Coral reef fisheries of the *Mzeina* Bedu in South Sinai

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ABSTRACT

Coral reef ecosystems around the world are declining as a result of human impacts including overfishing, pollution and climate change. The coral reefs of South Sinai are significant because of their unique biodiversity, the income that they generate for Egypt through diving tourism and their importance as traditional fishing grounds for the *Mzeina* Bedu. This PhD takes a multidisciplinary approach to evaluate the status of coral reefs in South Sinai, with particular emphasis on Bedouin fishers and their fisheries. In Chapter 1, I examine the importance of culture and traditional knowledge in the successful management of coral reefs and natural resources in general. In Chapter 2, I focus on the Bedouin tribes of South Sinai to understand their history and socioeconomic status; investigating their fishing techniques and sites as well as their traditional understanding of coral reef biodiversity and ecology. Chapter 3 focusses on the ecological status of finfish populations at sites along the South Sinai coastline, evaluating the status of stocks and potential Bedouin fishery impacts. Chapter 4 examines the invertebrate fishery, which is mainly practised by Bedouin women and targets *Tridacna* clams. Chapter 5 takes a fishery-dependent approach to assess the catch of Bedouin fishers, to understand important biological and socioeconomic parameters that influence the fishery. In Chapter 6, I use statistical modelling techniques on long-term coral reef ecosystem data sets from the study area to analyse longer-term trends in the ecological status of reefs in South Sinai and likely causes for these trends. Coral reefs provide income, tourism, food and coastal protection to local communities and indigenous people throughout the tropics. The socio-cultural facets of the *Mzeina* Bedu have been inextricably connected to the reefs and associated fisheries of South Sinai for generations. However, exploited finfish and invertebrate communities have declined in both size and abundance with increased fishing pressure, resulting in ecosystem-wide impacts. The *Mzeina* themselves should be integral to any proposed fisheries monitoring or management initiatives, and technological approaches may provide useful cost-effective tools. Fisheries ecosystem-level declines have been apparent over at least the last decade and sustained monitoring is essential to ensure that the impact of management initiatives may be measured. If urgent collaborative management and enforcement actions are implemented alongside a programme to develop livelihood opportunities for the *Mzeina*, the reefs of South Sinai could return to a state that supports both the socioeconomic needs of the Bedu and continues to generate substantial tourism income.

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ABBREVIATIONS

CPUE	Catch Per Unit Effort
EEAA	Egyptian Environmental Affairs Agency
EU	European Union
FAO	Food and Agriculture Organization (United Nations)
FL	Fork Length
GAFRD	General Authority for Fish Resources Development (Egypt)
IUCN	International Union for the Conservation of Nature (World Conservation Union)
MALR	Ministry of Agriculture and Land Reclamation (Egypt)
MSEA	Ministry of State for Environmental Affairs (Egypt)
MPA	Marine Protected Area
MSY	Maximum Sustainable Yield
NCS	Nature Conservation Sector (Egypt)
NGO	Non-Governmental Organization
NGS	National Geographic Society
NOAA	National Oceanic and Atmospheric Administration (USA)
NTZ	No Take Zone
PA	Protected Area
PERSGA	Programme for the Environment of the Red Sea and Gulf of Aden
SL	Standard Length
TEK	Traditional Ecological Knowledge
TL	Total Length
UNEP-	United Nations Environment Programme-
WCMC	World Conservation Monitoring Centre
UNESCO	United Nations Educational, Scientific and Cultural Organization
UVC	Underwater Visual Census

GENERAL INTRODUCTION

Following a century of rapid material development, the world's natural environment is now under increasing threat, and the most pervasive impacts are primarily anthropogenic in origin (Ellis et al. 2010). Growing evidence and real-world changes convincingly show that humanity is driving global environmental change and has pushed us into a new geological age - the Anthropocene, which suggests that we must fundamentally alter our relationship with the planet that we inhabit (Steffen et al. 2011). Water shortages, extreme weather, deteriorating conditions for food production, ecosystem loss, ocean acidification and sea-level rise are real dangers that could threaten development and trigger humanitarian crises around the globe (Griggs et al. 2013). To date, there are substantial gaps in understanding of the intricate links between humankind and nature, hindering well-informed management decisions to protect natural resources (Schmeller et al 2017) and we currently risk driving the Earth's ecosystems onto a trajectory toward more inhospitable environmental states from which there may be no recovery (Johnson et al. 2017).

Coral reefs are among the world's most unique and vital habitats, acting as 'blue carbon' sinks and providing food and livelihood to millions worldwide (Cinner et al. 2012; Pendelton et al. 2016). However, reefs around the world are under threat, primarily from human-generated carbon dioxide emissions that have driven ocean acidification and created a chemical environment in which it is impossible for reef-building corals to develop (Geraldi et al. 2020). Unusually high temperatures as a result of global warming have resulted in depressingly regular coral bleaching events, where corals lose their symbiotic algae, leading to substantial, widespread morbidity and mortality (Hughes et al. 2017). Compounding these threats are a host of other human impacts including pollution, unsustainable tourism and overfishing (Riegl et al. 2009).

The coral reefs of the Red Sea are among the most unique in the world, and, importantly, may hold the key to resilience in the face of climate change since they naturally exist at temperature extremes (Osman et al. 2018). The reefs of South Sinai face a variety of threats many of which have originated directly and indirectly from uncontrolled tourism development (Gladstone et al. 2013). The *Mzeina* Bedu of South Sinai are a socially-marginalized group who have fished the area's reefs for generations (Lavie 1990). Their nomadic lifestyles have been disrupted by the population growth in the region (Matrahazi 2019) and they have settled in coastal towns since the 1970s, working in tourism and continuing to fish for subsistence and sale to local markets (Goodman 2013). Managing this bioculturally-diverse region's coral reefs and associated fisheries resources is thus not a simple process of counting fish. The multidisciplinary dimensions of coral reef resources in South Sinai require a complex approach to their management that takes into account the culture and socioeconomic needs of local indigenous tribal peoples, works closely with them to make use of the untapped repository of traditional ecological knowledge and ultimately holds the key to discover feasible and sustainable management options (Johannes 1981).

This PhD aims to understand the coral reef fisheries of the *Mzeina* Bedu of South Sinai, not only from a biological perspective but also from the wider socioeconomic and cultural perspectives that have, to date been inadequately considered at a global scale in artisanal fisheries management. Ultimately, the work provides potential options to improve monitoring and management of coral reefs and their associated resources that are inclusive to the *Mzeina* and thus are likely to be more widely accepted than the generally unsuccessful approaches employed to date.

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CHAPTER 1. CORAL REEF MANAGEMENT IN REGIONS OF HIGH BIOCULTURAL DIVERSITY

Abstract

Sites of high biodiversity are often also sites of high cultural significance, particularly in coastal areas. Protected areas are a tool to conserve biodiversity, but many fail because they do not adequately consider the cultural and socioeconomic needs of local human populations. Indigenous peoples often have a good understanding of ecosystem dynamics and environmental conservation and their traditional ecological knowledge can complement scientific research, often improving acceptance of management measures. Coral reefs are among the world's most biologically, socioeconomically and culturally important ecosystems, but they are under serious threat from climate change, pollution and overharvesting. Artisanal coral reef fisheries are ubiquitous in the tropics, providing subsistence and livelihood to local communities, but many are declining because of overfishing and loss of coral habitat. Most coral reef fishers live in poverty and have little socioeconomic resilience to environmental changes. Coral reefs are further threatened by unsustainable tourism and recreational activities. Various approaches have been employed to manage coral reef fisheries including gear-based, effort-based, temporal, and spatial approaches. Traditionally, reef fisheries have been managed through a top-down approach, governed by state-generated legislation. However, there is a growing acceptance that in order to be successful, management approaches must adequately consider the needs of local communities and other stakeholders. Integral to this inclusive approach is the research, understanding and inclusion of customary management into contemporary coral reef fisheries approaches. Components of customary management often mirror those of contemporary management but can be more adaptable and easily accepted by local stakeholders.

1.1. Linking biological and cultural diversity

Human societies have interacted with nature over millennia, and in many cultures these interactions are apparent through a rich history of traditional management, protected areas and sacred sites (Stevens 1997; Wild and McCeod 2008). Thus, on a global scale, sites of high biodiversity are often also sites of high cultural diversity and significance (Posey and Dutfield 1997), particularly in coastal areas (Brown and Hausner 2017). In such societies, social and religious values have frequently coevolved over centuries alongside ecological factors (Gadgil et al. 1993). It is becoming more widely accepted that the diversity of life is made up of both living organisms (biodiversity) and human beliefs, values and worldviews (cultures) (Pretty et al. 2009). The UNESCO Universal Declaration on Cultural Diversity recognizes that cultural diversity is the common heritage of humanity and is as necessary for humankind as biodiversity is for nature (UNESCO 2001). Therefore, in such cases, effective nature conservation requires sufficient consideration of the corresponding cultural perspectives (Johannes 1978; Gavin et al. 2015; Blake et al. 2017; Gee et al. 2017) and a thorough understanding of the complex relationship between the two will improve the chances of success when attempting to formulate a realistic conservation strategy for such regions.

1.2. Protected Areas: concept and purpose

Definition: A protected area (PA) is a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values (Dudley 2008).

Protected areas have been widely utilized as a tool to conserve biodiversity and natural resources since the late 19th century, and nowadays the World Conservation Union (IUCN) recognizes over 200,000 PAs, protecting over 20 million km^2 and covering approximately 15 % of the world's land and 18 % of the oceans (UNEP-WCMC, IUCN and NGS 2018).

When adequately enforced, there is little doubt that PAs are a useful and effective tool for the conservation of biodiversity (Gormley et al. 2012; Coetzee et al. 2014; Gray et al. 2016). However, PAs vary considerably in their effectiveness (Leverington et al. 2010; Edgar et al. 2014; Watson et al. 2014) and are often reported as having negative impacts on local people (Adams et al. 2004; Colchester 2004; Wilkie et al. 2006). Effective enforcement depends on funding and resources available (Hilborn et al. 2006) while local stakeholder compliance depends on the level of their support, acceptance and participation (King and Faasili 1999; Evans and Birchenough 2001; Johannes 2002; Aswani and Hamilton 2004; Poonian et al. 2008; Andrade and Rhodes 2012; Oldekop et al. 2016; Hogg et al. 2017; Zafra-Calvo et al. 2017).

The social impacts of PAs include economic, livelihood and cultural impacts, and may result directly from management policies such as hunting regulations, or indirectly through wider social and economic changes, for example, the effects of increased tourism (Holmes and Brockington 2012). Local community responses to these social impacts are influenced by socio-political contexts at both local and regional scales (Brockington et al. 2006; Nelson and Agrawal 2008). Impacts are also unevenly distributed; they are most intense at local rather than national scales, and within communities along lines of class, gender, ethnicity, and caste; commonly benefits tend to accrue to the wealthiest and most powerful, and costs fall on the weakest and poorest (Holmes 2007). Additionally, increasing attention on human rights, particularly of culturally marginalized groups, has underlined the injustices of the past, such as restrictions imposed on traditional extractive practices, or removal of indigenous populations from PAs (Dahal et al. 2014; Devine 2014).

1.3. Traditional Ecological Knowledge

Definition: Traditional ecological knowledge (TEK) is the cumulative body of knowledge, practice and belief, evolving by adaptive processes and handed down through generations by

cultural transmission about the relationship of living beings (including humans) with one another and with their environment (Berkes et al. 2000).

Indigenous peoples often retain a strong understanding of ecosystem dynamics and sustainable management practices, crafted for centuries by the specifics of completing tasks in the environment with which they have co-existed for a long time (Berkes et al. 2000; Davis and Wagner 2003). Evidence of a detailed understanding of nature and its protection dates from the ancient Egyptians (1540 BC) and is further apparent in succeeding civilizations including the ancient Greeks, Romans and Ottomans (Abd-Alah 1999). Customary management of marine resources was first recorded by contemporary researchers in Pacific island communities during the 19th century (Somerville 1897; Malinowski 1922, 1935). Knowledge may be passed systematically, but more often through observation, imitation and individual experimentation (Hagen et al. 2017). For example, if a fisher was consistently able to maximize catches, that individual would pass their techniques down to subsequent generations; whereas techniques or fishing grounds that were not fruitful would fade out of memory (Drew 2005). Furthermore, because TEK is created in an iterative fashion, it can reflect modern changes in peoples' environment or culture (Ellis and West 2004) and thus TEK systems, as well as the institutions and customary management practices that they inform, represent an historical database of long-term social-ecological adaptations to dynamics and change (Berkes and Folke 2002).

Modern-day ecologists have welcomed TEK in their attempts to understand more fully properties of ecosystems such as nonlinearity, unpredictability and complexity, and have attempted to incorporate these ideas into the management of natural resources (Berkes and Folke 1998, 2002; Gadgil et al. 2003; Grafton and Kompas 2005). Communities with a good traditional understanding of the resources they are exploiting may be more likely to cooperate with conservation measures (Braga et al. 2017), and may even take responsibility for

environmental monitoring programmes (Saunders and Xuereb 2016). Management systems that recognize customary common property regimes based on kinship, ritual, social conventions, religion and supernatural beliefs are often better respected (Frazer 1922; Gadgil and Vartak 1976; Reichel-Dolmatoff 1976; Johannes 1978, 1981, 2002; Ruddle 1988, 1993, 1998; North 1990; Chapin 1991; Gadgil and Guha 1993; Osemeobo 1994; McClanahan et al. 1997; Hviding 1998; Asafu-Adjaye 2000; Aswani 2005; Aswani et al. 2007; Clarke and Jupiter 2010). Thus, management incorporating TEK and implemented by informal institutions can result in lower enforcement, monitoring and sanction costs since they do not rely on third parties for their dissemination or enforcement (North 1990; Posner and Rasmusen 1999), which could be of particular benefit in developing countries where conservation funding is often limited. However, there are significant methodological barriers to be overcome in translating TEK-based information into the technical language of western science (Huntington 2000; Gilchrist et al. 2005; Polfus et al. 2014; Table 1.1).

Particularly since the industrial revolution in Europe and its global proliferation, TEK and its associated customary natural resource management systems have been increasingly affected by external factors that cause stresses and often lead to a radical change in systems, including their erosion and demise (Johannes 1978; Ruddle 1993; Cox 2000; Maffi 2005). This erosion of TEK is a result of a number of complex factors including the compounding influences of formal schooling and loss of local languages (McCarter and Gavin 2011; Saynes-Vásquez et al. 2016), dominant religions (Tang and Tang 2010), changes in land use (Gray et al. 2008), market integration (Godoy et al. 2005), loss of access to resources through conservation programmes (Gómez-Baggethun et al. 2012), industrialization processes (Brodt 2001; Turner and Turner 2008) and socioeconomic changes and social network structure (Henrich and Henrich 2010; Reyes-García et al. 2013). However, TEK can still persist even as societies undergo these cultural, economic and ecological changes (Byg and Balslev 2001; Olsson and

Folke 2001; Lykke et al. 2004; Pieroni et al. 2004; Barthel et al. 2010). In this way, TEK is increasingly being viewed as having a hybrid, adaptive and dynamic nature, accommodating new forms of knowledge and discounting components as they become obsolete (Gómez-Baggethun and Reyes-García 2013).

Table 1.1. Comparison of Traditional and Scientific Ecological Knowledge

Traditional Ecological Knowledge	***Scientific Ecological Knowledge***
• qualitative	• quantitative
• intuitive	• purely rational
• holistic	• reductionist
• mind and matter considered together	• mind and matter considered apart
• moral	• value free
• based on empirical observations and accumulation of facts by trial and error	• systematic, experimental
• data generated by resource users themselves	• led by researchers
• based on long-time series information on one locality	• relatively short-time series

(Berkes 1993)

1.4. Coral reefs: global importance and current status

Coral reefs cover 255,000 km^2 of the Earth's surface and are among the most productive and biodiverse of the world's ecosystems (Crossland et al. 1991; Spalding and Grenfell 1997; Knowlton et al. 2010). Reefs supply goods and services including seafood, recreation, and coastal protection as well as aesthetic and cultural benefits to around 500 million of the world's poorest people (Moberg and Folke 1999; Donner and Potere 2007). Up to three

million species including almost a third of the world's marine fish species are found on coral reefs (McAllister 1991; Small et al. 1998) and around six million fishers rely on reef fisheries for their livelihoods (Teh et al. 2013). These complex ecosystems pose special problems for their sustainable use due to the diversity of species and the complexity of trophic interactions and the extended time scales over which degradation may manifest itself (McClanahan 1995; Hughes et al. 2010; McClanahan et al. 2011).

Coral reefs have been estimated to provide worldwide benefits and services to the value of US$ 29.8 billion every year, notably tourism (US$ 9.6 billion) and fisheries (US$ 5.7 billion) (Spurgeon 1992; Cesar et al. 2003). However, these activities have increasingly caused degradation on many reefs throughout the world (McManus 1997; Davenport and Davenport 2006; Brodie and Pearson 2016). People benefit directly or indirectly from coastal activities through food provision, artisanal fishing opportunities, natural products, carbon storage, coastal protection, tourism and recreation, coastal livelihoods and economies, clean waters and biodiversity (Moberg and Folke 1999; Donner and Potere 2007; Woodhead et al. 2019). Moreover, coral reefs provide non-material, cultural benefits such as spiritual interactions, inspirational experiences, cultural identity, sense of place, bequest and existence values (Rodrigues et al. 2017).

Assessments of the global area of coral reefs have suggested that at least 19 % has already been lost, and approximately 75 % of remaining reefs have been rated as threatened when local threats are combined with thermal stress (Carpenter et al. 2008; Wilkinson 2008; Burke et al. 2011). This serious decline of coral reefs is primarily because of over-harvesting (Jackson et al. 2001; Pandolfi et al. 2003; McClanahan et al. 2011; Morais et al. 2020), pollution (Maina et al. 2013; Mallela et al. 2013; Kroon et al. 2014), disease (Harvell et al. 2002; Rosenberg and Kushmaro 2011, Williams et al. 2020) and climate change (Hughes et

al. 2003; Hoegh-Guldberg et al. 2007; Glynn 2012; Cinner et al. 2013; Ricke et al. 2013; Hughes et al. 2019); for reviews, see Bellwood et al. 2004; Gross 2013.

1.5. Coral reef fisheries

Definition: 'Small scale' and 'artisanal' fisheries: although these two terms are frequently used interchangeably in the literature, there is a subtle difference of emphasis. Whereas artisanal fishing implies a degree of simplicity or tradition (artisanality) in the chosen fishing method (e.g., simple traps, poisons, harpoons, lures and, at the most basic level, hand collection), the term small-scale fisheries typically implies nothing about gear or methods, focusing almost exclusively on the size of the fishing unit/operation—in practical terms, often operationalized in terms of the size of the boat (Batista et al. 2014).

Marine fishes are the last remaining animal resources still taken in huge quantities from the wild (Pauly 2008). The shallow nearshore nature of coral reefs precludes their exploitation by large industrial fishing vessels, so these fisheries are usually the domain of small-scale and/or artisanal fishers (Dalzell et al. 1996; McManus 1997; Pauly 2006). Reef fisheries are ubiquitous along tropical coastlines throughout the world, mainly exploited by individual households and local communities to fulfil subsistence requirements and often to yield a modest marketable surplus (Ruddle 1996). The immense biodiversity of coral reefs and thus the variety of targeted species results in most reef-associated fisheries being categorized as multi-species and multi-gear, particularly in developing countries where there is often no such thing as a 'trash fish' (Munro 1996).

Expanding artisanal fisheries are a primary cause of reef fish population decline and loss of coral reef structure, function and resilience (McClanahan 2002; Newton et al. 2007). Their maximum sustainable yield is easily exceeded, even with low fisher density and primitive gear (Jackson et al. 2001; Wing and Wing 2001; Pauly et al. 2002; Pandolfi et al. 2003; Hardt

2009) especially when combined with other impacts (McClanahan et al. 2014). As yields have fallen, fishers have resorted to more and more efficient tactics that ensure a greater catch per unit effort, including dynamite, poisons and drive-net fishing (Silva 2006; Wells 2009; Muallil et al. 2013). The use of dynamite, particularly in Asia, has transformed reefs into lifeless rubble, further reducing reef productivity and locking fishers into a downward spiral of declining catches (Riegl and Luke 1999; Riegl 2001; Fox et al. 2005). Reef fisheries have a disheartening management record; in the most heavily populated regions of the world, overwhelming poverty has depleted resources to incredibly low levels (Munro 1996; McManus 1997; McClanahan et al. 2008). There exist notable examples where artisanal fisheries are sustainable (Dalzell 1998; Craig et al. 2008; Cinner 2011; Martin et al. 2017) and declines have been successfully reversed through effective management policies (Williamson et al. 2004; McClanahan and Hicks 2011; Erisman et al. 2017). However, the management of artisanal coral reef fisheries remains intrinsically socially and ecologically complex (Aswani et al. 2007; Cinner et al. 2009b). Furthermore, it is expected that coral mortality due to climate change will further reduce sustainable harvest levels for impacted reefs (Westmacott et al. 2000; Hoegh-Guldberg et al. 2007; Pratchett et al. 2008; Robinson et al. 2019).

1.6. Coral reef tourism

Coastal development can occur rapidly in areas experiencing a sudden spurt in tourism popularity, and inadequate planning can lead to the uncontrolled discharge of raw sewage and resultant eutrophication which supports excessive algal growth on reefs (Baker et al. 2013; Naumann et al. 2015; Gil et al. 2015; Gohar and Kondolf 2016; Mwaura et al. 2017). Solid waste such as plastics can cause further aesthetic damage (Williams et al. 2016; Krelling et al. 2017) and kill turtles and seabirds (Santos et al. 2015). The damage caused by recreational scuba divers and snorkellers touching coral reefs accidentally with their hands, fins and other

equipment has also been widely recognized (Hawkins and Roberts 1992; Rouphael and Inglis 1997; Zakai and Chadwick-Furman 2002; Hasler and Ott 2008; Poonian et al. 2010; Zhang et al. 2016; Renfro and Chadwick 2017). At high densities, the impact of divers can cause measurable degradation of reef habitat and carrying capacities of 4,000–7,000 divers site^{-1} year^{-1} have been proposed (Dixon et al. 1993; Hawkins and Roberts 1997; Schleyer and Tomalin 2000). Dive tourism can also cause further damage through inconsiderate anchoring (Giglio et al. 2017), standing on and trampling reefs (Leujak and Ormond 2008) and potentially behavioural modification of megafauna (Shackley 1999; Hayes et al. 2017).

1.7. Socioeconomic issues affecting reefs

Globally, of those people who rely on coral reefs for their livelihood or subsistence, a significant proportion lives in poverty (Whittingham et al. 2003). The entry cost of artisanal reef fisheries is very low because fishing sites can be accessed on foot or using simple boats with primitive gears (Belton and Thilsted 2014). Thus they provide multiple livelihood opportunities for poorer households, but this means that there is no real economic constraint on fishing effort (Munro 1996). In many tropical coastal communities, fishing and other extractive activities on reefs may be the primary or only source of income (Hughes et al. 2012), and poorer people often migrate to the coast in search of employment in the tourism and fishing sectors (Curran et al. 2002; Small and Nicholls 2003). Small-scale fishers are often 'part-timers' who participate in a range of livelihood activities, driven by seasonality or local demand (Brugère et al. 2008; Cinner et al. 2009a; Cinner 2014).

Coastal communities are vulnerable to environmental change, instability and natural disasters, and fishers are exposed to high degrees of risk and uncertainty in terms of personal safety as well as income (Andersson and Ngazi 1998; Dietz et al. 2003; Adger et al. 2005; Ferrol-Schulte et al. 2013; Hamza et al. 2017). Fishery-based livelihoods are also highly vulnerable because fisheries are an open-access resource: ownership of fishing grounds is

complex and there is a general lack of clarity in resource tenure (Berkes 2006; Mascia and Claus 2009). Thus reef fishery-dependent communities often rely on a suite of livelihoods to cope with this stochasticity (Cinner 2014). It can be difficult for fishers to adopt alternative livelihoods because of insufficient capital assets, together with the fact that fishers often highly value their social status, cultural traditions and adventurous lifestyle (Cinner et al. 2009a, 2009b; Cinner 2014; Satumanatpan and Pollnac 2017). Physical isolation can also restrict livelihood options and increase dependence on natural resources, especially for small island communities (Pomeroy et al. 2006; Bunce et al. 2009; Kramer et al. 2009). Poorer households often have a greater reliance on fishing as a primary source of income (Béné 2003; Cinner et al. 2010), but it is unclear whether the poor turn to fishing because they lack alternatives, or whether Malthusian overexploitation (Pauly 1990) is the precursor to poverty (Cinner et al. 2009b).

1.8. Reef fisheries management approaches

Any approach to fisheries management must inherently reduce fishing mortality or 'size at first capture', and thus prevent or limit growth or recruitment overfishing (Medley et al. 1993). The single-species biological yield approach that has dominated fisheries research since the 1950s cannot effectively be applied to multispecies reef fisheries, and so an adaptive ecosystem-based approach is essential (Hall 1999; Andrew et al. 2007). Most coral reefs are located in developing countries (McClanahan 1999; Donner and Potere 2007) and methods are thus required to cope with the limited resources available in such regions. Management regimes must be straightforward and easy to enforce (Medley et al. 1993), and approaches developed for industrial and commercial fishing (Beddington and Rettig 1983) are therefore generally inappropriate. Many coastal communities are so highly dependent on coral reef resources and face such significant socioeconomic challenges that extreme limitations on exploitation may pose unrealistic burdens and thus receive little support or

compliance (Sievanen et al. 2005; McClanahan et al. 2006, 2008; Cinner et al. 2009a; McClanahan 2010). Furthermore, since reef fish populations are strongly dependent on habitat quality, management approaches should take into account the threats of coral bleaching, disease and physical damage (Roberts 1995; McClanahan and Arthur 2001; Friedlander et al. 2003; Graham et al. 2007). Resource-user conflicts between fishers and other stakeholders such as tourists must also be considered (Oracion et al. 2005; Tuda et al. 2014). In many cases, a combination of approaches will be required for successful reef fisheries management, and there is a need to evaluate existing approaches empirically and explore alternatives, particularly in developing nations (Mumby and Steneck 2008).

Gear-based Approaches

Specific gears used by artisanal fishers differentially target fish sizes, species and thus functional groups (Dalzell 1996; Sluka and Sullivan 1998; Rueda and Defeo 2003; McClanahan and Mangi 2004; McClanahan and Cinner 2008; Cinner et al. 2009c; Davies et al. 2009). Limiting type or specifications (normally mesh size) of fishing gear alters the catchability of individual species and sizes, adjusting fishing mortalities and potentially increasing sustainable yield in multispecies fisheries (Medley et al. 1993). Elimination of destructive fishing methods such as dynamite and poison is particularly important but if alternative methods of equal efficiency are not available, enforcement may be difficult (Wells 2009; Javaid et al. 2017). Where coral and associated habitats need protection or restoration, managers should consider reducing or banning the use of nets (Mangi and Roberts 2006).

Fishers are often more supportive of restrictions on specific types of fishing gear compared to blanket closures of fishing areas (McClanahan et al 2005). Artisanal fishers often employ a range of fishing gears, and consequently gear-based regulations are less likely to threaten their livelihoods than full fishery closures, as long as their alternative gears are still permitted

(Cinner et al. 2009c). A key trade-off is gear profitability, which is influenced by the catch per unit effort, the value of the species targeted and the maintenance or operational costs (McClanahan 2010). Fishers are unlikely to adopt gears with reduced profitability without compensation. Thus, in some contexts, certain gear restrictions may be difficult to enforce and fishers' preference for them may be in part because they can be easy to circumvent (Cinner et al. 2009c). Enforcement can also be complicated because of the absence of clearly defined landing sites and fishing grounds in most artisanal reef fisheries (Medley et al. 1993).

Effort-based approaches

Limited entry and quotas are potential approaches to restrict the number of boats, fishers or catch landed. However, such approaches are difficult to enforce in reef fisheries, where large numbers of species are being taken with a variety of gears and landed at numerous points in space and time (Ruddle 2014). Lack of biological data may make it impossible to set quotas with any accuracy and quotas may be inappropriate where gears are unselective, because fishing may have to cease once the first quota is reached, even when other species remain virtually unexploited (Medley et al. 1993). Thus effort-based approaches are rarely used for coral reef fisheries in developing countries, but have been employed, for example, in Australia (Little et al. 2011) and the USA (Ault et al. 2005), or in fisheries targeting high-value invertebrates such as lobster or conch (Aiken et al. 1999).

Temporal approaches

Seasonal closures have been employed as a fisheries management tool for some time (Hilborn et al. 2006) by providing protection during vulnerable periods of the life history, such as spawning or migration (Guénette et al. 1998). This approach is limited by the multispecies nature of reef fisheries and difficulty of enforcement, but temporal restrictions have been successfully employed to protect spawning aggregations (Beets and Friedlander

1999; Pears et al. 2007) and have been widely used in reef invertebrate fisheries (Smith and Berkes 1991; Theile 2001; Cox and Hunt 2005; Oliver et al. 2015) because of their easier-studied, more predictable reproductive life histories.

Spatial approaches

Marine Protected Areas (MPAs) are a common prescription for ailing reefs (Roberts and Polunin 1993) and have been shown to be effective in both reducing overfishing and maintaining coral habitat (Selig and Bruno 2010). Marine Protected Areas can comprise areas where any of the previously mentioned measures apply, or they may be strict no-take zones where any exploitation of marine resources is prohibited. No-take areas reduce the risks associated with any ambiguity in ecological data, environmental stochasticity, and inaccuracies in measurement and control of fishing effort and catch (Lauck et al. 1998; Mangel 2000; Grafton and Kompas 2005; Grafton et al. 2006). Spillover from MPAs of larvae or adult fish can also increase catches in fished areas (Russ and Alcala 1996; Roberts et al. 2001; Fidler et al. 2017). The benefits of MPAs can depend on several factors including the types of fishes targeted, the relative size of the MPA and its position relative to unprotected areas to which larvae may disperse (DeMartini 1993; Halpern 2003; Krueck et al. 2017).

1.9. Co-management approaches and indigenous fisheries

Fisheries management in most of the world has conventionally been directed by national legislation, because of the complications involved in the sustainable management of common property such as oceans (Hardin 1968; Berkes 1985). However, this ‘top-down’ approach is becoming disfavoured as it is increasingly recognized that indigenous peoples have particular rights pertaining to their access to and participation in fisheries (Nielsen and Vedsmand 1999; Nielsen et al. 2004; Maxwell et al. 2020). The combination of a top-down approach and a

narrow focus on resource issues has led to management being based on formal biological science produced in specialized research institutions, disregarding or even being in direct contrast to the experiences of fishers. Until fairly recently, coral reef management was evaluated from a mainly biological perspective, focussing on, for example, increased coral cover or fish yields, with approaches meeting these narrowly-defined goals announced as 'successes' (Selig and Bruno 2010; Gill et al. 2017). However, these same approaches may, in fact, be social 'failures' when social evaluation criteria are applied (Christie 2004; Hargreaves-Allen et al. 2017). This is because modern fisheries management fails to address the core concerns of fishing communities, is insensitive to local conditions, lacks backing from fishing communities and is even inefficient in achieving its own objectives (Nielsen et al. 2004; Ban and Frid 2018; Bennett et al. 2018). Gear restrictions, no-take zones, closed seasons and other components of modern reef fisheries management are supposed to raise household incomes from reef fisheries and help prevent habitat degradation through the limitation of fishing activity (Hilborn et al. 2004). However, artisanal fishers are often highly reliant on reef fisheries and aim to obtain fresh fish to feed their families or sell for cash, and they often complain that management efforts do not function as promised (Poonian et al. 2008; Giakoumi et al. 2018).

It is now widely recognized that quite often MPAs represent an optimization of trade-offs between biodiversity benefits and socioeconomic values, and attempting biological conservation without addressing socioeconomic complexities is liable to be met with limited success (Mascia 2003; Cinner 2007). Community support for MPA regulations is thus seen as a key factor in achieving successful biological and socioeconomic outcomes (Pollnac et al. 2001). Recognition of fishing rights represents an affirmation of the unique political status of indigenous peoples, as well as a material opportunity that they can mobilize to improve their commonly abysmal employment, income and social conditions. However, recognizing and

incorporating indigenous fishing rights into management policy can be complicated by the often negative reaction of non-indigenous fishers, who view the approach as unfair (Davis and Jentoft 2001). Co-management agreements between indigenous people, other stakeholders and state agencies offer substantial promise as a way of dealing with natural resource-based conflicts (Cinner et al. 2016; Austin et al. 2017; Ban and Frid 2018). These collaborative arrangements for managing natural resources can foster a sense of community empowerment because local stakeholders participate meaningfully in decision-making and benefit-sharing (Castro and Nielsen 2001; Granek and Brown 2005).

1.10. Customary reef management

Tropical coastal ecosystems are dynamic, complex and heterogeneous (Jones and Syms 1998; Almany 2004; Salas and Gaertner 2004; Holstein et al. 2014). Fishers who hunt in these waters rely on an expert level of ecological knowledge, not only of the fishes themselves but also of the necessary gears, vessels and techniques required to catch them most efficiently, which depend on season, habitat characteristics and other dynamic factors (Johannes 1981; Ruddle 1994; Aswani and Hamilton 2004). For example, fishers often use gears precisely adapted to the season or moon phase, as well as to other aspects of a complex and cyclically changing biological and physical environment to target particular species and often capitalize on a sophisticated body of TEK (Johannes 1981). Their qualitative perceptions and knowledge about species and places, and their skills can be utilized to identify and understand key ecological events and long-term processes on coral reefs (Hamilton et al. 2012; Lopes et al. 2018). In addition, their feelings, attitudes and preferences about environmental issues can provide information from which to develop effective management strategies (Skubel et al. 2019).

Customary management approaches have employed temporal, spatial, gear and species restrictions similar to contemporary resource management approaches (Johannes 1978;

Colding and Folke 2001; Table 1.2). Life history-based customs often serve resource management ends: for example, size restrictions exist for slow-moving or sessile marine species particularly susceptible to overharvesting (Johannes 1978). Life history-based customs on finfish are rare (Johannes 1978), maybe because other conservation measures (e.g. closed seasons, and habitat and method taboos) are more effective in sustaining highly mobile populations of finfish. Customs may also relate to the morphological appearance and behaviour of species (Zann 1985; Begossi 1992).

