching (e.g. aerial surveillance, radar, night vision, etc.), and emphasizing that the likelihood of getting caught is high, as are the consequences of getting caught. Yet, this approach could also have perverse effects of increasing non-compliance if employed in contexts where little to no enforcement capacity exists. In these cases, fishers would know that the likelihood of getting caught is very low, so contradictive messages from management authorities would only serve to undermine trust in management, which is demonstrably critical for compliance (Stern 2008). A last message that could be further emphasised is one regularly emphasized to deter drink driving: ignorance is not an excuse. My findings demonstrate that fishers have very high levels of awareness of no-fishing reserve boundaries and knowledge of where to get zoning maps, so communications could emphasize that the burden of responsibility is now on the user not to break the law, rather than on law enforcement officers to prove intent.

As mentioned previously, I maintained a collaborative partnership with the Field Compliance Unit of the Great Barrier Reef Marine Park Authority throughout my thesis. Regular briefings on my research findings thereby allowed me to recommend pertinent compliance communication and outreach strategies that GBRMPA has since adapted and employed as part of their ongoing recreational fishing compliance blitz (GBRMPA 2016, 2017c, d, b). Thus, key results from this thesis have already had uptake by the relevant management agency. Future research that follows on and further develops the approach of this thesis would therefore be well positioned to assess the effects of these adapted communication strategies on fishers' perceptions and compliance behaviours.

4) Designing rules and interactions to shape behaviour

As explained in Chapter 5, many natural resource management agencies are attempting to bolster current shortcomings in enforcement capacity by harnessing the latent surveillance and enforcement capacity of resource users. Yet, these efforts could result in perverse outcomes such as increased conflict, violence, and in extreme cases, death (Witness 2017). Responsibly and effectively engaging resources users therefore necessitates understanding how institutional design aspects can be used to encourage and protect informal voluntary surveillance and enforcement behaviours. In Chapter 5, I demonstrated that most fishers chose to do nothing after observing poaching, often because they wanted to avoid conflict. Encouragingly, institutions can reduce conflict by offering avenues, incentives, and protection to voluntary reporters or 'whistleblowers'. Voluntary reporting programs such as Crime Stoppers or anti-poaching hotlines are used throughout the world to aid policing and conservation enforcement (e.g. Lippert 2002; McSkimming & Berg 2008; Green 2016). If designed and operated effectively, these institutions can create closer ties between resource users and enforcement authorities, as well as acting as deterrents for future crimes (Bursik & Grasmick 1993; Lippert 2002).

If financial incentives are made available to voluntary reporters of poaching upon *successful* prosecution, these programs could be particularly advantageous for combatting poaching (Kohn 2016). For instance, offering financial incentives contingent upon successful prosecution drastically increased reliable reporting of fraud, to the extent that whistleblowers are now the most important source of fraud detection worldwide (ACFE 2014). While I did demonstrate how voluntary surveillance and enforcement were related more to outcomes (rule agreement) than participation in decision-making (Chapter 5), considerable knowledge gaps exist in our understanding of how to use institutional design aspects and conditions to shape compliance and voluntary enforcement in conservation. I discuss how to address this gap, among others, in the future directions section below.

5) Developing a holistic understanding of poaching

Taking an interdisciplinary approach allowed me to use numerous lenses to view and appreciate the contribution of a range of social science disciplines to understand poaching behaviours. I found support for multiple theories and disciplinary perspectives; each provides important information on the different components of compliance, but would also have been incomplete if considered on its own. For example, the TPB is often used to investigate poaching behaviours, but even the extended framework (incorporating more normative dimensions) did not capture other aspects that were significant predictors of the proxy indicators of poaching. For instance, in the institutional literature, Ostrom discussed the importance of considering characteristics of resource users (Ostrom 2007). In this case, proxy indicators of poaching were significantly related to two such characteristics (i.e. fisher avidity and the importance of fishing), and an additional component of socialisation not typically covered in the TPB (whether a respondent actually fished with others). This multi-disciplinary approach was again useful for examining and understanding the conditions that influenced fishers' behaviours after observing poaching. Here, I found that rule agreement (often conceptualized as distributive justice in social psychology) and participation in decision-making processes (incorporated from Ostrom's seminal work on institutions) were both significant predictors of fishers' behaviours after observing poaching. By comparing the influence of these two predictors, I was able to demonstrate how the competing self-interests of fishers may lead to a culture of disharmony that discourages cooperation, thereby highlighting the need to for equitable management practices. Thus, tying together a range of disciplinary theories, perspectives, and approaches allowed me to provide a more holistic understanding of poaching-related behaviours, and a better understanding of how to address them, than would have been provided by a more traditional and compartmentalized single disciplinary approach.

6.3 Caveats and Critiques

Numerous caveats apply to this thesis. First and foremost, this thesis investigated an illegal and socially undesirable behaviour, so information yielded from this thesis is certainly subject to some level of bias and untruthful reporting. I addressed this specific issue by investing considerable efforts to ensure that I developed a rigorous survey and question design that used forgiving wording. All of these design aspects were guided and refined during qualitative pilot research and numerous pilot trials of survey application. I also employed multiple measurement methods/approaches to estimating and understanding poaching behaviours, which were useful for triangulating and comparing information from each source.

A second caveat is that this study (Chapter 4) used a correlative rather than causal investigatory approach. I chose this method because it allowed me to purposively sample fishers returning to access points directly after fishing in the GBRMP. This ensured that I sampled the relevant fishing population and reduced the potential for selection bias (i.e. poachers may be less inclined to voluntarily complete an online survey about compliance at a later date compared to a short survey when loading their boat). It is still possible that some poachers declined the invitation to survey, but my approach to introducing and explaining the survey used forgiving wording to reduce this possibility. Although I did ask people whether they thought others poached due to a low risk of detection, I did not specifically ask respondents to indicate their own personal perception of the likelihood of detection. It is therefore possible that this method masked the potential differences between these two types of perceived deterrence.

6.4 Future directions

As mentioned earlier, numerous gaps still exist in our understanding of compliance and enforcement behaviours in the context of poaching in protected areas. A specific shortcoming is the lack of knowledge about the effects of institutions on individual's behaviours. One way to address this knowledge gap could be to apply game theory experiments in the field that examine fisher's compliance and enforcement decisions under changing rules and designs. I also described numerous normative dimensions that seem to be influencing fishers' compliance behaviours and their perception of these behaviours in society. Yet, less is known about how these perceptions are diffused throughout the fishing population. For instance, poaching subcultures are likely to exist, but how are the perceived values or prevalence of poaching established and transmitted throughout the subgroup? These answers could potentially be established by using social network analysis to examine fisher's perceptions of poaching. By combining derelict fishing gear surveys and estimates of poaching provided via social surveys, I also highlighted how the estimated levels of poaching in a population of fishers does not necessarily capture the impact of that non-compliance in the water. In this regard, further advancements in techniques (e.g. the quantitative randomised response technique; Conteh *et al.* 2015) could be applied to assess both the prevalence of poaching, and its impact on the ecosystem.

6.5 Concluding remarks

The avenues described here are substantial departures from traditional views of enforcement and compliance management. Yet, the need for a sea change in compliance management is evident, given the growing number of studies that describe the relative inability of most marine reserves to effectively manage poaching (e.g. Mora *et al.* 2006; Gill *et al.* 2017). This transformation will undoubtedly be accompanied by growing pains, wicked problems, and continued failures in management, but as Albert Einstein once said, “Often in evolutionary processes a species must adopt to new conditions in order to survive” (Einstein 1946).

References

1.
Abrahamse, W. & Steg, L. (2013). Social influence approaches to encourage resource conservation: A meta-analysis. *Global Environmental Change*, 23, 1773-1785.
2.
ACFE (2014). Report to the nations on occupational fraud and abuse. Association of Certified Crime Examiners.
3.
Afflerbach, J.C., Lester, S.E., Dougherty, D.T. & Poon, S.E. (2014). A global survey of "TURF-reserves", Territorial Use Rights for Fisheries coupled with marine reserves. *Global Ecology and Conservation*, 2, 97-106.
4.
Ajzen, I. (1991). The theory of planned behavior. *Organizational behavior and human decision processes*, 50, 179-211.
5.
Ajzen, I. & Fishbein, M. (1980). Understanding attitudes and predicting social behaviour.
6.
Akers, R.L. (2011). *Social learning and social structure: A general theory of crime and deviance*. Transaction Publishers.
7.
Alexander, S.M., Armitage, D. & Charles, A. (2015). Social networks and transitions to co-management in Jamaican marine reserves and small-scale fisheries. *Global Environmental Change*, 35, 213-225.
8.
Arias, A. (2015). Understanding and managing compliance in the nature conservation context. *Journal of environmental management*, 153, 134-143.
9.
Arias, A., Cinner, J.E., Jones, R.E. & Pressey, R.L. (2015). Levels and drivers of fishers' compliance with marine protected areas. *Ecology and Society*, 20.
10.
Arias, A. & Sutton, S. (2013). Understanding recreational fishers' compliance with no-take zones in the Great Barrier Reef Marine Park. *Ecology and Society*, 18.
11.
Armitage, C.J. & Conner, M. (2001). Efficacy of the theory of planned behaviour: A meta-analytic review. *British journal of social psychology*, 40, 471-499.
12.
Bandura, A. & Walters, R.H. (1977). Social learning theory.
13.
Barrett-Howard, E. & Tyler, T.R. (1986). Procedural justice as a criterion in allocation decisions. *Journal of Personality and Social Psychology*, 50, 296.
14.
Becker, G.S. (1968). Crime and punishment: An economic approach. In: *The economic dimensions of crime*. Springer, pp. 13-68.
15.

Bell, S., Hampshire, K. & Topalidou, S. (2007). The political culture of poaching: a case study from northern Greece. *Biodiversity and Conservation*, 16, 399-418.
16.
Bergseth, B.J., Russ, G.R. & Cinner, J.E. (2015). Measuring and monitoring compliance in no-take marine reserves. *Fish and Fisheries*, 16, 240-258.
17.
Berkowitz, A.D. (2005). An overview of the social norms approach. *Changing the culture of college drinking: A socially situated health communication campaign*, 193-214.
18.
Bessey, K.M. (1985). Wildlife law revised: violator profiles and their implications for management. *Human Dimensions in Wildlife Newsletter*, 4, 10-16.
19.
Bova, C.S., Halse, S.J., Aswani, S. & Potts, W.M. (2017). Assessing a social norms approach for improving recreational fisheries compliance. *Fisheries Management and Ecology*, 24, 117-125.
20.
Buchanan, T. (1985). Commitment and leisure behavior: A theoretical perspective. *Leisure Sciences*, 7, 401-420.
21.
Bursik, R. & Grasmick, H. (1993). Neighborhoods and crime: The dimensions of effective social control. *New York: Lexington*.
22.
Büscher, B. & Ramutsindela, M. (2015). Green Violence: Rhino Poaching and the War to Save Southern Africa's Peace Parks. *African Affairs*.
23.
Byers, J.E. & Noonburg, E.G. (2007). Poaching, enforcement, and the efficacy of marine reserves. *Ecological Applications*, 17, 1851-1856.
24.
Cardenas, J.C., Stranlund, J. & Willis, C. (2000). Local environmental control and institutional crowding-out. *World development*, 28, 1719-1733.
25.
Cattell, R.B. (1966). The scree test for the number of factors. *Multivariate behavioral research*, 1, 245-276.
26.
Cepić, D. & Nunan, F. (2017). Justifying non-compliance: The morality of illegalities in small scale fisheries of Lake Victoria, East Africa. *Marine Policy*, 86, 104-110.
27.
Chen, X.-P., Pillutla, M.M. & Yao, X. (2009). Unintended consequences of cooperation inducing and maintaining mechanisms in public goods dilemmas: Sanctions and moral appeals. *Group Processes & Intergroup Relations*, 12, 241-255.
28.
Cialdini, R.B. (2003). Crafting normative messages to protect the environment. *Current directions in psychological science*, 12, 105-109.
29.
Cialdini, R.B. (2007). Descriptive social norms as underappreciated sources of social control. *Psychometrika*, 72, 263.
30.