In Palau, traditional reef and lagoon tenure, and resource and habitat taboos, declined significantly as a result of colonial modernization (Johannes 1978), but ultimately were revitalized, resulting in an improvement of ecological conditions on community-managed reefs (Johannes 2002). This renaissance of TEK and customary management of coral-reef management has brought promising results, with informants becoming collaborators in applied projects rather than just subjects of anthropological inquiry (Sillitoe 1998). There is now a strong focus on the integration of such practices into conservation planning (Johannes 1982; Zaan 1985; King and Faasili 1999; Glaesel 2000; Colding and Folke 2001; Davis and Wagner 2003; Cinner and Aswani 2007) since the management of such complex adaptive systems can benefit from the combination of different knowledge systems (McLain and Lee 1996; Johannes 1998; Holling 2001; Berkes and Jolly 2002; Gadgil et al. 2003). The adaptability of traditional management techniques to changing seasons in extreme environments may even provide the key to securing natural resources in the face of future climate change (Leonard et al. 2013; Ksenofontov et al. 2017).

Table 1.2. Customary reef fishery management approaches

Types of customary management	Description	Analog in modern fisheries management	Differences to modern fisheries management	Examples
Spatial	Areas closed to fishing. These can be temporary (i.e. closed for several months to provide supplies of fish for a feast) or permanent (where spirits reside)	Marine protected areas, temporary fisheries closures	Often temporary and almost always harvested. Maybe reactive to events (e.g. death in the village, declining catch)	Polunin 1984; McClanahan et al. 1997
Temporal	Restricting fishing/harvesting activities during specific days, weeks, months, etc. Often short in duration (e.g. Sabbath), species-specific (e.g. *Trochus spp.*), and around a specified event (e.g. spawning aggregation)	Closed seasons	Dates may be highly flexible and reactive to events (e.g. price fluctuations for commercial species, spawning aggregations) rather than set dates.	Johannes 1978; Thorburn 2001
Gear	Prohibiting/restricting certain harvesting technologies or techniques	Gear prohibitions	Maybe inherited rights to use certain gears. Often exclude non-owners	Johannes 1981; Cinner 2005
Effort	Limiting who can harvest certain species, use certain gears, fish certain areas, etc	Permitting	Often based on initiation rights, lineage, class, or gender	Veitayaki 1994; Mantjoro 1996
Species	Prohibiting the consumption of certain species. Often lineage-related dietary restrictions	Species-specific bans	Often the species may be caught or killed, but not eaten.	Carrier 1987; Hickey 2006
Catch	Restricting the quantity of harvest. Often social norms that stress the avoidance of waste. Very rare	Total allowable catch, quotas	Quotas not set *a priori*	Johannes 1981; Teulières 1992

Adapted from Cinner and Aswani 2007

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CHAPTER 2. SOCIOECONOMIC HISTORY OF THE *MZEINA* AND ITS INFLUENCE ON THEIR REEF FISHERIES

Abstract

Unconventional data sources have proven useful to fill knowledge gaps in historical baselines for marine ecosystems, including the traditional ecological knowledge (TEK) of older community members. Tribal fisheries in the Middle East are poorly studied, although evidence suggests that fish stocks have declined across the region over the last 40 years. This work aimed to explore the traditional fishing practices of the *Mzeina* Bedu of South Sinai, using TEK and *in situ* observation. The territory of the *Mzeina* Bedouin tribe extends along the South Sinai coastline, and they have traditionally spent the winter fishing and gathering dates along the coastal plain. Nowadays, they have settled permanently in coastal towns such as Dahab, resulting in substantial socioeconomic and cultural changes. The region hosts coral reefs and internationally-significant marine biodiversity. However, these natural resources are threatened by various impacts including uncontrolled tourism, pollution and overfishing. The *Mzeina* have recognized that fisheries stocks have declined in recent decades, and attribute this decline to the increase in demand from tourism and migration to the coast. This decline in fisheries has resulted in a change in gear from lines that target large piscivores in deeper water to various nets of finer and finer mesh to target schooling herbivores on the reef flat. The *Mzeina* possess a detailed understanding of fishing techniques, seasonal movements, and spawning patterns which should be integrated into fisheries management planning. Marine protected areas (MPAs) have been established since the Israeli Occupation in the 1970s; Ras Mohammed National Park has been widely recognized as a success, however, other MPAs such as Nabq and Abu Gallum, which overlap with *Mzeina* traditional fishing grounds, have proven less effective. Challenges facing these management efforts include inadequate funding, poor enforcement, and a lack of involvement of local stakeholders in decision-making and uncontrolled tourism activities.

2.1. Introduction

Modern marine conservation problems are urgent and oceans are notoriously difficult to study, particularly in the developing world where local research and management capacity can be limited, thus any useful sources of information on marine ecosystems should be seriously considered to facilitate their effective management (Hixon et al. 2001; Worm and Branch 2012). Degradation of coral reef ecosystems began centuries ago and reefs were no longer 'pristine' well before the advent of scientific literature or living memory (Jackson 1997; Pandolfi et al. 2003; Knowlton and Jackson 2008). *In situ* study of reefs only became a possibility relatively recently when modern SCUBA diving equipment was developed in the 1950s (Marx 1990). Individual perceptions of environmental baselines are often based on those that existed at the beginning of that individual's career experience and thus are constantly shifting with each new generation (Ainsworth et al. 2008; Pinnegar and Engelhard 2008; Lotze and Worm 2009; Turvey et al. 2010). However, anecdotal data from previous generations, passed orally and through historical literature are frequently dismissed even though they are potentially useful where formal analysis approaches are limited (Pauly 1995; Sáenz–Arroyo et al. 2005; Parsons et al. 2009).

Unconventional sources of data have been successfully employed in the past to evaluate historical baselines for marine ecosystems including narrative and archival documents (McClenachan et al. 2006; Romero et al. 2002), historical photographs (McClenachan 2009), early survey and monitoring records (Baum and Myers 2004; Jones et al. 2016), interviews with fishers and other resource users (Ames 2004), and zooarchaeological remains (Beech 2001; McNiven and Bedingfield 2008). About 40 % of the world's 6,000 to 8,000 indigenous peoples have homelands, territories and nations that encompass coastal ocean and island regions (Nietschmann 1997) and their traditional ecological knowledge (TEK) can provide an especially valuable tool to estimate historical environmental baselines, for example, in coral

reef fisheries, where empirical scientific data are often unavailable (García-Quijano 2007; Bunce et al. 2008; De Mitcheson et al. 2008; Eckert et al. 2017; Purcell et al. 2017). Traditional ecological knowledge research may be divided into three major approaches (Stepp 2005): (1) economic (how human society uses plants and animals), (2) cognitive (how humans know and conceptualize plants and animals) and (3) ecological (how human society interacts with plants and animals). Important aspects of fisheries management that can be clarified using TEK include: (1) the location of fish aggregation areas and explanations of aggregating behaviour; (2) the behaviour of various species in relation to time, tide, lunar phase and depth and (3) the use of appropriate fishing methods to capture particular species at particular times and places (Sabetian 2002).

Research methods employed to elucidate TEK have included participatory mapping (Lauer and Aswani 2008; Thiault et al. 2017), inventories of local biodiversity using native taxonomies (Berlin 1992) and interviews of key informants or experts to evaluate management approaches and socioeconomic and cultural values (Bunce et al. 2000; Newing 2010; Martin et al. 2012). The traditional systems that describe climatic phenomena, habitat composition, and biotic taxonomies often conveniently correspond with western scientific categories (Brookfield and Hart 1971). Moreover, such approaches can build an understanding of cultural and socioeconomic factors that may improve the success of management actions (Fiske 1992; Ban et al. 2009).

Traditional and tribal marine fisheries in the Middle East and North Africa are poorly studied, although they appear to be widespread, with examples in almost every country with a coastline including the United Arab Emirates (Wilkinson 1983; Hawker et al. 2005; Suwaed 2015), Oman (Webster 1991; Lancaster and Lancaster 1995; Al-Rashdi et al. 2007), Iran (Taheri 2009), Saudi Arabia (Wright 1838; Al-Haratani 1997), Yemen (Van Rensburg 2017), Sudan (Cameron 1887; Eltayeb 2004; Agius 2012a), Egypt (Ashworth and Ormond 2005)

and Mauritania (Campredon and Cuq 2001). Before the advent of the global oil industry, seafaring was of great cultural importance in the Arab world and fish was the principal source of protein (Feidi 1998; Agius 2012b; Agius et al. 2016). A significant proportion of the region's fisheries are artisanal, however, data on stock status and impacts of exploitation are limited (Bruckner et al. 2011; Sale et al. 2011; Grandcourt 2012; Tesfamichael and Pauly 2016; Jabado and Spaet 2017). The majority of the Red Sea's coastline belongs to five countries: Egypt, Sudan, Eritrea, Saudi Arabia and Yemen and c. 5 million live within this highly arid, sparsely populated area (Hinrichsen 2016). Red Sea reefs have supported artisanal fisheries for thousands of years and historically many of these fisheries were regarded as sustainable or even 'underexploited', more due to low population densities and logistical barriers rather than because of deliberate conservation efforts (Head 1987; Behairy et al. 1992; PERSGA 2001).

Artisanal fishers on Red Sea coral reefs use a variety of gears and catch composition and volume varies by fishing method and season (Bruckner et al. 2011; Tesfamichael and Pauly 2016). Despite the large variety of species landed, the bulk of the catch is composed of a small number of species (Sheppard et al. 1992; Bruckner et al. 2011). Reef fisheries throughout the Red Sea have historically targeted predatory piscivores (Jin et al. 2012), although total productivity of these stocks is relatively low (e.g., 0.4 metric tons km^{-2} y^{-1}; Kedidi 1984). Over at least the last 40 years, increased settlement and fishing effort along the Red Sea coastline has led to fish stock depletion (Jin et al. 2012), particularly in areas frequently visited by fishers (Tesfamichael 2001) or of resources specifically targeted by fishers such as sea cucumbers (Kotb et al. 2008) and sharks (Tesfamichael 2012; Spaet and Berumen 2015). As the preferred piscivorous species have been overfished and depleted, increased effort has been directed towards species lower down the food chain, such as herbivores and planktivores (Sheppard et al. 1992) and it is now estimated that 55 % of reefs

in the Middle East are overfished (Burke et al 2011; Fine et al. 2019). Artisanal fishers have reported declines in income, catch and size of fishes (Gladstone 2002) and local fishing skills and ecological knowledge have been irreversibly lost as fishing has been abandoned for more lucrative livelihoods such as tourism (Kotb et al. 2008; Bruckner et al. 2011). Although at least 75 Marine Protected Areas (MPAs) were established or recommended in the region (PERSGA 2001), management efforts have not kept pace with fisheries growth (Bruckner et al. 2011). In addition, there are issues with management, representation of regionally significant habitats and regional cooperation in MPA planning (Gladstone et al. 1999; Gladstone 2000; Gladstone et al. 2003).

Egypt now has one of the most advanced fisheries industries in the Red Sea (Tesfamichael and Pitcher 2006; Samy-Kamal 2015). Egyptian fisheries may be divided into four major sectors: industrial, artisanal, subsistence and recreational. The subsistence fishery, officially recognized by Egyptian authorities as the 'traditional fishery' includes fishers who fish for own consumption, but also may sell a small portion of their catch at local markets (Tesfamichael and Pauly 2016). The traditional fishery contributes about 3 % of Egypt's total fisheries catch but its contribution has remained fairly constant over the years because it is less affected by external factors and is required as long as there exists a continued locally-based demand for the animal protein it provides (Tesfamichael and Pauly 2016). All of the fisheries of Sinai are categorized as 'traditional' except in El Tur (Chakraborty et al. 1983; Mabrouk 2015) and only 0.75 % of the national motorized fishing fleet is registered in Dahab (Samy-Kamal 2015; Figure 2.1).

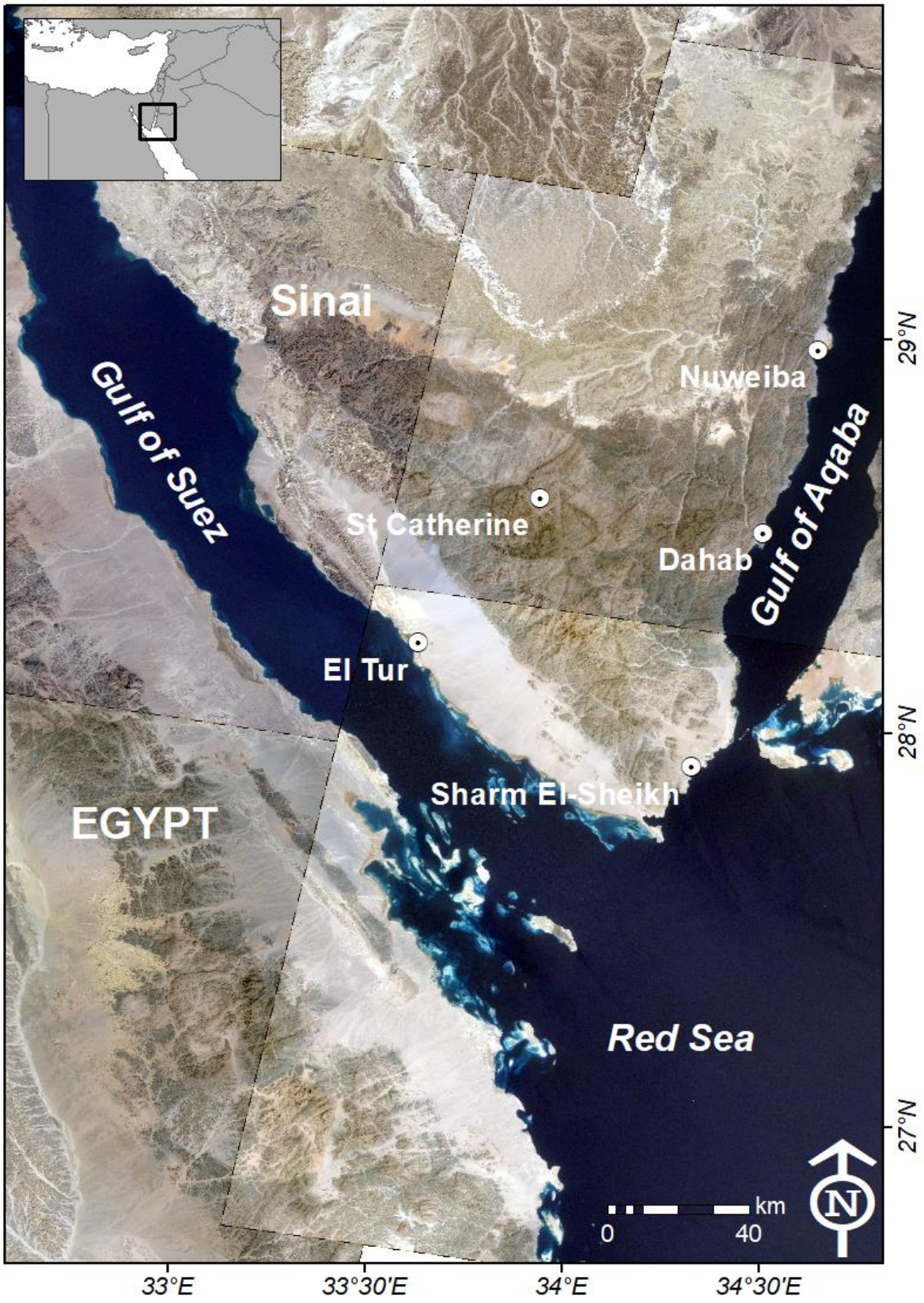


Figure 2.1. South Sinai, Egypt

The aim of this chapter is to describe the historical status of coral reefs and fishery impacts in South Sinai, Egypt as established through a review of historical literary accounts and documentation of TEK through discussion with *Mzeina* Bedouin elders and expert fishers in both formal and informal settings. By developing a baseline at the outset on which to evaluate changes, I discuss historical impacts and time scales of impacts of Bedouin fisheries in South Sinai. My approach aims to find common ground between different systems of knowledge and I will use the information presented in this chapter to provide a solid background to compare and contrast with ecological assessments of Bedouin fishing sites in future chapters. In particular, I present information on individual patterns of coastal resource use, tribal laws, access to fishing grounds, traditional stories, roles and gender responsibilities, the cultural and socioeconomic significance of fishing, Bedouin understanding of ecological processes, changes in employment and lifestyle, and socio-political factors affecting changes such as the proliferation of tourism and Israeli occupation. I also assess the socioeconomic and cultural significance of marine resources, examining persistence, current status and changes through time.

2.2. Research Approach

I was based in Dahab, South Sinai (Figure 2.1) from 2010 until 2015, a time of significant political change in Egypt (Vatikiotis 2013; Ketchley 2017). During this time, I became close friends with a number of Bedouin fishermen and their families; the trust that was developed enabled me to discover more detailed information about their fisheries than has previously been recorded. My aim was for participants to become partners in the study rather than just subjects. However, even though I had good relationships with many of the individuals I worked with, they were still suspicious of formal questionnaires, clipboards, dictaphones and involvement of Egyptians because of cultural tensions; so many of the details that I gathered were through informal discussions and participation in fishing trips. The bulk of the

information presented was provided from multiple detailed discussions with 10 experienced *Mzeina* fishers (eight men and two women) although additional information was gathered from more casual conversations with around 30 other individuals. The majority of fishers that I worked with were able to speak some English, and most discussions were conducted in a mixture of Bedouin Arabic and English, with assistance from proficiently bilingual Bedu as necessary. All fishers that provided data gave their informed consent; I ensured that they were aware of the aim of my work and that I was writing a PhD thesis and I worked with them on the understanding that they would remain anonymous.

To build up a picture of the historical socioeconomic and ecological factors that have contributed to the current status of *Mzeina* coral reef fisheries in the Sinai I conducted a thorough review of all available literature on the region, including travellers' accounts, anthropological research and marine biological studies. Access to these documents, many of which were unpublished, was difficult and required collaboration from Britiah, Israeli and Egyptian libraries and subsequent translation where necessary. Marine Protected Area management and planning documents were also collected from various Egyptian authorities and analysed to assess the level of inclusion of Bedouin needs.

Bedouin fishing practices are founded on local knowledge of the habits of fish species based on an awareness of the nature of the seas, coastline and reefs; as well as the changes induced by seasonality in winds, sea temperatures and currents (pers. comm.). Through personal contacts, I identified 'expert fishers' who had either been fishing for more than 20 years or for whom fishing was their primary livelihood and were identified as 'fishermen' among their peers. I worked with these expert fishers to identify fishing sites along the coastline using simple maps and landmarks. The exact locations of sites were confirmed in the field by fishers at each site. I also discussed various fish species and families known to exist in the area using photographic identification guides (Lieske and Myers 1999; Debelius 2001; Lieske

and Myers 2004), where possible, identifying the local Bedouin name, primary gears used, commercial value of species, and any species-specific fishing regulations, either tribal or state-enforced (Table 2.3; Appendix 2.1). All Arabic terms were simply transcripted phonetically since the correct Romanization of Arabic, particularly dialects, is highly complex (de Jong 2011), and most of the Bedu I spoke to were both wary of and not sufficiently literate to accurately write anything down. This process gave me a good knowledge of basic terms and provided a sound basis for future, more detailed, discussions.

Traditional practices were further investigated through semi-structured interviews and informal discussion with fishers, and observation of fishing activities to assess persistence, current status and changes through time. Due to gender segregation requirements in Islamic law, it was extremely difficult to speak with female fishers, although often they were willing to speak to me at remote fishing sites in the absence of Bedouin men. Issues discussed included important fishing sites, gear, species targeted, seasonality, tribal fishing laws and management, access to fishing grounds, traditional stories, roles and gender responsibilities, the cultural and socioeconomic significance of fishing, Bedouin understanding of ecological processes, changes in employment and lifestyle, and factors affecting these changes.

By accompanying fishers on trips, I gained a first-hand understanding of what it is really like to be a Bedouin fisherman and avoided many of the logistical complications involved in trying to organize social research with fishers in the field (Pollnac 1984). The true depth and breadth of Bedouin TEK only became fully evident when they were engaging directly with the environment, making standardized questionnaire-type approaches limited. My aim was to understand the biological features of reefs and fisheries as they were perceived in the context of everyday Bedouin life rather than in terms of marine science. Bedu generally clearly remembered straightforward facts such as geographical names, although historical activities often seemed embellished through superstition and fantasy (Jennings-Bramley 1906;

Arensburg et al. 1979). It was particularly difficult for me to ascertain details of drivers of fishing behaviour and methods for choice of fishing sites and times. While of great interest to my work, these matters seemed trivial to the Bedu. Often such details were only finally elucidated when we worked together in the field such that I would discover, for example, a spawning aggregation. As such, I am certain there is still a wealth of information about more subtle factors that affect Bedouin fishing behaviour that I was unable to gather. Many accounts of fisheries decline and wider changes in society were identical, even down to the choice of metaphors and other language used, illustrating the importance of storytelling and collective knowledge in Bedouin society.

2.3. Origin and history of the *Mzeina* Bedu

Settlement in the Sinai region has always been a dynamic process, influenced by geopolitical, socioeconomic and ecological factors (Finkelstein and Perevolotsky 1990). The *Mzeina* Bedouin tribe originated from a family group who split from the *Bani Harb* tribe of Saudi Arabia approximately 500 years ago because of a blood feud (Ginat 1984; Lavie 1990; Biallas 2013) and migrated to South Sinai where they formed an allegiance with six other tribes to forge the *Tawara* confederacy (Glassner 1974; Marx 2013). The *Mzeina*'s tribal territory now covers the coastline from El Tur in the Gulf of Suez north to Nuweiba on the Gulf of Aqaba, encompassing the settlements of Sharm el Sheikh and Dahab (Figure 2.1). During the 20th century, the successive colonial powers and national governments ruling Sinai promoted modern western ideals that came into conflict with the Bedu who only recognized the authority of their tribal leaders (Cordes and Scholz 1980; Lavie 1989) and consequently the Bedouin nomadic lifestyle has slowly declined as tribes have integrated with settled communities (Glassner 1974; Bocco 2006; Suwaed 2015). This process has been exacerbated by the fact that nomadism as a way of life and economic pursuit was lowly regarded by political decision-makers in the Middle East (Cordes and Scholz 1980). The

Bedu are traditionally pastoralists for goats and camels, but have never subsisted from a single-sector economy as they also grow dates, fish, garden and produce charcoal (Beadnell 1927; Marx 1999; Goodman 2013; Matrahazi 2019). Furthermore they have capitalized on their unrivalled knowledge of the Sinai wilderness to earn money escorting pilgrims to Mecca and even drug smuggling (Glassner 1974; Hobbs 1998; Marx 1999, 2008). They have worked as menial labourers under both the Egyptians and Israelis (Bernstein 1980; Rabinowitz 1985) and more recently, have become involved in tourism, initially at Saint Catherine and subsequently along the South Sinai coast (Sowers 2007; Goodman 2013).

In South Sinai, the political and economic changes since the 1960s have driven the Bedu into urbanisation, and the state has actively encouraged their settlement (Awad 1959: Cole 2003). A landmark change in Sinai Bedouin life occurred in 1971, when the Israeli administration built a major coastal highway southwards from Eilat, through Nuweiba and Dahab to Sharm el-Sheikh; developing diving centres, tourist facilities, schools and hospitals, providing access to advanced healthcare in Tel Aviv and training Bedu in hospitality and first aid (Glassner 1974; Zito 1982). The *Mzeina* were further encouraged to give up their pastoral lifestyles when the government distribution of fodder stopped in 1980 after several years of poor rainfall made pastoralism unsustainable as a livelihood (Cabahug 2002). As they settled, predominantly in the village of Assala, Dahab, they continued to keep their flocks in their townhouses, although there are now moves by the state to end this (pers. comm.). As the Bedu realized the income potential of tourism, they began providing taxi services, accommodation, coffee houses, camel and jeep trips, clothing, souvenirs and fish (Zito 1982, Matrahazi 2019). However, uncontrolled tourism development along the coastline has limited Bedouin access to traditional fishing sites (Ali 1998) and they have lost the integrity of their cultural values of simplicity, cohesiveness, mutual respect and traditional loose organization (El Bastawisi 1995; Cole 2003). Additionally, the Bedu are now plagued by the social

complications of a sedentary life, including prostitution, sexual relations and intermarriage with foreigners, drug dealing and addiction, commoditisation of their culture and a new materialism (Lavie 1990; Lavie 1993; Hobbs 1996; Abdalla 2004; Shaalan 2005; Goodwin 2007; Jacobs 2009; Fattah and Eddy 2018).

Sustainable tourism in indigenous territories is linked to several key parameters including land ownership, community control of tourism, government support for tourism development, restricted access to indigenous homelands and reclaiming natural or cultural resources used for tourism (Zeppel 1998). In Egypt, the state's tourism plans for South Sinai in the 1990s did not explicitly consider Bedouin participation (Homa 2007). As a result of the South Sinai Development Project (1994–2017), over 90 % of Bedouin land on the Aqaba coastline was, often forcibly, re-appropriated for development, allowing the Bedu to access only 7 % of the land they traditionally exploited because they lacked legally-recognized ownership (Lavie 1990; Homa 2007; Sowers 2007; Karkabi 2013). Furthermore, the sea was regarded as a collectively-owned resource and became state property (Ali 1998). Thus the Bedu were further forced to lead a sedentary life in order to prove rights to their ancestral land through construction or cultivation and thus be able to use the infrastructure facilities provided by the government (Karkabi 2013). The formalization of tourism in South Sinai over the last three decades has made it difficult for Bedu to participate in and benefit from the industry (Goodman 2013). Business ownership and employment in tourism and public services have been exclusively reserved for Egyptian migrants, while the Sinai Bedu have been stereotyped as disloyal to the state, and lazy, incompetent employees (Homa 2002; Shaalan 2005; Goodman 2013; Karkabi 2013). Consequently, the Bedu have become a minority population that has been pushed to the fringes of the tourism market to work informally as independent taxi drivers and as tourist guides in the desert (Aziz 2000; Gilbert 2011a; Goodman 2013). Meanwhile, fishing has changed from a subsistence activity to one of commercial value,

supplying tourists with fresh fish; in some way allowing the Bedu to integrate themselves more fully into local economies (Goodman 2013).

Virtually every aspect of Bedouin life is regulated by customary law, Islamic law is widely respected and Bedu are also, somewhat begrudgingly, subject to the law of the state (Cole and Altorki 1998; Stewart 2006; Matrahazi 2019) although they rarely pay taxes or recognize any government power (Zito 1982; Pelham 2012). The Bedu also hold Pre-Islamic beliefs and superstitions, based on star lore, animism and practical desert laws and customs (Bailey 1974; Bernstein 1980; Matrahazi 2019). In Egypt, environmental legislation has always been rooted in tradition and religion, for example, it was covered by a civil code within the body of Islamic law *(Mejellah)* when Egypt was part of the Ottoman Empire from 1517 until 1914 (Shackley 1999). In Islam, there are explicit teachings of the interconnectedness between man and nature and recognition that there are limits to resource exploitation (Child and Grainger 1990; Elmusa 2005; Khalid and Thani 2008). Pastoral nomads have often been regarded as uninterested in protecting their natural resources (Herrmann and Hutchinson 2005), however, the Bedu have a sophisticated understanding of the ecology their natural environment, with strict tribal laws and traditions governing the use of resources (Briggs et al 1998, 1999; Hobbs 2010; Hobbs et al. 2014). Their conservation ethic is deeply rooted in the tribal system of *al-hilf* (the agreement) to control seasonal use of pastures, or personal action *dakhl* (essence) normally to protect trees (Grainger 2003). These systems have been enforced by '*urf* (tribal law), so when a person pledges to uphold a principle that all of the tribe's peoples regard as just, acting against it violates both his personal honour and *'urf* itself (Grainger 2003). The Bedu have a use for almost every species of plant in their surroundings, including numerous medicinal uses (Lavie 1990; Abu-Rabia 2015; Halmy 2017; Matrahazi 2019). The importance of plants is illustrated by the fact that places are named after them and people's characteristics described by them; for example, a bitter person may be compared to

the bitter-tasting desert gourd *(Citrullus colocynthis)* (Bailey and Danin 1981). The Bedu believe that it is wrong to kill any animal without good reason and in sacred areas, the Bedu have enforced traditional forms of environmental protection such as permanent hunting bans around Mount Sinai (Hobbs 2010; Matrahazi 2019). Nowadays, Bedouin TEK is slowly being eroded and priorities have changed in their societies as less time is spent conducting traditional farming and fishing activities (Gilbert 2011a). However, in some cases, such as in the Wadi Allaqi Biosphere Reserve, in the Nubian Desert, Bedu are now combining new information from modern scientific research with their TEK (Belal et al. 2009).

2.4. Protected Areas in South Sinai

The impetus for creating the first protected areas in Sinai developed during the 1970s, just before Israel handed the territory back to Egypt in 1979. At this time a variety of environmental impacts had started to pervade the region including overgrazing and dynamite fishing (Baldwin et al. 1988). For example the Arabian leopard *(Panthera pardus jarvisi)* had disappeared from Sinai (Spalton and Al Hikmani 2006), the ranges of the dorcas gazelle *(Gazella dorcas)* and Nubian ibex *(Capra nubiana)* were reduced (Saleh 1987); and greater flamingos (*Phoenicopterus roseus)* were disturbed by egg-collecting (Baldwin et al. 1988). The introduction of marine protected areas (MPAs) to Sinai by Israel in the 1970s was documented by Fishelson (1980), who noted that the main aim of the MPAs was to preserve the diversity of coastal habitats, with clearly defined boundaries, the areas outside of which could be promoted for tourism. He described the development of the road from Sharm el Sheikh to Eilat, which was purposely routed away from the coast to ensure that large tracts of coral coastline were almost inaccessible. Within MPAs, fishing or collection of corals or other invertebrates and disposal of waste was prohibited (Randall 1969; Fishelson 1980; Table 2.1). Education of the Bedu was an important aspect of these early MPAs, and the link between healthy reefs and income was emphasized along with the importance of various

coastal habitats as nursery grounds for commercially-important fish species (Fishelson 1980). Bedu required permits to fish in limited areas using traditional and non-destructive methods and Bedouin ranger teams patrolled the MPAs (Fishelson 1980; Lavie 1990; Table 2.1). However, as Israel prepared to handover the Sinai territory back to Egypt, there was considerable fear that the reefs would not be adequately protected by the Egyptians (Zito 1982; Table 2.1).

Currently, the Egyptian Environmental Affairs Agency (EEAA) and the Nature Conservation Sector (NCS) are responsible for nature conservation issues in South Sinai under the Ministry of State for Environmental Affairs (MSEA) (Shehata 1998; Abd-Alah 1999). Law 102/1983 provides the legislative framework for establishing and managing protected areas in Egypt, which are defined as 'any area of land or coastal or inland water characterized by special flora, fauna and natural features having cultural, scientific, tourism or aesthetic value' (NCS 2006). In addition, Law 4/1994 complements Law 102/1983: although it is focused mainly on pollution issues. Several clauses have implications for nature conservation and hunting management. However, Law 102/1983 is widely regarded as inadequate as there is no legislated provision for management plans, operational budgets and demarcation of protected area boundaries (Shackley 1999).

Table 2.1. Illustrative quotes from discussions with *Mzeina* Bedu regarding socioeconomic and ecological changes in coastal South Sinai

Topic	Quotes
Relationship with authority	• During the Israeli occupation of Sinai, we needed permission to fish, particularly for Nabq or Tiran • The Israelis cared about the sea but the Egyptians don't care • The Israelis banned collection of spider conch and giant triton shells • Our *arishas* in Nabq were demolished by the government
Fishing gears	• We sometimes catch tuna by setting a net in deep water overnight • In calm conditions, we sometimes fish from boats • We can catch 5–10 kg in a day using a net and only up to 1 kg using a hook-and-line
Fishing grounds	• The best fishing grounds are found in Nabq, but there is less completion between fishers in the Dahab area • There is only occasional line fishing in at Ras Abu Gallum because it is for tourism • Around Tiran Island, you can catch more than 100 kg of fish in a single day
Fishing seasons	• Rabbitfishes migrate along the coastline and we catch a lot in the summertime • Now there are not so many fish so we fish all year round rather than just in the summer • There is more fishing in the summertime because it is too cold to go fishing in the winter

Topic	Quotes
Changes in catch	• We always throw back butterflyfishes, pufferfishes and other small reef fishes • We used to catch lobsters by hand during the daytime, now we can only find them at night by snorkelling • More and more people are fishing and there are less and less fish • There used to be only 2 or 3 nets in the whole of Dahab, now everyone fishes with nets • Now lines are used much more rarely • We only ever used to catch big fish, there was never any reason to catch small fish
Changes in livelihood	• Before ice was available, we used to catch parrotfishes and salt and dry them in the summertime • We sell our catch to restaurants and shops, and eat some ourselves • Families go fishing with their kids during the summer holidays, professional fishermen fish between 8–10 days per month all year round • Some families go camping on the beach and just catch fish to eat during their stay • Some Bedu just fish because they enjoy fishing, others have no other income and must fish to earn money
Conservation and management	• National Parks staff have worked with us in the past, for example, they paid us to clean the beach • We always respect the No-Take-Zones at Nabq, I get angry when others break the rules • National Parks did nothing when all the lobsters were being caught for tourist restaurants • The Egyptian Government have even allowed some boats to fish in Ras Mohamed National Park

To date, the social, economic and cultural impacts of protected area development on the Bedu in South Sinai have been relatively well-studied in the Saint Catherine Protectorate (Figure 2.2) compared to its marine counterparts (e.g. Grainger 2003; Grainger and Gilbert 2008; Hussein 2010; Gilbert 2011b, 2013). More than 7,000 Bedu belonging to six tribes live within or close to the borders of St Catherine's Protectorate (Grainger 2003). Here, Bedu work mainly in tourism enterprises as guides or cameleers and some continue to cultivate mountain gardens while women herd livestock and produce traditional craft items (Grainger 2003; Marx 2012) although conflict with tourism exists because of the limited water supply (Grainger 2003). One of the objectives of the protected area was 'to provide for the full integration and protection of indigenous people resident in, or adjacent to declared protectorates' (Shehata 1998). However, in reality, the Bedu only had minimum contact and a feeble working relationship with the Egyptian state authorities (Hobbs 1996; Aziz 2000). In the mountains of South Sinai, the indigenous Bedu have changed rapidly during the past 50 years from semi-nomads to a largely settled population struggling to replace their traditions with viable livelihoods of standard waged work and to cope with significant difficulties in their relationship with the Egyptian State (Gilbert 2011a). These difficulties affect their relationship with the St Catherine Protectorate that has engulfed their land, and have enhanced their perception of their role as protectors of the environment (Gilbert 2011b).

2.1. South Sinai's coral reefs

The coral reefs of the Gulf of Aqaba were noted for their natural resources as long ago as the fourth century:

'Its fish are excellent and unusually sweet…there are trumpet-shells and oysters of various kinds, white shells, and several kinds of large snail…there is a great deal of coral on the shore.' (Egeria 384 in Manley and Abdel-Hakim 2009).

Modern science has continued to recognize the Red Sea as a region of high biodiversity (Stehli and Wells 1971; Berumen et al. 2013) that includes c. 300 hard corals, representing over 50 genera; (Sheppard and Sheppard 1991; Riegl and Velimirov 1994); in the Egyptian jurisdiction of the Red Sea, 209 hard corals and 16 soft corals have been recorded (Kotb et al. 2008) along with 1,078 fish species (Golani and Bogorodsky 2010). The Red Sea also harbours one of the highest degrees of endemism for marine organisms (Ormond and Edwards 1987) and includes 14 % of fishes (Randall 1994), 33 % of crustaceans, 15 % of echinoderms, and up to 25 % of corals (Cox et al. 2016). Endemism can be even higher for specific taxa. For example, up to 50 % of Red Sea Chaetodontidae (butterflyfishes) are endemic to the region (Roberts et al. 1992). The Red Sea also supports diverse marine megafauna, some of which are threatened with extinction including the vulnerable dugong (*Dugong dugon)* (Lipkin 1975; Hanafy et al. 2006)*;* five threatened species of sea turtle, primarily the endangered green turtle (*Chelonia mydas)* and critically endangered hawksbill (*Eretmochelys imbricata)* that nest on offshore islands (Frazier and Salas 1984; Seminoff 2004; Mortimer and Donelly 2008; Hanafy 2012a); migratory birds (Frumkin et al. 1995; Ibrahim 2011); sharks (Baranes and Ben-Tuvia 1978; Spaet et al. 2012) and cetaceans (Baldwin 2003; Notarbartolo di Sciara et al. 2017).



Figure 2.2. Protected Areas in South Sinai

Three broad physiographic zones occur along the South Sinai shoreline (Galal 1999): (1) shallow marine zone dominated by fringing reef coral reefs; (2) a slightly inclined coastal plain covered by fluvial sands/gravel and sand dune vegetation and (3) the steep-sided mountains of South Sinai, which rise inland towards St Catherine. Periodic flooding of small *wadis* (dry, seasonally flooded valleys) maintains patchy vegetation which provide fodder for the Bedu's camels and goats, and native mammals, including the fennec fox *(Vulpes zerda),* Cape hare *(Lepus capensis),* and the vulnerable dorcas gazelle *(Gazella dorcas),* and Nubian ibex *(Capra nubiana)* (IUCN SSC Antelope Specialist Group 2017; Alkon et al 2008). Halophytic shrubs found on the coastal plain include *Salvadora persica, Limonium axillare, Suaeda fruticosa* and *Nitraria retusa* growing in zones parallel to the shoreline depending on local topography and hydrological conditions (El-Gazzar et al. 1995). Nabq MPA also contains the northernmost *Avicennia marina* mangrove communities in the world (Por et al. 1977; Galal 1999; Gab-Alla et al. 2010).

A shallow reef flat, originally formed from raised fossilised Pleistocene reef, extends seaward from the coastline and varies from 100 m to 800 m in width (Strasser and Strohmenger 1997; Hassan et al. 2013). The landward side of the reef is covered with a thin layer of sandy mud, and algal mats with patches of macroalgae including *Padina, Caulerpa, Laurencia* and *Gracilaria.* The central and outer reef flat includes small pools and shallow lagoons with larger macroalgae, such as *Turbinaria* and *Sargassum*; the seagrasses *Thalassia hemprichii*, *Halophila spp*. and *Halodule spp.*; gastropods including *Strombus* and *Lambis spp*.; echinoderms *Echinometra mathaei*, *Diadema setosum* and *Holothuria spp*. and small colonies of hard corals. Wave action from the prevailing northeast wind can create a shallow groove and spur formation along the reef edge in places where the reef slopes steeply to between 3 and 20 m (Galal 1999). On the reef slope, hard coral growth is dominated by branched (*Acropora spp.* and *Pocillopora spp.)* and massive species (*Porites spp., Favia spp.)* and soft

corals of the genus *Sinularia* are also common. Below the reef slope, a wide sandy bottom interspersed with coral patches (*Favia, Acropora*, and *Porites spp.)* and mixed seagrass beds gently slopes to 50 m and deeper.

2.2. Threats to coral reefs in South Sinai

Coral cover in the Red Sea has decreased markedly and negative changes have become clearly evident over the last 30 years (Riegl et al. 2012; Price et al. 2014). Almost two decades ago, the 'Red Sea Riviera' (Coral reefs in the Gulf of Aqaba off Jordan, Israel and Egypt plus the reefs of the Egyptian Red Sea) was noted as one of twelve 'reefs at risk' globally, due to the considerable threat from unfettered development of coastal tourism (Bryant et al. 1998). The Egyptian government previously pursued an ambitious target of 16 million tourists in the Red Sea by 2017 (Wahab 1997; Shaalan 2005) although this aspiration has been significantly affected by political turmoil in the country since 2012 (Vatikiotis 2013; Ketchley 2017) and a likely Islamic terrorist attack on a Russian charter flight in 2015 (Lowe 2015). However, tourism continues to be the major source of local anthropogenic disturbances on these reefs (Fine et al. 2019; Table 2.2).

The rich marine life of South Sinai has made it one of the world's premier SCUBA diving destinations and the area is also popular for other watersports, particularly kite surfing and windsurfing (Shaalan 2005). Coastal tourism in South Sinai and the rest of the Red Sea coast has been noted to account for half of all Egypt's tourism income (Sowers 2007) and the South Sinai coastline has experienced a massive increase in tourism over the last ten years with measurable intense impacts on coral reefs and other coastal habitats (Shackley 1999; Shaalan 2005; Davenport and Davenport 2006; Hasler and Ott 2008; Tawfik 2010; Hannak et al. 2011; Hilmi et al. 2012; El Naggar et al. 2017; Mona et al. 2019; Table 2.2). Although Egypt has shown aspirations for sustainable tourism (Eraqi 2007), the majority of tourism development in South Sinai occurred within 50–300 m of the shore (Shaalan 2005) and thus

has created major environmental impacts on the narrow fringing reef and social implications for the people who rely on its resources. These threats are further exacerbated by climate change (Cantin et al. 2010; Kvitt et al. 2011; Furby et al. 2013), although the reefs of the northern Red Sea are relatively tolerant to thermal stress (Fine et al. 2013; Osman et al. 2018). Furthermore, all these impacts make reefs less resilient to the negative effects of compound stressors such as fishing (Hughes and Connell 1999; Wooldridge et al. 2005; Pratchett et al. 2008). Endangered Red Sea megafauna are threatened by bycatch in net fisheries and turtle egg consumption (Rouphael et al. 2015). Egypt's lack of institutional capabilities to monitor and control these pressures has magnified them further (El Raey 2010), although tourism development in the Gulf of Aqaba may have actually improved awareness of the value of environmental assets and thus effective enforcement of environmental laws (Mabrouk 2015).

Diving tourism encourages conservation, generates revenue, and supports local communities (Lucrezi et al. 2017), but it has also measurably impacted coral reefs in the Red Sea causing hard coral mortality (Wielgus et al. 2004) and smothering of corals through disturbance of sediments (Hanafy 2012b). The reefs of South Sinai have been noted as the most heavily dived in the world (Shaalan 2005) with more than 30,000 dives y^{-1} estimated at Dahab alone (Hasler and Ott 2008) and diver carrying capacity has certainly been exceeded on a number of Egyptian Red Sea reefs (Jameson et al. 1999). Even complete 'no-use' closures may not be sufficient to protect corals on narrow reefs supporting intense recreational use (Epstein et al. 1999).

Table 2.2. Impacts on natural resources along the Gulf of Aqaba coastline in South Sinai

Impact	Source	Status	Description
Tourism development- hotels and resorts	Egyptian and international companies	Permits required	Environmental guidelines often ignored, unsustainable
Sewage discharge	Coastal towns	Limited regulations, poor monitoring and enforcement	Changes in bacterial communities, algal growth
Waste and littering	Visitors on beach, shipping	Clean-ups, garbage collection	Most washes onto the shore, mainly of local origin
Oil pollution	Shipping	Offender is fined	Tarballs on shore, seabirds etc
Vessel groundings	Shipping	Offender is fined	Passing cargo and smaller ships
Vehicles damaging vegetation and eroding coastline	Visitors, Bedu	Tourist vehicles required to keep to marked tracks	4x4 vehicles used to access remote shorelines or for convenience
Physical damage to reefs	Tourists and Fishers	Forbidden by law	Reef-walking, snorkelling and diving
Collection of marine curios	Tourists	Forbidden by law	Souvenir collection
Fishing	Bedouin Fishers	Traditional fishing permitted except in No-Take Zones (NTZs)	Generally traditional and small-scale using nets, spears and handlines
Gleaning of reef organisms	Bedu (predominantly women)	Traditional rights, forbidden inside NTZs	Collection of molluscs for local consumption
Collection of vegetation for fuel *(Salvadora spp.)*	Bedu	Commercial exploitation is forbidden	Local Bedu for own use
Coastal construction	Bedu	Small villages and encampments (permit required)	Light constructions – arishas for tourism and fishing
Mangrove harvesting	Bedu	Prohibited	Mostly by local Bedu for fuel and poles
Grazing	Goats and camels	Permitted	Owned by local Bedu

(Galal 1999; Gladstone et al. 2013; Fine et al. 2019)

The Red Sea has been impacted by pollution from marine litter and discarded fishing gear (Abu-Hilal and Al-Najjar 2004), heavy metals (Ali et al. 2011), fishing port development (Sharaan et al. 2017), eutrophication resulting from nutrient run-off (Walker and Ormond 1982; Fishelson 1995; Abelson et al. 2005; Ali and Hamed 2006; Naumann et al. 2015), ballast water discharges and oil pollution from commercial ships (Dicks 1987; Sheppard et al. 1992) and desalinization plants (Hoepner and Lattemann 2002). Pollution can travel several kilometres and affect even remote reefs (Abelson et al. 1999) resulting in depletion of soft-bottom communities, declines and trophic changes in fish populations (Khalaf and Kochzius 2002), low coral recruitment, death of corals and associated ichthyofauna, and slow recovery from disturbances (Wielgus et al. 2003). Apart from the negative environmental impact, these issues result in negative publicity, which in turn reduces the quality and image of the tourist destination (Shaalan 2005).

2.3. *Mzeina* Bedouin artisanal fishing practices

Historical and current status

The earliest scientific data on the marine environment of the Gulf of Aqaba and Bedouin exploitation of marine resources was gathered by the Israelis during the 1970s (Fishelson 1980, 2009). However, this study found that Bedouin TEK of marine ecosystems extended well before this time. Until the early 1980s, 80 % of coastal Bedouin communities in South Sinai relied on fishing as their main source of livelihood or subsistence (Glassner 1974; Ali 1998). Historically Dahab contained some of the most extensive date palm groves in the region, and Bedu migrated there, primarily from Bir Ogda, for the bountiful date harvest and seasonal fishing in the summer (Melamid 1957; Glassner 1974; Bernstein 1980). The fishing season traditionally began with the arrival of Lethrinidae (emperors) and Lutjanidae (snappers) in Ras Mohammed in April and thereafter continued for three months (Cabahug 2002). In the winter, the Bedu would return to their pastoral lifestyle in the mountains, taking

dried and salted fish as an additional source of protein or to sell or exchange in Wadi Feran (Matrahazi 2019; Table 2.1). During the Israeli occupation, Israeli security guards would take 8–10 *Mzeina* in a large sailboat with ice boxes to Tiran Island where they would catch 8–10 t of fish in 5–6 days (Table 2.1). In the 1990s, boats were distributed by the state in an attempt to improve local incomes through the exploitation of offshore reefs, although many of these vessels were ultimately repurposed for the tourism industry (Cabahug 2002). The *Mzeina* continue to fish the South Sinai coastline targeting shallow waters and coral reefs using handlines, gillnets, trammel, seine nets and cast nets (Samy et al. 2011; Tesfamichael and Pauly 2016; Table 2.3) and jeeps, pickups and camels are used to transport fishing equipment and supplies along the coast (Chakraborty 1984a, 1984b). The total fisheries yield in the area has been estimated at 1.9–6.2 t km^{-2} yr^{-1} (Mabrouk 2007).

Table 2.3. Marine organisms targeted by *Mzeina* Bedu on coral reefs

Method	Families and species targeted
Hook-and-line	Epinephelinae, Hemiramphidae, Lutjanidae, Platycephalidae, Sphyraenidae, Synodontidae
Net	Apognidae, Caesonidae, Carangidae, Fistulariidae, Haemulidae, Holocentridae, Labridae, Mullidae, Platycephalidae, Priacanthidae, Scaridae, Synodontidae
Spear	Epinephelidae, Haemulidae, Labridae, Scaridae
Collected for food	*Lambis lambis*, Littorinidae, *Octopus spp.*, *Strombus tricornis*, *Tridacna spp.*, Trochidae
Collected for curio trade	*Charonia tritonis*, Cypraeidae
Collected for bait	*Cellana rota*
Not eaten	Chaetodontidae, Cheloniidae, Ephippidae, Muraenidae, Pomacanthidae, Scorpaenidae, Tetraodontidae
Targeted at night	Apognidae, Haemulidae, Hemiramphidae, Holocentridae, Palinuridae

(pers. obs.; pers. comm.)

Changes in the fishery

The general opinion of expert Bedouin fishers suggested that tourism had become the main economic activity in contemporary South Sinai, driving change in: (1) demand for fishery resources; (2) socioenvironmental and landscape changes; (3) diversification of jobs for income; and (4) changes in socio-cultural values (Table 2.1). Bedouin fishers unanimously recalled that fishery resources were far more abundant and diverse before the growth of tourism (Zito 1982; Table 2.1) and that now fishers can barely 'catch enough for lunch' in one trip even though they used to easily catch 50–60 kg. Nowadays, not only many more Bedu are fishing but also Egyptians and even Russians. Species that have shown most notable declines included Haemulidae (sweetlips), Kyphosidae (rudderfishes), Lethrinidae, Scaridae (parrotfishes), Sparidae (breams) and especially Palinuridae (spiny lobsters). The decline in lobsters was widely noted to be a result of high demand and prices from tourist restaurants which encouraged Bedouin fishers to catch as many as possible (Table 2.1). One expert fisher exclaimed that there used to be so many lobsters on the reef that 'you could see them from the mountains' and *Panulirus penicillatus* were apparently common in 1986 (Plaut and Fishelson 1991). It was mentioned that 'all the fish caught by line' have now become rare and that nets were only rarely used in the past. There used to be only two or three nets in the whole of Dahab, and they were large 'eight finger' mesh, since the Bedu only targeted large fish because there was no reason to catch small ones (Table 2.1). Using a line would yield a maximum catch of 1 kg day^{-1} but it was estimated that net fishing could catch up to 15 kg day^{-1} (Table 2.1), so this gear rapidly depleted finfish stocks. Lobsters were historically always caught by hand, but now snorkelling and diving were necessary to reach them (Table 2.1). Declining catches (Galal 1999; Galal et al. 2012; Mabrouk 2015; Advani et al. 2015) and a decline in the use of hook-and-line and loss of predatory species (Mabrouk 2015) in *Mzeina* fisheries have been noted by previous studies.

Fishing gears

Nets are the most widely-used fishing gear by *Mzeina* (Galal 1999) and take a wide variety of species, with selectivity determined by the mesh size (Acosta 1994, Mbaru et al. 2019). This gear has been linked to bycatch of marine megafauna in the Red Sea (Rouphael et al 2013) and worldwide (Moore et al 2010). Nets can damage corals through entanglement or through direct contact and trampling by fishers (Mangi & Roberts 2006; McClanahan & Cinner 2008). Nets are also a major contributor to ghost fishing since their three-dimensional perspective is maintained when tangled around the reef so impacts can continue for a substantial time (Donohue 2001; Matsuoka 2005). The *Mzeina* primarily use nets from the shore, swimming out with snorkel, mask and fins and carrying the net floating on an old inflated inner tube or polystyrene platform (pers. obs.). Boats are rarely used. Two basic types of net are employed: gillnets and trammel nets, which appear to be used interchangeably, depending on availability and fishers' preferences. Nets are generally 18–25 m in length and 1.4–1.8 m deep (Galal 1999; pers. obs.). *Mzeina* use a variety of techniques to catch fish using nets, summarized in Table 2.3. Fishers displayed environmental awareness in their fishing behaviour, for example, they carefully removed unwanted species from nets and took care to avoid snagging their gear on corals (pers. obs.). The common nature of the resource was illustrated by the fact that fishers were happy for other *Mzeina* to harvest from their set nets at any time since 'the sea belongs to everyone'.

Table 2.4. Net fishing techniques of the *Mzeina*

	Technique	**Families targeted**
Drive net	Net set on the reef flat and fishers (often children) scare the fish into the net by splashing and sometimes hitting the reef with sticks over 30 mins – 2 hours	Acanthuridae, Scaridae, and Siganidae.
Set Net	Net set at the reef crest and fishers swim and attempt to herd schools of herbivores into the net and encircle them	Scaridae, Siganidae, Acanthuridae
Long – term / Overnight Set	Weighted net up to 50 m long set along the edge of the reef crest or across channels overnight. Dependent on the lunar phase. May be set for 6 hours - several days, with fish being harvested daily using mask and snorkel.	Muglidae, Siganidae , Gerreidae Holocentridae, Haemulidae, Lethrinidae and Sparidae

(Galal 1999; pers. obs.)

Handline fishing, where a single baited hook is attached to a line and held by hand, is among the least expensive fishing gears and thus for many fishers represents an entry point into coral reef fishery-based subsistence and livelihood (Béné 2006). On reefs, line fishing tends to target high trophic groups such as Epinephelidae (groupers) and Lutjanidae (Cinner and McClanahan 2006; Cinner et al. 2009). Line fishing is widely regarded as one of the least destructive and selective gears (Ralston 1990), although lines may occasionally snag on and abrade sessile invertebrates on coral reefs (Chiappone et al. 2005). Line fishing is the commonest fishing gear in the Red Sea but has been reported as the least frequently used gear by the *Mzeina* (Galal 1999). The *Mzeina* usually use this gear in deeper lagoons and reef pools on the back reef, off the reef edge or from small wooden canoes or *hassakas* over the reef slope (Galal 1999). Nowadays, old surfboards or slabs of polystyrene are also commonly used as fishing platforms (pers. obs.). *Mzeina* also use lines to fish offshore in calm seas, at depths of 50–200 m using 3–6 m motorized boats aiming to catch larger Lutjanidae (Mabrouk 2015). In deeper water, multiple lines and hooks or large-mesh overnight-set gillnets are occasionally deployed to catch Carangidae (jacks), Scombridae (mackerels and tunas), Carcharhinidae (reef sharks) and Sphyraenidae (barracudas), primarily in the winter, notably around Gabr el Bint (pers. comm.).

The *Mzeina* regularly use modern spearguns for fishing, often as a supplementary gear while nets are in the water or at nighttime, but it is unclear if any variation of this gear was ever traditionally used (pers. obs.). Nowadays, although classed as a controlled firearm, spearguns are smuggled in from Europe and Saudi Arabia, and are even sold openly in Cairo and Alexandria. This highly selective and efficient gear can skew species composition and the size distribution since it allows the targeting of keystone species and larger individuals (Sluka and Sullivan 1998; Pavlowich and Kapuscinski 2017; Mbaru et al. 2019; Skinner et al. 2019). Spearfishing using SCUBA also has the potential to eliminate populations in deep water refuges, (Lindfield et al. 2014; McClanahan and Cinner 2008) although Bedu only occassionally use SCUBA for spearfishing (pers. obs.).

Fishing grounds

Mzeina fishing grounds along the Gulf of Aqaba coastline from Sharm el Sheikh to Nuweiba have remained constant for generations, and this was the case even before Dahab was settled (Figure 2.3). Fisher density in Abu Gallum (Figure 3.1) has been estimated at around 10 fishers day^{-1} (Samy et al. 2011) and in Ghargana village in Nabq, there are approximately 25 permanent fishers, which increase to 43 in the summertime when they move down from Khriza village in the mountains where they overwinter (Mabrouk 2007, 2015). Visiting fishers in Nabq primarily originate from Dahab, with a few from Sharm el Sheikh and only occasional individuals from Nuweiba (Table 2.1; pers. comm.). The current density and distribution of fishing effort are almost impossible to estimate, since fishers fish over wide areas on foot, so commonly used approaches such as vessel counts (Miller et al. 2007) or gear counts (Horta e Costa et al. 2013) are not possible. However, there certainly is considerable variation between the summer and winter seasons; for example, at Amoyed (Figure 3.1), the 40 or so *arishas* (palm frond huts) are deserted in the winter, but are completely full of *Mzeina* families during the hottest summer months, and pigeons are kept on-site in order to provide a protein source when fishing is poor. In the winter, it is too cold to sleep in *arishas* and the fish are believed to move to deeper waters (Table 2.1). In the summertime, Bedu traditionally target Lutjanidae and Lethrinidae as well as aggregations of Scaridae and Siganidae (rabbitfishes), where it is easy to catch large numbers of fish at once; whereas in the winter, Mugilidae (mullets), Haemulidae, Fistulariidae (cornetfishes) and Zenarchopteridae (halfbeaks) are also caught. However, these seasonal patterns are fading now as fish stocks have simply become low all year round so there is no longer a clear peak season (Table 2.1). Fishing grounds vary in accessibility since the coastal paths often do not allow vehicle access even by jeeps although *Mzeina* access some sites by driving directly across the reef flat (pers. obs.). Fishing is not permitted at any recreational dive sites while divers are in the water. Ras Abu Gallum is only occasionally fished because the area is primarily used for small tourist restaurants and basic accommodation (Table 2.1). Until recently, Bedu used to

travel by boat to Tiran for up to 9 days where fish stocks were more abundant and they could catch up to 100 kg day^{-1} (Table 2.1). Now Bedu need special permission from the police to travel to Tiran and overnight fishing camps on the island are no longer permitted because it has now become a territory of Saudi Arabia (Arafat 2019). During the Israeli Occupation, it was not permitted to fish further south than Gabr el Bint *(Aemaileh)* (Table 2.1; Figure 2.3).

Socio-cultural aspects of the fishery

Mzeina fishers generally fish in family-based groups of two to four individuals, often with the assistance of younger relatives, who play in the sea and slowly begin to learn traditional fishing techniques. If fishing from a boat, groups tend to be larger with up to 6 individuals. Fishing trips can last for 2–4 days; with the fish caught being eaten directly to sustain longer trips. The fishery is truly multi-gear, targeting a wide variety of species (Table 2.3) and most groups use two or three types of gear during the same season and even during the same diel period or individual fishing trip (Galal 1999; pers. obs). The gear used depends on weather and oceanographic conditions, tidal patterns, and fishers' relative needs to meet the subsistence requirements of their families or to generate income (Galal 1999) but also personal preference and specialist skills are also of significance. Different levels of fishing professionalism exist, for example, some fishers exclusively fish year-round and are regarded as 'fishermen' by their peers; while other fishers have multiple livelihoods, for example, restaurant owners at Ras Abu Gallum who may catch fish to order when necessary (Table 2.1). There was a clear distinction between these different types of fisher, with the more professional certainly having more regard for longer-term sustainability and environmental issues (pers. obs). It has been reported that green turtle (*Chelonia mydas)* penis was regarded as an aphrodisiac among the *Mzeina* (Levy 1978) but this was never noted by any informants in this study although it was mentioned that *Mzeina* have eaten turtles in the past. Evidence of a hawksbill turtle *(Eretmochelys imbricata)* entangled in a net and abandoned was observed at Laguna Beach in Dahab (pers. obs.). Shellfish, particularly *Tridacna* (giant clams) were widely noted for their aphrodisiac properties.



Figure 2.3. Bedouin fishing grounds between Sharm el Sheikh and Nuweiba, South Sinai

In most cases, the *Mzeina* catch is destined for consumption by friends and family. When sold, catch is generally destined for shops in Dahab, three or four of which are owned by Bedu and four or five which are Egyptian-owned; catch is also sold directly to other households or tourist restaurants. Prices can range from c. US$ 2.50–4 kg^{-1} depending on the species, with better prices normally obtained if sales are made directly to restaurants. Occasionally fish are still dried for sale at a later date, particularly Siganidae and Scaridae, although this practice has declined with the increased availability of ice (Table 2.1).

Gender roles

In traditional Bedouin society, men and women both make important labour contributions to support household livelihoods (Cole 2003). With respect to fishing, men are generally responsible for catching finfish and lobster, while women occasionally fish with lines in the winter but primarily glean for molluscs *(Tridacna* spp., *Octopus spp., Lambis* spp. and *Strombus* spp.). The method involves using 'spears' fashioned out of approximately 1 m of iron rebar, with one end slightly bent and sharpened and the other end formed into a circular handle (Ashworth et al. 2004). These organisms are easily harvested because they are abundant on the reef flat and simple to find. Women carry shoulder bags and sometimes use floating polystyrene boxes in which to store their catch (pers. obs). Octopus are tenderized by beating them against rocks on the shore before being taken home (Lavie 1990). Women are normally transported by jeep or pickup by a male family member then walk to access more remote sites (pers. obs.). Some may use wetsuits and snorkelling gear (pers. comm.). Peak fishing activity corresponds with low spring tides. Octopus are more commonly targeted in the winter. *Tridacna* are caught by sticking a spear into the open clam then wiggling it until the clam is broken free from its surrounding rock or hard coral (pers. obs.) Octopus are caught by searching for their holes, characterized by a small pile of red and white stones at the entrance then the octopus is speared in its hole (pers. comm). Fishing is a social activity and groups of several women normally glean together and take a picnic of biscuits and cakes to share (pers. obs.). Men will

also occasionally collect molluscs while their other fishing gear is in the water (pers. obs.). It is worth noting that such reef gathering can cause considerable physical damage to the reef (Galal 1999). *Mzeina* gleaning, particularly for *Tridacna* has historically been intense, and it has been suggested that overexploitation was responsible for the possible local extinction of *T. squamosina* (Richter et al. 2008; Huber and Eschner 2010).

2.4. Marine Protected Areas (MPAs) in South Sinai

An estimated 38 % of Egypt's population lives in the coastal zone and the coastal population is growing rapidly (Neumann et al. 2015). Fisheries make a significant contribution to the country's food security, providing an affordable source of protein (Samy-Kamal 2015), so sustainable management of marine resources is of huge importance. However, in recent years, national policy has focussed on aquaculture development, rather than the sustainable management of wild stocks and MPAs have received insufficient financial support, particularly in the Red Sea (Samy et al. 2011; Samy-Kamal 2015). There is an urgent need for management of small-scale fisheries because they have not been adequately considered in fisheries management plans (Feidi 1998; Pramod and Pitcher 2006). In addition, baseline data and reference points for Egyptian Red Sea fisheries are limited and socioeconomic data are virtually non-existent (Pramod and Pitcher 2006; Tesfamichael and Pauly 2016).

Three of Egypt's six MPAs are located in South Sinai (Table 2.5): Ras Mohammed National Park, Nabq Multiple Use Management Area and Abu Gallum Multiple Use Management Area, which presently cover the whole of the Gulf of Aqaba coastline. Ras Mohamed was declared a Protected Area in 1983, encompassing some of the most extensive coral reefs on the Egyptian Red Sea, and the other two followed in 1992, linked together in 1994 to form the Ras Mohamed National Park Sector (UNEP-WCMC and IUCN 2019). They were established through technical and financial assistance from the European Union (Clarke 1992) and were initially hailed as some of the most successful MPAs in the Middle East (Eid and Fawzi 1991; Shehata 1998; Ashworth and Ormond 2005). One of

the main objectives for establishing this network of MPAs was to maintain local traditional fisheries while assuring the ecological sustainability of the area's coral reefs (Pearson and Shehata 1998) and the Bedu have always been permitted to fish in MPAs outside of No-Take Zones (NTZs), with restrictions on net mesh size (Galal 1999). However, although there was a concerted effort to establish appropriate environmental legislation, the development of coastal tourism overtook the rate at which effective baseline studies and environmental impact assessments could be completed and conservation measures were quickly compromised to cope with the steep rises in tourism demand (Shackley 1999; Shaalan 2005; Jobbins 2006; Tabet and Fanning 2012). Nowadays, the administration and management of most protected areas in Egypt is widely regarded as ineffective, barring a few exceptions, and many designated sites do not exist except on paper as management is virtually non-existent. For example all MPAs except Ras Mohammed lack detailed management and annual operational plans and budgets (Abd-Alah 1999; Rouphael et al. 2015).

Table 2.5. Egyptian Red Sea Marine Protected Areas

	Year established	IUCN category	Area / km^2	Marine area / km^2
Ras Mohammed National Park	1983	II	850	337
Elba Multiple Use Management Area	1995	VI	35,600	2,330
Nabq Multiple Use Management Area	1996	VI	600	122
Abu Gallum Multiple Use Management Area	1996	VI	400	320
Wadi El Gemal–Hamata National Park	2003	II	7,450	2,100
El-Salum Marine Protected Area	2010	NA	383	293

(UNEP-WCMC and IUCN 2019)

The main management objectives developed by the EEAA for the Nabq MPA in a provisional management plan (Mabrouk 2007) were to: (1) maintain the natural and cultural resources and conserve Nabq MPA biodiversity; (2) enhance the sustainable use of natural resources in the MPA;

(3) promote Nabq MPA as a focal point for ecotourism in the region, and support socio-economic benefits to the local community and the nation; and (4) increase public understanding and appreciation of Nabq MPA natural and cultural heritage. The marine portion of Nabq MPA falls under four management zones of varying protection levels: strict natural zone, NTZ, recreational zone and multiple-use zone; activities in the Dahab section of the MPA are unregulated (Mabrouk 2007); Abu Gallum MPA still lacks a formal management plan (Table 2.6).

Table 2.6. Status of Nabq and Abu Gallum MPAs

	Abu Gallum	**Nabq**
Declared	1992: Law 102/1983, Prime Ministerial Decree 1511/1992 adjusted by Decree 33/1996 to cover the Gulf of Aqaba coastline and include Dahab MPA within Nabq Multiple Use Management Area	
Staff	15	19
Infrastructure	Visitor centre, staff house, pickup, no boat	Boat, pickup, visitor centre, 3 staff houses, handicraft workshops
Funding	c. US$ 9,500 y^{-1}	
Entry Fee	c. US$ 3 Foreigner / c. US$ 1 Egyptian	
Resident population	< 50 (pers. obs.)	452 (Mabrouk 2007)
Mean no. fishermen	10 day^{-1}	20 (40 in summertime) day^{-1}
Tourism potential	400 day^{-1}	18,000 y^{-1} 3,000 $month^{-1}$ (Mabrouk 2015)

(Samy et al. 2011)

The *Mzeina* Bedu's fishing activity is focussed along the coastline from the northern border of Ras Mohamed National Park, to the southern border of the Tarabin tribe's territory in Nuweiba (Figure 2.1; Figure 2.2) thus their traditional fishing grounds are totally encompassed by the Nabq and Abu Gallum MPAs. Although these MPAs were established with consideration for Bedouin traditional fishing activities, there was a disconnect between the aspirations of MPA managers on the ground and the wider political climate in Egypt, which focused on tourism development (Sowers 2007). This meant insufficient consideration for Bedouin livelihoods, leaving them marginalized with a poor understanding of environmental issues and being unsupportive of MPA regulations (El Bastawisi 1995; Gladstone et al. 1999; Cabahug 2002; Marshall et al. 2010; Tabet and Fanning 2012). The Bedouin community was consulted on the demarcation of NTZs, but it is not clear if they were chosen with the objective to appease the Bedu or to actually optimize marine biodiversity conservation impacts (Hansen et al. 2011; Teh et al. 2013).

It was widely recognized to me among the *Mzeina* that compliance within MPAs was low and many Bedu felt that they had not been sufficiently involved in determining the regulations. The Bedu acknowledged accountability for their fishing behaviour to the National Park Sector and God with no responsibility or support from Bedouin tribal justice systems. When Bedu have been caught fishing illegally, National Parks officers would confiscate any illegal gear and issue a fine, but patrols were rare and easy to avoid. Older fishers believed that God was watching them and so they must follow the regulations and thus were angered by younger fishers who did not respect the rules (Table 2.1). Younger fishers said that they saw others breaking the rules, for example, using spearguns, and so they just did the same to ensure their catch was not disadvantaged.

State regulations that were widely known by the Bedu included a minimum net mesh size of 'three fingers' and prohibition of spearguns and fishing using SCUBA. Such restrictions have been shown to significantly increase fish biomass (Cinner et al. 2006; McClanahan et al. 2006). Bedu also help the National Park Sector on beach clean-ups (Table 2.1). The capture of Siganidae is prohibited

during the peak spawning season from June to August (Popper and Gundermann 1975), a rule created by the National Park Sector. According to local custom and food preferences, small reef fish such as Chaetodonidae are always thrown back; turtles are never caught intentionally although sharks are occasionally consumed (Table 2.3). During the Israeli occupation, it was not permitted to collect spider conch *(Lambis lambis)*, giant tritons *(Charonia tritonis)* or Panaluridae (Table 2.1). It was felt that the Israelis cared much more for the marine environment and its resources than the Egyptian State and when the Israelis left, they told the Bedu to 'take great care of the unique marine environment and freshwater sources in South Sinai' (Table 2.1).

Although the NTZs in Nabq MPA initially appeared to be effective, given there was increased target species'abundance on the reef within their boundaries and increased Catch Per Unit Effort (CPUE) outside (Ashworth 2004; Ashworth et al. 2004; Ashworth and Ormond 2005); over the years, fishing pressure and non-compliance has increased with a corresponding decrease in abundance of target species and CPUE (Galal et al. 2012; Mabrouk 2015; Advani et al. 2015). Poor waste management and pollution have also been noted (Mona et al. 2019). Fishing along the coastline of Dahab remains unregulated and has been shown to cause a conflict with watersports activities (Ashworth and Ormond 2005). Improved enforcement and a stronger participatory approach to management have been recommended to resolve these issues (Advani et al. 2015; Mabrouk 2015). Although MPAs are often justified on the grounds of delivering fishery benefits, these benefits are rarely realized within the time periods expected by local stakeholders, if at all, leading to poor compliance with regulations in the long-term (Oracion et al. 2005; Kareiva 2006; McClanahan et al. 2008). Where dependency on marine resources is very strong, the incentives to continue extractive activities within NTZs for reasons such as cultural needs, subsistence, or poverty, often outweigh disincentives such as fines or jail, and so compliance will likely be low (Marshall et al. 2010).

A further complication is created by the fact that Bedu are permitted to fish within MPAs using 'traditional methods' without any clear definition of what constitutes a 'traditional method'.