Cialdini, R.B. & Goldstein, N.J. (2004). Social influence: compliance and conformity. *Annu Rev Psychol*, 55, 591-621.
31.
Cialdini, R.B., Kallgren, C.A. & Reno, R.R. (1991). A focus theory of normative conduct: A theoretical refinement and reevaluation of the role of norms in human behavior. In: *Advances in experimental social psychology*. Elsevier, pp. 201-234.
32.
Cinner, J.E., Daw, T., Huchery, C., Thoya, P., Wamukota, A., Cedras, M. *et al.* (2014). Winners and Losers in Marine Conservation: Fishers' Displacement and Livelihood Benefits from Marine Reserves. *Society & Natural Resources*, 27, 994-1005.
33.
Cinner, J.E. & McClanahan, T.R. (2015). A sea change on the African coast? Preliminary social and ecological outcomes of a governance transformation in Kenyan fisheries. *Global Environmental Change*, 30, 133-139.
34.
Cinner, J.E., McClanahan, T.R., MacNeil, M.A., Graham, N.A., Daw, T.M., Mukminin, A. *et al.* (2012). Comanagement of coral reef social-ecological systems. *Proc Natl Acad Sci U S A*, 109, 5219-5222.
35.
Cohen-Charash, Y. & Spector, P.E. (2001). The role of justice in organizations: A meta-analysis. *Organizational behavior and human decision processes*, 86, 278-321.
36.
Conner, M. & Armitage, C.J. (1998). Extending the theory of planned behavior: A review and avenues for further research. *Journal of applied social psychology*, 28, 1429-1464.
37.
Conteh, A., Gavin, M.C. & Solomon, J. (2015). Quantifying illegal hunting: a novel application of the quantitative randomised response technique. *Biological Conservation*, 189, 16-23.
38.
Cox, M., Arnold, G. & Tomás, S.V. (2010). A review of design principles for community-based natural resource management.
39.
Cross, P., St John, F.A., Khan, S. & Petroczi, A. (2013). Innovative techniques for estimating illegal activities in a human-wildlife-management conflict. *PLoS One*, 8, e53681.
40.
Dalton, D.R., Wimbush, J.C. & Daily, C.M. (1994). Using the unmatched count technique (UCT) to estimate base rates for sensitive behavior. *Personnel Psychology*, 47, 817-829.
41.
Davis, K.J., Kragt, M.E., Gelcich, S., Burton, M., Schilizzi, S. & Pannell, D.J. (2015). Why are Fishers not Enforcing Their Marine User Rights? *Environmental and Resource Economics*, 67, 661-681.
42.
Davis, K.L.F., Russ, G.R., Williamson, D.H. & Evans, R.D. (2004). Surveillance and Poaching on Inshore Reefs of the Great Barrier Reef Marine Park. *Coastal Management*, 32, 373-387.
43.

Dawes, R.M., McTavish, J. & Shaklee, H. (1977). Behavior, communication, and assumptions about other people's behavior in a commons dilemma situation. *Journal of personality and social psychology*, 35, 1.
44.
DeCaro, D.A., Janssen, M.A. & Lee, A. (2015). Synergistic effects of voting and enforcement on internalized motivation to cooperate in a resource dilemma. *Judgment and Decision Making*, 10, 511.
45.
Ditton, R.B., Loomis, D.K. & Choi, S. (1992). Recreation specialization: Re-conceptualization from a social worlds perspective. *Journal of Leisure Research*, 24, 33.
46.
Droitcour, J., Caspar, R.A., Hubbard, M.L., Parsley, T.L., Visscher, W. & Ezzati, T.M. (1991). The item count technique as a method of indirect questioning: A review of its development and a case study application. *Measurement errors in surveys*, 185-210.
47.
Einstein, A. (1946). The real problem is in the hearts of men. (ed. Amrine, M) The New York Times Magazine.
48.
Eliason, S.L. (2004). Accounts of Wildlife Law Violators: Motivations and Rationalizations. *Human Dimensions of Wildlife*, 9, 119-131.
49.
Epstein, G. (2017). Local rulemaking, enforcement and compliance in state-owned forest commons. *Ecological Economics*, 131, 312-321.
50.
Fabiano, P.M., Perkins, H.W., Berkowitz, A., Linkenbach, J. & Stark, C. (2003). Engaging men as social justice allies in ending violence against women: evidence for a social norms approach. *J Am Coll Health*, 52, 105-112.
51.
Feary, D.A., Cinner, J.E., Graham, N.A. & Januchowski-Hartley, F.A. (2011). Effects of customary marine closures on fish behavior, spear-fishing success, and underwater visual surveys. *Conserv Biol*, 25, 341-349.
52.
Festinger, L. (1954). A theory of social comparison processes. *Human relations*, 7, 117-140.
53.
Filteau, M.R. (2012). Deterring defiance:'Don’t give a poacher a reason to poach'.
54.
Fischer, C.S. (1995). The subcultural theory of urbanism: A twentieth-year assessment. *American Journal of Sociology*, 101, 543-577.
55.
Fishbein, M. & Ajzen, I. (1975). *Belief, attitude, intention and behavior: An introduction to theory and research*.
56.
Fisher, R.J. (1993). Social desirability bias and the validity of indirect questioning. *Journal of consumer research*, 20, 303-315.
57.
Garoupa, N. (1997). The theory of optimal law enforcement. *Journal of economic surveys*, 11, 267-295.

58.
Gavin, M.C., Solomon, J.N. & Blank, S.G. (2010). Measuring and monitoring illegal use of natural resources. *Conserv Biol*, 24, 89-100.
59.
GBRMPA (2015). Fishing offences on the rise. Great Barrier Reef Marine Park Authority.
60.
GBRMPA (2016). Public urged to report suspected illegal fishing and help protect the Reef. . Great Barrier Reef Marine Park Agency.
61.
GBRMPA (2017a). Annual report 2016-2017. Great Barrier Reef Marine Park Authority.
62.
GBRMPA (2017b). Illegal recreational fishers are catch of the day. Great Barrier Reef Marine Park Authority.
63.
GBRMPA (2017c). Illegal recreational fishing fine increased to $2100. Great Barrier Reef Marine Park Authority.
64.
GBRMPA (2017d). Ramped-up patrols to target illegal fishing hotspots this school holidays. Great Barrier Reef Marine Park Authority.
65.
Gelman, A. & Hill, J. (2007). *Data analysis using regression and multilevel/hierarchical models*. Cambridge University Press New York, NY, USA.
66.
Gelman, A., Jakulin, A., Pittau, M.G. & Su, Y.-S. (2008). A weakly informative default prior distribution for logistic and other regression models. *The Annals of Applied Statistics*, 2, 1360-1383.
67.
Gezelius, S.S. (2004). Food, Money, and Morals: Compliance Among Natural Resource Harvesters. *Human Ecology*, 32, 615-634.
68.
Gigliotti, L.M. & Peyton, R.B. (1993). Values and behaviors of trout anglers, and their attitudes toward fishery management, relative to membership in fishing organizations: a Michigan case study. *North American Journal of Fisheries Management*, 13, 492-501.
69.
Gill, D.A., Mascia, M.B., Ahmadia, G.N., Glew, L., Lester, S.E., Barnes, M. *et al.* (2017). Capacity shortfalls hinder the performance of marine protected areas globally. *Nature*, 543, 665-669.
70.
Grasmick, H.G. & Bryjak, G.J. (1980). The deterrent effect of perceived severity of punishment. *Social forces*, 59, 471-491.
71.
Grasmick, H.G. & Green, D.E. (1980). Legal punishment, social disapproval and internalization as inhibitors of illegal behavior. *J. Crim. L. & Criminology*, 71, 325.
72.
Green, E.K. (2016). Judging the Effectiveness of Anti-Poaching Hotlines. *Journal of Rural Social Sciences*, 31, 1.

73.
Gurney, G., Cinner, J., Sartin, J., Pressey, R., Ban, N., Marshall, N. *et al.* (2016). Participation in devolved commons management: Multiscale socioeconomic factors related to individuals' participation in community-based management of marine protected areas in Indonesia. *Environmental Science & Policy*, 61, 212-220.
74.
Gurney, G.G., Cinner, J., Ban, N.C., Pressey, R.L., Pollnac, R., Campbell, S.J. *et al.* (2014). Poverty and protected areas: An evaluation of a marine integrated conservation and development project in Indonesia. *Global Environmental Change*, 26, 98-107.
75.
Gurney, G.G., Pressey, R.L., Cinner, J.E., Pollnac, R. & Campbell, S.J. (2015). Integrated conservation and development: evaluating a community-based marine protected area project for equality of socioeconomic impacts. *Phil. Trans. R. Soc. B*, 370, 20140277.
76.
Gutierrez, N.L., Hilborn, R. & Defeo, O. (2011). Leadership, social capital and incentives promote successful fisheries. *Nature*, 470, 386-389.
77.
Haggarty, D.R., Martell, S.J.D. & Shurin, J.B. (2016). Lack of recreational fishing compliance may compromise effectiveness of Rockfish Conservation Areas in British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences*, 73, 1587-1598.
78.
Haines, M.P. & Barker, G.P. (2003). The NIU experiment: a case study of the social norms approach. In: *The Social Norms Approach to Preventing School and College Age Substance Abuse: A Handbook for Educators, Counselors, and Clinicians* (ed. Perkins, HW). Jossey-Bass San Fransciso.
79.
Ham, S., Brown, T., Curtis, J., Weiler, B., Hughes, M. & Poll, M. (2009). Promoting persuasion in protected areas: A guide for managers who want to use strategic communication to influence visitor behaviour. Technical report.
80.
Hancock, L. & Henry, N. (2003). Perceptions, norms and tobacco use in college residence hall freshmen: evaluation of a social norms marketing intervention. In: *The Social Norms Approach to Preventing School and College Age Substance Abuse: A Handbook for Educators, Counselors, and Clinicians* (ed. Perkins, HW). Jossey-Bass San Francisco.
81.
Hardeman, W., Johnston, M., Johnston, D., Bonetti, D., Wareham, N. & Kinmonth, A.L. (2002). Application of the theory of planned behaviour in behaviour change interventions: A systematic review. *Psychology and health*, 17, 123-158.
82.
Hatcher, A., Jaffry, S., Thébaud, O. & Bennett, E. (2000). Normative and social influences affecting compliance with fishery regulations. *Land Economics*, 448-461.
83.
Hauck, M. & Sweijd, N. (1999). A case study of abalone poaching in South Africa and its impact on fisheries management. *ICES Journal of Marine Science*, 56, 1024-1032.
84.

Heberlein, T.A. (2012). Navigating environmental attitudes. *Conservation Biology*, 26, 583-585.

85.
Henrich, J., Mcelreath, R., Barr, A., Ensminger, J., Barrett, C., Bolyanatz, E. *et al.* (2006). Costly punishment across human societies. *Science*, 312, 1767-1770.

86.
Hofstede, G. (1980). Culture's consequences: National differences in thinking and organizing. *Beverly Hills, Calif.: Sage*.

87.
Horvitz, D.G., Greenberg, B.G. & Abernathy, J.R. (1976). Randomized response: A data-gathering device for sensitive questions. *International Statistical Review/Revue Internationale de Statistique*, 181-196.

88.
Januchowski-Hartley, F.A., Graham, N.A., Cinner, J.E. & Russ, G.R. (2015). Local fishing influences coral reef fish behavior inside protected areas of the Indo-Pacific. *Biological Conservation*, 182, 8-12.

89.
Januchowski-Hartley, F.A., Graham, N.A., Feary, D.A., Morove, T. & Cinner, J.E. (2011). Fear of fishers: human predation explains behavioral changes in coral reef fishes. *PLoS One*, 6, e22761.

90.
Johannes, R.E. (1981). *Words of the lagoon: fishing and marine lore in the Palau district of Micronesia*. Univ of California Press.

91.
Kahan, D.M., Peters, E., Wittlin, M., Slovic, P., Ouellette, L.L., Braman, D. *et al.* (2012). The polarizing impact of science literacy and numeracy on perceived climate change risks. *Nature climate change*, 2, 732.

92.
Kahler, J.S. & Gore, M.L. (2012). Beyond the cooking pot and pocket book: Factors influencing noncompliance with wildlife poaching rules. *International Journal of Comparative and Applied Criminal Justice*, 36, 103-120.

93.
Keizer, K. & Schultz, P. (2011). The role of social norms in understanding and changing pro-environmental behaviour. In press in L. Steg, A. van den Berg, and J. de Groot, editors. Environmental psychology. Wiley-Blackwell, Malden, Massachusetts.

94.
Kelleher, G., Bleakley, C. & Wells, S. (1995). A global representative system of marine protected areas. Washington DC.