Problematically, 'traditional' fishing is often considered benign when in reality it can often have serious ecological impacts (Russ and Alcala 1989; Hawkins and Roberts 1992; Marsh et al. 2004; Mangi and Roberts 2006). Fishing methods such as use of nylon nets or boats with outboard motors can be considered 'traditional' when this development has only been introduced in recent decades (Rouphael et al. 2015). Additionally, Bedu are likely to continue to utilize new fishing tools in the future in order to maximize their CPUE (Sheppard 2014) and until 'traditional' fishing gears are clearly defined, enforcement surrounding them will continue to be problematic (Rouphael et al. 2015). Some studies have suggested that artisanal fishing activity in the Gulf of Aqaba could negatively impact coral reefs and fish communities (Ammar et al. 2011; Galal et al. 2012; Advani et al. 2015; Mabrouk 2015), although others suggest these impacts are minimal, especially compared to those from tourism development (Naumann et al. 2015).

South Sinai's MPAs have been chronically underfunded since the end of the initial projects to establish them from 1993 until 2002, which were estimated to cost close to US$ 20 m (Cesar 2003) with follow-up funding from the European Union South Sinai Regional Development Programme (2005–2010) of around US$ 7.5 m (Mabrouk 2015). It has been estimated that globally, MPAs cost an average of US$ 2,698 km^{-2} y^{-1} to run effectively, with costs increasing with proximity to human settlements (Balmford et al. 2004). Nabq currently receives US$ 11.66 km^{-2} y^{-1} and Abu Gallum US$ 17.50 km^{-2} y^{-1} (Table 2.6), and it is not clear if even this support is even still allocated. This level of funding is derisory when compared to the estimated income from Egypt's reefs (US$ 472.5 M y^{-1} from tourism and US$ 40.4 M y^{-1} from fisheries (Cesar 2003)). In contrast to the relatively successful user fee schemes of Ras Mohammed (Cesar 2003, Jobbins 2006) and St Catherines, Nabq and Abu Gallum rarely charge visitors because of a lack of staff and internal corruption, although Nabq does collect fees from official visitors arriving from Sharm el Sheikh (pers. obs.). Fines for violations are remitted directly to the central fund of the NCS (Samy et al. 2011) so the money is not necessarily available locally. Options for non-governmental organization (NGO) support are limited in Egypt as

the process of establishing civil society organizations is extremely convoluted and they are seen as a threat to the state (Herrold and Atia 2016). There are no signs that this will change in the near future, and in fact, the situation is worsening (Shaalan 2005; Gilbert 2009). Specifically, the absence of participatory planning processes and accompanying conflict prevent any resolution mechanisms between sectors which is a major barrier to progress in addressing coastal zone conflicts (Tabet and Fanning 2012). Incorporation of Bedouin TEK and socio-cultural needs into the management of South Sinai's coastal ecosystem and fisheries management through participatory planning is thus essential to ensure a better understanding of the socioecologically-dynamic situation and wider acceptance of regulations.

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Appendix 2.1. Bedouin Arabic names of reef organisms

Bedouin name	**Common name**	**Latin name**
Nagil	Coral grouper, Malabar grouper	*Plectropomus spp., Epinephelus spp.*
Ota Shegah 'sister'	Peacock grouper	*Cephalopholis spp.*
Sheenia	Spotted Grouper	*Epinephelus spp*
Dakisi (Dahab), Nagil (Nuweiba)	Greasy Grouper	*Epinephelus tauvina*
Boursoula	Marbled Grouper	*Epinephelus spp*
Taween	White blotched Grouper	*Epinephelus spp*
Dharana	Blacktip Grouper	*Epinephelus fasciatus*
Gateren	Sweetlips	Haemulidae
Shahoor	Gibbus Sweetlips	*Plectorinchus gibbosus*
Shahoor, Mesina	Snappers	Lutjanidae
Abu raghedi	Bluelined snapper	*Lutjanus coeruleolineatus*
Asimohne	Humpback snapper	*Lutjanus gibbus*
Harameya	Jobfish	*Aprion virescens*
Shahoor	Longnose emperor	*Lethrinus olivaceus*
Mesina	Emperors,Seabreams	Lethrinidae, Sparidae
Gamar 'moon'	Bigeye emperor	*Monotaxis grandoculis*
Amoreh	Goatfishes	*Mullidae*
Sebloheh	Yellow saddled goatfish	*Parupeneus cyclostamus*
Harari, Tahalah	Wrasses	Labridae
Fadeh	Blue razor wrasse, Hogfishes	*Iniistius pavo,* Bodianinae
Robadea	Broomtail wrasse	*Cheilinus lunulatus*
Hagab	Sand wrasses	Corinae
Zibrebeah 'penis'	Cigar wrasse	*Cheilio inermis*
Mesi	Klunzinger's wrasse	*Thalassoma klunzingeri*
Abu zangor	Bird wrasse	*Gamphosus caeruleus*
Gaban	Humphead Wrasse	*Cheilinus undulatus*
Fadah	Parrotfishes	Scaridae
Harid, Warga	Longnose parrotfish	*Hipposcarus harid*
Shaaram	Triggerfishes	Balistidae
Hegem	Picasso Triggerfishes	*Rhinecanthus assasi*

Bedouin name	**Common name**	**Latin name**
Saheli	Surgeonfishes	Acanthuridae
Geneah	Small surgeonfishes	Acanthuridae
Rahawa	Unicornfishes	Nasonidae
Harafi	Rabbitfishes	Siganidae
Shizen	Red sea rabbitfish	*Siganus rivulatus*
Bayadaat	Trevallies	Carangidae
Bahra	Fusiliers	Caesonidae
Robugh	Twobar seabream	*Acanthopagrus bifasciatus*
Shahoor	Rudderfishes	*Kyphosidae*
Zaguuri	Rainbow runner, Mackerels	*Elagatis bipinnulata,*
		Scombridae
Balamiper	Tunas	Scombridae
Agama	Barracudas	*Sphyraena spp.*
Zumacat musa	Flounders	Bothidae
Raghedeh	Lizardfishes, Crocodilefishes	Synodontidae
		Platycephalidae
Kharem	Halfbeaks	Hemiramphidae
Balooleh	Soldierfishes	Holocentridae
	Cardinalfishes	Apogonidae
Abu alane	Bigeyes	Priacanthidae
Rogotoi	Crescent-banded grunter	*Terapon jarbua*
Ghasbah	Cornetfishes	Fistulariidae
Arabi, Bouri (Egyptian)	Mullets	Mugilidae
Tahamal	Rudderfishes	Kyphosidae
Mamomosei	Rays	Batoidea
Abu nefra	Porcupinefishes, Pufferfishes,	Diodontidae,
	Boxfishes	Tetraodontidae, Ostracidiae
Rogoleh	Damselfishes	Pomacentridae
Mushti	Batfishes, Angelfishes,	Ephippidae, Pomacanthidae,
	Butterflyfishes	Chaetodontidae
Oranhi	Lionfishes	*Pterois spp.*
Abu leban	Stonefishes	Scorpaenidae

Bedouin name	**Common name**	**Latin name**
Jowba, Igmun (large ones)	Moray eels	Muraenidae
Abu galumbo	Crabs	Brachyura
Hanarbish	Hermit crabs	Diogenidae
Gernen	Spiny lobsters, Slipper lobsters	Palinuridae, Scyllaridae
Saefila	Cuttlefishes	*Sepia spp.*
Garneta	Octopuses	*Octopus spp.*
Busra	Giant clams	*Tridacna spp*
Goob	Giant triton	*Charonia tritonis*
Wadeh	Cowries	Cypraeidae
Jaas	Topshells	Trochidae
Hybol	Three horned strombus	*Strombus tricornis*
Ardfar	Spider conch	*Lambis lambis*
Leseg	Limpets	Patellidae
Recba ajooze ‘old woman’s knee’	Chitons	Chitonidae
Besei	Marine turtles	Chelonidae

CHAPTER 3. FISH ASSEMBLAGES ON SOUTH SINAI REEFS ALONG A FISHING PRESSURE GRADIENT

Abstract

Coral reef fisheries are primarily located primarily in developing countries and are widely overexploited, but the socioeconomic status and cultural needs of reef fishers can make it ethically-difficult for managers to restrict fishing activity. Furthermore, the diversity of fish species and the complexity of trophic interactions on coral reefs generate complications when attempting to identify the ecological impacts of fishing pressure. The coral reefs of South Sinai have been fished by the *Mzeina* Bedu for generations, but increases in demand due to migration to the coast and exponential tourism growth in recent decades have resulted in a rapid rise in fishery pressure on the region's coral reefs. This work aimed to evaluate reef fisheries assemblages using underwater visual census along gradients of fishing pressures, based on accessibility of fishing sites. South Sinai's reefs were found to host total fish biomass comparable to heavily fished sites in other regions of the Indo-Pacific. Target fish biomass (R^2=0.517, F(1,6)=6.430, p<0.05), total fish abundance (R^2=0.533, F(1,6)=6.846, p<0.05), mean individual weight of target species (R^2=0.684, F(1,6)=13.000, p<0.05) and % contribution of herbivores to total biomass (R^2=0.560, F(1,6)=7.630, p<0.05) were all negatively correlated with fishing pressure, providing evidence that fishing was severely impacting the natural structure of the reef fish community. Fish biomass (Univariate analysis of Variance F(1)=27.882, p<0.001) and species richness (t(4)=-3.818, p<0.05) were found to be higher during the summer, the *Mzeina's* traditional fishing season and higher in deeper water (biomass: Univariate Analysis of Variance: F(1)=6.773, p<0.01; species richness: t(2)=4.992, p<0.05), suggesting a depth refugia effect from fishing. The too-small fishery closures and poor enforcement that have characterized the management of South Sinai Marine Protected Areas to date are unlikely to be sufficient to arrest the apparent decline in fishery resources. Large fishery closures are essential in order to promote recovery to biomass and species richness that approach unfished levels which would likely take decades, even under perfect enforcement. If large fishery closures are unacceptable to the *Mzeina*, then gear restrictions could be a viable alternative, but would require strict self-regulation due to the length and remoteness of the coastline. Development of supplementary livelihoods, to increase socioeconomic resilience of the *Mzeina* and reduce their dependence on reef fisheries should be a component of any proposed fisheries management programme.

3.1. Introduction

Coral reef fisheries account for a seemingly insignificant proportion (2–5 %) of global catch (Pauly et al. 2003; Johnson et al. 2012), however, they are labour-intensive and focussed in developing countries, providing animal protein and income to millions of people in the world's poorest communities (Russ 1991; Moberg and Folke 1999; Whittingham et al. 2003; Sadovy 2005). In these situations, local socioeconomic pressures often mean that immediate food security takes priority over long-term sustainability and reef fisheries are widely overexploited (Pauly et al. 2002; Pandolfi et al. 2003; Bellwood et al. 2004). Some estimates suggest that current total landings of coral reef fisheries are 64 % higher than can be sustained (Newton et al. 2007) and there is widespread documentation of severe declines in reef fish biomass due to fishing mortality (Williams et al. 2011; Valdivia et al. 2017). In many cases, inadequate monitoring, unreliable baselines and the absence of reference points for management mean that managers are reluctant to make the socially and politically difficult decision to restrict fishing activities before the situation becomes critical (McClanahan et al. 2011).

A significant obstacle to the sustainable management of coral reef fisheries is the lack of good information on fish population abundance at spatial scales commensurate with the exploitation of the resource (Hilborn et al. 2005; Cinner et al. 2012). Direct and indirect influences of fishing mortality on coral reef ecology are complex due to the diversity of species, the complexity of trophic interactions and the extended time scales over which degradation may manifest itself (McClanahan 1995; Hughes et al. 2010; McClanahan et al. 2011). One approach to tease apart these impacts is to compare the structure of fish communities using fishery-independent sampling across time or space to compare areas subjected to varying levels of fishing pressure (Russ and Alcala 1989; McClanahan et al. 2007). Coral reef fisheries have been shown to influence fish density (Alcala 1988), biomass (Polunin and Roberts 1993), species richness (Nañola et al. 2011), community composition (Pet-Soede et al. 2001) body size (Graham et al. 2005; Wilson et al. 2010) and relative abundance of trophic guilds (Friedlander and DeMartini 2002; Campbell and Pardede 2006). Reefs closed to

fishing tend to exhibit from twice up to 10 times the biomass of targeted fishes when compared to neighbouring fished areas (Russ and Alcala 1989, 2003; Polunin and Roberts 1993; Friedlander et al. 2007; McClanahan et al. 2007).

When a reef is initially subjected to fishing pressure, species diversity is reduced as fishers generally prefer to target larger-bodied individuals, which are predominantly predatory species (Russ and Alcala 1989). As fishing pressure increases, the density of target fish decreases, leading to a downward trend in the trophic level of targeted species (Pauly et al. 1998b; Pauly and Palomares 2005). Thus the trophic composition of catches slowly changes, with predatory Chondrichthyes (sharks and rays), Serranidae (groupers) and Lutjanidae (snappers) being replaced by opportunistic and lower trophic level species (Koslow et al. 1988; Jennings and Polunin 1996, 1997; Jennings and Kaiser 1998; Friedlander and DeMartini 2002; Bhathal and Pauly 2008). For example, more than 54 % of the total fish biomass in the unfished remote Northwestern Hawaiian Islands consisted of apex predators, whereas this trophic level accounted for less than 3 % of the fish biomass in the more heavily exploited Main Hawaiian Islands (Friedlander and DeMartini 2002) and reef shark populations have declined in Chagos Archipelago by 90 % sincee the 1970s (Graham et al. 2010). Often changes may be imperceptible day to day and can only be elucidated from the knowledge and experience of lifetime fishers, historical photographs or archaeological evidence (McClenachan 2009; McClanahan and Omukoto 2011). It is essential to document such information, not only to monitor declines in fishery target stocks, but also because loss of top predators can have ramifications for the rest of the ecosystem, including trophic cascades (Myers and Worm 2003; Bascompte et al. 2005; Heithaus et al. 2008; Ruttenberg et al. 2011; Walsh et al. 2012), changes in the timing of prey life history parameters (DeMartini et al. 2005), and a reduction in key ecosystem functions (Worm et al. 2002).

Generally, hook-and-line fisheries target large-bodied fishes and piscivores (Mumby et al. 2014), nets catch a diversity of fishes (Mangi & Roberts 2007), and spearguns and traps target herbivores

and piscivores (Hawkins et al. 2007; Frisch et al. 2012). As predatory species decline, herbivorous fishes become the new targets, and fishers may transition from hook-and-line to other, less selective gears such as nets and traps (Selgrath et al. 2018). On reefs, algal grazing by herbivores plays an important role by mediating competition between corals and macroalgae; when herbivores are overfished, macroalgae dominate the substrate, smothering and killing corals by preventing photosynthesis leading to a transition from a coral-dominated substrate to an algal-dominated substrate (Lewis 1986; Mumby et al. 2006; Fox and Bellwood 2007; Jessen et al. 2012; Cheal et al. 2013; Jessen and Wild 2013). The loss of herbivorous fishes through fishing has been shown to trigger these 'coral-algal phase shifts' on coral reefs worldwide (Hughes 1994; Hughes et al. 2007) and there are almost no documented cases of reversals of this phenomenon (Graham et al. 2013). Loss of herbivorous fish can also lead to reduced competition with other algivores such as urchins, resulting in increases in their populations and damaging bioerosion of corals (McClanahan et al. 1994). Furthermore, macroalgal overgrowth can be compounded by eutrophication from coastal development as fishing communities grow along the coast (Naumann et al. 2015; Wenger et al. 2015).

In addition to fishing impacts, localized differences in coral reef fish communities can result from a number of environmental factors including depth, geomorphology, hard coral cover, reef complexity and productivity (Luckhurst and Luckhurst 1978; Roberts and Ormond 1987; Roberts et al. 1992; Newman and Williams 2001; Wilson et al. 2008, 2009; Pinca et al. 2012; Williams et al. 2015; Heenan et al. 2016; Darling et al. 2017; Richardson et al. 2017; Smallhorn-West et al. 2017). Faced with multiple stressors of increasing magnitude over the last 30–40 years, coral reef health has significantly declined worldwide with substantial (50–75 %) reductions in living coral cover (Gardner et al. 2003; Bruno and Selig 2007; Schutte et al. 2010; De'ath et al. 2012; Jackson et al. 2014; Hughes et al. 2018) and loss of associated structural complexity (Öhman and Rajasuriya 1998; Alvarez-Filip et al. 2009, 2011). As coral cover and habitat complexity decline, so does the diversity

of associated fish and invertebrate assemblages (Öhman and Rajasuriya 1998; Jones et al. 2004; Idjadi and Edmunds 2006; Pratchett et al. 2008, 2018). Specifically, reef degradation alters food availability (Tootell and Steele 2016; Hempson et al. 2017) and reduces the diversity and density of refuges for prey (Hixon and Beets 1993; Giffin et al. 2019). These changes affect predator-prey interactions and can have cascading impacts on food webs and fisheries productivity (Rogers et al. 2018). Thus, the impacts of fishing on reef fishes may be exacerbated by environmental impacts on habitat quality from pollution (Baum et al. 2015), coral bleaching (Pratchett et al. 2018), diver damage (Hawkins and Roberts 1997), ship grounding and anchor damage (Giglio et al. 2017) and destructive fishing gears (Chiappone et al. 2005). The composition of reef fish assemblages has also been found to change with depth; herbivores are often most abundant on the reef flat while predatory species inhabit deeper waters on the reef slope (McGehee 1994; Friedlander and Parrish 1998; Brokovich et al. 2006, 2008; Ayalon et al. 2010). Within protected areas, the effect of protection has been noted to be more pronounced in shallow waters than in deep, assumed to be due to increased fishing pressure in shallow waters (Polunin and Roberts 1993; Ashworth and Ormond 2005; Tyler et al. 2009). Seasonal effects on reef fish populations have also been noted (Sale and Douglas 1984; López-Pérez et al. 2013).

The coral reefs of South Sinai are globally noted for their species diversity (Stehli and Wells 1971; Golani and Bogorodsky 2010; Golani and Fricke 2018) and endemism (Ormond and Edwards 1987). *Mzeina* Bedouin tribes have fished these reefs for centuries, catching a diversity of coral reef fish species with a variety of artisanal gears (Galal 1999; Table 2.3). Traditionally, the *Mzeina* seasonally migrated to the coast during the summer to fish and returned to their nomadic pastoralist lifestyles in the wintertime (Glassner 1974). However, in recent years, as a result of pressure from the Egyptian state and the development of income opportunities from the growth of international tourism, the *Mzeina* have become increasingly sedenterized in the towns of Dahab and, to a lesser extent, Nuweiba (Homa 2007). This increase in coastal population, coupled with increased demand from

tourism, has resulted in a rapid increase in the exploitation of coral reef fishery resources in the area (Chapter 2).

To date, fisheries management approaches have been focussed on spatial restrictions, through Nabq MPA and Abu Gallum MPA, both established in 1992 when a network of no-take zones was established and only the Bedu were allowed to fish using 'traditional methods' in other areas along the coastline (Mabrouk 2007). However, although these measures showed some promise in their early years (Galal et al. 2002; Ashworth 2004; Ashworth et al. 2004; Ashworth and Ormond 2005), a lack of capacity and funding for enforcement eventually resulted in reduced effectiveness, measurable fisheries impacts, and sustainability concerns (Samy et al. 2011; Galal et al. 2012; Advani et al. 2015; Mona et al. 2019). The no-take zones were also relatively small, and unlikely to be effective for highly mobile fish species or those that exploit a wide variety of habitats during their life history (Roberts et al. 2001; Malcolm et al. 2016).

This study aimed to investigate the influence of fishing on reef fish community composition, biomass, and trophic characteristics across study sites which varied in their accessibility and convenience, and thus were likely to be subjected to varying levels of artisanal fishing pressure from the *Mzeina*. To achieve this aim, I compared various reef fish community metrics collected using Underwater Visual Census (UVC) and correlated them with the travel time from the nearest urban centre, Dahab. I also investigated the effects of season and depth to evaluate how these factors may influence with Bedouin fishing behaviours and practices.

3.2. Materials and Methods

The research was conducted at study sites along the South Sinai coastline, around the Bedouin town of Dahab (Figure 3.1) and all had been previously identified as *Mzeina* fishing grounds (Figure 2.3). Accessibility of each site varied substantially, some were accessible by tarmac road, while others could only be accessed by a 4-wheel drive car, boat or on foot and thus it was hypothesized that they

would represent a range of fishing pressures (Table 3.1). All sites except for Laguna were comparable in terms of biophysical characteristics. Laguna was a shallow gently-sloping site with a sandy-gravel substrate and intermittent coral heads; whereas all other sites were steeper slopes, dominated by hard coral, with the reef crest around 3–5 m and the bottom of the slope at 20 m or more.

Table 3.1. Accessibility of each study site to the *Mzeina*

Site name	**Means of access (travel times are approximate)**	
	from Dahab	*from Nuweiba*
Nakula Amoyed Ras Abu Gallum	4x4 via Bir Ogda (1 h) Boat (1 h)	4x4 along the coast (2 h) 4x4 via Bir Ogda (2 h) Boat (3 h)
Lighthouse Laguna	Tarmac road (10 min) On foot (10 min)	-
Um Sid	4x4 (30 min)	-
Shahira	4x4 with the last km on foot (1 h) Boat (1 h)	-
Gabr el Bint	Boat only (1.5 h)	-

The key metrics used to evaluate fish communities, ecological impacts and potential sustainable yields include biomass, species richness, body length and growth metrics (Rochet and Trenkel 2003; Hall et al. 2006; Newman et al. 2006; Taylor et al. 2015; Nash et al. 2016). The approach used here aimed to assess some of these variables through the collection of fishery-independent data (Shepherd 1984) using Underwater Visual Census (UVC) (Harmelin-Vivien et al. 1985; Thresher and Gunn 1986), applying recommendations for optimal transect size (50 m x 5 m) and number of replicates per site (10–15) from the literature (Samoilys and Carlos 2000; Sayer and Poonian 2007).



Figure 3.1. Reef fish and invertebrate survey sites along the South Sinai coastline

The majority of surveys were conducted in shallow (1–3 m) water along the reef crest using snorkelling because almost all Bedouin fishing is conducted from the shore (Galal 1999; Chapter 2) and it was thus likely that these depths would be most impacted by fishing activities. Time of day and tidal state often varied between surveys due to the logistical issues associated with access to more remote sites, which may have some influence on fish activity and thus observed density (Krumme 2009). However, all surveys were conducted at least two hours after sunrise and two hours before sunset. No local closed areas existed that could be used as control sites. Surveys were conducted at eight sites during summer and at only six sites during the winter, because of persistent bad weather. In addition three sites were also surveyed during summer using SCUBA along the 10–15 m contour on the reef slope (Table 3.2) in order to assess if depth provided any refuge from fishing impacts (Tyler et al. 2009; Goetze et al. 2011; Lindfield et al. 2016).

Table 3.2. Distribution of surveys across sites, seasons and depths

Site	Winter (2016)	Summer (2017)	Deep (10 - 15 m; 2017)
Nakula	20 Dec	12 Jul	-
Amoyed	-	18 Jul	-
Ras Abu Gallum	-	11 Jul	-
Lighthouse	11 / 16 Dec	8 Jul	30 Jul
Laguna	8 Dec	10 Jul	-
Um Sid	12 Dec	9 Jul	28 Jul
Shahira	13 Dec	15 Jul	-
Gabr el Bint	14 Dec	24 Jul	27 Jul

Fish surveys were conducted while swimming at a constant speed of 10 m min^{-1} for 5 min, resulting in a transect length of 50 m. The surveys did not employ a fixed transect line such as a measuring tape because it would be difficult to lay on the reef crest because of water movement and the resulting disturbance could potentially repel or attract fishes (Patterson et al. 2007; Dickens et al. 2011). All fishes encountered in a visually-estimated 5 m-wide band (2.5 m either side of the

observer) were recorded to species level and lengths of each individual were recorded to the nearest 5 cm. The families encountered are noted in Table 3.3 and included those both targeted and not targeted by the *Mzeina* fishery because non-target catches can still be critical for local consumption and food security (Pauly and Zeller 2016; Wamukota and McClanahan 2017). I surveyed a total of ten transects at each site, swimming a random distance of at least 10 m before starting subsequent transects to avoid overlap and pseudoreplication.

Counts of large schools (>20 individuals) were visually estimated. Female and juvenile Scaridae (parrotfishes) were occasionally impossible to identify to species level. Swimming speeds were well-practised by timing a swim along a known distance until 10 m was completed in 1.0±0.013 min (mean±SE). Length estimation was practised through the use of artificial fish silhouettes of known length until lengths were always estimated correctly (Yulianto et al. 2015). Underwater visual census of fishes is well-known to underestimate densities, particularly of cryptic species (Willis 2001) although such species are not generally targeted by Bedu (Table 3.3).

It was impossible to gauge fishing pressure through direct observation because of the remote, informal and widespread nature of fishing sites and reserved nature of Bedouin fishers. I considered using aerial imagery and direct observation to count the number of *arishas* (palm frond huts) at each site and use the count as a proxy for fishing pressure. However, at some sites such as Ras Abu Gallum, a number of the *arishas* were used for tourist accommodation or at Amoyed they were used for housing goats and pigeons. Community-level fish biomass at remote fishing sites is often comparable to that at sites closed to fishing, because fishing pressure decreases with distance to human population (Graham and McClanaham 2013; Advani et al. 2015; Campbell et al. 2018). However, *Mzeina* fishers could originate from Nuweiba, Dahab or Ras Abu Gallum or also stay overnight in *arishas* along the coast, thus complicating the calculation of distance as a proxy for accessibility. Ultimately, the metric of approximate direct travel time from Dahab (Table 3.1) was

used as the best proxy available since only minimal numbers of fishers would be likely to originate from Nuweiba (due to distance) or Ras Abu Gallum (due to the small population living there).

From the UVC, fish length estimates (TL) were converted to weights by using the allometric length-weight equation: $W=aL^b$; where a and b are constants obtained from the literature and L is length and W is Weight. The values for a and b were selected from the literature by giving precedence to estimates derived from the highest sample size (Appendix 3.1). If there were no experimental estimates available, then a Bayesian estimate was used (Froese et al. 2014). For unknown Scaridae species, a and b were estimated based on the values for the rest of the family. Where the estimates were based on Standard Length (SL) or Fork Length (FL), conversion factors were obtained from the literature (Appendix 3.1).

Statistical analysis was conducted using IBM SPSS Statistics v23. The dataset from summer 2017 was used for all between-site comparisons. Where biomass was not normally distributed, it was ln-transformed to permit the use of parametric statistical analysis (ANOVA) to test variation between sites. Other data could not be normalized through transformation, so non-parametric statistical analyses (Kruskal-Wallis tests) were utilized. Correlation analysis was performed to assess relationships between the fish community metrics and travel time from Dahab. The effects of depth and season were tested using Univariate analysis of variance for biomass and paired t-tests for species richness. Data were converted to biomass per unit area of reef (kg ha^{-1}) through simple extrapolation for comparison with other studies.

Table 3.3. Diet and fishery significance of fish families counted during UVC

Family		Primary diet	Targeted by Bedu
Acanthuridae	Surgeonfishes	Herbivore	Yes
Anthiadinae	Anthiases	Planktivore	No
Apogonidae	Cardinalfishes	Piscivore	No
Balistidae	Triggerfishes	Macroinvertivore	No
Caesionidae	Fusiliers	Planktivore	Yes
Carangidae	Trevallies	Piscivore	Yes
Chaetodontidae	Butterflyfishes	Microinvertivore	No
Cirrhitidae	Hawkfishes	Piscivore-Invertivore	No
Diodontidae	Porcupinefishes	Macroinvertivore	No
Epinephelinae	Groupers	Piscivore	Yes
Fistulariidae	Cornetfishes	Piscivore-Invertivore	Yes
Grammistinae	Soapfishes	Piscivore-Invertivore	No
Haemulidae	Sweetlips	Piscivore-Invertivore	Yes
Holocentridae	Soldierfishes and Squirrelfishes	Macroinvertivore	Yes
Labridae	Wrasses	Macroinvertivore	Yes
Lethrinidae	Emperors	Macroinvertivore	Yes
Lutjanidae	Snappers	Piscivore	Yes
Monacanthidae	Filefishes	Microinvertivore	No
Mullidae	Goatfishes	Macroinvertivore	Yes
Ostraciidae	Boxfishes	Microinvertivore	No
Pomacanthidae	Angelfishes	Microinvertivore	No
Pomocentridae	Damselfishes	Small Omnivore	No
Priacanthidae	Bigeyes	Planktivore	Yes
Pseudochromidae	Dottybacks	Microinvertivore	No
Scaridae	Parrotfishes	Herbivore	Yes
Scorpaenidae	Scorpionfishes	Piscivore-Invertivore	No
Siganidae	Rabbitfishes	Herbivore	Yes
Tetraodontidae	Pufferfishes	Omnivore	No

3.3. Results

A total of 25,150 individual fish from 105 species and 28 families were recorded during the UVC of 170 transects across 8 sites, 2 depths and 2 seasons (Appendix 3.1). One *Himantura uarnak* and one *Taeniura lymma* were seen on the transects but not included in the analysis. Total biomass across all sites, seasons and depths varied from 28 kg ha^{-1} at Laguna Bay during the winter up to 250 kg ha^{-1} at Shahira in the summer with an overall average of 132±16.0 kg ha^{-1} (mean±SE). The mean biomass in summer was 147±21.3 kg ha^{-1}, in winter 91.1±22.8 kg $ha^{-1.}$ and at depth 174±43.2 kg ha^{-1}.

Herbivores dominated the fish assemblage at all sites and the percentage composition of herbivores by biomass correlated with travel time from Dahab (R^2=0.560, F(1,6)=7.630, p<0.05; Figure 3.2; Figure 3.5). Biomass (ANOVA F(7)=8.164, p<0.001) and mean individual weight (Kruskal-Wallis H χ2(7)= 36.563, df=7, p < 0.001) of targeted species varied significantly across sites (Figure 3.3; Figure 3.4). The sites located in Dahab itself, Laguna and Lighthouse, hosted lower fish biomass and smaller fish than most other sites and Shahira had higher biomass than other sites.

Mean biomass of targeted species (R^2=0.517, F(1,6)=6.430, p<0.05), total fish abundance (R^2=0.533, F(1,6)=6.846, p<0.05) and mean individual weight of target species (R^2=0.684, F(1,6)=13.000, p<0.05) correlated with travel time from Dahab (Figure 3.5). Mean biomass of target species was also strongly correlated with total fish biomass (R^2=0.738, F(1,6)=16.942, p<0.01).

At the sites that were surveyed using SCUBA, depth was found to have a significant effect on total biomass, with higher biomass (Univariate Analysis of Variance: F(1)=6.773, p<0.01; Figure 3.6) and species richness (Paired t-test: t(2)=4.992, p<0.05; Table 3.4) on the deeper surveys. At sites that were surveyed during summer and winter, it was found that overall, fish biomass (Univariate analysis of Variance F(1)=27.882, p<0.001; Figure 3.7), and species richness (excluding Lighthouse; Paired t-test: t(4)=-3.818, p<0.05; Table 3.4) were higher during the summer.

Table 3.4. Reef fish species richness at each site, season and depth

Site	Winter (2016)	Summer (2017)	Deep (10 - 15 m; 2017)
Nakula	43	50	-
Amoyed	-	50	-
Ras Abu Gallum	-	41	-
Lighthouse	61	33	59
Laguna Bay	26	47	-
Um Sid	29	35	67
Shahira	29	43	-
Gabr el Bint	41	48	98

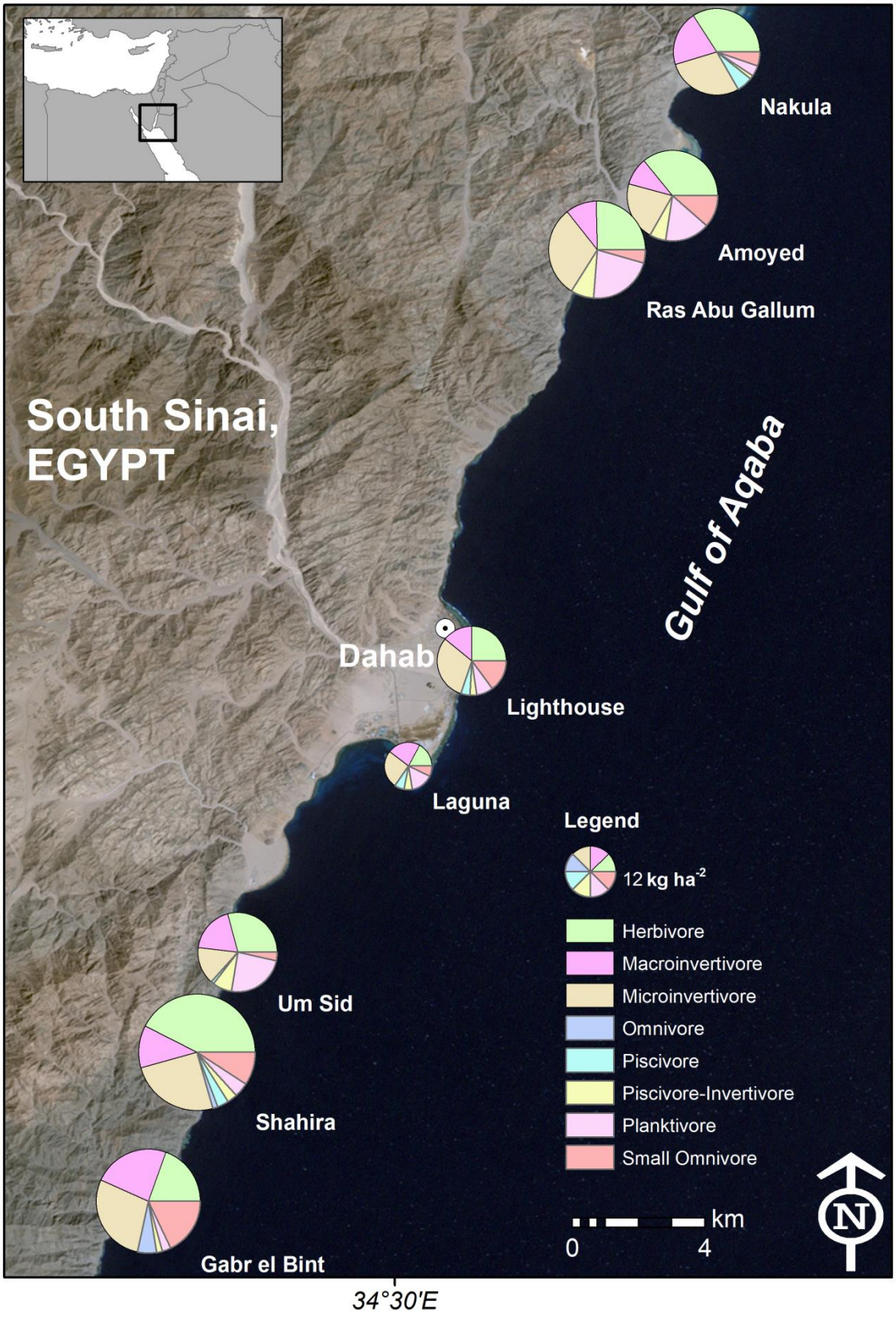


Figure 3.2. Distribution of trophic groups by biomass across study sites in South Sinai

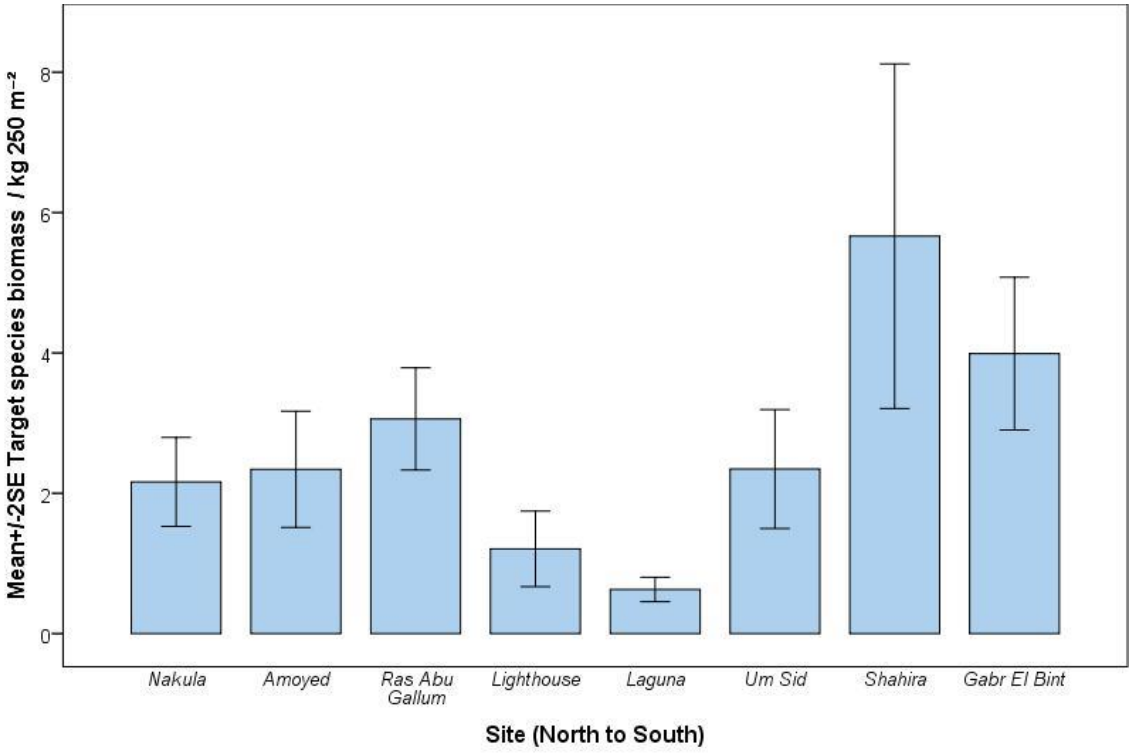


Figure 3.3. Mean biomass of targeted reef fishes at each site in South Sinai

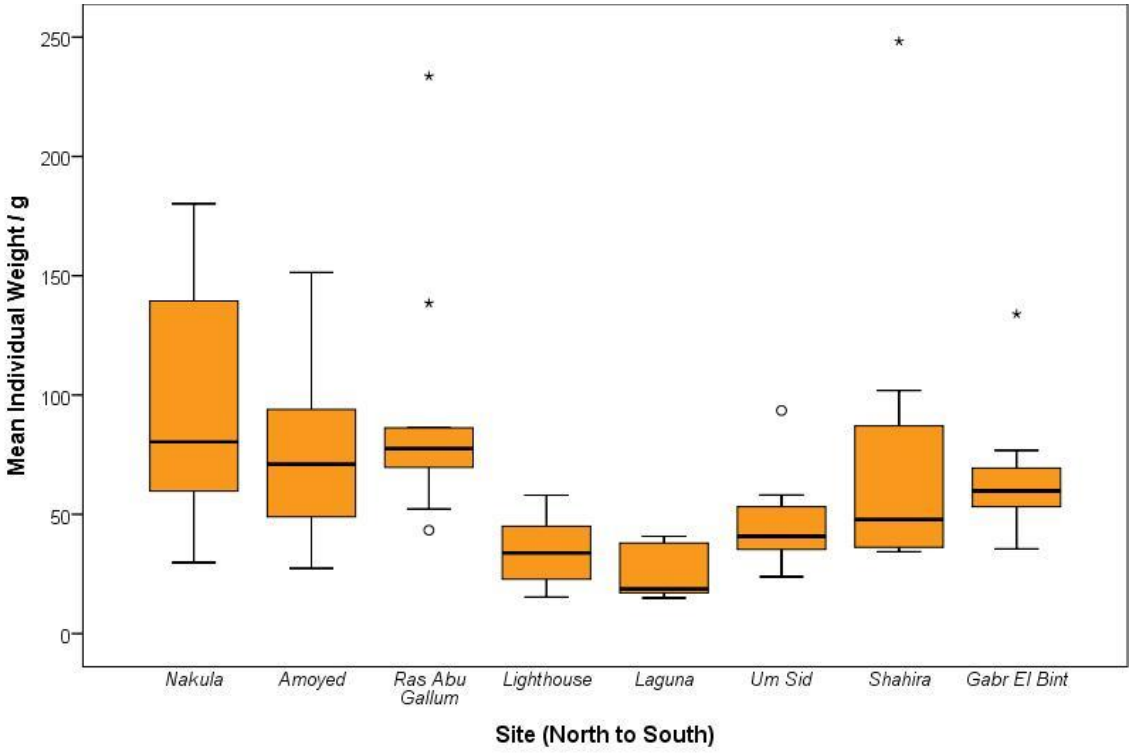
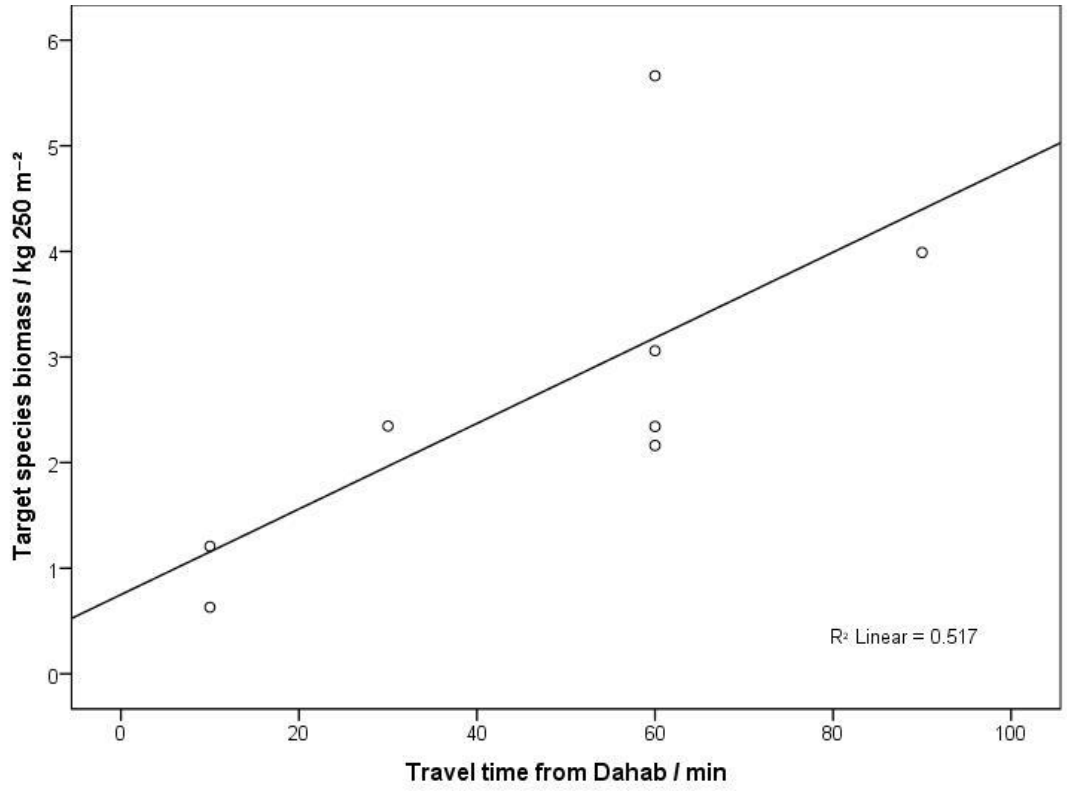


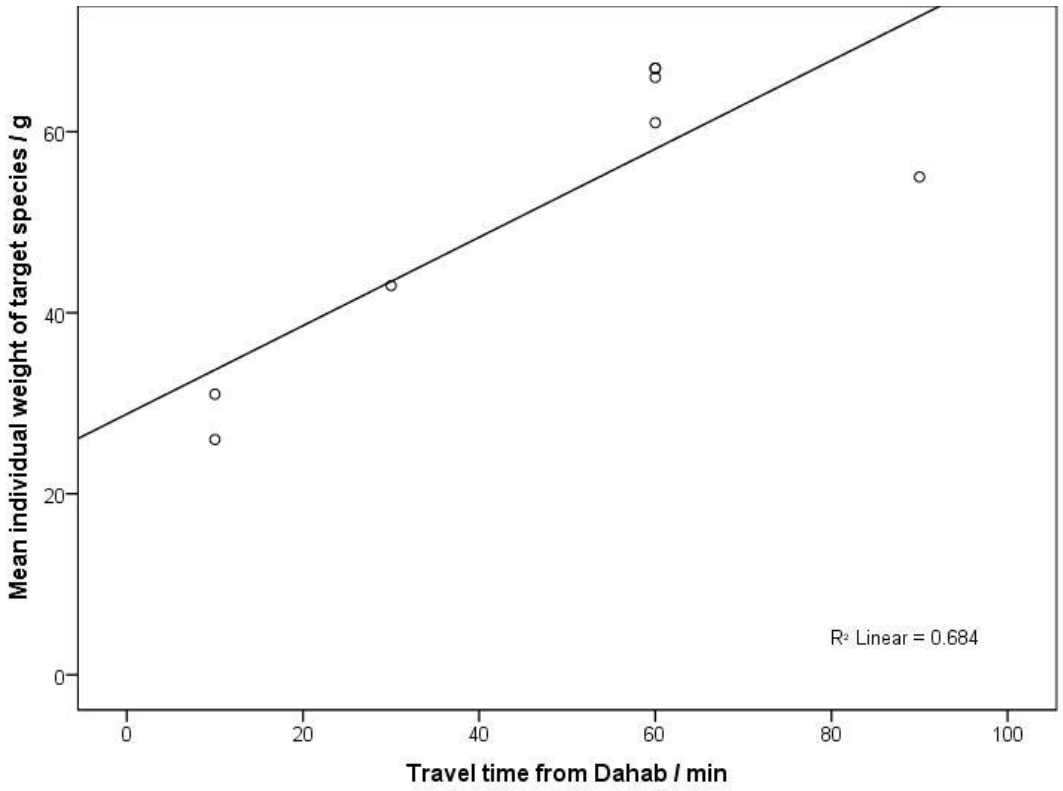
Figure 3.4. Mean individual weight of targeted reef fishes at each site in South Sinai

Boxplots show median ± quartiles and outliers

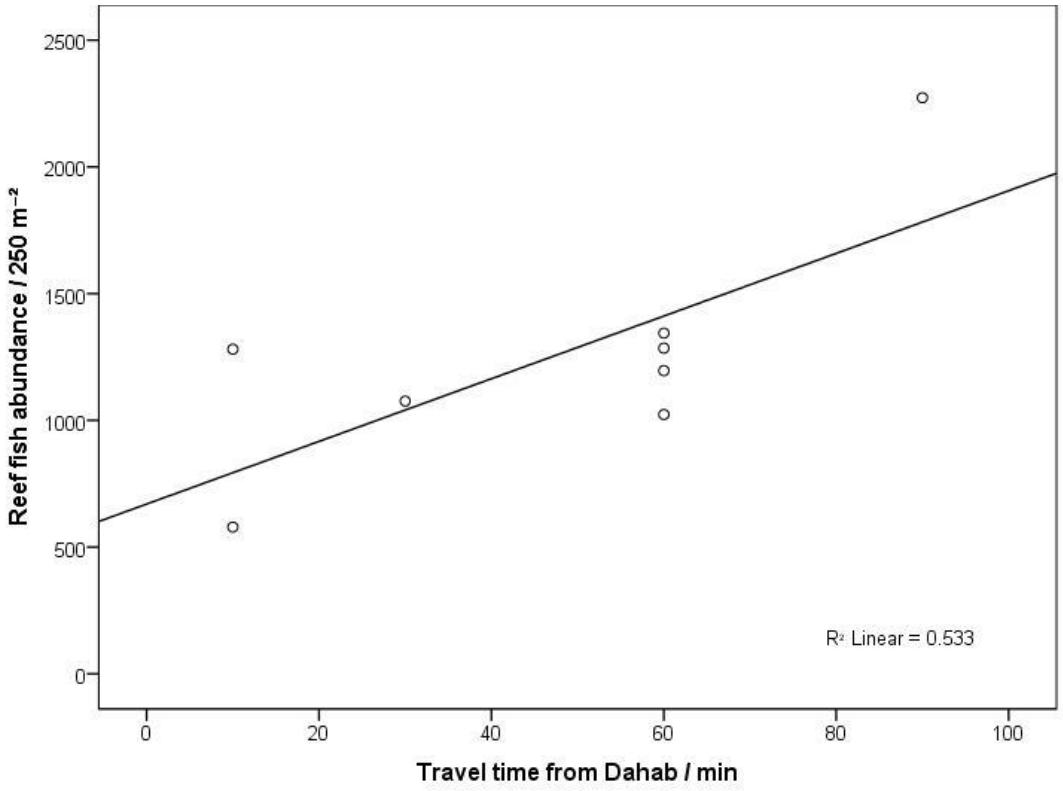
(a) Target species biomass



(b) Mean individual weight of target species



(c) Reef fish abundance



(d) Percentage contribution of herbivore biomass to total biomass

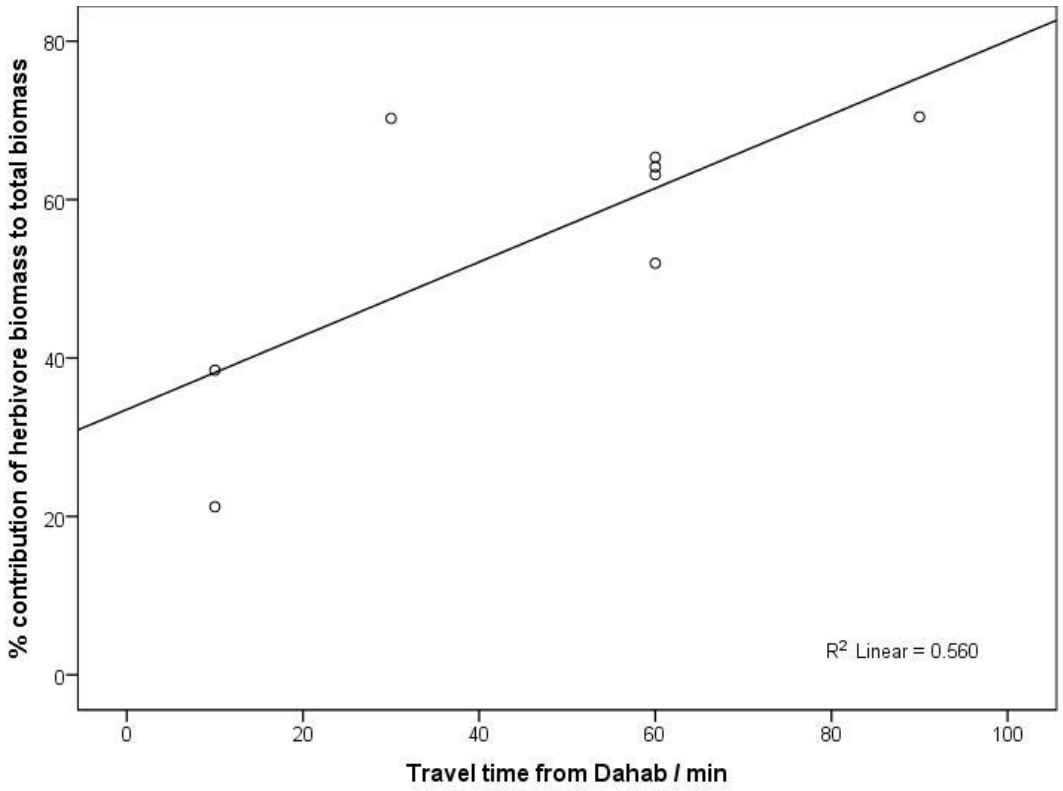


Figure 3.5. Correlations between travel time and various reef fish community metrics

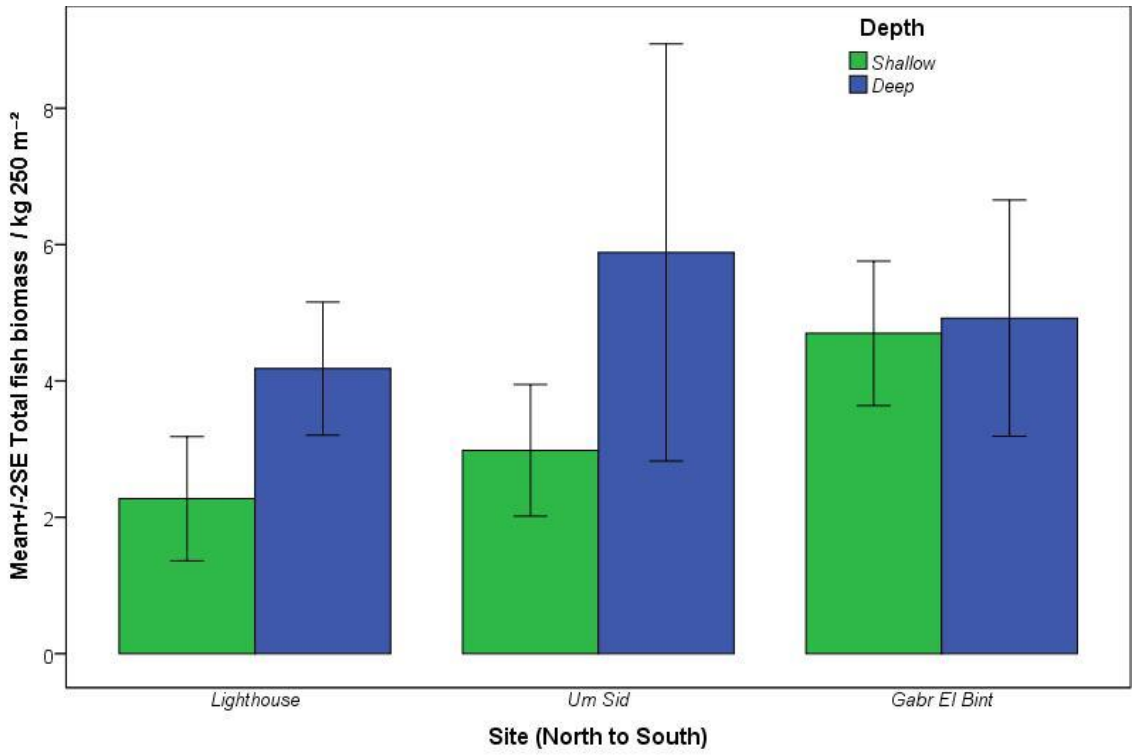


Figure 3.6. Effect of depth on reef fish biomass at Bedouin fishing sites in South Sinai

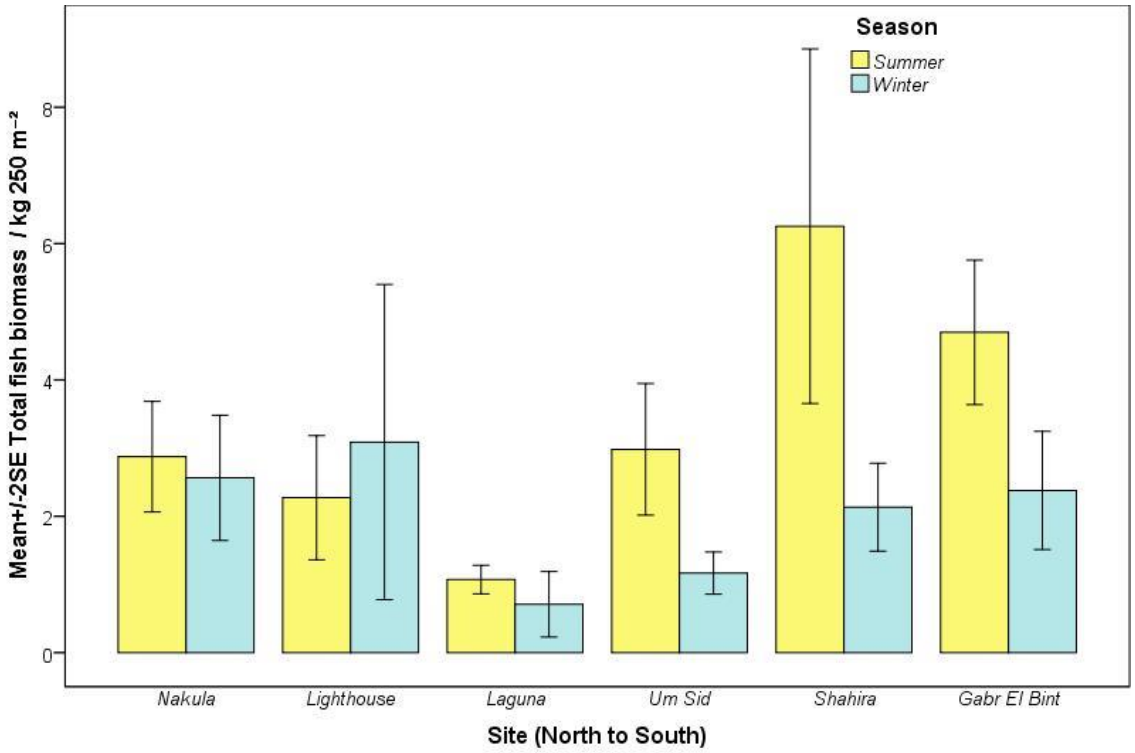


Figure 3.7. Effect of season on reef fish biomass at Bedouin fishing sites in South Sinai

3.4. Discussion

This research has shown that shallow South Sinai reefs support a mean fish biomass of 132±16 kg ha^{-1} and reef fish biomass is driven by herbivore biomass. Target fish biomass was correlated with total fish biomass, indicating that fishing impacts not only the targeted species but also the wider fish assemblage through ecosystem effects. Targeted fish biomass, total fish abundance, mean individual weight of target species and % composition of herbivores increased with the inaccessibility of the fishing site. Fish biomass and species richness were also found to be lower in shallow water and during the winter.

The cascading impacts of reef fisheries often reflect the state of coral reef ecosystems (McClanahan et al. 2011; Karr et al. 2015; Soler et al. 2015), therefore fish biomass is one of the better metrics to assess fishing pressure, and ecosystem status (MacNeil et al. 2015; Cinner et al. 2016). Heavily fished reefs in Kenya have been found to host fish biomass of 150 kg ha^{-1} (McClanahan and Graham 2005) and reefs located outside MPAs in Madagascar support 251–665 kg ha^{-1} (Gilchrist et al. 2020). Thus the biomass estimates from this study are characteristic of unprotected Indo Pacific reefs under sustained fishing pressure. Although it should be noted that the fringing reef in Dahab is very narrow, and could naturally have lower fish biomass, simply due to the reduced diversity of habitat niches. The average biomass of Egyptian reefs has previously been assessed at 552.73 kg ha^{-1} (Cinner et al. 2016), although this figure was estimated from southern Red Sea reefs which generally have higher coral cover (Kotb 2004).

Fishing pressure on coral reefs is known to reduce total fish biomass and average individual size (Chapman and Kramer 1999; Friedlander et al. 2002; Chirico et al. 2017; Nugraha et al. 2017; Campbell et al. 2018; Friedlander et al. 2018). Mean individual weight and total biomass of targeted species was negatively correlated with accessibility to fishers, indicating that assemblages on reefs with higher fishing pressure were generally composed of fewer and smaller individuals from lower trophic levels. These findings strongly suggest that Dahab's reef fish stocks are suffering from

growth-, recruitment- and ecosystem-overfishing (McManus 1997). Unless stocks are heavily overfished, fishers tend to be highly selective, targeting larger-bodied individuals, through mesh size (nets and traps), hook size and bait choice, or visual selection while spearfishing (Ralston 1990; Dalzell1996; McClanahan and Mangi 2004; McClanahan and Cinner 2008; Cinner et al. 2009; Hicks and McClanahan 2012). Certain life-history characteristics make some reef fish species particularly vulnerable to fishing. Many coral reef fish species live for more than 10 years (Choat and Robertson 2002) and such species with slow reproductive rates and late maturity are the least able to cope with increases in fishing mortality (Buxton 1993; Jennings et al. 1998, 1999). In addition, fishing pressure can skew the sex ratios of protogynous hermaphroditic species by selectively taking larger, exclusively male fish (Sadovy 1996; Armsworth 2001; McBride et al. 2007). Where fishing has substantially reduced biomass, it is predominantly fast-growing, high-turnover species that remain in the ecosystem (Jennings and Lock 1996; Russ and Alcala 1998; Houk et al 2018). The size spectra of the fish community may thus be a useful way to judge the intensity of and recovery from fishing (Dulvy et al. 2004, Graham et al. 2005). The substantial contribution of fast-growing herbivorous species to total biomass, particularly at the most inaccessible fishing sites is further evidence that ecosystem overfishing on South Sinai reefs is at an advanced stage.

Biomass was relatively high at Ras Abu Gallum (Figure 3.3), considering its accessibility from the nearby *Mzeina* settlement (Samy et al. 2011), and this may be because many Bedu do not fish here since they view it as a tourism site. This finding suggests that the Bedu are willing to manage their fishery sustainably as long as there is some means of livelihood compensation and that fishing impacts could be reduced through more inclusive benefits for Bedu from tourism. It is of particular note that Ras Abu Gallums's tourism facilities are completely managed by the Bedu, so any benefits go directly to their community without involvement from any Egyptians. Similar findings in other regions have suggested that development of tourism as a sustainable livelihood can be a successful complementary activity in an integrated fisheries management programme (Porter et al. 2015; Masud

et al. 2017). However, *Mzeina* who are full-time 'fishermen' receive substantial personal satisfaction and status in wider society from fishing (Chapter 2) and are unlikely to completely give up their livelihood (Pomeroy et al. 1997; Carter and Garaway 2014; Cinner 2014), but tourism may still provide a supplementary income to reduce overall fishing pressure and incentivize conformity with fishing regulations (Peterson and Stead 2011; Wright et al. 2016).

The diversity of reef fish species observed in the vicinity of Dahab was low when compared to other reefs in the Red Sea, but diversity at the family level was relatively high (Golani and Fricke 2018; Atta et al. 2019), although UVC methods probably underestimate species diversity since hidden individuals and cryptic species are less likely to be detected (Sale and Douglas 1981; Brock 1982). The fish community was dominated by herbivores and small species such as Chaetodonidae (butterflyfishes) and Pomocentridae (damselfishes), while predatory and larger-bodied species were largely absent, indicating that they had already been preferentially extirpated through fishing. Species richness may also have an effect on overall biomass by increasing efficiency of resource use and production (Duffy et al. 2016; Mora et al. 2014).

Biomass increased with depth at all sites investigated. This might indicate a depth-refuge effect from fishing pressure since most Bedouin fishing gears do not reach depths of more than a few metres (Tyler et al. 2009; Goetze et al. 2011). Moreover, it has been previously found that the shallow reef flat usually has the highest abundance of fish (Edwards and Rosewell 1999) and that overall abundance does not change with depth on unfished reefs in the Red Sea (Brokovich et al 2008). The effect of depth is also the least in the most inaccessible site with the lowest fishing pressure, Gabr El Bint, further suggesting that difference is a result of more intense fishing pressure in the shallows. There could be future implications if the Bedu use more advanced gears to reach deeper fish stocks in the future as shallow-water populations decline (Morato et al. 2006; Rijnsdorp et al. 2008; Lindfield et al. 2014). Alternatively, the deeper areas of the reef are less susceptible to anthropogenic

impacts and coral bleaching (Bongaerts et al 2010) and so the increased fish biomass may simply be due to better habitat quality at depth.

Coral reef fish assemblages often display seasonal changes in size, composition and abundance (Bellwood 1988; López-Pérez et al. 2013; Scott et al. 2019), and this study showed an increase in biomass during the summer at almost all sites. Bedu traditionally fish during the summer, noting that catches are better at this time of year (Chapter 2) and the UVC data support the theory. Lighthouse was the only site that did not follow this trend, possibly because it is located in a shallower bay that maintains a more stable temperature year-round and is further from deeper waters where the fish presumably migrate to in winter. These seasonal variations should be carefully considered when planning survey timetables for long term data collection programmes such as Reef Check (Hodgson 2001).

Increased human population density and coastal development in the town of Dahab could also result in a gradient of impacts such as increases in sediment, nutrients and other pollutants as well as recreational diver and anchor damage which could cause degradation of nearby coral reefs (Tomascik and Sander 1987; Hunter and Evans 1995; Hasler and Ott 2008; Naumann et al. 2015). Total abundance of all fish families increased with distance from Dahab and the effects of habitat degradation usually has much broader effects on fish assemblages compared to fishing (Jones et al. 2004; Wilson et al. 2006). However, where significant habitat or environmental degradation occurs around heavily populated locations, it is most likely to exacerbate the impacts of intensive fishing rather than act as a main driver of local fish stock declines and thus it will be essential to build consensus between stakeholders, including the *Mzeina*, on the underlying cause of fishery resource declines in order to take appropriate remedial action and avoid a ‘blame-game’ situation (Weible 2007; Williams et al 2008).

Unfished biomass for reefs in the western Indian Ocean and potentially globally is estimated at 1,000–1,200 kg ha^{-1} (McClanahan et al. 2007, 2015; MacNeil et al. 2015; McClanahan 2018). According to these estimates, even the highest biomass recorded at Shahira is still only a quarter of the site's potential carrying capacity in an unfished state. However, extrapolation to other reefs should firstly consider other variables such as regional, habitat, and productivity differences, light fishing, or recovery from previous fishing (Williams et al. 2008). For example, biomass for unfished areas in the Seychelles and Fiji has been estimated at between 500 and 1,000 kg ha^{-1} (Jennings et al. 1995, Jennings & Polunin 1997, Dulvy et al. 2004). However, when considering potential management interventions in South Sinai, it would not be unreasonable to aim for mean fish biomass of over 1000 kg ha^{-1}, since the fished biomass is comparable to that for the western Indian Ocean. The timescale for recovery is sobering, however, since it would probably take over 20 years to recover to such targets, even within permanent closures (McClanahan and Kaunda-Arara 1996; McClanahan 2000; Russ and Alcala 2004; McClanahan and Graham 2005; Edgar et al. 2009). Species richness would likely reach an asymptote after around 10 years, although species composition would probably be dynamic through time, as competition changes and predation increases (Hixon 1991, McClanahan 1992; Graham et al. 2003, Graham et al. 2007).

In reality, evidence shows that most coral reef MPAs are poorly managed (McClanahan 1999; Kelleher et al. 1995) and even light or restricted fishing, lax enforcement or temporally-controlled exploitation are all likely to result in setbacks to recovery (Russ and Alcala 1998; McClanahan and Mangi 2000). The current approach to MPAs in South Sinai is thus likely inadequate because traditional fishing by Bedu is permitted and the long, remote coastline is impossible to patrol effectively within current government budget allocations (Samy et al. 2011, Chapter 2). Well-enforced, large (>10 km^2), permanent closures are probably the only approach that will successfully maintain ecological states that are representative of unfished ecosystems (McClanahan et al. 2007; Graham and McClanahan 2013) and will also provide a useful baseline from which to evaluate the

impacts of other management approaches (Edgar et al. 2009). Closures can also increase the density of target fish species outside the MPA, through spillover (McClanahan and Mangi 2000; Goñi et al. 2008).

Small closures in heavily fished landscapes rarely achieve ecological or life-history conservation targets because of slow recovery, even in well-enforced closures, (McClanahan & Humphries 2012; Humphries et al. 2014, McClanahan 2014), where fishing in the surrounding seascape, human populations, and distance from markets reduce their biomass potential (Daw et al. 2011; Cinner et al. 2013; Januchowski-Hartley et al. 2013). Thus it is likely that the no-take zones previously established in Nabq MPA were far too small to generate any long-term benefits for fisheries stocks. If large-scale 'wilderness' areas, closed to fishing are unacceptable to the Bedu, adopting culturally and socially acceptable fishing gear regulations may provide gains in biomass, catch and associated ecological functions (Johnson 2010; Costello et al. 2012; McClanahan et al. 2012; Mbaru & McClanahan 2013; McClanahan et al. 2014, 2015; Campbell et al. 2018). Smaller-scale closures coupled with wider gear management could receive more community support and keep biomass levels within or above the fisheries target window and prevent broader ecological changes (Cinner et al. 2009; Stockwell et al. 2009; Babcock et al. 2010; McClanahan et al. 2011). It would be essential to develop a clear definition of allowable gears including mesh sizes and materials rather than relying on the currently-used, ambiguous term 'traditional methods' (Rouphael et al. 2015). Historically, Bedu only fished with hook-and-line and large mesh nets (Chapter 2) so this may be a starting point for negotiation, while fully recognizing the Bedu's traditional fishing rights. In areas such as South Sinai, where enforcement is difficult and compliance is low, bans on destructive and exploitative gears can provide an initial step to demonstrate management benefits to fishing communities before more targeted or nuanced fishing restrictions are achievable (Campbell et al. 2012). Such approaches would help managers to overcome some of the complexity and stakeholder scepticism inherent in multispecies, artisanal reef fisheries management.