95.
Kohn, S.M. (2016). Monetary rewards for wildlife whistleblowers: a game-changer in wildlife trafficking detection and deterrence. *Environmental Law Reporter*, 46, 10054-10070.

96.
Kuperan, K. & Sutinen, J.G. (1998). Blue water crime: deterrence, legitimacy, and compliance in fisheries. *Law and Society Review*, 309-338.

97.
Lakens, D. (2013). Calculating and reporting effect sizes to facilitate cumulative science: a practical primer for t-tests and ANOVAs. *Frontiers in psychology*, 4, 863.

98.
Lancaster, D., Dearden, P. & Ban, N.C. (2015). Drivers of recreational fisher compliance in temperate marine conservation areas: A study of Rockfish Conservation Areas in British Columbia, Canada. *Global Ecology and Conservation*, 4, 645-657.
99.
Landsheer, J.A., Van der Heijden, P. & Van Gils, G. (1999). Trust and understanding, two psychological aspects of randomized response. *Quality and Quantity*, 33, 1-12.
100.
Lensvelt-Mulders, G.J. & Boeije, H.R. (2007). Evaluating compliance with a computer assisted randomized response technique: a qualitative study into the origins of lying and cheating. *Computers in Human Behavior*, 23, 591-608.
101.
Lensvelt-Mulders, G.J.L.M., Hox, J.J., van der Heijden, P.G.M. & Maas, C.J.M. (2005). Meta-Analysis of Randomized Response Research: Thirty-Five Years of Validation. *Sociological Methods & Research*, 33, 319-348.
102.
Levi, M., Sacks, A. & Tyler, T. (2009). Conceptualizing Legitimacy, Measuring Legitimating Beliefs. *American Behavioral Scientist*, 53, 354-375.
103.
Li, O., Sutton, S.G. & Tynan, L. (2010). Communicating scientific information to recreational fishers. *Human Dimensions of Wildlife*, 15, 106-118.
104.
Linares, C., Garrabou, J., Hereu, B., Diaz, D., Marschal, C., Sala, E. *et al.* (2012). Assessing the effectiveness of marine reserves on unsustainably harvested long-lived sessile invertebrates. *Conserv Biol*, 26, 88-96.
105.
Lind, E.A. & Tyler, T.R. (1988). *The social psychology of procedural justice*. Springer Science & Business Media.
106.
Lippert, R. (2002). Policing property and moral risk through promotions, anonymization and rewards: crime stoppers revisited. *Social & Legal Studies*, 11, 475-502.
107.
Little, L., Smith, A., McDonald, A., Punt, A., Mapstone, B., Pantus, F. *et al.* (2005). Effects of size and fragmentation of marine reserves and fisher infringement on the catch and biomass of coral trout, Plectropomus leopardus, on the Great Barrier Reef, Australia. *Fisheries Management and Ecology*, 12, 177-188.
108.
Lundquist, C.J. & Granek, E.F. (2005). Strategies for Successful Marine Conservation: Integrating Socioeconomic, Political, and Scientific Factors. *Conservation Biology*, 19, 1771-1778.
109.
Magee, C., Voyer, M., McIlgorm, A. & Li, O. (2018). Chasing the thrill or just passing the time? Trialing a new mixed methods approach to understanding heterogeneity amongst recreational fishers based on motivations. *Fisheries Research*, 199, 107-118.
110.

Martin, C.L., Momtaz, S., Jordan, A. & Moltschaniwskyj, N.A. (2016). Exploring recreational fishers' perceptions, attitudes, and support towards a multiple-use marine protected area six years after implementation. *Marine Policy*, 73, 138-145.
111.
Matthies, E., Klöckner, C.A. & Preißner, C.L. (2006). Applying a modified moral decision making model to change habitual car use: how can commitment be effective? *Applied Psychology*, 55, 91-106.
112.
Mbaru, E.K. & Barnes, M.L. (2017). Key players in conservation diffusion: Using social network analysis to identify critical injection points. *Biological Conservation*, 210, 222-232.
113.
McCauley, D.J., Woods, P., Sullivan, B., Bergman, B., Jablonicky, C., Roan, A. *et al.* (2016). Ending hide and seek at sea. *Science*, 351, 1148-1150.
114.
McClanahan, T.R. (1999). Is there a future for coral reef parks in poor tropical countries? *Coral reefs*, 18, 321-325.
115.
McClanahan, T.R., Graham, N.A.J., Wilson, S.K., Letourneur, Y. & Fisher, R. (2009). Effects of fisheries closure size, age, and history of compliance on coral reef fish communities in the western Indian Ocean. *Marine Ecology Progress Series*, 396, 99-109.
116.
McConney, P. & Pena, M. (2012). Capacity for (Co)Management of Marine Protected Areas in the Caribbean. *Coastal Management*, 40, 268-278.
117.
McCook, L.J., Ayling, T., Cappo, M., Choat, J.H., Evans, R.D., De Freitas, D.M. *et al.* (2010). Adaptive management of the Great Barrier Reef: a globally significant demonstration of the benefits of networks of marine reserves. *Proceedings of the National Academy of Sciences*, 107, 18278-18285.
118.
McSkimming, M.J. & Berg, B.L. (2008). Motivations for Citizen Involvement in a Community Crime Prevention Initiative: Pennsylvania's TIP (Turn in a Poacher) Program. *Human Dimensions of Wildlife*, 13, 234-242.
119.
Milgram, S., Bickman, L. & Berkowitz, L. (1969). Note on the drawing power of crowds of different size. *Journal of personality and social psychology*, 13, 79.
120.
Miller, J.D. (1984). *A new survey technique for studying deviant behavior*. George Washington University.
121.
Mora, C., Andréfouët, S., Costello, M.J., Kranenburg, C., Rollo, A., Veron, J. *et al.* (2006). Coral reefs and the global network of marine protected areas. *SCIENCE-NEW YORK THEN WASHINGTON-*, 2006, 1750.
122.
Mulder, L.B., van Dijk, E., De Cremer, D. & Wilke, H.A.M. (2006). Undermining trust and cooperation: The paradox of sanctioning systems in social dilemmas. *Journal of Experimental Social Psychology*, 42, 147-162.

123.
Mullen, B. & Hu, L.t. (1988). Social projection as a function of cognitive mechanisms: Two meta-analytic integrations. *British Journal of Social Psychology*, 27, 333-356.
124.
Muth, R.M. & Bowe, J.F. (1998). Illegal harvest of renewable natural resources in North America: Toward a typology of the motivations for poaching. *Society & Natural Resources*, 11, 9-24.
125.
Nicholson, J. & Higgins, G.E. (2017). Social Structure Social Learning Theory: Preventing Crime and Violence. In: *Preventing Crime and Violence*, pp. 11-20.
126.
Nielsen, J.R. & Mathiesen, C. (2003). Important factors influencing rule compliance in fisheries lessons from Denmark. *Marine Policy*, 27, 409-416.
127.
Nuno, A. & St. John, F.A.V. (2015). How to ask sensitive questions in conservation: A review of specialized questioning techniques. *Biological Conservation*, 189, 5-15.
128.
Ostrom, E. (1990). *Governing the commons*. Cambridge university press.
129.
Ostrom, E. (1998). A behavioral approach to the rational choice theory of collective action: Presidential address, American Political Science Association, 1997. *American political science review*, 92, 1-22.
130.
Ostrom, E. (2005). *Understanding Institutional Diversity*. Princeton University Press.
131.
Ostrom, E. (2007). A diagnostic approach for going beyond panaceas. *Proc Natl Acad Sci U S A*, 104, 15181-15187.
132.
Oyserman, D., Coon, H.M. & Kemmelmeier, M. (2002). Rethinking individualism and collectivism: Evaluation of theoretical assumptions and meta-analyses. *Psychological bulletin*, 128, 3.
133.
Perkins, H.W. (1997). College student misperceptions of alcohol and other drug norms among peers: Exploring causes, consequences, and implications for prevention programs. *Designing alcohol and other drug prevention programs in higher education: Bringing theory into practice*, 177-206.
134.
Petróczi, A., Mazanov, J., Nepusz, T., Backhouse, S.H. & Naughton, D.P. (2008). Comfort in big numbers: Does over-estimation of doping prevalence in others indicate self-involvement? *J Occup Med Toxicol*, 3, 19.
135.
Pollnac, R., Christie, P., Cinner, J.E., Dalton, T., Daw, T.M., Forrester, G.E. *et al.* (2010). Marine reserves as linked social-ecological systems. *Proc Natl Acad Sci U S A*, 107, 18262-18265.
136.
Pomeroy, R.S. & Berkes, F. (1997). Two to tango: the role of government in fisheries co-management. *Marine policy*, 21, 465-480.

137.
Prentice, D.A. & Miller, D.T. (1996). Pluralistic Ignorance and the Perpetuation of Social Norms by Unwitting Actors. In: *Advances in Experimental Social Psychology Volume 28*, pp. 161-209.
138.
Rife, A.N., Erisman, B., Sanchez, A. & Aburto-Oropeza, O. (2013). When good intentions are not enough… Insights on networks of "paper park" marine protected areas. *Conservation Letters*, 6, 200-212.
139.
Robbins, W.D., Hisano, M., Connolly, S.R. & Choat, J.H. (2006). Ongoing collapse of coral-reef shark populations. *Curr Biol*, 16, 2314-2319.
140.
Robertson, C.J., Ralston, D.A. & Crittenden, W.F. (2012). The relationship between cultural values and moral philosophy: A generational subculture theory approach. *AMS review*, 2, 99-107.
141.
Ross, L., Greene, D. & House, P. (1977). The "false consensus effect": An egocentric bias in social perception and attribution processes. *Journal of experimental social psychology*, 13, 279-301.
142.
Russ, G.R. & Alcala, A.C. (2010). Decadal-scale rebuilding of predator biomass in Philippine marine reserves. *Oecologia*, 163, 1103-1106.
143.
Rustagi, D., Engel, S. & Kosfeld, M. (2010). Conditional cooperation and costly monitoring explain success in forest commons management. *science*, 330, 961-965.
144.
Salvatier, J., Wiecki, T.V. & Fonnesbeck, C. (2016). Probabilistic programming in Python using PyMC3. *PeerJ Computer Science*, 2, e55.
145.
Salz, R.J. & Loomis, D.K. (2005). Recreation specialization and anglers' attitudes towards restricted fishing areas. *Human Dimensions of Wildlife*, 10, 187-199.
146.
Sandström, A., Crona, B. & Bodin, Ö. (2014). Legitimacy in Co-Management: The Impact of Preexisting Structures, Social Networks and Governance Strategies. *Environmental Policy and Governance*, 24, 60-76.
147.
Schroeder, C.M. & Prentice, D.A. (1998). Exposing pluralistic ignorance to reduce alcohol use among college students. *Journal of Applied Social Psychology*, 28, 2150-2180.
148.
Schultz, P.W. (2011). Conservation means behavior. *Conserv Biol*, 25, 1080-1083.
149.
Schultz, P.W., Nolan, J.M., Cialdini, R.B., Goldstein, N.J. & Griskevicius, V. (2007). The constructive, destructive, and reconstructive power of social norms. *Psychological science*, 18, 429-434.
150.
Sherman, L.W. (1993). Defiance, Deterrence, and Irrelevance: A Theory of the Criminal Sanction. *Journal of Research in Crime and Delinquency*, 30, 445-473.