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Appendix 3.1. Reef fish species list including diet and biometric parameters

Family	Genus	Species	A	b	Length-Weight	Diet	Length-Length
Acanthuridae	*Acanthurus*	*nigrofuscus*	0.02291	2.96	Froese et al. 2014	Herbivore	-
		sohal	0.02291	2.96	Froese et al. 2014	Herbivore	-
	Ctenochaetus	*striatus*	0.02313	3.063	Kulbicki et al. 2005b	Herbivore	Sano 1984
	Naso	*elegans*	0.02291	2.97	Froese et al. 2014	Herbivore	-
		hexacanthus	0.04240	2.854	Choat and Axe 1996	Planktivore	Hobson 1974
		unicornis	0.02344	2.96	Froese et al. 2014	Herbivore	-
	Zebrasoma	*desjardinii*	0.02344	2.97	Froese et al. 2014	Herbivore	-
		xanthurum	0.02344	2.97	Froese et al. 2014	Herbivore	-
Anthiadinae	*Pseudanthias*	*squamipinnis*	0.01514	2.92	Froese et al. 2014	Planktivore	-
		taeniatus	0.01277	3.0	Pauly et al. 1998a	Planktivore	-
Apogonidae	*Cheilodipterus*	*arabicus*	0.01096	3.07	Pauly et al. 1998a	Piscivore	-
		quinquelineatus	0.01607	2.999	Kulbicki et al. 2005b	Piscivore	Froese and Pauly 2018
Balistidae	*Balistapus*	*undulatus*	0.03548	3.00	Froese et al. 2014	Macroinvertivore	-
	Rhinecanthus	*assasi*	0.02570	2.94	Froese et al. 2014	Macroinvertivore	-
	Sufflamen	*albicaudatum*	0.02570	2.94	Froese et al. 2014	Macroinvertivore	-
Caesionidae	*Caesio*	*suevica*	0.01259	3.09	Froese et al. 2014	Planktivore	-
Carangidae	*Caranx*	*sexfasciatus*	0.0501	2.71	Kamikawa et al. 2015	Piscivore-Invertivore	Salini et al. 1994
Chaetodontidae	*Chaetodon*	*auriga*	0.0404	2.829	Kulbicki et al. 2005b	Microinvertivore	Froese and Pauly 2018
		austriacus	0.02344	3.01	Froese et al. 2014	Microinvertivore	-
		fasciatus	0.02344	3.01	Froese et al. 2014	Microinvertivore	-
		melannotus	0.02669	3.049	Kulbicki et al. 2005b	Microinvertivore	Froese and Pauly 2018
		paucifasciatus	0.02344	3.01	Froese et al. 2014	Microinvertivore	-
		pictus	0.02344	3.01	Froese et al. 2014	Microinvertivore	-
		semilarvatus	0.02344	3.01	Froese et al. 2014	Microinvertivore	-
		trifascialis	0.02578	2.969	Kulbicki et al. 2005b	Microinvertivore	Froese and Pauly 2018
	Heniochus	*intermedius*	0.03981	3	Pauly et al. 1998a	Microinvertivore	-
Cirrhitidae	*Paracirrhites*	*forsteri*	0.01122	3.07	Froese et al. 2014	Piscivore-Invertivore	-
Diodontidae	*Diodon*	*hystrix*	0.19300	2.472	Kulbicki et al. 2005b	Macroinvertivore	Froese and Pauly 2018
Epinephelinae	*Aethaloperca*	*rogaa*	0.01445	3.09	Froese et al. 2014	Piscivore	-
	Cephalopholis	*argus*	0.01190	3.130	Kamikawa et al. 2015	Piscivore	-
		miniata	0.02800	2.850	Palla et al. 2018	Piscivore	-
	Epinephelus	*fasciatus*	0.0118	3.08	Kamikawa et al. 2015	Piscivore	-
		tauvina	0.01440	3.024	Mathews and Samuel 1987	Piscivore	-
	Variola	*louti*	0.01300	3.090	Kamikawa et al. 2015	Piscivore	-
Fistulariidae	*Fistularia*	*commersonii*	0.01120	2.540	Erguden et al. 2009	Piscivore-Invertivore	-

Family	Genus	Species	A	b	Length-Weight	Diet	Length-Length
Grammistinae	*Diploprion*	*drachi*	0.01698	3.03	Froese et al. 2014	Piscivore-Invertivore	-
Haemulidae	*Plectorhinchus*	*gaterinus*	0.01349	3.01	Froese et al. 2014	Piscivore-Invertivore	-
Holocentridae	*Myripristis*	*murdjan*	0.01930	3.034	Murty 2002	Macroinvertivore	-
Labridae	*Anampses*	*meleagrides*	0.01072	3.04	Froese et al. 2014	Macroinvertivore	-
		twistii	0.01148	3.04	Froese et al. 2014	Macroinvertivore	-
	Bodianus	*anthioides*	0.01202	3.05	Froese et al. 2014	Macroinvertivore	-
	Cheilinus	*abudjubbe*	0.01950	3.00	Froese et al. 2014	Macroinvertivore	-
		lunulatus	0.01950	3.00	Froese et al. 2014	Macroinvertivore	-
	Cheilio	*inermis*	0.00400	3.010	Kochzius 1997	Macroinvertivore	-
	Coris	*aygula*	0.00266	3.489	Kulbicki et al. 2005b	Macroinvertivore	Froese and Pauly 2018
	Epibulus	*insidiator*	0.01614	3.081	Kulbicki et al. 2005b	Piscivore-Invertivore	Froese and Pauly 2018
	Gomphosus	*caeruleus*	0.00955	3.03	Froese et al. 2014	Macroinvertivore	-
	Halichoeres	*hortulanus*	0.01190	3.064	Murty 2002	Microinvertivore	-
	Halichoeres	*marginatus*	0.00526	3.414	Murty 2002	Microinvertivore	-
	Hemigymnus	*fasciatus*	0.01122	3.04	Froese et al. 2014	Macroinvertivore	-
	Hologymnosus	*annulatus*	0.01072	3.04	Froese et al. 2014	Piscivore	-
	Labroides	*dimidiatus*	0.00585	3.231	Kulbicki et al. 2005b	Microinvertivore	Froese and Pauly 2018
	Macropharyngodon	*bipartitus*	0.01072	3.04	Froese et al. 2014	Macroinvertivore	Froese and Pauly 2018
	Oxycheilinus	*diagramma*	0.03890	2.847	Gumanao et al. 2016	Macroinvertivore	Froese and Pauly 2018
		mentalis	0.01778	2.93	Froese et al. 2014	Macroinvertivore	-
	Paracheilinus	*octotaenia*	0.01023	3.05	Froese et al. 2014	Planktivore	-
	Pseudodax	*moluccanus*	0.01122	3.04	Froese et al. 2014	Microinvertivore	-
	Thalassoma	*klunzingeri*	0.01072	3.03	Froese et al. 2014	Macroinvertivore	-
Lethrinidae	*Lethrinus*	*mahsena*	0.01200	3.160	Samboo and Mauree 1988	Macroinvertivore	Froese and Pauly 2018
		nebulosus	0.01871	2.996	Kulbicki et al. 2005b	Macroinvertivore	Froese and Pauly 2018
	Monotaxis	*grandoculus*	0.04300	2.840	Kamikawa et al. 2015	Macroinvertivore	Froese and Pauly 2018
Lutjanidae	*Lutjanus*	*bohar*	0.01563	3.059	Kulbicki et al. 2005b	Piscivore	Froese and Pauly 2018
		ehrenbergii	0.04700	2.640	Palla et al. 2018	Piscivore-Invertivore	Froese and Pauly 2018
	Macolor	*niger*	0.02170	2.970	Kamikawa et al. 2015	Piscivore-Invertivore	Froese and Pauly 2018
Monacanthidae	*Amanses*	*scopas*	0.02138	2.91	Froese et al. 2014	Microinvertivore	-
Mullidae	*Mulloides*	*flavolineatus*	0.00886	3.060	Murty 2002	Macroinvertivore	Wahbeh and Ajiad 1985
		vanicolensis	0.00990	3.015	Jehangeer 2003	Macroinvertivore	Froese and Pauly 2018
	Parupeneus	*cyclostomus*	0.01360	3.110	Kamikawa et al. 2015	Piscivore-Invertivore	Randall 1985
		forsskali	0.01148	3.08	Froese et al. 2014	Macroinvertivore	Hobson 1974
		macronemus	0.00546	3.297	Murty 2002	Macroinvertivore	Froese and Pauly 2018
Ostraciidae	*Ostracion*	*cubicus*	0.12900	2.519	Kulbicki et al. 2005b	Microinvertivore	Froese and Pauly 2018
		cyanurus	0.01122	3.04	Froese et al. 2014	Microinvertivore	Froese and Pauly 2018

Family	Genus	Species	A	b	Length-Weight	Diet	Length-Length
Pomacanthidae	*Pomacanthus*	*imperator*	0.09590	2.770	Chung and Woo 1999	Microinvertivore	Froese and Pauly 2018
		maculosus	0.02900	3.000	Grandcourt et al. 2010	Small Omnivore	Froese and Pauly 2018
	Pygoplites	*diacanthus*	0.0276	3.0	Pauly et al. 1998a	Microinvertivore	-
Pomacentridae	*Abudefduf*	*vaigiensis*	0.00989	3.267	Murty 2002	Small Omnivore	-
	Amblyglyphidodon	*flavilatus*	0.02089	2.99	Froese et al. 2014	Planktivore	-
		leucogaster	0.02974	2.936	Kulbicki et al. 2005b	Planktivore	Froese and Pauly 2018
	Amphiprion	*bicinctus*	0.01622	2.96	Froese et al. 2014	Planktivore	-
	Chromis	*dimidiata*	0.01698	3.00	Froese et al. 2014	Planktivore	-
		pembae	0.01698	3.00	Froese et al. 2014	Planktivore	-
		viridis	0.02445	2.784	Murty 2002	Herbivore	-
	Dascyllus	*aruanus*	0.09640	2.333	Murty 2002	Small Omnivore	
		marginatus	0.0166	2.97	Froese et al. 2014	Planktivore	-
		trimaculatus	0.03132	3.043	Kulbicki et al. 2005b	Planktivore	Froese and Pauly 2018
	Neoglyphidodon	*melas*	0.02089	2.99	Froese et al. 2014	Microinvertivore	-
	Plectroglyphidodon	*lacrymatus*	0.02089	2.99	Froese et al. 2014	Microinvertivore	-
	Pomacentrus	*sulfureus*	0.01318	2.98	Froese et al. 2014	Small Omnivore	-
Priacanthidae	*Priacanthus*	*hamrur*	0.01750	2.856	Thomas et al. 2003	Planktivore	Philip 1998
Pseudochromidae	*Pseudochromis*	*fridmani*	0.00589	3.10	Froese et al. 2014	Microinvertivore	-
Scaridae	*Calotomus*	*viridescens*	0.01995	3.01	Froese et al. 2014	Herbivore	-
	Cetoscarus	*bicolor*	0.01349	3.03	Froese et al. 2014	Herbivore	-
	Chlorurus	*sordidus*	0.02000	3.040	Kamikawa et al. 2015	Herbivore	Sano 1984
	Hipposcarus	*harid*	0.0061	3.2501	Chapter 5	Herbivore	-
	Scarus	*ferrugineus*	0.01380	3.03	Froese et al. 2014	Herbivore	-
		fuscopurpureus	0.01445	3.04	Froese et al. 2014	Herbivore	-
		ghobban	0.01650	3.041	Kulbicki et al. 2005b	Herbivore	Sano 1984
		gibbus	0.01413	3.04	Froese et al. 2014	Herbivore	-
		niger	0.0316	3.09	Choat et al. 1996	Herbivore	Gumanao et al 2016
		unknown `	0.01	3.04	Estimate	Herbivore	-
Scorpaenidae	*Pterois*	*volitans*	0.00521	3.240	Edwards et al. 2014	Piscivore-Invertivore	Sano 1984
Siganidae	*Siganus*	*luridus*	0.01100	3.040	Abdallah 2002	Herbivore	Stergiou 1988
		rivulatus	0.01100	3.060	Gabr et al. 2018	Herbivore	Lundberg and Golani 1995
Tetraodontidae	*Arothron*	*diadematus*	0.03388	2.87	Froese et al. 2014	Omnivore	-
		stellatus	0.09150	2.672	Kulbicki et al. 2005b	Omnivore	Kulbicki et al. 2005a

CHAPTER 4. INVERTEBRATE FISHERIES OF *MZEINA* BEDOUIN WOMEN

Abstract

Invertebrate fisheries have expanded in recent years as finfish fisheries have become overfished. Although these fisheries are often perceived as resistant to overfishing, they can often be highly vulnerable as a result of both biological and socioeconomic factors. Invertebrate fisheries are poorly studied, and a number of historical invertebrate fisheries are now declining or collapsed. Invertebrates including *Tridacna spp., Octopus spp.* and other molluscs of value for both food and the curio market are often gathered in the intertidal zone on fringing coral reef flats, primarily by women. The women of the *Mzeina* Bedouin tribe in South Sinai participate in this activity throughout the coastal areas of their territory. Substrate composition and invertebrate density were assessed at eight coral reef sites along the South Sinai coastline in the region of the town of Dahab. It was found that densities of *Tridacna* correlated closely with hard coral cover (R^2=0.869, $F(1,6)=39.655$, $p<0.005$) and both abundance (Kruskal-Wallis H: $\chi2(7)= 42.137$, $p < 0.001$) and size (Kruskal-Wallis H: $\chi2(7)= 48.879$, $p < 0.001$) varied across sites and generally declined with distance from the town and the rate of collection thus probably varied with the difficulty of access. *Tridacna* are chronically overfished in the area since approximately half of the discarded clam shells on the coastline from historical fishing activities were below the size of sexual maturity. *Mzeina* women reported declines in their catch and several species known to be of fishery importance to the Bedu were absent at the study sites, indicating likely overexploitation of a wide variety of reef invertebrates. No evidence was found to suggest damage to hard coral or other substrates from trampling by gleaners. Diademitid urchin distribution did not follow clear patterns of distribution, indicating the complexity of ecological interactions that involve these species on the coral reef. Fleshy algae were abundant at several reef sites, primarily a result of nutrient pollution from coastal development. Invertebrate fisheries could recover relatively rapidly if fishing pressure were reduced. A successful conservation programme would need to intrinsically involve the *Mzeina* women at all stages including data collection, development and enforcement of any management measures.

4.1. Introduction

While many finfish fisheries worldwide are declining or have already collapsed (FAO 2006), marine invertebrate fisheries have grown rapidly in recent times (Anderson et al. 2011). As global markets have expanded, invertebrate fisheries have provided an alternative source of animal protein, employment and economic opportunities (Berkes et al. 2006). Invertebrates have conventionally been regarded as particularly resistant to overfishing because of their wide geographic ranges and short life cycles (Carlton 1993; Jamieson 1993). However, they are in fact highly vulnerable, particularly where they are easy to capture by shore-gleaning, reef-walking, snorkelling or diving (Barnes and Rawlinson 2009). Invertebrate species that are particularly vulnerable to overfishing include those unable to escape depredation, with a large body size at maturity, and rare or stressed species (de Boer et al. 2000). Socioeconomic factors such as rapid human population growth and high economic or traditional dependence on invertebrates may exacerbate their overexploitation (Pauly 1990; Kronen et al. 2010; Bao and Drew 2017). Many invertebrates are broadcast spawners so successful reproduction is density-dependent, and reducing densities may lead to a disproportionate reduction in fertilization and subsequent recruitment (Uthicke and Benzie 2000). Knowledge of the ecological consequences of fishing these animals is sketchy, even though some have been shown to have important ecological functions (Davis et al. 2015; Purcell et al. 2016) and declines of invertebrate populations may have significant trophic effects on wider marine ecosystems (Eddy et al. 2017).

Although overfishing in tropical coral reef finfish fisheries has been studied in some depth (Graham et al. 2017; Valdivia et al. 2017; Ruppert et al. 2018), tropical invertebrate fisheries are rarely monitored and often unregulated (Leiva and Castilla 2002; Andrew et al. 2003) with data on their stock status limited (Perry et al. 1999; Anderson et al. 2011). Furthermore, the deleterious effects of fishing are compounded by other impacts such as climate change (Harley et al. 2006) and pollution (Youssef et al. 2017). As a result, an increasing number of historical and recent invertebrate fisheries

are in decline or have already collapsed (Tegner et al. 1996; Karpov et al. 2000; Uthicke and Benzie 2001; Linnane et al. 2010; Nordlund et al. 2010; Fröcklin et al. 2014; Lane and Limbong 2015). However, it is heartening that concerted management efforts adapted to local conditions have been shown to reduce exploitation and increase stock biomass, reversing such declines (Breen and Kendrick 1997; Castilla and Fernandez 1998; Oliver et al. 2015).

Invertebrates have been gathered on foot in the intertidal zone throughout the tropics since prehistoric times, primarily by women (Chapman 1987; Dalzell et al. 1996; McLachlan et al. 1996; de Boer et al. 2000; Bird and Bird 2002; Kronen 2002; Siason et al. 2002; Jimenez et al. 2011; Bose et al. 2013; Hauzer et al. 2013; Douglass 2017; Harris and Weisler 2017). The most common foods gathered from coral reefs include octopus, sea cucumbers, sea urchins, various crustaceans, finfish, seaweed and shellfish. In developing countries, these fisheries often represent a major source of animal protein or financial income, for a low capital investment, since no gear, except maybe a homemade spear, is required (Ruddle 1996). Shells may also be collected for the marine curio trade (Newton et al. 1993; Nijman et al. 2016). As well as the potential impacts of overexploitation on target species themselves (Villanoy et al. 1988; Foale 1998; Ashworth et al. 2004), collection can cause significant physical coral reef damage through the use of destructive fishing tools and trampling (Andréfouët et al. 2013).

Worldwide, Tridacnidae (giant clams) are exploited for food, aquaria and curios (Wabnitz et al. 2003; bin Othman et al. 2010) and their shallow distribution and conspicuousness makes them easy to catch (Van Wynsberge et al. 2012). Tridacnidae are especially vulnerable to stock depletion because of their late sexual maturity, sessile adult phase, and broadcast spawning strategy (Neo et al. 2015). Juvenile recruitment is patchy, so uncontrolled harvest of adults quickly leads to local extinctions (Yamaguchi 1977). Tridacnidae are restricted to limited areas of the tropical Indo-Pacific (Copland and Lucas 1988; bin Othman et al. 2010) and significant population declines and localized extinctions have been reported (Wells et al. 1983; Lucas 1994). *Tridacna maxima* and *T. squamosa*

have been included in the Convention of International Trade of Endangered Species (CITES) Appendix II since 1985 (UNEP-WCMC 2015); and *T. squamosina* is proposed as Critically Endangered on the IUCN Red List (Richter et al. 2008; Huber and Eschner 2010). More broadly, the Mollusca are the phylum most affected by extinction worldwide yet are largely neglected by conservationists (Régnier et al. 2009). It has been recommended that proactive stock management is vital for such species with low growth rates and densities and erratic recruitment (Gwyther and Munro 1981; Lucas 1994).

Coral reefs are complex and diverse ecosystems and decline or removal of a species or functional group can lead to unpredictable repercussions throughout the trophic system (Dulvy et al. 2004; Graham et al. 2017; Hammerschlag et al. 2018). As reef fisheries decline, there is commonly a downward trend in the trophic level of targeted species (Jennings and Polunin 1997; Pauly et al. 1998). When herbivorous fishes are removed by fishing, their trophic function is often replaced by sea urchins, so on heavily-fished reefs urchins become the dominant herbivore while herbivorous fishes dominate less-fished reefs (McClanahan and Shafir 1990; Rodríguez-Barreras et al. 2015; Steneck et al. 2017). Herbivores play an essential role in coral reef ecology by limiting the development and establishment of algal communities that impede coral recruitment (Mumby et al. 2006). However, excessively high densities of urchins may reduce coral recruitment and increase bioerosion (Glynn et al. 1979; Sammarco 1980), resulting in a loss of: live coral, calcium carbonate accretion, associated faunal diversity, and the coral reef ecosystem services of fisheries productivity and coastal protection.

Recently, gender aspects have been highlighted in small-scale fisheries (Bennett 2005) within a diversity of areas such as: empowerment and agency (Onyango and Jentoft 2011); well-being approaches (Weeratunge et al. 2014); women's contribution to fisheries (Harper et al. 2013; Fröcklin et al. 2014; Kleiber et al. 2015); recognition of the gendered aspects of knowledge (Iniesta-Arandia et al. 2014); need for policy inclusion and way of life considerations (Santos 2015); health issues in

aquaculture (Fröcklin et al. 2012); and women entering fish trade arenas (Béné and Merten 2008; Fröcklin et al. 2013). Men and women use coastal areas in different ways and thus generate distinct environmental impacts, particularly in artisanal fisheries where the diversity of gears, species, habitats and utilization of catch, as well as complex social, cultural and political structures may augment these differences (Oliver 1989; Weeratunge et al. 2010; Kleiber et al. 2015; Torre-Castro et al. 2017).

The most noticeable differences between women's and men's fishing noted in the Pacific are that women: prefer daytime fishing, focus on shallow waters close to shore and fish without using watercraft (Chapman 1987). The often abundant, nearshore resources can provide a safety net in times of food scarcity or poor fishing weather, as gleaning is generally less risky, requires less energy and is more predictable than offshore fisheries or agriculture (Nietschmann 1973; Luomala 1980; Dahlberg 1981; Tucker et al. 2013). Moreover, inshore areas such as reef flats, mangroves and seagrass beds are often nursery habitats for offshore species, so their proper management is also key to the sustainability of finfish stocks (Nagelkerken et al. 2000; Kleiber et al. 2015).

Although women represent between 5 and 73 % of the total capture fisheries workforce (Weeratunge et al. 2010) the ability to fish is often synonymous with manhood (Matthews 1993) and gleaning is rarely regarded as real fishing (Harper et al. 2013). In some parts of the world, historical studies have shown that women's invertebrate fisheries may account for as much catch as the activities of men fishing from boats: for example, 32 % in American Samoa and 25–50 % in Papua New Guinea (Chapman 1987) while on Kosrae, Micronesia, women have been observed to conduct the majority of fishing activities (Des Rochers 1992). However, the importance of women in fisheries is widely overlooked and under-acknowledged in fisheries management, leading to underestimates of fishing pressure and an undervaluation of socioeconomic benefits potentially jeopardizing invertebrate populations and associated livelihoods (Fröcklin et al. 2014; Teixeira and Campos 2019).

Women are rarely included in fishery management decisions because governments, industry, banks and society in general do not realize the importance of their contributions (Aswani and Weiant 2004; Frangoudes et al. 2008). Data gathered on women in fisheries are often qualitative and not incorporated into management plans and can be difficult to obtain because of societal constraints (Johannes 1981; Johannes et al. 1993). Many women possess detailed knowledge of the fauna with which they interact, including knowledge on the spawning seasonality, feeding habits, and periodicity of many invertebrates (Aswani and Weiant 2004). Women have been integral in shaping customary governance systems and developing new networks to address changing circumstances (Chuenpagdee et al. 2006) and thus should be consulted adequately to ensure the success of coastal resource management efforts (Chapman 1987).

Tridacna species found in the Gulf of Aqaba (Figure 2.1) include: *T. squamosa, T. maxima*, (Roa-Quiaoit 2005) and *T. squamosina* (Richter et al. 2008; Huber and Eschner 2010). Although it is the northwestern limit of their range, *Tridacna* densities in South Sinai are the highest in the Red Sea (Roa-Quiaoit 2005). Palaeolithic studies suggest that modern humans have been exploiting Red Sea molluscs for at least 125,000 years (Bailey 2004; Walters et al 2000). *T. squamosina* represents less than 1 % of the present stocks but up to >80 % of fossil shells with the decline in proportion and shell size (20x) probably due to overharvesting (Richter et al. 2008).

Throughout the study area in South Sinai (Figure 3.1), Bedouin women harvest invertebrates at low tide on shallow reef flats using a traditional metal spear (Ashworth et al. 2004), mainly targeting *Tridacna* spp. and *Octopus* spp. but also *Lambis lambis, Tectus dentatus* and *Strombus tricornis* (Benzoni et al. 2006) which form a traditional part of their diet. Clams are generally only eaten by the Bedu, although occasionally more adventurous Egyptians may try them and they are believed to possess aphrodisiac properties (pers. comm.). The nacreous layer of *Trochus* shells has also been used as a source of mother-of-pearl (Ashworth 2004). Fishermen opportunistically take invertebrates as a snack while fishing for finfish, cooking them *in situ* on the campfire. Women reach more remote

sites by travelling with a male family member in a jeep or pickup as far as possible depending on the quality of the track, and then continuing on foot.

The South Sinai coastline around the Bedouin settlement of Dahab is subject to varying degrees of environmental protection legislation. To the North and South lie Ras Abu Gallum marine protected area (MPA) and Nabq MPA respectively, where only fishing using traditional methods is permitted (Samy et al. 2011; Figure 2.2). Nabq MPA extends along the coast of Dahab, but this area does not have any restrictions on fishing (Mabrouk 2007). There is no official definition of what constitutes a 'traditional method' (Rouphael et al. 2015) though it seems reasonable to assume that collection of reef invertebrates by hand is permitted as it has been conducted along this coastline for centuries (Richter et al. 2008) . Additionally, enforcement of regulations in these MPAs has been shown to be poor to non-existent (Galal et al. 2012; Advani et al. 2015). Thus, at all survey sites, reef gleaning is legal and freely conducted without fear of interference from the state and so any differences in exploitation are likely a result of other social and environmental factors.

This study aimed to evaluate the nature and extent of the ecological impacts of the invertebrate fishery on the South Sinai coastline with a focus on the Tridacnidae, and identify factors affecting exploitation that could be incorporated into management. My approach involved the collection of habitat and invertebrate community data through in-water surveys and comparison across a number of sites that varied in accessibility, fishing frequency and environmental characteristics. In addition, I evaluated the impacts of historical collection by measuring *Tridacna* shells discarded by fishers along the shoreline and comparing them with the size distribution of living individuas on the reef.

4.2. Materials and Methods

Reef flat invertebrate and substrate surveys were conducted at eight sites extending north and south of Dahab, located within Nabq and Ras Abu Gallum MPAs during November and December 2016 and July 2017 (Figure 3.1). Sites were selected to provide a geographical spread and varying degrees

of fishing pressure. The logistics involved in surveying these more remote sites meant that camping was essential and the survey methodology was devised to require minimal equipment and time while still providing comparative and detailed data. Surveys were conducted on the reef flat, towards the reef crest because almost all Bedouin fishing is conducted from the shore and the reef flat is one of the primary habitats for *Tridacna* and other marine invertebrates (Ullmann 2013). At these depths (1–2 m), snorkelling provided an effective means to estimate invertebrate population densities and substrate composition. Time of day and tidal state often varied between surveys due to the logistical issues associated with access to the more remote sites. However, surveys were always carried out when the tide was at a suitable height to survey by snorkelling, i.e. not too deep that the substrate could not clearly be seen, but not so shallow that shallow rocks and corals created risk of injury.

A total of ten haphazardly distributed transects were surveyed at each site where the surveyor swam at least 5 m in a random direction before starting subsequent transects to avoid overlap and pseudoreplication. For each transect, a 10 m fibreglass tape was laid along the substrate and weighted at both ends after which macroinvertebrates were recorded to species level along a band within 1 m either side of the tape. The area surveyed on each transect was thus 20 m^2, and the total area surveyed at each site was 200 m^2. The species recorded are listed in Table 4.1. There was no indication of any systematic environmental variation across the sites at this scale and there are no known examples of such variation occurring on reefs in this region (Galal 1999). All surveys were carried out by the author to minimize the effects of inter-observer variation.

All *Tridacna* counted were also measured along the longest length using a plastic ruler. The most common species included the giant clams *T. maxima* and *T. squamosa*, which are the focus of this chapter, because of their importance to the Mzeina as noted in Chapter 2. However, these species could not easily be distinguished because many of the discerning features were hidden, so numbers have been pooled for analysis, although *T. maxima* appeared to be the more common. *Tridacna* were also difficult to measure because they were often embedded in the rocky or coral substrate, so

precision was generally estimated to be +/- 1 cm. To evaluate historical exploitation of *Tridacna*, up to 100 shells that had been discarded by fishers on the shore were collected at random and measured. Only shells without visible wear were collected. The age of these shells was difficult to evaluate. However, it was noted that some mounds of *Tridacna* shells had with time become heavily covered in sand, an indication that at some sites, clam collection was a long-enduring practice for the *Mzeina*.

Substrate composition was evaluated using underwater videography and photography to provide a permanent record of surveys and to save valuable fieldwork time (Leujak and Ormond 2008). The transect tapes used for the invertebrate surveys were video-recorded using a digital underwater camera. Three frames were selected from the digital video at the start, midway and end of the transect (representing 0.5 m x 0.5 m photo quadrats) and percentage cover of each of 12 substrate categories (Table 4.2) was visually estimated from the digital images. The mean of the three quadrats was taken as the substrate composition for that individual transect, and all correlation analyses were conducted at the site scale.

Statistical analysis was carried out using IBM SPSS Statistics v23. The data collected did not follow normal distributions, and could not be transformed to normal distributions using the usual approaches and so non-parametric tests were used throughout. Variation between sites in both abundance and size of invertebrate species, and habitat percentage cover was tested using Kruskal-Wallis tests, followed by a post-hoc Dunn's test with Bonferri correction to identify where the variation occurred through pairwise comparisons. Correlation analysis was conducted to identify the strength of relationships between invertebrate species abundance and the percentage cover of each habitat type. Mann-Whitney U tests were used to assess the significance of differences in length between *Tridacna* shells found on the shore and those observed on the reef.

Table 4.1. List of invertebrates surveyed and their importance to the *Mzeina*

Species	Taxonomic authority	Phylum	Common name	Exploited by Bedu
Tridacna maxima	Röding 1798	Mollusca	Elongate giant clam	Yes
Tridacna squamosa	Lamark 1819	Mollusca	Fluted / scaly giant clam	Yes
Tectus dentatus	Forskål 1775	Mollusca	Red Sea topshell	Yes
Lambis lambis	Linnaeus 1758	Mollusca	Spider conch	Yes
Strombus tricornis	Lightfoot 1786	Mollusca	Three-knobbed conch	Yes
Strombus gibberulus	Mörch 1850	Mollusca	Humpback conch	No
Strombus fasciatus	Born 1778	Mollusca	Lineated conch	No
Charonia tritonis	Linnaeus 1758	Mollusca	Giant triton	Yes
Octopus sp	Gray 1849	Mollusca	Octopus	Yes
Diadema setosum	Leske 1778	Echinodermata	Diadema sea urchin	No
Echinothrix diadema	Linnaeus 1758	Echinodermata	Diadema sea urchin	No
Actinopyga mauritiana	Quoy and Gaimard 1833	Echinodermata	Surf redfish	No
Holothuria atra	Jaeger 1833	Echinodermata	Black cucumber / Lollyfish	No
Holothuria edulis	Lesson 1830	Echinodermata	Pinkfish	No
Holothuria leucospilota	Brandt 1835	Echinodermata	Lollyfish	No
Synapta maculata	Chamisso and Eysenhardt 1821	Echinodermata		No

(Ashworth 2004; see Appendix 2.1 for Bedouin Arabic names)

Table 4.2. Substrate categories recorded during surveys of South Sinai reef flats

Substrate	Code	Description	Importance	Identification criteria	Common species
Hard Coral	HC	Phylum Cnidaria, Class Anthozoa, Subclass Hexacorallia, Order Scleractinia Subdivided into Branching, encrusting, Foliaceous, Plate like, Columnar, Free-living and massive forms	Dominant part of healthy reef habitat, provides bulk of reef structure; shelter for reef fish, food for corallivores	Rigid calcareous skeleton, variety of structures and colours	*Acropora* spp. *Porites* spp.
Soft Coral	SC	Phylum Cnidaria, Class Anthozoa, Subclass Octocorallia	Dominant on current exposed reefs	Pale grey, brown, yellow or blue soft, firm tissues with embedded polyps	*Sarcophyton* spp.
Coralline algae	CA	Encrusting, rigid form of algae, within the red seaweeds	Contributes to reef structure and stabilises coral rubble	Red or pink smooth crusts on dead coral	*Neogoniolithon brassica-florida*
Calcareous algae	CCA	Form of seaweed with calcified segments	Contributes to marine sediments and sand production	Green plants with discoid calcareous segments	*Halimeda opuntia*
Turf algae	TA	Early successional stage algae	Important for herbivorous reef fish, dominant on hard substrates	Soft, fine, short covering of green algae on substrate	Mixture of early stage algal species
Fleshy algae	FA	Large macroscopic seaweeds (non-vascular plants)	Compete with hard coral for light and space, indicators of pollution and overfishing	Red, brown and green algae	*Sargassum* spp. *Ulva* spp.
Broken Coral	BC	(Normally) Branching coral with breakages			
Rubble	R	Fragments of dead coral	Indication of reef degradation	Fragments of dead coral	NA
Rock	RC	Underlying reef structure	Provides substrate for coral recruits and turf algae	NA	NA
Sand	S	Fine yellow grains	Feeding substrate for fish, esp. invertivores	NA	NA
Seagrass	SG	Vascular angiosperms anchored on seabed in shallow water	Seagrass beds used as nursery grounds by fish	Grass-like plants with creeping rhizome	*Cymodocea serrulata* *Thalassodendron ciliatum*
Sponge	SP	Phylum Porifera	Compete with hard corals for space. Stressed/polluted reefs have higher sponge biomass (Aerts 1998)	Spongy tissue with ossicle variety of colours	*Stylotella aurantium* *Liosina paradoxa*

4.3. Results

Giant clams (*Tridacna spp.*) were the most abundant exploited invertebrate and occurred at all sites (Table 4.3). The most common sea urchins were *Diadema setosum* and *Echinothrix diadema,* while other species only occurred at certain sites. The Holothurian community was varied but rare, and so were pooled for analysis. An octopus was recorded on one occasion, although the underwater visual census method is not suited to an accurate evaluation of this genus because of their cryptic and shy nature (Willis 2001). Many of the target species to be surveyed were not seen on any transects.

Hard coral cover generally increased with distance from Dahab and varied significantly between sites (Kruskal - Wallis H: $\chi 2(7)= 30.075$, df=7, $p < 0.001$, Figure 4.1). Post-hoc pairwise comparisons showed that hard coral cover was significantly higher at Gabr el Bint than at Shahira ($p < 0.001$), Nakula ($p < 0.005$), Lighthouse ($p < 0.005$) and Ras Abu Gallum ($p < 0.05$). Fleshy algae was particularly dominant on reef flats at Um Sid, Lighthouse, Nakula and Shahira (Kruskal - Wallis H: $\chi 2(7)=31.028$, $p < 0.001$, Figure 4.1). Post-hoc pairwise comparisons showed that fleshy algal cover was significantly higher at Um Sid than at Laguna ($p < 0.005$), Ras Abu Gallum ($p < 0.01$) and Amoyed ($p< 0.05$).