151.
Smallwood, C.B. & Beckley, L.E. (2012). Spatial distribution and zoning compliance of recreational fishing in Ningaloo Marine Park, north-western Australia. *Fisheries Research*, 125-126, 40-50.
152.
St. John, F.A.V., Edwards-Jones, G., Gibbons, J.M. & Jones, J.P.G. (2010). Testing novel methods for assessing rule breaking in conservation. *Biological Conservation*, 143, 1025-1030.
153.
St. John, F.A.V., Keane, A.M., Edwards-Jones, G., Jones, L., Yarnell, R.W. & Jones, J.P.G. (2012). Identifying indicators of illegal behaviour: carnivore killing in human-managed landscapes. *Proceedings of the Royal Society of London B: Biological Sciences*, rspb20111228.
154.
St. John, F.A.V., Mai, C.-H. & Pei, K.J.C. (2015). Evaluating deterrents of illegal behaviour in conservation: Carnivore killing in rural Taiwan. *Biological Conservation*, 189, 86-94.
155.
Steg, L. & Vlek, C. (2009). Encouraging pro-environmental behaviour: An integrative review and research agenda. *Journal of Environmental Psychology*, 29, 309-317.
156.
Steinmetz, R., Srirattanaporn, S., Mor-Tip, J. & Seuaturien, N. (2014). Can community outreach alleviate poaching pressure and recover wildlife in South-East Asian protected areas? *Journal of Applied Ecology*, 51, 1469-1478.
157.
Stern, M.J. (2008). Coercion, voluntary compliance and protest: the role of trust and legitimacy in combating local opposition to protected areas. *Environmental Conservation*, 35.
158.
Sutinen, J.G. & Kuperan, K. (1999). A socio-economic theory of regulatory compliance. *International Journal of Social Economics*, 26, 174-193.
159.
Sutinen, J.G., Rieser, A. & Gauvin, J.R. (1990). Measuring and explaining noncompliance in federally managed fisheries. *Ocean Development & International Law*, 21, 335-372.
160.
Sutton, S. (2003). Personal and situational determinants of catch-and-release choice of freshwater anglers. *Human Dimensions of Wildlife*, 8, 109-126.
161.
Sutton, S. & Hallett, R. (1988). Understanding the effects of fear-arousing communications: the role of cognitive factors and amount of fear aroused. *Journal of Behavioral Medicine*, 11, 353-360.
162.
Team, R.D.C. (2016). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. 2014.
163.
Thomas, A.S., Milfont, T.L. & Gavin, M.C. (2016). A New Approach to Identifying the Drivers of Regulation Compliance Using Multivariate Behavioural Models. *PLoS One*, 11, e0163868.

164.
Ting-Toomey, S. (1991). Intimacy expressions in three cultures: France, Japan, and the United States. *International Journal of Intercultural Relations*, 15, 29-46.
165.
Tourangeau, R. & Yan, T. (2007). Sensitive questions in surveys. *Psychol Bull*, 133, 859-883.
166.
Turner, R.A., Addison, J., Arias, A., Bergseth, B.J., Marshall, N.A., Morrison, T.H. *et al.* (2016). Trust, confidence, and equity affect the legitimacy of natural resource governance. *Ecology and Society*, 21.
167.
Tyler, T.R. (2000). Social justice: Outcome and procedure. *International journal of psychology*, 35, 117-125.
168.
Tyler, T.R. (2006a). Psychological perspectives on legitimacy and legitimation. *Annu. Rev. Psychol.*, 57, 375-400.
169.
Tyler, T.R. (2006b). *Why people obey the law*. Princeton University Press.
170.
Tyler, T.R. (2010). *Why people cooperate: The role of social motivations*. Princeton University Press.
171.
Vira, V. & Ewing, T. (2014). Ivory's curse: the militarization and professionalization of poaching in Africa.
172.
Vollan, B. (2008). Socio-ecological explanations for crowding-out effects from economic field experiments in southern Africa. *Ecological Economics*, 67, 560-573.
173.
Von Essen, E., Hansen, H.P., Nordström Källström, H., Peterson, M.N. & Peterson, T.R. (2014). Deconstructing the poaching phenomenon: a review of typologies for understanding illegal hunting. *British Journal of Criminology*, 54, 632-651.
174.
Voyer, M., Gladstone, W. & Goodall, H. (2014). Understanding marine park opposition: the relationship between social impacts, environmental knowledge and motivation to fish. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 24, 441-462.
175.
Warner, S.L. (1965). Randomized response: A survey technique for eliminating evasive answer bias. *Journal of the American Statistical Association*, 60, 63-69.
176.
Warner, T.E. & Pomeroy, R.S. (2012). Creating compliance: A cross-sectional study of the factors associated with marine protected area outcomes. *Marine Policy*, 36, 922-932.
177.
Webley, J., McInnes, K., Texeira, D., Lawson, A. & Quinn, R. (2015). Statewide recreational fishing survey, 2013-2014. State of Queensland, Australia. Department of Agriculture, Fisheries and Forestry.
178.

White, K.M., Smith, J.R., Terry, D.J., Greenslade, J.H. & McKimmie, B.M. (2009). Social influence in the theory of planned behaviour: The role of descriptive, injunctive, and in-group norms. *British Journal of Social Psychology*, 48, 135-158.

179.
Williamson, D.H., Ceccarelli, D.M., Evans, R.D., Hill, J.K. & Russ, G.R. (2014). Derelict fishing line provides a useful proxy for estimating levels of non-compliance with no-take marine reserves. *PLoS One*, 9, e114395.

180.
Witness, G. (2017). Defenders of the earth: global killings of land and environmental land defenders in 2016.

181.
Witte, K. & Allen, M. (2000). A meta-analysis of fear appeals: Implications for effective public health campaigns. *Health education & behavior*, 27, 591-615.

182.
Wittemyer, G., Northrup, J.M., Blanc, J., Douglas-Hamilton, I., Omondi, P. & Burnham, K.P. (2014). Illegal killing for ivory drives global decline in African elephants. *Proc Natl Acad Sci U S A*, 111, 13117-13121.

183.
Yanovitzky, I., Stewart, L.P. & Lederman, L.C. (2006). Social distance, perceived drinking by peers, and alcohol use by college students. *Health communication*, 19, 1-10.

APPENDIX 1: Protected areas preserve natural behaviour of a targeted fish species on coral reefs.

Synopsis

Marine protected areas are increasingly being implemented to attain a variety of conservation and fisheries management objectives. Although rarely considered, protection of targeted species within these areas may also conserve behaviours (e.g. boldness) that are often the first removed by human exploitation. Here, I examine fish behaviour in fished, no-take/no-fishing, and no-entry management zones for a highly targeted reef fish species (coral trout; *Plectropomus leopardus*) on coral reefs in two regions of the Great Barrier Reef Marine Park, Australia. Using three behavioural metrics (flight-initiation distance, pre-flight behaviour, and escape trajectories), I demonstrate how protected areas, particularly no-entry zones, can effectively conserve naïve or bold behavioural traits in fish populations. Flight-initiation distance was consistently highest in fished zones, but the effects of protection afforded by no-fishing and no-entry reserves varied by study region. Flight-initiation distance was consistently higher for fish above the minimum legal retention size limit, except in no-entry reserves of the southern region. This indicates that no-entry reserves may be maintaining near-natural, pre-exploitation behaviour, which could have considerable implications for the genetic and social structure of a highly valuable commercial species. Conservation and fisheries management would therefore benefit from an increased understanding of how fish behaviour can influence population structures, and how these populations may be influenced by fishing and other human interactions.

Introduction

Protected areas have been used for centuries to attain a range of natural resource management outcomes, including conservation and sustainable harvest. These desired outcomes often encompass the protection of targeted or threatened species, as well as ecosystem functions or processes. Numerous studies have documented changes in animal behaviour due to human exploitation and interaction from terrestrial (de Boer et al. 2004; Thiel et al. 2007), freshwater (Sutter et al. 2012) and marine systems (Januchowski-Hartley et al. 2011), but few studies have examined what occurs in the absence of these pressures (but see Feary et al. 2011; Januchowski-Hartley et al. 2015).

In the context of marine ecosystems, it may be expected that the exclusion of extractive activities in no-take or no-entry marine reserves may lead to modifications in animal behaviour towards natural, pre-disturbance states, characterised by naïve or bold behaviour. For example, Charles Darwin documented "extreme tameness" (i.e. naivety) for birds of the Galapagos Islands in his Journal of Researches (1845), even though they had already been subject to hunting by humans, and may not have been as tame as they naturally would be (see Darwin 1845). The maintenance of, or shift towards natural, bold behaviour could have important management implications, considering that these behavioural changes may affect sexual selection (e.g. Biro & Post 2008), habitat usage (e.g. Cleveland et al. 2012), or the foraging behaviour of key species (e.g. Fortin et al. 2005; Madin et al. 2010; Rizzari et al. 2014). Thus, the relative paucity of research examining fish behaviour in the absence of human pressures constitutes a critical knowledge gap for conservation biology.

Animal behaviour can be modified substantially through interaction with humans, whether non-lethal (e.g. coexistence, tourism viewing or feeding) or lethal (e.g. hunting, fishing, or collection). An extensive literature has documented these changes in many of the world's ecosystems (reviewed by Stankowich & Blumstein 2005; Cooper & Blumstein

2015), and documents many similarities in altered behaviour of animals in terrestrial, freshwater, and marine environments. For example, the non-lethal presence of humans in terrestrial environments often results in increased flight distance for a variety of species, including birds and large-bodied ungulates (e.g. De Boer et al. 2004, Thiel et al. 2007). This trend of increased flight distance also occurs when humans hunt animals, often inducing greater changes in behaviour compared to non-lethal interactions with humans, regardless of the species or ecosystem (Jayakody et al. 2008; Guidetti et al. 2008).

Flight-initiation distance (FID) is regularly used as a behavioural measurement or proxy of fear in animals towards predators and humans (Frid & Dill 2002; Stankowich & Blumstein 2005), and is defined as the distance an animal will allow a potential predator to approach before fleeing. FID can be influenced by numerous biological and environmental factors, including habitat complexity, visibility, trophic position of the animal affected (e.g. predator vs. herbivore), and body size (Kulbicki 1998; Gotanda et al. 2009; Januchowski-Hartley et al. 2011). However, the effects of environmental or biological factors are typically of secondary importance compared to the effects of hunting or fishing (Thiel et al. 2007; Jayakody et al. 2008; Januchowski-Hartley et al. 2015), especially if these anthropogenic pressures are intense and/or sustained. As noted previously, studies of FID have occurred in most of the worlds' ecosystems, but the emphasis is often on terrestrial settings rather than aquatic environments. Documenting changes in fish behaviour due to fishing is inherently challenging, but recent years have seen an expansion of this topic, especially in coral reef ecosystems (e.g. Gotanda et al. 2009; Feary et al. 2011; Januchowski-Hartley et al. 2011, 2012, 2015).

Australia's Great Barrier Reef Marine Park (GBRMP) is a large multi-use marine park that generates gradients of fishing pressure and human interaction (Rizzari et al. 2015), making it an ideal system in which to investigate resultant changes in fish behaviour. The management system of the GBRMP includes areas open to fishing and permanent spatial closures, which comprise two different levels of protection from humans: no-fishing and no-entry reserves. Fishing is prohibited in both closure types, but no-entry reserves are strictly enforced human exclusion areas, whereas non-extractive activities (e.g. diving) are permitted in no-fishing reserves. The most heavily targeted reef fish species in the GBRMP is the common coral trout (*Plectropomus leopardus*, otherwise known as leopard coral grouper), which comprises approximately 52% of spearfishers' catch (Frisch et al. 2012; Leigh et al. 2014). Coral trout are thus an ideal study species to document changes in behaviour due to fishing pressure and varying degrees of human interaction. The aim of this study was to determine the effect of fishing and human interaction on behaviour of coral trout. Specifically, I investigated two research questions: 1) How does protection from fishing influence target species behaviour; and 2) Does fish behaviour differ between no-entry and no-fishing reserves?

Methods

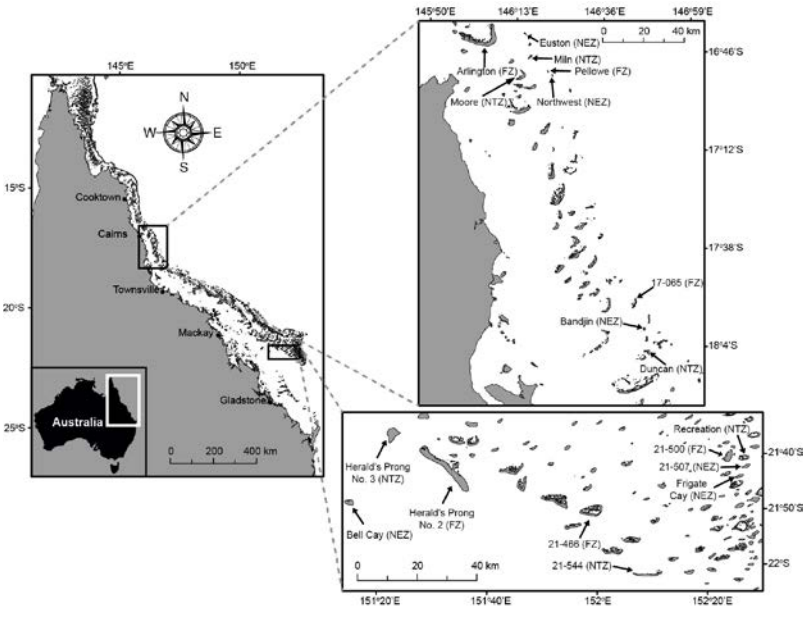

Fig. 1. Map of study sites in the Great Barrier Reef Marine Park, Australia.

Study site and design

This study was conducted on 18 outer-shelf coral reefs, in two regions (northern and southern) of the Great Barrier Reef, Australia between March and May 2014; the northern region included outer-shelf reefs of the Cairns and Innisfail management regions, while the southern region included the Swains reefs, located ~140 NM offshore of Mackay (Fig. 1). I surveyed three reefs per management zone in both regions (fished, no-fishing, and no-entry; total per region = 9, Table S1). The two regions surveyed in this study also receive different types and amounts of human pressure. For instance, although located ~ 140 NM offshore, the Swains reefs in the southern region receive considerable commercial line fishing pressure, and some charter line fishing pressure, but relatively few divers or spearfishers (Mapstone et al. 2004). The northern region reefs off Innisfail and Cairns receive less commercial line fishing pressure (Mapstone et al. 2004) but higher numbers of recreational hook and line and spear fishers, divers, and tourists due to their relatively proximity to the coast and to a major population centre (city of Cairns; pop. 150,920). I also included non-target or control species from the family Chaetodontidae, the selection of which was dependent on local abundances; *Chaetodon baronessa* was used as the non-target species in the northern region, while *Chaetodon rainfordi* was used in the southern region. Although chaetodontids have smaller body sizes and occupy a lower trophic level than *P. leopardus*, they are a locally abundant group that are unequivocally not targeted by fishers, whereas other species of similar size and trophic level are sometimes targeted by spearfishers or line fishers, or caught as bycatch (Frisch et al. 2008). Thus, chaetodontids are an appropriate control group for investigating the effects of protection from fishing on coral trout behaviour.

Flight-initiation distance

Individual fish were first identified and their size (TL) estimated visually from a horizontal distance of >8 m. Starting distance, or the distance between an observer and the study species before an experimental approach or trial is known to affect FID (Tran et al. 2015). Therefore, I used a starting distance of 8 – 10 m for all approaches, which was the minimum distance as determined by visibility. This approach does not fully control for the potential effect of alert distance (the distance at which the study species is alert of an approaching threat) on FID, but I did include visibility as a covariate to accommodate the potential influence of alert distance (i.e. alert distance likely increases with increasing visibility). Both SCUBA and free-dive methods were used to collect data, as previous research (Januchowski-Hartley et al. 2012) did not find significant differences in FID between these methods; findings that were further validated by a non-significant result of collection method on FID reported in this study (Table 1). After initial observation (~15 sec) to ensure normal fish behaviour, each fish was approached at a steady speed (~0.75 $m.s^{-1}$) by a single observer (BJB). “Flight” was defined as the moment when a fish performed a “C-start”, or swam away faster than the diver’s approach speed, as in Januchowski-Hartley et al. (2012). FID was estimated to the nearest centimetre using weighted markers deployed on the benthos in the following manner: When a fish fled, a marker was dropped directly under the observers’ head, followed by a second marker at the initial point from which the fish fled. If a fish fled to cover in the benthos (rather than to open water), a third marker was placed to also estimate the distance to cover. Fish were not approached if they were being chased, cleaned, courted, or displaying territorial or abnormal behaviour. Finally, the starting angle, or orientation of fish prior to the diver’s approach was random across treatments and is therefore unlikely to be the cause of observed differences.

Pre-flight behaviour was also recorded to provide a second measure of fish response to the diver's approach. During the approach, the observer categorized fish behaviour into one of six pre-flight categories: Approach diver (A), Crypsis (C), No change (N), Re-orientation (R), Tacking (T), or Watching (W) (for detailed descriptions, see Table S2). These pre-flight categories are thus a proxy measure that allows another comparison of wariness among zones. A third measure of behaviour recorded during the study was the escape trajectory of each fish (relative to the diver), which was estimated to the nearest 45°. While the mechanisms producing escape trajectories of fish are unknown, previous research suggests that they may correspond to the threat of predation (Domenici et al. 2011a), making them potentially useful behavioural metrics for cross-validation.

Table 1. Model averaged coefficients, standard errors, and associated P-values for flight-initiation distance (FID) of target species (Plectropomus leopardus). All estimates are relative to FID of fish in fished zones of the northern region (see Table S4). Interactions are represented with colons. P-values in bold are significant at the 0.05 level. Table S5 summarizes the top models used in averaging (all models of ΔAICc $\leq$ 2).

Variable	**Estimate**	**Std. Error**	***P*-value**
Intercept	173.306	41.382	<0.001
Rugosity	-8.657	3.401	**0.011**
Southern region	-45.829	11.931	**<0.001**
Size	1.386	0.248	**<0.001**
Visibility	1.800	0.912	**0.049**
No-take zone	-80.572	12.512	**<0.001**
No-entry zone	-91.999	14.646	**<0.001**
Tourism	-15.311	8.220	0.090
Distance to nearest fished reef	-1.279	0.881	0.191
Southern region: No-take zone	32.313	18.706	0.120
Southern region: No-entry zone	6.287	19.117	0.767
Coral cover	-0.234	0.232	0.313
Collection method	-11.695	13.838	0.443
Distance to city	0.086	0.131	0.549
Slope	-4.349	4.302	0.313
Distance to shore	-0.118	0.169	0.533

Benthic cover

Benthic habitat was measured for each corresponding measure of FID, by estimating the angular slope of the reef, rugosity, and percent coral cover for a 1 m^2 area surrounding the location from which each fish fled. This enabled a detailed, micro-level approach to control for the effect of the benthos on fish flight behaviour, which may confound the principal factors of interest (see above). Reef slope and rugosity were assessed in five categories as described in Williamson et al. (2014b): 1) reef slope 0-10°, expanses of rubble and sand; 2) reef slope <45°, bommies dispersed among mostly rubble and sand; 3) reef slope ~45°, small patches of rubble and sand among some coral structure; 4) reef slope >45°, complex coral structure, few small ledges, holes and caves; and 5) reef slope ~ 90°, high reef complexity with large over-hangs, holes and caves.

Analysis

I investigated the influence of protection (management zones) on the FID of target and non-target fish using linear mixed effects models (LMEs). A chi-squared homogeneity test was used to investigate whether each pre-flight behaviour (e.g. tacking) differed among management zones (null hypothesis = the relative frequency of each behaviour is not different among zones). To examine the influence of protection on target fish escape trajectories, I used a general additive mixed model (GAMM) with a supine cyclical spline smoother to account for the circular nature of the data. Further distribution parameters of escape trajectories were assessed using Anscombe-Glynn and Angostino tests (Anscombe & Glynn 1983; D'Agostino et al. 1990). In all models, management zone was treated as a fixed effect, whereas each behavioural or FID replicate was treated as a random effect nested within reef. Fish size (TL) was analysed as a continuous variable in all models, but categorised into two size classes for the target species (above and below minimum legal retention size, 38 cm) to aid graphical interpretation of the effect of management regime on FID.

Model selection for the LMEs was done via multi-model averaging (Burnham & Anderson 2002) based on minimization of corrected Akaike's Information Criterion (AICc), using the *dredge* function in the *MuMIn* package in R. This involves creating a global model with all possible models and combinations of factors. In this method, the smallest AICc value indicates the model of best fit, or the model supported most by the data, given the models considered. Relative support for one model is determined by calculating the differences between AICc and the smallest AICc (ΔAICc). These differences are then scaled into model weights (wAICc), which are used to calculate model-averaged coefficients with associated standard error values and P-values for each predictor variable. Selection of a subset of candidate models to be used in model averaging included all models with ΔAICc values of ≤2, as suggested by Burnham and Anderson (2004). Analyses were performed in R (R Development Core Team 2016) using the *moments, nlme, mcgv,* and *MuMIn* packages for distribution parameters, LMEs, GAMMs, and multi-model inference, respectively.

Results

Effect of management zone on flight-initiation distance

LMEs indicated that marine reserve protection significantly influenced the FID of target fish, *P. leopardus* (Table 1), but this was not evident for the behaviour of non-target species (Family: Chaetodontidae) (Fig. 2, Table S3). However, the effect of protection via management zones on coral trout behaviour varied among study regions (Fig. 2b, Table 1). In general, FID of coral trout was higher in the northern region (Table S4), which is located closer to shore and receives considerably higher levels of human usage (Table S1). In the southern region, mean FID of coral trout exhibited a step-down trend as protection increased, and was substantially different between each zone; FID was highest in fished zones (mean of 150 cm, SE=11), lower in no-fishing reserves (mean of 97 cm, SE=6), and lowest in no-entry reserves (mean of 73 cm, SE=5) (Fig. 2b). This step down trend was no apparent in the northern region; although coral trout FID was highest in fished zones (mean of 212 cm, SE=9), it was lowest in no-fishing reserves (mean of 109 cm, SE=5), instead of no-entry reserves (mean of 140 cm, SE=8) (Fig. 2b). Model averaging indicated that protection status (i.e. management zone) had a significant effect on coral trout FID, and was consistently selected as an important factor in all ranked models included in model averaging (ΔAICc values of ≤2; Table S5). Fish size and visibility were also positively related to coral trout FID in ranked models, while habitat rugosity was negatively related (Table 1). Overall, FID of non-target chaetodontids was positively related to fish size, and higher in the northern region compared to the southern region (Table S3).

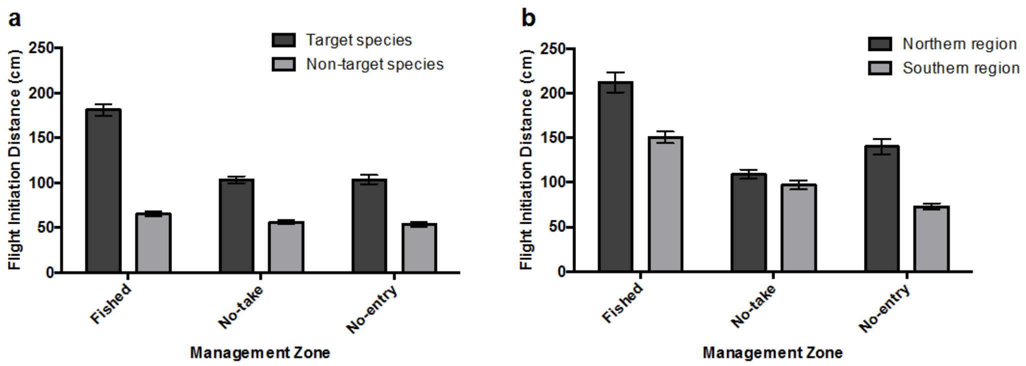

Fig. 2. Flight-initiation distance (FID) of a targeted fish species (Plectropomus leopardus) and non-target species (Chaetodontidae) in the Great Barrier Reef, Australia. a). FID of P. leopardus (target) and Chaetodon spp. (non-target) among fished, no-take, and no-entry zones. b). FID of P. leopardus in the northern and southern GBR study regions. Values represent mean FID for each species and error bars indicate standard errors.

Effect of fish size on flight-initiation distance

Fish size was positively related to FID in both target (Table 1) and non-target fish (Table S3), with the exception of target fish in no-entry reserves of the southern region (Fig. 3). For coral trout, the relationship between fish size and FID is especially apparent when comparing fish above and below the minimum legal size limit (38 cm total length); fish above the legal size limit consistently displayed greater FIDs than fish below the legal size limit in each management zone (Fig. 3). The only exception to this trend occurred in no-entry reserves of the southern region, where there was no difference in FID between size classes (Fig. 3).