When all sites were pooled, numbers of *Tridacna* counted in each transect ranged from 0 to 41, mean=7.89, SE=1.18, giving a density of 0.394±0.0592 clams m^{-2} (mean±SE). In general, abundances of *Tridacna* on the reef flat increased with distance from Dahab (Figure 4.2) although there are two sites that were notable exceptions: Nakula and Shahira which both hosted very low densities. *Tridacna* abundance varied across sites (Kruskal-Wallis H: $\chi 2(7)= 42.137$, $p < 0.001$, Figure 4.3). Post-hoc pairwise comparisons showed that there were more clams at Gabr el Bint than Nakula ($p < 0.001$), Lighthouse ($p < 0.001$), Shahira ($p < 0.001$) and Um Sid ($p < 0.05$). *Tridacna* density was positively correlated with hard coral cover ($R^2=0.869$, $F(1,6)=39.655$, $p<0.005$).

Table 4.3. Distribution of invertebrates surveyed at each site (presence-absence)

	Nakula	Amoyed	Ras Abu Gallum	Lighthouse	Laguna	Um Sid	Shahira	Gabr El Bint
Tridacna spp	✓	✓	✓	✓	✓	✓	✓	✓
Echinothrix diadema	✓	✓		✓	✓	✓	✓	
Diadema setosum			✓	✓	✓		✓	
Echinometra mathaei	✓	✓	✓	✓	✓	✓	✓	✓
Heterocentrotus mamillatus		✓		✓		✓		✓
Strombus fasciatus		✓		✓				
S.gibberulus						✓		
Holothuria spp.	✓		✓		✓		✓	
Octopus cyanea							✓	

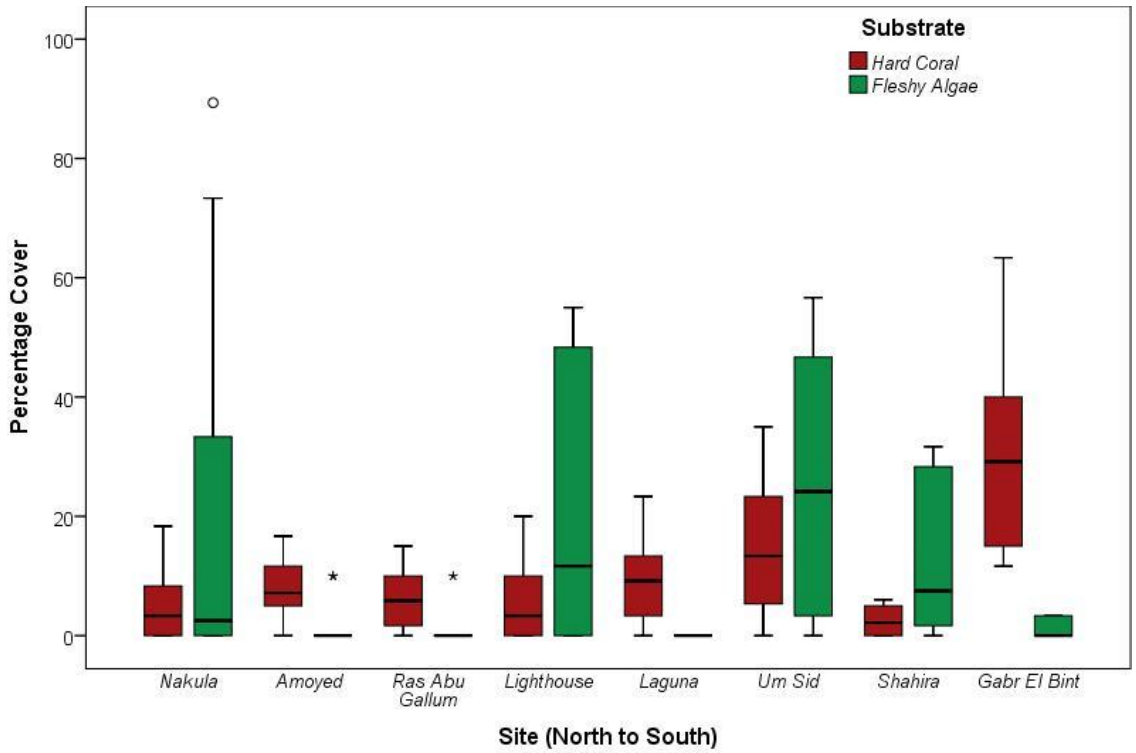


Figure 4.1. Percentage cover of Hard Coral and Fleshy Algae on reef flats in South Sinai

Boxplots show median ± quartiles and outliers.

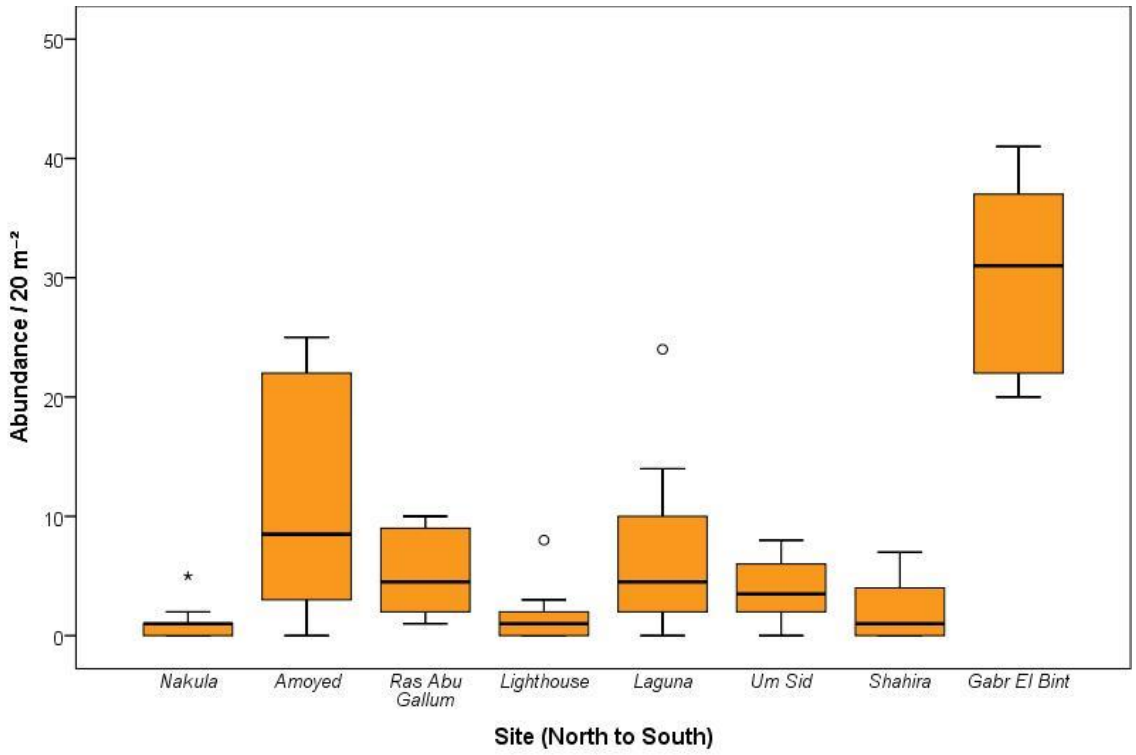


Figure 4.2. Abundance of *Tridacna* clams on eight reef flats exploited by the *Mzeina*

Boxplots show median ± quartiles and outliers.

When pooled across all sites, clam shell length ranged from 2–20 cm and sizes generally increased further away from Dahab (Figure 4.3). When all sites were pooled, *Tridacna* shells collected on the shore ranged in size from 7.5 to 26 cm; mean=11.0, SE=0.180, Median=10.3. *Tridacna* shells on shore were larger than on the reef at almost all sites, with the exception of Ras Abu Gallum (Figure 4.3, Table 4.4). The largest *Tridacna* (20cm) found on the reef in this study also occurred at Ras Abu Gallum. *Tridacna* length varied significantly across sites (Kruskal-Wallis H: $\chi 2(7)= 48.879$, $p < 0.001$). Post-hoc pairwise comparisons showed that clams at Ras Abu Gallum were larger than those at Um Sid ($p < 0.001$), Gabr El Bint ($p < 0.001$), Amoyed ($p < 0.001$), Lighthouse ($p<0.05$) and Laguna ($p<0.05$). Clams at Nakula were larger than those at Um Sid ($p < 0.005$), Amoyed ($p<0.05$) and Gabr El Bint ($p<0.05$).

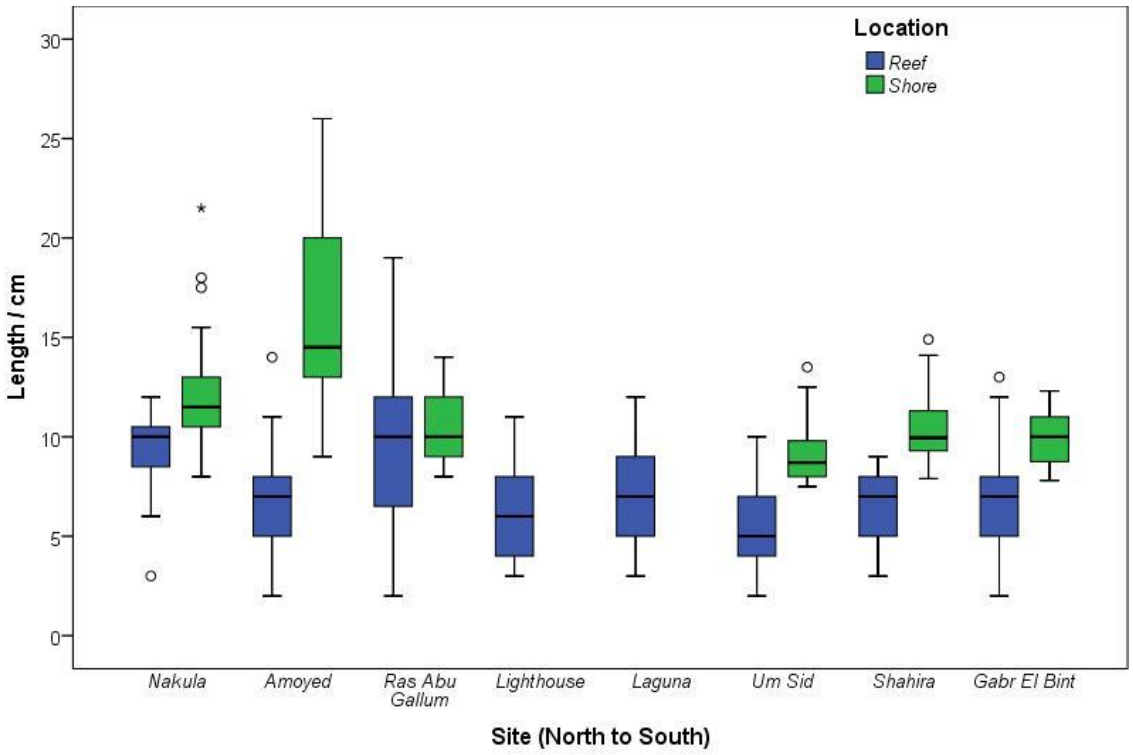


Figure 4.3. Size distribution of *Tridacna* lengths on the reef and shells discarded on shore

Boxplots show median ± quartiles and outliers

Table 4.4. Comparison between *Tridacna* shell size distributions on reef and shore

(Mann-Whitney U test: * p<0.05; **p<0.005, ***p<0.001, ns not significant)

Site	Median / cm		n		U
	Reef	Shore	Reef	Shore	
Nakula	*10.0*	*11.5*	*12*	*38*	*89.5***
Amoyed	*7.00*	*14.5*	*115*	*32*	*38.5****
Abu Gallum	*10.0*	*10.0*	*52*	*33*	*739.0^{ns}*
Um Sid	*5.00*	*8.70*	*40*	*30*	*156.5****
Shahira	*7.00*	*9.95*	*21*	*82*	*50.5****
Gabr El Bint	*7.00*	*10.0*	*304*	*60*	*2165.5****

Diadematid urchin density generally decreased as distance of the site from Dahab increased, with the exception of Lighthouse, where numbers were low (Kruskal-Wallis H=23.068, df=7, p < 0.005, Figure 4.1). Post-hoc pairwise comparisons showed that there were significantly fewer Diadematid

urchins at Gabr el Bint than Ras Abu Gallum ($p<0.01$) and Laguna Bay ($p<0.05$). *Echinometra mathaei* abundance was correlated with hard coral cover ($R^2=0.535$, $F(1,6)=6.897$, $p<0.05$). Other invertebrate species were found at less than half of the sites and so no in-depth statistical analysis was conducted on their abundance.

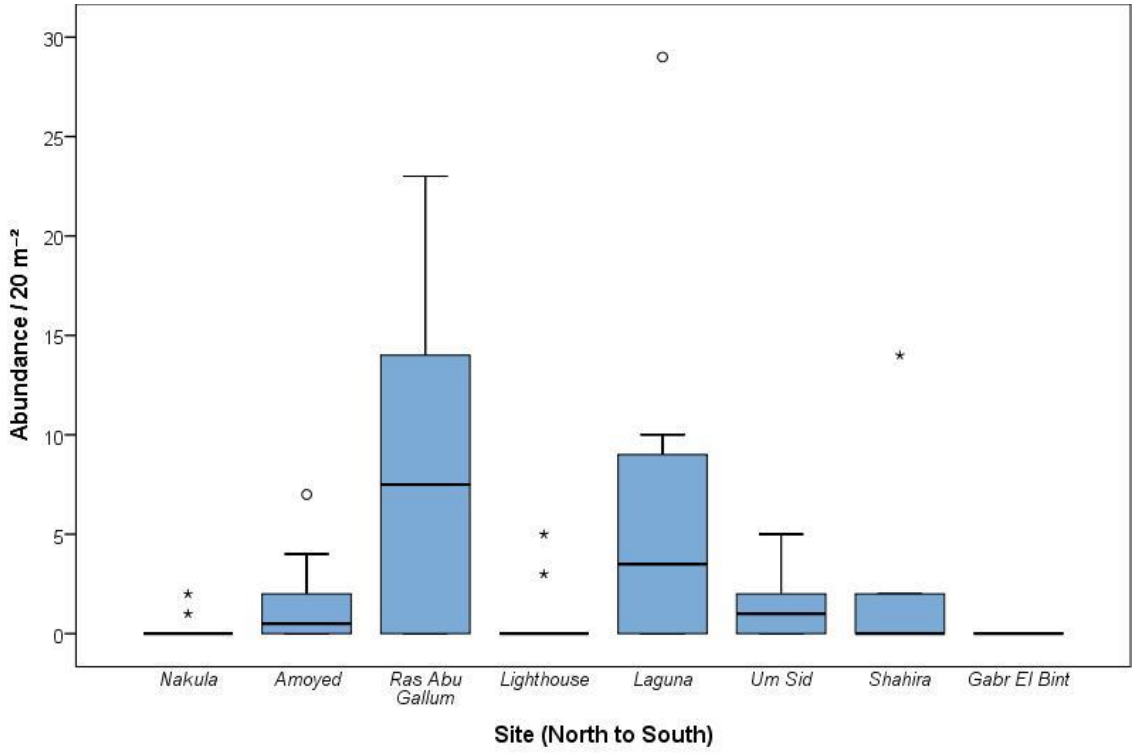


Figure 4.4. Diademitid urchin density at eight sites along the South Sinai coastline

Boxplots show median ± quartiles and outliers

4.4. Discussion

Apart from *Tridacna* and urchins, few other invertebrate species were recorded during snorkel surveys. Even species of importance in Bedouin fisheries such as *Tectus dentatus* and *Lambis lambis* (Ashworth et al. 2004; Benzoni et al. 2006) were absent on the reef flat at the study sites. However, it was noted that Bedouin men and occasionally women captured these species by snorkelling and free diving in deeper waters (Chapter 2); so it is possible that heavy exploitation by gleaning on the reef flat has resulted in the extirpation of these species in the shallows. Octopus were also rarely seen on

transects, but this is more likely due to their cryptic and evasive nature since Bedouin women still gathered substantial numbers of this genus (Chapter 2).

Hard coral cover generally increased with distance from Dahab and this is likely to be a result of higher intensities of impacts closer to the urban centre. These impacts could include trampling from reef gleaners but also other anthropogenic factors such as diver and snorkeler impacts, pollution from coastal development, fishing and anchor damage (Tomascik and Sander 1987; Hunter and Evans 1995; Hasler and Ott 2008; Naumann et al. 2015). Fleshy algae was noticeably dominant at four sites: Nakula, which was the historical location for a National Parks office; Lighthouse, in the centre of Dahab (Figure 3.1), Um Sid, close to recent major hotel developments in the Wadi Gnai area (Naumann et al. 2015), and Shahira, an important Bedouin gleaning and fishing site (Chapter 2). Nutrient pollution is therefore likely to have caused the high algal growth in the first three cases, while trampling of corals, allowing algae to outcompete hard coral is more likely in the last case.

The densities of *Tridacna* recorded are similar to others from the region: 0.32±0.032 clams m^{-2} (Ullmann 2013); 0.76±2.12 clams m^{-2} (Zuschin et al. 2001); max. 0.22 m^{-2} (Bodoy 1984) but considerably higher than those found in Nabq: 0.0804 m^{-2} in No Take Zones (NTZs) and 0.0434 m^{-2} in fished areas (Ashworth 2004). The low numbers found in Nabq may be because there were major decreases in giant clam populations between 1997 and 2002, with many of the small clams seen in 1997 not surviving through to 2002, attributed to sediment from major constructions in South Sinai over the previous 5 years (Hassan et al. 2002). The stochastic nature of oceanic larval and recruitment process may also lead to patchiness in clam distribution between sites (Black et al 2011), but it is likely that exploitation by Bedu is a major factor determining clam population densities in the area (Ashworth et al. 2004). Low densities of giant clams have also been noted from heavily fished areas in the Philippines (Juinio et al. 1989) and Saudi Arabia (Bodoy 1984).

A strong correlation was apparent between hard coral cover and clam density which could potentially be due to three reasons: (1) hard corals have been damaged by women as they collect clams through trampling and use of extraction tools; (2) hard corals and clams both grow in areas where there are suitable hard substrates to attach to or (3) the general environmental conditions and water quality at each site determine both the density of clams and the cover of hard coral. The actual cause is most likely to be a combination of these factors, although it was noted that there was little evidence of recently broken coral, so if damage during harvesting was a factor, it is likely to be historical. Giant clams are hard coral-associated species, they bysally attach among or within corals (Zuschin et al. 2001) and loose sediment may hinder byssal attachment (Kilada et al 1998). *Tridacna* even have physiological similarities with hard corals because they also possess symbiotic zooxanthellae (Trench 1987; DeBoer et al 2012) and therefore are subject to some mutual threats including bleaching (Buck et al. 2002).

Tridacna at the study sites were notably smaller than those recorded in previous studies from the area where lengths ranged from 2 to 33 cm in Nabq MPA with the largest individuals found in Ras Mohamed MPA and within the NTZs at Nabq (Kilada et al 1998; Ashworth 2004). At Abu Sautir, El Quseir, it has been noted that although clams ranged from 1 to 30 cm (median 5 cm), individuals >11 cm and ≤2 cm occurred on the reef slope but not on the reef flat (Ullmann 2013). In the majority of international studies, it has been noted that invertebrates are approximately 20 % larger in unfished reserves (Halpern 2003). The overall size range for *T. maxima* in the Philippines was recorded as 1.2–37.7 cm with the largest number occurring between 4–14 cm (Juinio et al. 1989). It is likely that significant exploitation of *Tridacna* in South Sinai prevents most individuals from reaching their maximum size.

Large species such as *Tridacna* are often harvested before they reach sexual maturity as indicated by the small size of harvested shells at the study sites. *Tridacna* grow slowly, although the actual rate can vary from 2.8 to 5.4 cm y^{-1} (Calumpong 1992). They are protandric hermaphrodites with full

sexual maturity occurring around 3–4 years of age (Lucas 1994) at a length of 110–200 mm for *T. maxima* and *T. squamosa* (Morton 1983). It appears that the *Mzeina* currently harvest clams as small as 7.5 cm with a median size of 10.3 cm, thus around half are being taken before they have spawned even once, although larger shells may be kept for their curio value as ashtrays etc. and not represented on the shore. A further indicator of over-exploitation is the reduction in mean shell length in molluscs when harvested shells are compared with living individuals on the reef (de Boer et al. 2000). Enforcement of size limits may be one approach to ensure that the *Tridacna* fishery is managed sustainably and has proved successful in numerous other invertebrate fisheries (Richards et al. 1994). Although I was regularly assured by Bedu that 'they only collected the big ones', the perception of 'big' and 'small' *Tridacna* is likely to have drifted with each generation as overexploitation progressed (Pauly 1995; Knowlton and Jackson 2008).

At most sites in the study area, larger-bodied, higher-yield clams were relatively rare whereas low-yield smaller individuals were abundant. This size-abundance relationship is often cited as an effect of fishing pressure (de Boer et al. 2000; Hawkins and Roberts 2004). The rate of *Tridacna* collection on Red Sea reef flats has been noted to vary with the difficulty of access (Bodoy 1984). The general increase in *Tridacna* abundance and size with distance from Dahab is most likely due to the fact that fisherwomen normally need to walk, or rely on close male relatives for lifts by jeep or boat and thus are limited in the range to which they can travel and fish, so impacts of gleaning are lower on more distant reefs. It has also been noted that women are deterred from walking on the reef flat if there are too many urchins (Ashworth 2004). High densities of *Tridacna* on the reef at Gabr el Bint may be because the site is too far to travel to by foot and a Bedouin woman told me that she 'feared being murdered' if she travelled all the way to that site on foot. It was widely noted that the gleaning harvests at Shahira had declined in recent years and this is supported by the very low density of *Tridacna* at that site. The unusually high density and large clam size at Laguna Bay may be due to the fact that this area is constantly frequented by groups of windsurfing tourists so Bedouin women

feel uncomfortable in their presence, particularly participating in an extractive activity which is frowned on by environmentally-conscious tourists. The lack of shells found on the shore at this site further supports this idea. In general, Bedouin women prefer not to fish in areas that are frequented frequently by tourists such as Lighthouse because they prefer to keep their privacy and 'people talk too much'. However, the reef flat at Lighthouse is exploited by younger girls, who do not face the same cultural restrictions as mature women. Ras Abu Gallum is reserved for tourists, and thus fishing at this site is informally restricted by the Bedu, explaining the large size of clams on the reef at this site and the lack of discrepancy between the sizes of clams on the reef and harvested shells.

Diademitid urchin density did not follow any clear patterns of distribution, although there was a general decrease in density with distance from Dahab. This illustrates the complexity of ecological interactions likely affecting them. They compete with other herbivorous fishes and invertebrates and also influence the interactions between algae and coral and as such are of significance to reef ecology and fisheries management (Muthiga and McClanahan 2007). *Echinometra mathaei* distribution was correlated with hard coral cover; which was predictable since this species is a bioeroder that burrows into calcareous substrates and could contribute to the breakdown of hard coral on the reef flat (Bak 1994).

The traditional nature of many artisanal fishing communities means that women are often viewed as housewives, whose primary role is associated with reproduction (Vunisea 1997; Torre-Castro et al. 2017). They are often denied access to the best fishing grounds and high-value species leading to limited social mobility and fewer chances to accumulate livelihood assets (natural, physical, financial, social and human capital) thus impacting livelihood strategies (Fröcklin et al. 2014). For example, in Bedouin society, women are limited by the need to always be escorted by a male family member as a chaperone *(mahram)*. Women often participate in a wider diversity of livelihoods than men and by restricting their fishing activity to shallow, nearshore areas close to home, women are able to participate in productive work whilst also raising children (Torre-Castro et al.

2017). Because the harvesting of marine invertebrates is mostly carried out by women, the decline in these resources may have the dual effect of altering their position within the household and community and causing a decline in household levels of food security and income on the other (Aswani and Weiant 2004).

Mzeina Bedouin women gleaned almost exclusively at low spring tides; however it has been noted that in Nabq, women glean daily (Ashworth 2004) as in Micronesia (Murai 1954) compared with 6–7 day periods over spring tides in southwest Madagascar (Barnes and Rawlinson 2009). Whether the shorter time spent gleaning in the Dahab area is due to a productive fin-fishery providing enough food for the remainder of the month or to the inaccessibility of suitable gleaning habitats during neap tides, or a combination of both factors, is unclear. Yet the discovery of patterns explaining the distribution of small-scale fishers and the dynamics of resource use could help with designing management that promotes sustainable harvests and predict fisher responses to management (Abernethy et al. 2007). Gender issues must be reconciled in order to instigate any effective management measures; thus recognition of the importance of the fishery, empowerment of women and development of social capital are essential (Frangoudes et al. 2008) and close involvement of women and inclusion of their ecological knowledge is required in monitoring and management (Paul et al. 2016). Additionally, developing alternative livelihoods for Mzeina women would be useful to compensate economically for any restrictions imposed (Aswani 2000).

There is strong evidence that *Tridacna* fisheries may recover rapidly when fishing pressure is reduced. For example, in Nabq MPA, greater densities of *Tridacna* were recorded only 5–8 years after the establishment of no-take zones (Ashworth 2004). Similar recoveries and positive impacts of MPAs have been noted in the Pacific both for *Tridacna* and other molluscs (Green and Craig 1999; Dumas et al. 2010; Thaman et al. 2017). MPAs may also potentially export larvae to repopulate surrounding areas, much in the same way as they have been shown to spillover finfish populations (Harrison et al. 2012; Kough et al. 2019). However, evaluating MPAs' effects on invertebrate

fisheries is complicated because it can be difficult to separate fishing effects from the inherent levels of variability in invertebrate populations (Gray 2016).

Further research into the fishing effort exerted on giant clams in South Sinai is required in order to develop management recommendations. Much of the detail required will be difficult to collect considering the remote nature of many of the sites and the societal constraints on Bedouin women for communicating with non-Bedu and relaying information. To date, formal management of intertidal areas remains rare, and where present remains largely unregulated and unmonitored (Anderson et al. 2011). However, enforcement would be difficult, particularly at the remote fishing sites used by the *Mzeina* and a top-down regulatory approach would be unlikely to be successful due to historical mistrust between the Bedu and Egyptian state (Glaser at al. 2010; Lopes et al. 2013; Katikiro et al. 2015). The most realistic option would be to develop a locally-supported, self-imposed management regime through building awareness about the biology of the target species and the relationship between resource management and sustainability among the *Mzeina* women (Richards et al. 1994; Barnes and Rawlinson 2009).

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CHAPTER 5. MONITORING AND MANAGEMENT OF THE ARTISANAL CORAL REEF FISHERY IN SOUTH SINAI

Abstract

Coral reef fisheries are an important source of animal protein and food security in developing countries. However, fisheries are declining worldwide and fishing can impact coral reef ecosystems through multiple indirect and direct effects. Coral reef fisheries further present unique issues for management because of their data-deficient, remote, informal and multispecies nature. The *Mzeina* have fished South Sinai coral reefs for generations, however no management approaches have been successful in the sustainable management of the fishery to date. This work aimed to assess the catch of *Mzeina* Bedouin fishers in South Sinai and develop recommendations for the long-term monitoring and management of these reef fisheries. The fishery employed a variety of gears, primarily nets but also spearguns, hook-and-line and freediving. Only 15 species accounted for 86 % of the catch which consisted primarily of herbivores (57.7 % by weight), indicating that the reef fish community structure had already been altered through overfishing and preferential removal of carnivorous species. *Mzeina* also targeted large aggregations of parrotfish migrating along the coastline. The Length-Weight relationship for *Hipposcarus harid*, caught in such an aggregation was calculated as $W=0.0061\ L^{3.2501}$ (n=16; TL=26.1–43.7 cm; $R^2=0.9312$) and will be submitted to Fishbase since there is not yet any reliable estimation recorded. In order to prevent further deleterious impacts of fishing, it is recommended that a monitoring programme is established, potentially administered by the Bedu themselves. Modern technology such as smartphones could provide an opportunity for data collection in remote areas at a relatively low cost. Marine protected areas could potentially be an option for effective fisheries management although they must be of sufficient size to ensure adequate protection of fisheries stocks and implicitly involve the Bedu in their design and management. In particular, management and monitoring strategies that focus on herbivorous species or fish aggregations may result in higher levels of success considering the limited financial means for fisheries management in the region.

5.1. Introduction

Fish and fishery products are one of the most traded food commodities globally and provide about 16.6 % the world's animal protein (FAO 2012, Bellmann et al. 2016). Fish is a potential solution for food security issues in developing countries because it is high in protein and essential micronutrients but low in saturated fats, carbohydrates and cholesterol (Roos et al. 2007; Belton and Thilsted 2014; Thilsted et al. 2016). Coral reef fisheries, most of which are located in developing countries, are believed to be particularly productive (Marten and Polvina 1981; Russ 1991; Dalzell 1996; McClanahan 2006) with yields >5 t km^{-2} yr^{-1} and even up to >20 t km^{-2} yr^{-1} in the Philippines and American Samoa (Craig et al. 1993; Maypa et al. 2002). Thus coral reef fisheries could play a significant role in poverty alleviation amongst the world's poorest (Newton et al. 2007; Cabral and Geronimo 2018) and furthermore have great potential to act as engines of social and economic development through the provision of jobs, cultural identity and life satisfaction (Andrew et al. 2007; Loring et al. 2019).

Worldwide, marine fisheries catch has increased markedly from 16.8M t in 1950 to a peak of 86.4M t in 1996, when it declined and stabilized at around 80M t (FAO 2018). However the number of overexploited stocks has simultaneously increased, from 10 % in 1974 to 26 % in 1989, to 29.9 % in 2009 (FAO 2012, 2018). Failure to arrest this overexploitation has resulted in severe stock declines, particularly in predatory species (Myers and Worm 2003, 2005), and is thought to have caused 55 % of marine extinctions over the last two centuries (Dulvy et al. 2003) with resultant risks to food security (McClanahan et al. 2015a). However, comprehensive scientific assessment and effective fisheries management measures have the potential to reverse these trends (Hilborn et al. 2020). Coral reef fisheries are particularly susceptible to overfishing, resulting in population declines and shifts in trophic functions; even in cases where the catch is destined only for subsistence or sale at local markets (Jackson et al. 2001; Valentine and Heck 2005; Muallil et al. 2014; Graham et al. 2017; Samoilys et al. 2017). This vulnerability is compounded by the array of socioeconomic

pressures that impact coral reef fishers and their associated coastal communities, most of whom are located in developing countries (McClanahan 1999; Cinner and McClanahan 2006; Cinner et al. 2009).

It is well-accepted that fishing can substantially affect the composition of coral reef fish assemblages through several direct and indirect effects (Koslow et al. 1988; Russ and Alcala 1989; Friedlander and DeMartini 2002). Most obviously, fishing removes target species (and non-target species through non-selective gears) particularly larger, more desirable individuals resulting in decreases in abundance and average size and thus biomass of these species (Ashworth and Ormond 2005; Unsworth et al. 2007; Rolim et al. 2019). Secondly, destructive fishing techniques such as dynamite can damage benthic habitat, thereby affecting the abundance of fishes indirectly (Mangi and Roberts 2006; Wells 2009). Thirdly, targeting of specific trophic groups can lead to cascading effects through food webs, resulting in indirect effects on non-target species (Dulvy et al. 2004; Mumby et al. 2006). Fishing most strongly impacts slow-growing, larger species which are often characterized by high trophic status, late maturity, low rates of population increase, large sizes at sexual maturity and low rates of natural mortality (Russ and Alcala 1998; Jennings et al. 1999; Rochet et al. 2000; Cheung et al. 2005). In this way, fishing effort, gear and closures as well as size, life history, taxonomic and trophic composition of the catch are all likely to influence reef fisheries yields and sustainability (McClanahan 2006).

Although coral reef fisheries are known to be vulnerable, they have not been adequately assessed. Information on the harvesting modes and magnitude of specific reef fisheries has remained a knowledge gap for scientists and managers, particularly in developing countries (Fisher et al. 2011) even though small scale fisheries are thought to comprise between 25 % and 33 % of global landings (Chuenpagdee et al. 2006). More than 80 % of global fisheries are classed as data deficient, and information on small-scale fisheries is particularly scarce, poorly reported, less accessible and more difficult to interpret (Worm et al. 2009; Costello et al. 2012). Traditional fishery stock assessment

methods can cost tens of thousands to millions of dollars per stock (Pauly et al. 2013) which makes them infeasible for most coral reef fisheries because of the low financial and technical capacity for monitoring in many of the regions where they occur (Rosenberg et al. 2018). Assessment of the socioeconomic value of reef fisheries is further complicated because of seasonal use of fishing resources and part-time fishers who supplement their fishing income with other livelihood activities (Cinner and Bodin 2010; Cinner 2014; Dacks et al. 2018).

Clearly, assessing the impact of small-scale or artisanal fisheries can be logistically challenging, especially at remote locations, in developing countries (Salas et al. 2007; Moore et al. 2010). Although fishery-independent approaches such as underwater visual census provide data on reef fish assemblages, strong inferences on fishing impacts can only be made with a comprehensive understanding of the nature of the fishing process itself (Shepherd 1984; Daw et al. 2011). Developed nations normally have strict governance and enforcement structures, such as compulsory logbooks and vessel monitoring (Bastardie et al. 2010; Campbell et al. 2014), however effective fishery data collection in developing countries almost always requires the cooperation of fishers themselves (Obura 2001; Evans et al. 2011). It can be particularly difficult to gain the trust of fishers and encourage them to become involved in the collection of data which may ultimately result in fishing restrictions and negative impacts on their own livelihoods. While attempts have been made to involve fishing communities in the design of monitoring and management processes (Obura 2001; Obura et al. 2002; Scholz et al. 2004; Humber et al. 2011; Kittinger 2013) this is often resource-intensive to implement and therefore necessarily limited in spatial and temporal scope, precluding the recognition of larger-scale trends (Weijerman et al. 2016).