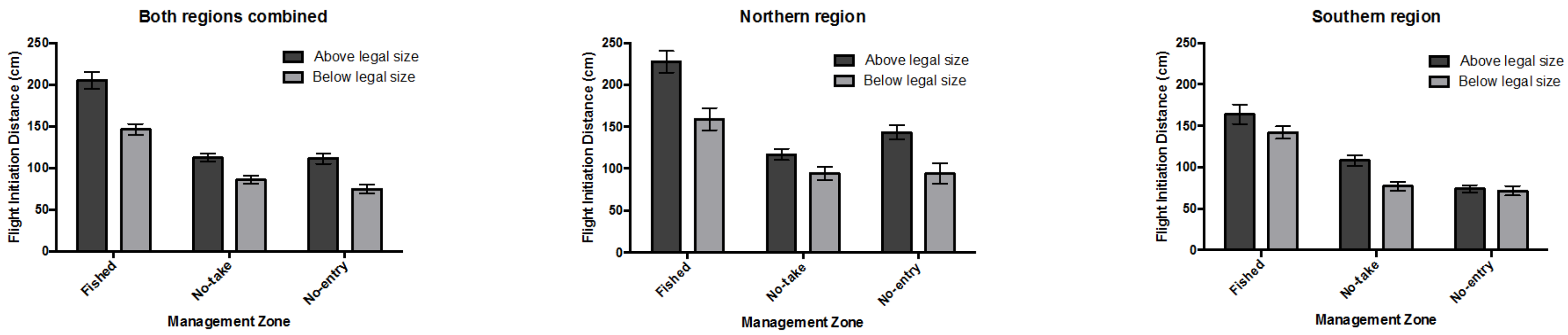

Fig. 3. Flight-initiation distance (FID) of coral trout (*Plectropomus leopardus*) between regions, size classes, and management zones. Values represent mean FID, and error bars indicate standard errors.

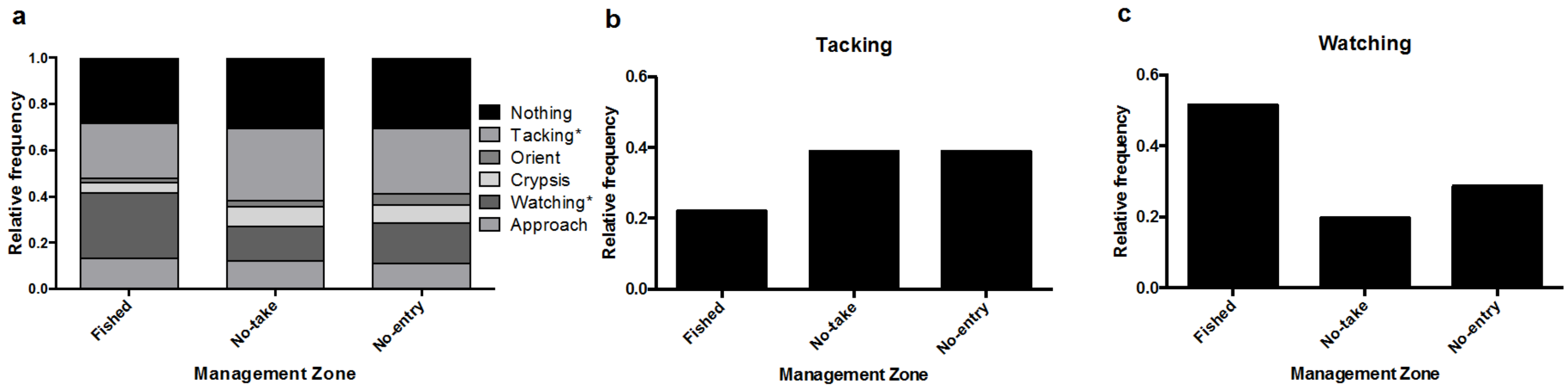

Fig. 4. Pre-flight behaviours of coral trout (*Plectropomus leopardus*) between management zones, a) relative frequency of all recorded behaviours, b) relative frequency of tacking behaviour, and c) relative frequency of watching behaviour. Behaviours with an asterisk in a) denote significant differences in behavioural occurrence between management zones, which are expanded in b) and c).

Effect of protection on pre-flight behaviour

Chi-squared tests also indicated that pre-flight behaviour of coral trout was significantly different between protected and fished zones for two types of pre-flight behaviour: tacking and watching (Table 2, Fig. 4). Tacking behaviour was almost twice as prevalent in protected zones (39% no-fishing and 39% no-entry) compared to fished zones (22%) (Fig. 4b), while watching behaviour was observed nearly twice as often in fished zones (52%) compared to no-fishing and no-entry reserves (20% and 30%, respectively) (Fig. 4c).

Table 2. Chi-squared results for pre-flight behaviours of target species (*Plectropomus leopardus*). P-values in bold indicate a significant difference for the frequency of each observed behaviour among three management zones (fished, no-take, and no-entry).

Pre-flight behaviour	**Chi-squared value**	**d.f.**	***P*-value**
Approach	1.153	2	0.562
Crypsis	1.474	2	0.479
Nothing	5.017	2	0.081
Re-orientation	1	2	0.607
Tacking	7	2	**0.03**
Watching	19.885	2	**<0.001**

Effect of protection on escape trajectory

In all management zones, coral trout fled predominantly directly away from the approaching diver (180°; Fig. 5). While GAMMs were unable to detect a significant difference (Table S6), fish in protected reserves of the southern region fled more often at lateral angles compared to those in fished zones, and lateral flight was more common in no-entry reserves compared to no-fishing reserves (Table S4, Fig. 5). Lateral flight trajectories were observed regularly for target fish in the no-entry reserves of the southern region, as indicated by a platykurtic distribution of escape trajectories (Fig. 5f), whereas all other zones had leptokurtic distributions (Table 3, Fig. 5).

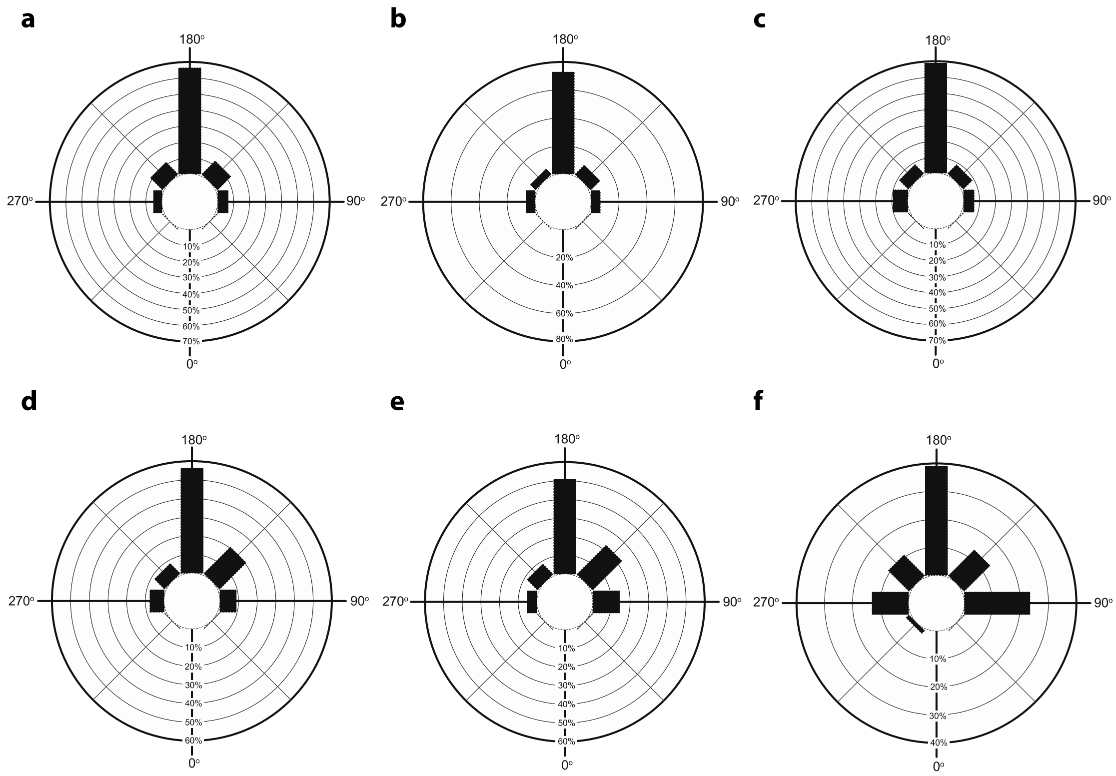

Fig. 5. Escape trajectory distributions of coral trout (*Plectropomus leopardus*) between regions and management zones. Each compass shows escape trajectories for: a) northern region fished zones, b) northern region no-fishing reserves, c) northern region no-entry reserves, d) southern region fished zones, e) southern region no-fishing reserves, and f) southern region no-entry reserves. Escape trajectories (ETs) are defined as the angle of flight, relative to the observer's approach (0°). Concentric circles represent frequency intervals; ETs have bin intervals of 45°. Note that the frequency intervals and maximum frequencies (concentric rings) are not constant for every figure, but instead are dependent on the overall distribution and frequencies of ETs for each zone.

Discussion

Effect of fishing and management zone on fish behaviour

Coral trout behaviour consistently differed between protected and fished reefs. Differences were apparent for each of three behavioural measures in this study (FID, pre-flight behaviour, and escape trajectory (ET)). FID was consistently highest on fished reefs, confirming previous research that describes increased wariness of targeted species due to human exploitation. Interestingly, FID levels were lowest on no-entry reefs in the southern region, and there was no discernible difference in behaviour between size classes (above and below legal limit). This suggests that these no-entry reserves are effectively conserving natural or near-natural “naïve” behaviour with few or any modifications due to fishing or human presence.

The relative frequency of pre-flight behaviours was significantly different between protected and fished zones for two of six behaviours measured: tacking and watching. Specifically, coral trout displayed twice the frequency of tacking behaviour on no-fishing and no-entry reefs compared to fished reefs. The only other study to examine pre-flight behaviour (Januchowski-Hartley et al. 2011) classified tacking as a wary behaviour, but our results suggest this may not be the case for *P. leopardus* on the GBR. Indeed, *P. leopardus* displayed markedly different behaviour from those previously described. However, Januchowski-Hartley et al. (2011) examined pre-flight behaviour at the family rather than the species level, and in a different geographic region (Papua New Guinea), so the reported differences in behaviour likely reflect geographic- or species-specific influences, and highlight the importance of considering context in behavioural studies. Future research examining the influence of pre-flight behaviours on fish survivorship would be beneficial for further elucidating this topic.

Targeted fish also displayed a much higher degree of vigilance in fished zones than in protected zones. Fish stopped what they were doing to watch the approaching diver almost twice as often in fished zones compared to protected no-fishing and no-entry reserves. No-fishing and no-entry reserves do generate a strong gradient in predator density (opposite to the gradient of human pressure) (Rizzari et al. 2015), which may have counteracting effects on coral trout behaviour. Thus, the increased vigilance displayed by fish in fished zones is likely indicative of modified behaviour due to human exploitation rather than natural predator densities (see also Jayakody et al. 2008; Januchowski-Hartley et al. 2011).

In the southern region, targeted fish increasingly fled at lateral ETs (relative to the diver) as the level of protection increased; lateral escape of fish was most prominent in no-entry reserves and least prominent in fished zones, with intermediate amounts of lateral flight recorded in no-fishing reserves. Previous studies have shown that animals often display preferred angles of escape (Domenici et al. 2011b), but the mechanisms producing these escape trajectories in fish are still unknown. A recent synthesis of animal escapology studies revealed that while ETs are variable, most fish prefer ETs between 90° and 180° when the threat or stimulus is positioned at 0° (Domenici et al. 2011a). These preferences may be explained by the fact that lateral ETs (~90°) allow fish to keep the threat within the sensory discriminating zone comprised by vision and the lateral line, whereas medial ETs or flight directly away (180°) would maximize distance from the threat (Domenici et al. 2011a).

Furthermore, previous research has demonstrated that ETs of 150-180° had the highest escape success when attacked by real predators (Walker et al. 2005). In the context of fishing, medial ETs (~180°) would also be advantageous for species targeted by spearfishers, because this minimizes a fish’s body profile (target size) and maximizes the chances of escape, relative to lateral ETs. Functional lateralization, or the localization of function on one side of the body or brain, may also explain the behavioural asymmetry of ETs displayed by coral trout (preference to escape at angles between 90° and 180° instead of between 180° and 270°). For instance, strongly lateralized fish often display higher escape reactivity, which is likely to afford higher escape success from predation (Dadda et al. 2010). When viewed in conjunction with the other behavioural measures in this study, my results suggest that both evolutionary syndromes (lateralization) and exposure to fishing are likely to influence ETs in coral trout. Finally, medial ETs are likely indicative of high wariness, because fish increase the distance between themselves and a potential threat as quickly as possible. Considering that fish in the southern no-entry reserves displayed the lowest prevalence of medial ETs, this lends further support to my conclusion that southern no-entry reserves are preserving natural or near-natural behaviour.

Effects of fish size and management zone on behavioural traits

I consistently found differences in FID between size classes of coral trout among management zones, with the exception of no-entry reserves in the southern region. The positive relationship between fish size and FID supports the theory of optimal fitness (Ydenberg & Dill 1986; Stankowich & Blumstein 2005; Fleming & Bateman 2015) and previous research on coral reef fishes (Gotanda et al. 2009; Januchowski-Hartley et al. 2011, 2015). One study (i.e. Feary et al. 2011) did not find an effect of fish size on FID, but this could have been due to the methodological approach of clustering fish size for most species, and a corresponding lack of resolution to examine the effect of fish size on FID.

Although the mechanisms responsible for the consistent differences in FID between size classes in protected areas are yet to be fully elucidated, suggested mechanisms and theories include: 1) selective pressure of the fishery; 2) optimal fitness theory; 3) direct movement of fishes with learned experience into no-fishing reserves; and/or 4) the social transmission of learned avoidance behaviour throughout fish populations (Kelley & Magurran 2003; Brown et al. 2006; Manassa et al. 2014; Cooper & Blumstein 2015). Yet, none of the latter three mechanisms satisfactorily explains why FID was not different between size classes of fish in no-entry reserves of the southern region. If optimal escape theory, social mechanisms, or direct movement of non-naïve fish into closed reserves were responsible for increased FID in no-fishing reserves, one would also expect to see this same pattern manifested in the no-entry reserves, especially considering the proximity of no-fishing and no-entry reserves in the southern region. In addition, if direct movement or social learning were the mechanisms affecting FID inside protected reserves, I would expect the distance to the nearest fished reef to influence my results, but LMEs did not indicate a relationship of any significance. Furthermore, coral trout rarely move between reefs (Davies 1996; Zeller et al. 2003), and instead display high site fidelity, remaining in small home ranges of ~0.5 km^2 (Bunt & Kingsford 2014; Matley et al. 2015), rather than the kilometre-plus distances that separated the reefs surveyed in this study. An alternative explanation may be that no-entry reserves in the southern region are preserving genetically influenced behavioural traits (i.e. animal personalities) and/or an absence of social learning or direct experience with fishing.

Numerous studies have documented the removal of bold or naïve species from wild populations in both terrestrial (Ciuti et al. 2012) and marine ecosystems (Biro & Post 2008; Sutter et al. 2012; Alós et al. 2015), which may generate exploitation-induced evolutionary changes (Olsen et al. 2004; Walsh et al. 2006; Uusi-Heikkilä et al. 2015). Despite these potential ramifications, few studies have considered whether protected areas can effectively conserve these behavioural traits (but see Januchoswki-Hartley et al. 2015). In the present study, two measures of flight behaviour suggest that no-entry reserves in the southern region of the GBRMP may be maintaining naïve and bold populations of target species, indicated by a lack of difference in FID between size classes and the platykurtic distribution of ETs. Thus, these southern no-entry reserves may provide regional baseline levels of this species' behaviour in the relative absence of human influences. FID of fishes was also markedly lower for unfished and isolated fish populations such as those in the Chagos wilderness area, when compared to other protected areas (Januchowski-Hartley et al. 2015). This further supports the hypothesis that human exclusion areas, or no-entry reserves, protect natural or near-natural fish behaviour to a greater extent than no-fishing reserves.

My findings demonstrate how protected areas, most notably human exclusion areas (i.e. no-entry zones), may preserve natural or near-natural behavioural traits in a population of large predatory reef fishes. However, the implications of preserved natural behaviour in a species have rarely been studied in this context (but see Biro & Post 2008; Sih et al. 2012). Previous research on assortative mating in fish populations has described how females preferentially paired with bolder males if given a chance to observe the males' behaviours towards a predator (Godin & Dugatkin 1996). Coral trout are polygynous and aggregate to spawn, wherein male individuals establish temporary breeding territories and attract females for mating through courtship displays (Samoilys & Squire 1994). Accordingly, boldness towards competitors and other predators may be beneficial for maintaining a breeding territory and for the transfer of genetic information to successive generations. Removal of these bold individuals may therefore modify both the natural social and genetic structure of coral trout, with implications that are yet unknown.

The primary focus of this study was to describe how fishing and two distinct levels of marine reserve protection influence the flight behaviour of fishes. As such, this is the first study to simultaneously examine escape trajectory, pre-flight behaviour, and flight-initiation distance in the context of marine fishing and marine protected areas. However, numerous other factors can influence flight behaviour in animals, notably indicators of physiological condition, such as sex and reproductive state (Stankowich & Blumstein 2005; Cooper & Blumstein 2015). Whilst important, I was unable to evaluate these parameters due to the non-extractive sampling design of this study. Secondly, latitudinal gradients in predation have also been described as affecting FID, whereas animals at lower latitudes display larger FIDs because of increased predation (e.g. Moller et al. 2015). I am unaware of any data suggesting any increased predation risk for coral trout on the GBR with latitude. Moreover, I am confident that any variation in predation risk would be much higher between management zones than between latitudinal gradients, and is therefore unlikely to have influenced our results. A last assumption made in this study is that fish in no-entry zones are unaffected or minimally affected by past fishing, and display natural or near-natural behaviour. My justification for this assumption is two-fold: 1) the mean years of protection for no-entry reserves in this study is 22 years, which is considerably larger than accepted estimates of coral trout longevity on the GBR (14 years; Ferreira & Russ 1994); and 2) selective pressure for “boldness” is not strong on the Great Barrier Reef compared to most other reef systems. Thus, it is unlikely that larvae recruiting to these zones are the progeny of individuals previously subjected to strong selective pressure, or are individuals that have experienced fishing by humans, unless in the form of illegal fishing, or non-compliance with no-fishing reserves.

Non-compliance with no-fishing reserves does occur within the GBRMP (Arias & Sutton 2013; Williamson et al. 2014a; GBRMPA 2015), and is believed to explain increased biomass of large predators in no-entry reserves compared to no-fishing reserves (e.g. Robbins et al. 2006; Rizzari et al. 2015). Thus, non-compliance in no-take or no-entry reserves could influence fish behaviour, and might explain the observed differences in trout behaviour between no-fishing and no-entry reserves between my two study regions. Specifically, non-compliance in the more accessible and heavily fished northern (Cairns) region could have modified fish behaviour more in northern no-fishing and no-entry reserves, than in the geographically isolated southern region. However, a detailed evaluation of non-compliance and its effects on fish behaviour are beyond the scope of this study, and any further discussion would thus be speculative. Therefore, future research on non-compliance and its effects on fish behaviour would be beneficial to determine whether fish behaviour could be used to assess fisher compliance, as suggested by Januchowski-Hartley et al. (2011, 2012).

This study illustrates how protected areas can conserve natural, or near-natural behaviour that may be indicative of pre-exploitation behaviour in a targeted species. Animal behaviour demonstrably impacts the reproductive success of individuals, and therefore shapes the genetic structure of populations. Conservation and fisheries management would therefore benefit from an increased understanding of how fish behaviour can influence population structures, and how these populations may be influenced by fishing and other human interactions.

Supplementary material

Table S1. Description of study sites, located within the Great Barrier Reef Marine Park (GBRMP). Tourism usage levels are based off GBRMPA Whitsundays management plans and communications with GBRMPA tourism department (Cairns & Innisfail reefs).

Reef	Zone	Years of protection (as of 2015)	GBRMP management sector	Distance to nearest fished reef (km)	Distance to shore (km)	Distance to urban centre (>50k people)	Tourism usage
Arlington	Fished	NA	Cairns	NA	18.5	31.4	High
Bandjin	No-entry	28	Innisfail	11.3	72	140.1	None
Bell Cay	No-entry	11	Swains	27	84.4	224.6	None
Duncan	No-take	28	Innisfail	10.9	73.6	128	Low
Euston	No-entry	32	Cairns	4.7	39.5	57.2	None
Frigate Cay	No-entry	28	Swains	4.6	190	261.2	None
Herald’s Prong 2	Fished	NA	Swains	NA	116.3	231.9	Low
Herald’s Prong 3	No-take	11	Swains	9	108.9	245.9	Low
Milln	No-take	19	Cairns	5.9	35.1	55	Moderate
Moore	No-take	26	Cairns	2.3	28.5	48.3	High
No-Name 17-065	Fished	NA	Innisfail	NA	65.9	147.2	Low
No-Name 21-466	Fished	NA	Swains	NA	144.4	233.1	Low
No-Name 21-500	Fished	NA	Swains	NA	193.8	265.7	Low
No-Name 21-507	No-entry	28	Swains	4.3	195.8	266.2	None
No-Name 21-544	No-take	35	Swains	11.0	153.8	228.6	Low
Northwest	No-entry	32	Cairns	4.3	46.6	66.7	None
Pellowe	Fished	NA	Cairns	NA	42.3	62.3	Low
Recreation	No-take	35	Swains	2.4	196	268.4	Low

Table S2. Summary of pre-flight behaviour categories.

Pre-flight behaviour	Definition
Approach	Fish approaches and inspects observer
Crypsis	Fish attempts to conceal itself and minimize profile, either through change in pigmentation or by sinking closer to the benthos without fleeing
Nothing	No change – fish fled without changing behaviour
Re-orient	Re-orienting towards refugia or towards approaching observer
Tacking	Fish halted activity and began slowly swimming away from (but not faster than) the observer in a tacking (side to side or zig zag) pattern before fleeing
Watching	Fish either stopped current activity and turned toward observer or continued to watch approaching observer if stationary and already facing observer

Table S3. a) Model averaged coefficients, standard errors, and associated *P*-values for flight-initiation distance (FID) of non-target species (f: Chaetodontidae). All estimates are relative to FID of fish in fished zones of the northern region (see Table 2). Interactions are represented with colons. *P*-values in bold are significant at the 0.05 level. b) Summary of top models for flight-initiation distance of non-target species (f: Chaetodontidae). Shown are degrees of freedom (df), model maximum log-likelihood (logLik), AICc, changes in AICc with respect to the top ranked model (ΔAICc) and AICc weights (wAICc). The top ranked model according to the lowest AICc value is in bold. Models of ΔAICc < 2 were included in model averaging to produce model-averaged coeffecients shown in a).

a)

Variable	Estimate	Std. error	*P*-value
Intercept	62.845	13.526	<0.001
Rugosity	-3.081	1.74	0.077
Southern region	-25.905	8.236	**0.005**
Size	1.21	0.525	**0.022**
Visibility	0.856	0.499	0.087
No-take zone	-9.168	7.771	0.289
No-entry zone	-13.146	8.428	0.158
Southern region:No-take zone	0.886	11.104	0.943
Southern region:No-entry zone	4.829	11.658	0.709

b)

Model	df	logLik	AICc	ΔAICc	wAICc
Rugosity+Region+Size+Visibility+Zone+Region*Zone	11	-1950.42	3923.5	0.00	0.40
Rugosity+Region+Size+Zone+Region*Zone	10	-1952.10	3924.7	1.26	0.21
Cover+Rugosity+Region+Size+Visibility+Zone+Region*Zone	12	-1951.01	3926.8	3.30	0.08

Table S4. Target species (*Plectropomus leopardus*) behaviour among management zones for three behavioural measures of wariness; flight-initiation distance (FID), pre-flight behaviour (PFB), and escape trajectory (ET). Note that the frequency of each pre-flight behaviour was computed separately, across management zones (sum of each row =100%). The prevalence of escape trajectories was also computed across zone, but in a cumulative fashion (sum of each column = 100%).

Behavioural measure	Fished zones	No-take reserves	No-entry reserves
Flight-initiation distance (FID)			
Overall mean FID (cm)	180.87	103.25	103.50
Mean FID Northern region	212.27	109.31	140.22
Mean FID Southern region	150.64	97.19	72.99
Pre-flight behaviour (PFB)			
Prevalence of tacking behaviour (n = 126 observations)	22% (n = 28)	39% (n = 49)	39% (n = 49)
Prevalence of watchful behaviour (n = 122 observations)	51% (n = 63)	20% (n = 24)	29% (n = 35)
Escape trajectory (ET)			
Medial escape trajectory (directly away from diver)	61%	61%	54%
Lateral escape trajectory	39%	39%	46%

Table S5. Summary of top models used for obtaining model-averaged coefficients (Table 1) for flight-initiation distance of target species (Plectropomus leopardus). Shown are degrees of freedom (df), model maximum log-likelihood (logLik), AICc, changes in AICc with respect to the top ranked model (ΔAICc) and AICc weights (wAICc). The top ranked model according to the lowest AICc value is in bold.

Model	df	logLik	AICc	ΔAICc	wAICc
Rugosity+Region+Size+Visibility+Zone	9	-2454.06	4926.53	0.00	0.11
Rugosity+Region+Size+Tourism+Zone	9	-2454.19	4926.79	0.26	0.09
Dist_Fished+Rugosity+Region+Size+Tourism+Zone	10	-2453.18	4926.86	0.32	0.09
Rugosity+Region+Size+Tourism+Visibility+Zone	10	-2453.41	4927.32	0.79	0.07
Rugosity+Region*Zone+Size+Visibility	11	-2452.52	4927.64	1.11	0.06
Cover+Rugosity+Region+Size+Visibility+Zone	10	-2453.61	4927.73	1.19	0.06
Rugosity+Method+Region+Size+Visibility+Zone	10	-2453.71	4927.93	1.39	0.05
Cover+Dist_fished+Rugosity+Region+Size+Tourism+Zone	11	-2452.71	4928.02	1.49	0.05
Cover+Rugosity+Region+Size+Tourism+Zone	10	-2453.83	4928.17	1.63	0.05
Dist_city+Rugosity+Region+Size+Visibility+Zone	10	-2453.84	4928.19	1.65	0.05
Rugosity+Region+Size+Slope+Visibility+Zone	10	-2453.88	4928.26	1.73	0.04
Rugosity+Region+Size+Zone	8	-2456.01	4928.35	1.82	0.04
Cover+Region+Size+Slope+Visibility+Zone	9	-2453.94	4928.39	1.85	0.04
Region+Size+Slope+Visibility+Zone	11	-2455.00	4928.41	1.87	0.04
Dist_fished+Dist_shore+Rugosity+Region+Size+Tourism+Zone	10	-2452.93	4928.46	1.93	0.04
Rugosity+Region+Size+Slope+Tourism+Zone	10	-2453.98	4928.46	1.93	0.04
Flight+Rugosity+Region+Size+Visibility+Zone	10	-2454.00	4928.51	1.98	0.04
Dist_fished+Rugosity+Region+Size+Slope+Tourism+Zone	11	-2452.95	4928.51	1.98	0.04

Table S6. Coefficients, standard errors, and associated P-values for GAMMs modelling escape trajectories of target species (*Plectropomus leopardus*). All estimates are relative to fished zones in the northern region (see Table S4). Interactions are represented with colons.

Variable	**Estimate**	**Std. error**	***P*-value**
Intercept	178.193	5.996	<0.001
Southern region	-4.382	8.427	0.567
No-take zone	0.007	8.519	0.999
No-entry zone	0.827	8.806	0.925
Southern region: No-take zone	-9.567	12.011	0.426
Southern region: No-entry zone	-2.735	12.189	0.823

References

Alós J, Palmer M, Trías P, Díaz-Gil C, Arlinghaus R. 2015. Recreational angling intensity correlates with alteration of vulnerability to fishing in a carnivorous coastal fish species. Canadian Journal of Fisheries and Aquatic Sciences **72**:217–225.

Anscombe FJ, Glynn WJ. 1983. Distribution of the kurtosis statistic b_2 for normal statistics. Biometrika **70**(1):227–234.

Arias, A, Sutton SG. 2013. Understanding recreational fishers' compliance with no-take zones in the Great Barrier Reef Marine Park. Ecology and Society **18**(4):18.

Biro PA, Post JR. 2008. Rapid depletion of genotypes with fast growth and bold personality traits from harvested fish populations. Proceedings of the National Academy of Sciences of the United States of America **105**:2919–2922.

Brown GE, Ferrari MCO, Chivers DP. 2006. Learning about danger: chemical alarm cues and threat-sensitive assessment of predation risk by fishes. Pages 59–80 in Brown C, Laland K, and Krause J, editors. Fish Cognition and Behaviour. 2nd edition. Blackwell Publishing Ltd.

Bunt CM, Kingsford MJ. 2014. Movement, habitat utilization and behaviour of coral trout *Plectropomus leopardus* during and after the reproductive period on the southern Great Barrier Reef. Marine Ecology Progress Series **496**:33–45.

Burnham KP, Anderson DR. 2002. Model selection and multimodel inference. Springer Science & Business Media.

Burnham KP, Anderson DR. 2004. Multimodel inference: Understanding AIC and BIC in model selection. Sociological Methods & Research **33**:261–304.

Caro T. 2007. Behaviour and conservation: a bridge too far? Trends in Ecology and Evolution **22**:394–400.

Ciuti S, Muhly TB, Paton DG, McDevitt AD, Musiani M, Boyce MS. 2012. Human selection of elk behavioural traits in a landscape of fear. Proceedings of the Royal Society B: Biological Sciences **279**:4407–4416.

Cleveland SM, Hebblewhite M, Thompson M, Henderson R. 2012. Linking elk movement and resource selection to hunting pressure in a heterogeneous landscape. Wildlife Society Bulletin **36**:658–668.

D'Agostino RB, Belanger A, D'Agostino RB Jr. 1990. A suggestion for using powerful and informative tests of normality. The American Statistician **44**(4):316-321.

Dadda M, Koolhaas WH, Domenici P. 2010. Behavioural asymmetry affects escape performance in a teleost fish. Biology Letters **6**:414-417.

Darwin, C. 1845. Journal of researches into the natural history and geology of the countries visited during the voyage of H.M.S. Beagle round the world, uder the command of Capt. Fitz Roy, R.N. 2nd edition. London: John Murray (The voyage of the Beagle).

Davies CR. 1996. Inter-reef movement of the common coral trout, *Plectropomus leopardus*. Report. Great Barrier Reef Marine Park Authority, Townsville, Queensland, Australia.

de Boer HY, van Breukelen L, Hootsmans MJM, van Wieren SE. 2004. Flight distance in roe deer *Capreolus capreolus* and fallow deer *Dama dama* as related to hunting and other factors. Wildlife Biology **10**:35–41.

Domenici P, Blagburn JM, Bacon JP. 2011a. Animal escapology II: escape trajectory case studies. The Journal of Experimental Biology **214**:2474–2494.

Domenici P, Blagburn JM, Bacon JP. 2011b. Animal escapology I: theoretical issues and emerging trends in escape trajectories. The Journal of Experimental Biology **214**:2463–2473.

Feary DA, Cinner JE, Graham NAJ, Januchowski-Hartley FA. 2011. Effects of customary marine closures on fish behaviour, spear-fishing success, and underwater visual surveys. Conservation Biology **25**:341–349.

Fortin D, Beyer HL, Boyce MS, Smith DW, Duchesne T, Mao JS. 2005. Wolves influence elk movements: behaviour shapes a trophic cascade in Yellowstone National Park. Ecology **86**:1320–1330.

GBRMPA. 2015. Fishing offences on the rise. Great Barrier Reef Marine Park Authority, Media Department. Published 23/09/15.

Godin JGJ, Dugatkin LA. 1996. Female mating preferences for bold males in the guppy, *Poecilia reticulate*. Proceedings of the National Academy of Sciences of the United States of America **93**:10262-10267.

Gotanda KM, Turgeon K, Kramer DL. 2009. Body size and reserve protection affect flight-initiation distance in parrotfishes. Behavioural Ecology and Sociobiology **63**:1563–1572.

Guidetti P, Vierucci E, Bussotti S. 2008. Differences in escape response of fish in protected and fished Mediterranean rocky reefs. Journal of the Marine Biological Association of the United Kingdom **88**(3):625–627.

Januchowski-Hartley FA, Graham NAJ, Cinner JE, Russ GR. 2015. Local fishing influences coral reef fish behaviour inside protected areas of the Indo-Pacific. Biological Conservation **182**:8–12.

Januchowski-Hartley FA, Graham NAJ, Feary DA, Morove T, Cinner JE. 2011. Fear of fishers: human predation explains behavioural changes in coral reef fishes. PLoS ONE **6**(8):e22761.

Januchowski-Hartley FA, Nash KL, Lawton RJ. 2012. Influence of spear guns, dive gear and observers on estimating fish flight-initiation distance on coral reefs. Marine Ecology Progress Series **469**:113–119.

Jayakody S, Sibbald AM, Gordon IJ, Lambin X. 2008. Red deer *Cervus elephus* vigilance behaviour differs with habitat and type of human disturbance. Wildlife Biology **14**:81–91.

Kelley JL, Magurran AE. 2003. Learned predator recognition and antipredator responses in fishes. Fish and Fisheries **4**:216–226.

Kulbicki M. 1998. How the acquired behaviour of commercial reef fishes may influence the results obtained from visual censuses. Journal of Experimental Marine Biology and Ecology **222**:11–30.

Leigh GM, Campbell AB, Lunow CP, O'Neill MF. 2014. Stock assessment of the Queensland east coast common coral trout (*Plectropomus leopardus*) fishery. Report. Department of Agriculture, Fisheries and Forestry, Brisbane, Queensland, Australia.

Madin EMP, Gaines SD, Warner RR. 2010. Field evidence for pervasive indirect effects of fishing on prey foraging behaviour. Ecology **91**:3563–3571.

Manassa RP, McCormick MI, Dixson DL, Ferrari MCO, Chivers DP. 2014. Social learning of predators by coral reef fish: does observer number influence acquisition of information? Behavioural Ecology and Sociobiology **68**:1237-1244.

Mapstone et al. 2004. The effects of line fishing on the Great Barrier Reef and evaluations of alternative potential management strategies. CRC Reef Research technical report No 52. CRC Reef Research Centre, Townsville, Australia.

Matley J, Heupel M, Simpfendorfer CA. 2015. Depth and space use of leopard coralgrouper *Plectropomus leopardus* using passive acoustic tracking. Marine Ecology Progress Series 521:201–216.

Olsen EM, Heino M, Lilly GR, Morgan MJ, Brattey J, Ernande B, Dieckmann U. 2004. Maturation trends indicative of rapid evolution preceded the collapse of northern cod. Nature **428**:932–935.

R Core Team. 2012. R: a language and environment for statistical computing. R Foundation for Statistical Computing. Vienna, Austria.

Rizzari JR, Bergseth BJ, Frisch AJ. 2015. Impact of conservation areas on trophic interactions between apex predators and herbivores on coral reefs. Conservation Biology **29**:418–429.

Rizzari JR, Frisch AJ, Hoey AS, McCormick MI. 2014. Not worth the risk: apex predators suppress herbivory on coral reefs. Oikos **123**:829-836.

Robbins WD, Hisano M, Connolly SR, Choat HJ. 2006. Ongoing collapse of coral-reef shark populations. Current Biology **16**:2314-2319.

Samoilys MA, Squire LC. 1994. Preliminary observations on the spawning behaviour of coral trout, *Plectropomus leopardus* (Pisces: Serranidae), on the Great Barrier Reef. Bulletin of Marine Science **54**:332–342.

Sih A, Cote J, Evans M, Fogarty S, Pruitt J. 2012. Ecological implications of behavioural syndromes. Ecology Letters **15**:278–289.

Stankowich T, Blumstein DT. 2005. Fear in animals: a meta-analysis and review of risk assessment. Proceedings of the Royal Society B **272**:2627–2634.

Sutter DAH, Suski CD, Philipp DP, Klefoth T, Wahl DH, Kersten P, Cooke SJ, Arlinghaus R. 2012. Recreational fishing selectively captures individuals with the highest fitness potential. Proceedings of the National Academy of Sciences of the United States of America **109**:20960–20965.

Thiel D, Ménoni E, Brenot J-F, Jenni L. 2007. Effects of recreation and hunting on flushing distance of Capercaillie. Journal of Wildlife Management **71**:1784–1792.

Uusi-Heikkilä S. et al. 2015. The evolutionary legacy of size-selective harvesting extends from genes to populations. Evolutionary Applications **8**:597-620.

Walker JA, Ghalambor CK, Griset OL, McKenney D, Reznick DN. 2005. Do faster starts increase the probability of evading predators? Functional Ecology **19**:808–815.

Walsh MR, Munch SB, Chiba S, Conover DO. 2006. Maladaptive changes in multiple traits caused by fishing: impediments to population recovery. Ecology Letters **9**:142–148.

Williamson DH, Ceccarelli DM, Evans RD, Jones GP, Russ GR. 2014. Habitat dynamics, marine reserve status, and the decline and recovery of coral reef fish communities. Ecology and Evolution **4**(4):337-354.

Zeller D, Stoute SL, Russ GR. 2003. Movements of reef fishes across marine reserve boundaries: Effects of manipulating a density gradient. Marine Ecology Progress Series **254**:269–280.