Most fishery management programmes and stock assessments are refined for single-species fisheries where controls on fishing can be highly specific to the behaviour, feedings habits, seasonal cycles and migrations of the target species (Begg et al. 1999). However, many of the world's fisheries, including most reef fisheries, are mixed or multi-species, where assorted species are caught

simultaneously (Säterberg et al. 2019). Reef fisheries are also difficult to monitor from a practical point of view because they typically use informal landing sites dispersed across a wide area, and are usually multi-species and multi-gear (Ban et al. 2009; Fenner 2012; Gill et al. 2017). Reef fishers often catch fish irrespective of size or function (Hicks and McClanahan 2012) making species-specific approaches to management difficult to implement (Johannes 2002; Worm & Branch 2012). Segregating, assessing and managing each species individually would simply be impractical and unrealistic (Salas et al. 2007). In reef fisheries, it is rare to have much catch data, except for high-value species such as giant clams in Fiji (Lewis et al. 1988) or *Conchilepas concholepas* in Chile (Leiva and Castilla 2002). Furthermore, traditional, single-species methods to estimate maximum sustainable yield (MSY) and thus quota-based management are ineffective for coral reef fisheries (McClanahan et al. 2015b; McClanahan 2018). Artisanal fishing gears are often home-made and improvised to suit a particular fisher's style of fishing and so can be difficult to adequately classify for an efficient monitoring or management programme. For example, in Kenya, gill nets are only reported as one gear type yet their sub-types span >40 cm in mesh size (Samoilys et al. 2017). Ecosystem-based fishery management where management is focussed on the ecosystem rather than the target species provides a solution for some of these issues (Pikitch et al. 2004).

Accurate information on biomass and growth rates of individuals is vital for stock assessments and management. Length-weight relationships are primarily useful to estimate biomass from length-frequency data particularly because direct weight measurements can be time-consuming or impossible in the field. Length-weight relationships can also be used to determine fish condition, compare fish growth among areas, and as a complement to species-specific reproduction and feeding studies (Froese 2006). Adequate local biometric data are sparse for most tropical and subtropical fish species (Horn 1989). Moreover, growth parameters for reef fish should not be applied too broadly, since considerable inter- and intra-specific variation are likely to occur based on food availability, temperature, and population density (Weatherley and Gill 1987; Clifton 1995).

Mzeina Bedouin tribes have fished the shallow coral reefs along the coastline of South Sinai for generations using a variety of artisanal gears (Glassner 1974; Galal 1999; Table 2.3). To date, fisheries management approaches have been focussed on spatial restrictions, through Nabq Managed Resource Protected Area and Abu Gallum Protected Area, both established in 1992 where a network of no-take zones was established and Bedu were only allowed to fish using 'traditional methods' (Mabrouk 2007; Figure 2.2). However, although these measures seemed effective in their early years (Galal et al. 2002; Ashworth 2004; Ashworth et al. 2004; Ashworth and Ormond 2005), a lack of enforcement has resulted in reduced effectiveness and measurable fisheries impacts and sustainability concerns (Galal et al. 2012; Advani et al. 2015; Mona et al. 2019). Any definition of 'sustainability' must include an element that remains similar over time, thus it is impossible to evaluate the sustainability of a fishery and implement effective management strategies without time series data on fish populations or catch (Pauly and Zeller 2014). To date, the Bedouin fisheries of South Sinai have not been adequately monitored, as reflected in Egypt's national fisheries statistics reported to the FAO. Bedu rarely, if ever, land at official landing sites and so the official figures reported by Egypt of 122 t y^{-1} for Dahab and 155 t y^{-1} for Nuweiba are likely to be significant underestimates (FAO 2003). A similar situation is apparent for many of the world's small scale fisheries, with an estimated 46 % of African catch unreported (Pauly and Zeller 2016; Belhabib et al. 2019). Furthermore, there is a lack of collaboration between the Egyptian Environmental Affairs Agency (EEAA) and National Park Sector (NCS) staff on the ground and the General Authority for Fish Resources Development (GAFRD) that is responsible for reporting of national fisheries catches (Samy-Kamal 2015).

The conservation status of Bedouin coral reef fisheries in South Sinai is unknown, although a number of studies have reported local declines attributable to fishing impacts (Advani et al. 2015; Mabrouk 2015; Chapter 2,3,4,6). Nevertheless, fishing is culturally important to the Bedu and part of

their ethnic identity in South Sinai (Lavie 1990). To ensure the sustainability of these fisheries, it is essential that a monitoring programme is implemented to guide their effective management.

The aim of this work was to work closely with Bedouin fishers to conduct surveys of landed catches to assess species composition and biometric characteristics. Issues relating to fisheries sustainability were also informally discussed with the fishers. This information was then assessed to evaluate pressures on the sustainability of fishery and potential options for long-term monitoring and management.

5.2. Materials and Methods

Bedouin fishing expeditions were usually spontaneous, landing sites were indistinct and gears such as nets were left in the water overnight or for indeterminate lengths of time. Thus it was difficult to always be present when fish were landed and catch sampling was necessarily opportunistic and depended on fishers' willingness to cooperate as well as being in the right place at the right time (Pollnac 1984). Several approaches were employed to overcome such issues. The first was a roving technique, where I surveyed fishing grounds at the times of highest fishing activity as determined through interviews with experienced fishers. The predominant factors that affected Bedouin fishing activity include season (higher in the summer and during school holidays), day of the week (higher on weekends) and sea state (calm seas). Fishing activity was also widely regarded to be higher during Ramadan when fish were caught for evening feasts to break the fast. However, even taking account of these factors, fishing remained quite unpredictable in space and time, so I often had to camp and wait at remote fishing sites for several days waiting for fishers to arrive. Secondly, I maintained regular contact with fishers and they called me by telephone when they were going fishing, often with only minutes of notice. Thirdly, I employed several young Bedu to record fishing activity while they were staying near fishing sites. They would take photos of fish with a ruler scale using their smartphones and send them to me by WhatsApp with additional details in text or voice

messages. Although all of these approaches resulted in opportunistically sampled data, it was the optimal survey approach for the available personnel, geographies, and resources.

To adequately assess the fishery, it was essential to build trust and personal relationships with a number of Bedouin fishers, who were naturally very suspicious of the Egyptian state and non-Bedu in general. I was regularly asked if I worked for 'National Parks' and Bedu often actively tried to avoid me when they saw me measuring fish on the shore. Consequently, much of the data collected were skewed in favour of fishers with whom I had developed a good working relationship and other fishers in their circle of friends or family. It is likely that fishers who knowingly took part in unsustainable fishing practices would be unwilling to participate in the study. I ensured that all fishers that provided data were aware of the aim of my work and that I was writing a PhD thesis, they gave their informed consent and I worked with them on the understanding that they would remain anonymous.

Once fishers had set their gear, often with my assistance, we waited and chatted, drinking tea, and informally gathering general information orally on the fishing process. When fish were landed, the majority of fishers, particularly at more remote locations, were keen to get them onto ice as soon as possible so the time window when I could have access to the fish was often very limited. Where possible, I identified and directly measured fish at the landing site but, as a permanent record, all fish in a sample were photographed on a white plastic slate with a 10 cm scale next to the specimen. This approach enabled me to later identify and measure fish digitally at my leisure and allowed me to optimize time in the field. When the whole catch could not be sampled because fishers did not want to be delayed, a random stratified sub-sample was measured to ensure that a good representation of species and sizes landed was recorded. I also tried to weigh as many individuals as possible using a small digital scale, although fishers were highly suspicious of this process, which made it difficult to do very often. Also noted where possible were: fishing gear (including measurements of net mesh sizes, hook sizes etc.), number and composition of the fishing team, fishing location, time spent

fishing, the likely fate of the catch and other general details about the fishing process. Fishers commonly used multiple gears; for example, they would collect fish from set nets while spearfishing or fish with a hook-and-line while nets were soaking, so the specific gear used to catch a particular fish was often impossible to determine.

Photographs of fish were taken using various digital cameras and smartphones, but primarily with a Canon Digital IXUS 85 IS set to 'Superfine' quality. Images were analysed using ImageJ software (Abràmoff et al. 2004) by using the 'Set Scale' function on the known length (10 cm scale) to determine the number of pixels cm^{-1} and then measuring the required dimensions of the fish using the 'Measure' function. Measurements of total length, fork length and standard length were taken for all fish (Figure 5.1). Some species did not have pronounced forks in the caudal fin so this measurement was not recorded for them. Other species did not have a clear terminal point of the vertebral column, in which case standard length was not measured. This approach was unsatisfactory for measuring non-finfish species (e.g. lobsters) because it was impossible to measure accurately from two-dimensional digital images due to variations in the curvature of the body of the photographed specimens. When it was not possible to weigh fish, weights were derived from the length measurement by using the allometric length-weight equation: $W=aL^b$; where a and b are constants obtained from the literature and L is length and W is weight. The values for a and b were selected from the literature by giving precedence to estimates derived from the highest sample size. If there were no experimental estimates available, then a Bayesian estimate was used (Froese et al. 2014).

For species where sufficient specimens had been weighed in the field, length-weight relationships were also derived using the allometric length-weight equation: $W=aL^b$, where a is the intercept and b is the slope of the log-transformed relation (Le Cren 1951; Froese 2006). When b=3, all fish dimensions increase at the same rate and its growth is termed isometric; when b<3, the fish is hypoallometric meaning it increases less in weight than predicted by its increase in length and

becomes more elongated as it grows; when b>3, the fish is hyperallometric meaning it increases more in weight than predicted by its increase in length and becomes more rounded as it grows. Length-length relationships were also determined by conducting regression analysis on the three different length measurements.

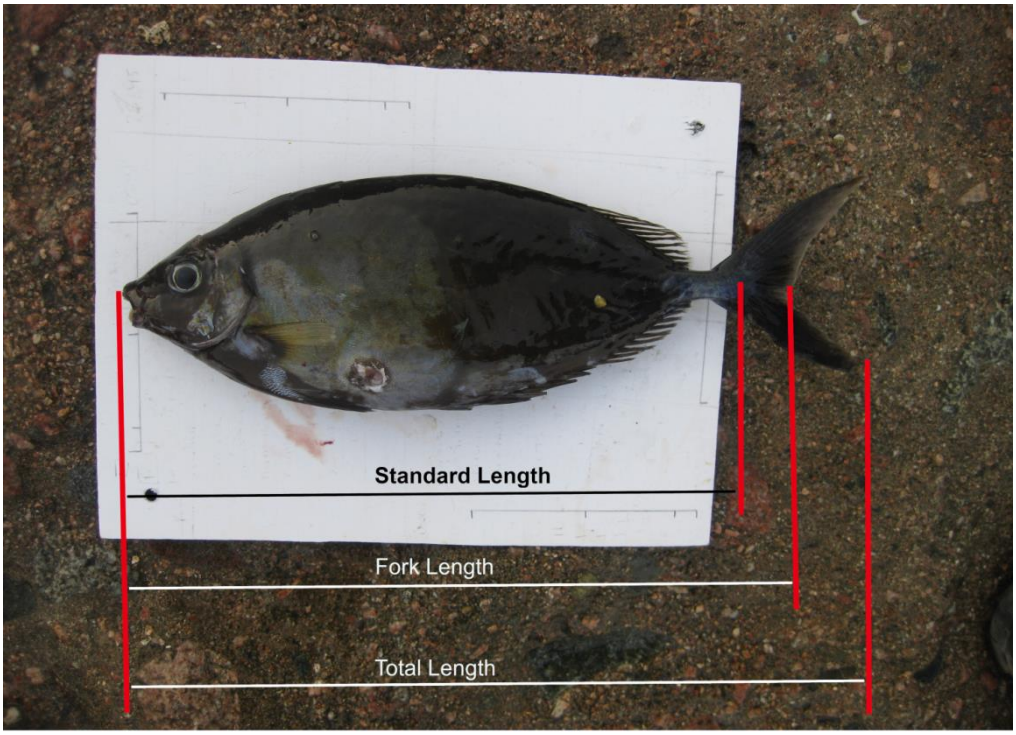


Figure 5.1. Measurement of Standard Length, Fork Length and Total Length

5.3. Results

Groups of fishers normally comprised family members such as brothers or father and son/s, and were often accompanied by young school-age children. The children would learn about fishing from their older relatives and sometimes play catching fish on the reef flat with a small net, usually releasing the often undersized reef fish they caught. This respect for fisheries sustainability was generally widespread, and adult fishers were careful not to damage coral and had a strong connection to the coastal ecosystem. All fishers were good swimmers, and set nets by swimming out with a mask and snorkel, floating the net itself on a modified surfboard or inflated rubber inner tube. There was a

clear division of labour, with certain members of the fishing team allocated to particular gears. When deployed on the reef flat, nets were held by hand and fishers aimed to scare fish into them. However, more experienced fishers preferred to set their nets on the reef slope and leave them for several hours, often overnight (pers. obs.). Both gill and trammel nets were employed with varying mesh sizes. Often, while a net was set, fishers would fish using spears or hook-and-line, or collect shellfish. Shellfish were normally cooked and eaten immediately on the fire that was constantly burning for tea preparation. Fishers used existing palm shelters *(arishas)* for shelter or would make impromptu shelters from tarpaulins if required. Some, more remote sites were located in security-restricted areas, and although police would stop vehicles and ask questions, they could easily be talked round to allow access for fishing, often in return for a portion of the catch.

A total of 33 species were recorded in Bedouin catch samples (Table 5.1). Scaridae (parrotfishes) composed 41.7 % of the catch by number and 51.9 % by weight. Only fifteen species accounted for 86 % of the catch (Table 5.1). Herbivores (Scaridae, Siganidae (rabbitfishes) and Acanthuridae (surgeonfishes)) accounted for 57.7 % of the catch and the mean trophic level by biomass was calculated at 2.8. Epinephelidae (groupers) were the only piscivorous family that contributed significantly to the catch (13.4 %). Four non-finfish species were also caught by hand, metal pole and speargun (Table 5.1).

Table 5.1. Summary of species caught by *Mzeina* Bedouin fishers in South Sinai

All L-W relationships used were as in Fishbase (Froese and Pauly 2018), estimated using the Bayesian approach given in Froese et al. 2014 except [1]Figure 5.2 and [2]El-Sayed Ali et al. 2011

L-L relationships were calculated for species where n≥3 and are given in Appendix 5.1

Estimates of vulnerability to fishing given in Fishbase (Froese and Pauly 2018), scored / 100 using the approach given in Cheung et al. 2005

Trophic levels calculated using Fishbase Ecology table

Family	Genus	Species	n	TL min	TL max	Trophic Level	Vulnerability	% N	% N (family)	Total biomass sampled / g	% total biomass	% total biomass for family
Acanthuridae	*Acanthurus*	*nigrofuscus*	1	13.0	13.0	2	27	0.6		51	0.1	
	Ctenochaetus	*striatus*	3	14.5	24.0	2	18	1.8	2.4	845	1.1	1.2
Apogonidae	*Cheilodipterus*	*macrodon*	1	21.9	21.9	4	25	0.6	0.6	221	0.3	0.3
Caesonidae	*Caesio*	*suevica*	3	17.8	21.2	3.4	26	1.8	1.8	374	0.5	0.5
Epinephelidae	*Cephalopholis*	*argus*	3	23.9	27.4	4.5	49	1.8		733	0.9	
	Cephalopholis	*miniata*	3	23.6	32.2	4.3	65	1.8		912	1.2	
	Epinephelus	*fasciatus*	2	19.4	21.8	3.7	53	1.2		221	0.3	
	Epinephelus	*tauvina*	3	27.9	39.4	4.1	59	1.8		2,005	2.6	
	Variola	*louti*	3	26.9	77.0	4.3	49	1.8	8.3	6,516	8.4	13.4
Fistulariidae	*Fistularia*	*commersonii*	1	62.9	62.9	4.3	68	0.6	0.6	465	0.6	0.6
Haemulidae	*Plectorhinchus*	*gaterinus*	5	25.4	28.5	4	41	3.0	3.0	1,378	1.8	1.8
Holocentridae	*Myripristis*	*murdjan*	8	16.9	31.6	3.4	35	4.8		3,345	4.3	
	Sargocentron	*spiniferum*	3	21.77	48.32	3.6	40	1.8	6.5	2,405	3.1	7.4
Labridae	*Cheilinus*	*abudjubbe*	1	23.9	23.9	3.5	36	0.6		272	0.4	
	Cheilinus	*lunatus*	4	24.7	40.0	3.6	48	2.4	3.0	3,089	4.0	4.3
	Coris	*aygula*	1	24.7	24.7	3.7	72	0.6	0.6	104	0.1	0.1
Lethrinidae	*Lethrinus*	*mahsena*	7	15.2	41.9	3.4	65	4.2	4.2	3,008	3.9	3.9

Family	Genus	Species	n	TL min	TL max	Trophic Level	Vulnerability	% N	% N (family)	Total biomass sampled / g	% total biomass	% total biomass for family
Lutjanidae	*Lutjanus*	*quinquelineatus*	1	24.2	24.2	3.7	34	0.6	0.6	181	0.2	0.2
Mullidae	*Mulloides*	*vanicolensis*	6	18.4	21.5	3.6	23	3.6		568	0.7	
	Parupeneus	*cyclostomus*	4	16.8	41.3	4.2	37	2.4		2,203	2.8	
	Parupeneus	*macronema*	8	15.3	25.2	3.5	33	4.8	10.7	965	1.2	4.8
Platycephalidae	*Papilloculiceps*	*longiceps*	1	49.8	49.8	4	46	0.6	0.6	742	1.0	1.0
Priacanthidae	*Priacanthus*	*hamrur*	3	19.9	28.6	3.6	24	1.8	1.8	682	0.9	0.9
Scaridae	*Calotomus*	*viridescens*	7	19.7	27.6	2	24	4.2		1,622	2.1	
	Chlororus	*sordidus*	8	23.1	34.9	2.6	19	4.8		3,510	4.5	
	Hipposcarus	*harid*	10	27.0	53.1	2	50	6.0		8,825[1]	11.4	
	Scarus	*ferrugineus*	22	20.8	37.4	2	61	13.1		14,516[2]	18.7	
	Scarus	*ghobban*	2	28.4	30.3	2	36	1.2		842	1.1	
	Scarus	*gibbus*	4	20.9	35.1	2	39	2.4		1,659	2.1	
	Scarus	*niger*	17	23.2	33.8	2	20	10.1	41.7	9,218	11.9	51.9
Siganidae	*Siganus*	*luridus*	17	17.4	26.6	2	39	10.1		2,255	2.9	
	Siganus	*rivulutus*	4	25.2	33.3	2	36	2.4	12.5	1,270	1.6	4.6
Sparidae	*Acanthopagrus*	*bifasciatus*	2	24.0	45.3	3.4	37	1.2	1.2	2,418	3.1	3.1
Total finfish			168							77,422		
Palinuridae	*Panulirus*	*penicillatus*	10									
Loliginidae	*Sepioteuthis*	*lessoniana*	1									
Strombidae	*Lambis*	*lambis*	1									
Trochidae	*Tectus*	*dentatus*	1									

Although Bedu often mentioned that they were waiting for large groups of fish, particularly Scaridae and Siganidae to arrive, they could rarely give any clear reason why the fish would be aggregating. The Bedu believed that Scaridae migrated in large groups around the Gulf of Aqaba, from Suez to Saudi Arabia and that the aggregations were linked to the lunar cycle. They indicated that Siganidae spawned during the summertime, and there was talk of restrictions on their exploitation during that season although there was limited knowledge on the detail of the implementation of these restrictions. On one fishing expedition in July 2017, just south of Dahab, we encountered an aggregation of long-nosed parrotfish *(Hipposcarus harid),* estimated by the Bedu at 150–200 kg of fish. The fishers aiming to catch them had been returning to the same spot regularly for several days, and were aware that the aggregation was due to arrive based on the moon phase. A team of five spent the whole afternoon in the water with several nets, catching as much of the aggregation as possible, and filling up the back of their pick-up truck. The fish were sold locally at US$ 3.90 kg^{-1} for terminal males and US$ 1.70 kg^{-1} for other colour phases. Interestingly, it appeared that the Bedouin fishers were unaware that the colour phases were of the same species, and regularly referred to them as 'the blue one' and 'the white one'. Sufficient fish from the aggregation were weighed to estimate the Length-Weight equation for *H. harid* as follows:

$W=0.0061\ L^{3.2501}$ (n=16; TL=26.1–43.7 cm; $R^2=0.9312$; Figure 5.2).

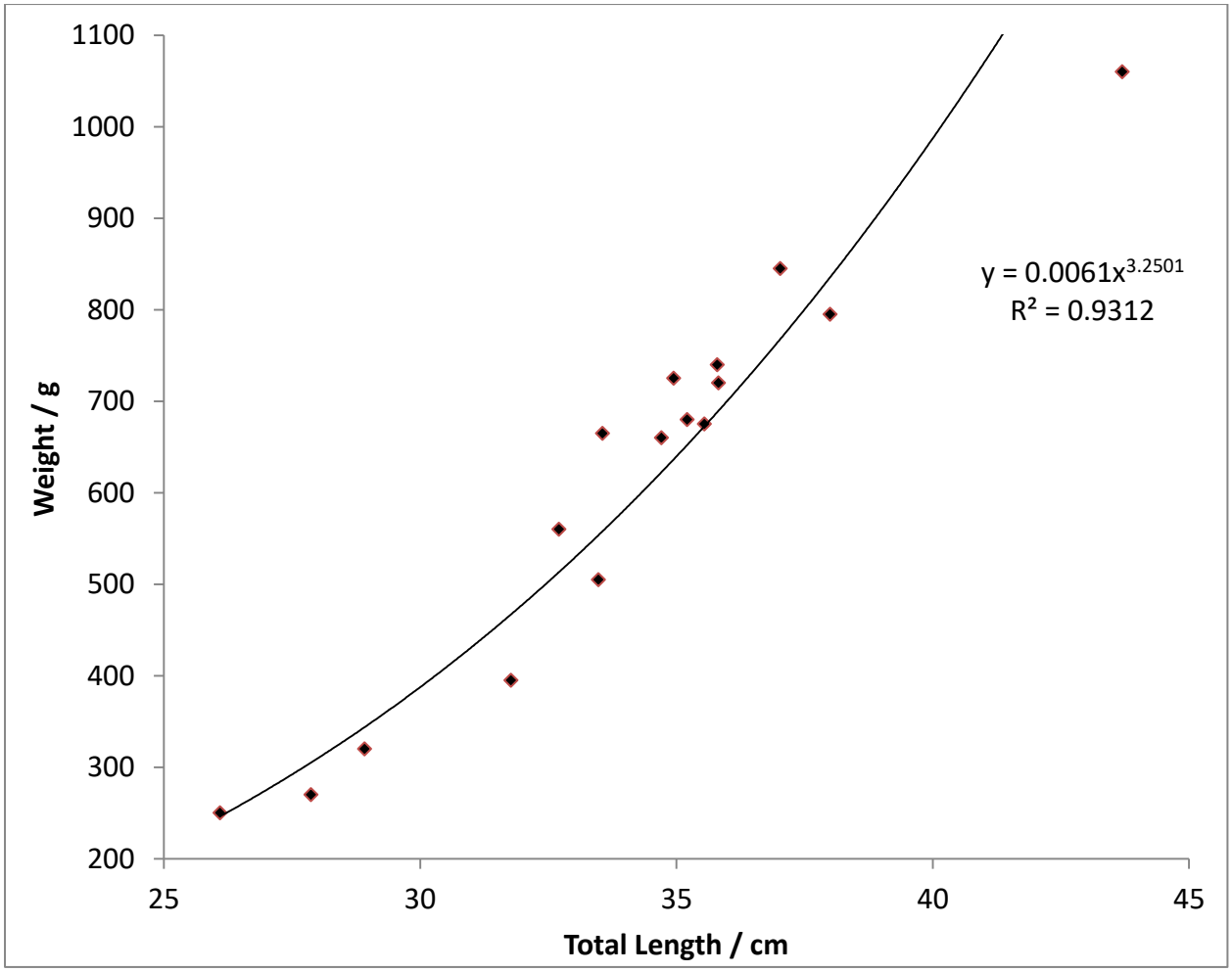


Figure 5.2. Length-Weight curve for *Hipposcarus harid*

Sample caught on the South Sinai Gulf of Aqaba coastline by *Mzeina* Bedouin artisanal fishers.

5.4. Discussion

Reef fisheries take about 200–300 species in the Pacific and about 100 in the Caribbean, but usually fewer than 20 species make up more than 75 % of the weight of the catch (Munro 1996). For example, only ten species of reef fish comprised 52 % of landings in Palau (Perron et al. 1983), five families accounted for 50–90 % of the catch in the Philippines (Maypa et al. 2002); six families represented 80 % of the catch in Madagascar (Laroche and Ramananarivo 1995) and 15 species made up 90 % of the catch in Kenya (McClanahan and Hicks 2011). This study only provides an opportunistically-sampled snapshot of Bedouin fishing activities in South Sinai, but it is clear that, although the fishery targets a variety of species and taxa, most of the catch is comprised of relatively

few species. The main fish families caught were seagrass and coral reef-associated species reflecting the habitats where fishing was focused. The valuable spiny lobster *(Panulirus penicillatus),* was a frequent component of the catch, most commonly speared during the night and it has been previously noted that a shift to such a nocturnal-only fishery often occurs as stocks of this species decline (Hearn and Murillo 2008). Also of note is the occurrence of commercially undesirable species in the catch including bluespotted cornetfish *(Fistularia commersonii)* and tentacled flathead *(Papilloculiceps longiceps)* which indicate that the fishery could be becoming less selective as preferred species decline. Many of the reef fish species caught were protogynous hermaphrodites which are particularly susceptible to overfishing, because disproportionately targeted fishing of larger, male individuals may result in a skew in sex ratios towards females, reducing the population's reproductive potential (Buxton 1993; Alonzo and Mangel 2004; Sato et al. 2018).

Herbivores dominated the Bedouin coral reef fishery, indicative of an overfished reef fish community where large piscivores have already been extirpated (Russ and Alcala 1989; Jennings et al. 1995; Jennings and Polunin 1996, 1997; Friedlander and DeMartini 2002; Mumby et al. 2012). Scaridae comprised over 50 % of the catch by biomass and Scaridae, Siganidae and Acanthuridae combined represented almost 60 % of the catch as also recorded at Nabq between 1995 -1997 (Galal 1999). Scaridae are widely exploited by artisanal fishers for the local market in the wider Red Sea and are one of the most popular fishes targeted on the coast of Saudi Arabia (Burger et al. 2014). If the aggregation of *H. harid* were included, the percentage composition of Scaridae would obviously have been even higher. Similar catch composition has been reported from heavily fished reefs in East Africa (McClanahan et al. 2008) and the Caribbean (Hughes 1994). Alternatively, Bedouin fisheries may disproportionately target herbivores since they are concentrated on the coastal shallow areas on the reef flat where herbivore populations are most abundant (Bouchon-Navaro and Harmelin-Vivien 1981). Small herbivorous fishes generally have rapid population doubling times (1.5– 4 y), which means they are less vulnerable to over-fishing than larger-bodied predators (e.g. Serranidae,

Lutjanidae (snappers), and Chondrichthyes (sharks and rays)) that grow more slowly and are longer-lived. The short life span and rapid growth of the Siganidae makes them particularly resilient to fishing pressure (Robinson et al. 2011; Samoilys et al. 2013). However, higher predators rely on smaller fish species for food, and fisheries that remove prey fish can, therefore, indirectly affect populations of their predators (Smith et al. 2010). Furthermore, the Scaridae and other herbivores are important in reducing macroalgal abundance and enhancing the dominance of crustose corallines, which are necessary conditions for the maintenance of healthy reef communities (McManus et al. 2019). Fishing restrictions that protect large, high-grazing-rate species, or designation of no-take areas, are likely to maximise algal removal (Nash et al. 2016) and controls on the exploitation of herbivores, particularly Scaridae have been successfully implemented in Hawaii (Williams et al. 2016) and the Caribbean (Cox et al. 2013; O'Farrell et al. 2016).

The discovery of an aggregation of *H. harid* was a first record for South Sinai, although this species is already known to aggregate annually in the Saudi Arabian Red Sea, where local fishers enjoy huge catches at the yearly '*Harid* festival' (Gladstone 1996; Spaet 2013). *H. harid* has been observed in groups consisting of a terminal phased individual accompanied by numerous individuals in the initial phase (Lieske and Myers 1999). However, the species is believed to be oviparous and protogynous with distinct pairing during breeding (Breder et al. 1966) and has never been confirmed in spawning aggregations like its close relative *H. longiceps* (Colin and Bell 1991). Species that aggregate at specific locations and times, often for spawning, face additional risk, because the predictable nature of these aggregations means that fishers can easily target them, and remove a large number of individuals from the population in a single event (Domeier and Colin 1997; Sadovy and Domeier 2005; Robinson et al. 2011). Thus a highly precautionary approach is recommended in their management (de Mitcheson 2016).

The relatively rare, and often highly localized nature of spawning aggregations makes it difficult to initially identify locations and timing, thus traditional ecological knowledge of local fishers is

commonly the main source of baseline data (Silvano et al. 2006; Hamilton et al. 2012) and means that unfished aggregations have rarely been studied (Domeier and Colin 1997) since they are rarely found by chance. Fishers' knowledge about spawning aggregations is often extremely detailed since regular aggregations have often been fished for centuries, or seasonal gluts in landings have subsequently been recognized as spawning aggregations (Johannes 1978). However, these details are often seen as 'privileged information' and to acquire this it is essential to progressively build trust (Colin et al. 2003; Hamilton et al. 2012). Bedouin fishers stated that they target large aggregations of both Scaridae and Siganidae and fisheries management strategies should be developed through detailed consultations on the environmental factors that influence these aggregations.

Currently, no reliable estimation of the Length-Weight relationship of *H. harid* is recorded in the global species database of fish species, Fishbase (Froese and Pauly 2018). However, two other evaluations have been published, both from the Red Sea (Table 5.2). Variation in the relationships derived from this study versus these previous evaluations may be due to the location and gear used to collect the samples resulting in different growth rates and size-ranges (Gabr and Mal 2016).

Table 5.2. Previous evaluations of the L-W relationship of *H. harid* in the Red Sea

Location	Fishery	n	Size range/cm	a	b	R^2	Reference
Jeddah, Saudi Arabia	Commercial longline	175	15.9–31.0 TL	0.0250	2.99		El-Sayed Ali et al. 2011
Hurghada, Egypt	Commercial	1,000	17–50 TL	0.0182	2.9323	0.9747	Mehanna et al. 2014

Low-intensity subsistence fishing in remote areas is unlikely to significantly alter coral reef fish communities (Martin et al. 2017), however, as tourism has grown in South Sinai, so too has demand for fresh fish to feed tourists as well as the local Bedouin population and immigrant Egyptian and international workers (Pearson and Shehata 1998). As the tourist industry has developed, even more remote sites have been made easily accessible, including regular boats to Ras Abu Gallum where

Bedouin hosts catch local fish for guests on demand. Bedu frequently lamented the decline of their reef fisheries, explaining how they used to only catch predatory Lutjanidae and Serranidae with hook-and-line, and only in more recent years, they had started using nets to catch herbivorous species such as Scaridae and Siganidae (Chapter 2). The Malthusian explanation of overfishing suggests that fishing effort and unselective gear use increase in proportion to human population growth and declining resources, particularly as the fishery transforms from being primarily subsistence to commercial: in this case, due to rapid tourism growth (Pauly et al. 1989; Pauly 1990; McManus 1997; Cinner and McClanahan 2006). However, human social organization and the development of economic alternatives can potentially solve such social-ecological tragedies (Dietz et al. 2003; Berkes and Seixas 2005; Turner et al. 2007).

In South Sinai, as for many coral reefs around the world, there is limited financial and technical capacity for fisheries monitoring and management (Pauly and Zeller 2014; McClanahan 2018). Furthermore, Egyptian fisheries authorities' consistent failure to follow management recommendations has been further compounded by poor enforcement of rules and lack of integration of local fishers in decision-making (Pramod and Pitcher 2006; Samy-Kamal 2015). These realities make it essential to identify cost-effective monitoring tools to track ecological and social aspects of artisanal fisheries, to inform adaptive state and community-level fisheries management. The most successful initiatives are often comprised of multi-pronged, collaborative approaches between the state and local stakeholders (Granek and Brown 2005). However, in Egypt, a strong history of top-down governance of not only environmental issues, but also of wider societal interactions (Hassan 2011; Gilbert and al Jebaali 2012) complicates the necessary decentralization to revitalize local-level responsibility for fisheries management. Furthermore, the state fears insurgency from Bedouin tribes that could impact tourism income from South Sinai (Dyer and Kessler 2014; Aftandilian 2015) and so is likely loathe to introduce controversial fishing regulations. However, local fisheries management could be enhanced through recognition of the *Mzeina*s' right to South Sinai's reef

fisheries, as an expression of self-determination and independence from the Egyptian state (Ban and Frid 2018; Jentoft et al. 2019). Illegal, unregulated, and undocumented (IUU) fishing activities are a major cause of negative fishing impacts (Sumaila et al. 2006) with a lack of awareness of local regulations, as well as weak law enforcement, acting as key contributing factors for lack of compliance (Bergseth et al. 2015). Involving local stakeholders in monitoring and management can help detect and quantify the extent of illegal activities and potentially improve compliance with regulations (Poh and Fanning 2012; Arias et al. 2015).

Managers and scientists cannot monitor the entire geographic areas of most coral reef fisheries as intensely as needed. Modern technology can provide opportunities to improve natural resource management in remote areas; for example, smartphones and the Internet of Things have been successfully employed, not only for fisheries monitoring (Paul et al. 2016) but also for endangered species surveys by non-specialists (Heigl et al. 2017; Jedersberger et al. 2018; Ožana et al. 2019). Mobile internet networks are reliable along much of the South Sinai coastline and Bedu regularly use smartphones to communicate with each other. Many *Mzeina* would be keen to show off their catch by taking digital photographs similar to those used in this study. However, there would certainly be suspicions that the images could be utilized to provide evidence of illegal fishing or to initiate draconian restrictions in the future. The low species diversity of the Bedouin catch might mean that only monitoring one or two indicator species or families could provide adequate data to inform management. For example, average individual Scarid weight has been shown to act as a simple indicator of the influence of fishing effects on moderate-to-heavily fished Caribbean reefs, where fish groups of high commercial value might be too rare to be used as reliable indicators (Vallès and Oxenford 2014). Furthermore, monitoring the Scaridae assemblage, particularly the dominance of resilient species, can give an insight into fishing pressure (Clua and Legendre 2008). Such data collection could enable comparisons of catch and effort on a finer scale (e.g. within a survey area, better differentiation of gear types) and variability across space and time.

Locations under community-based management have similar amounts or greater fish biomass compared to no-take protected areas thus sustaining both ecosystems and ecosystem benefits (Friedlander et al. 2013). However, community-based marine protected areas (MPAs) are often too small to effectively conserve wider fish stocks and tropical coastal habitats (Weeks et al. 2010), especially for large and mobile species (Sale et al. 2005; Filous et al 2017). Sites where endangered species aggregate in space or time (e.g. seabird nesting colonies, sea turtle rookeries) should thus be a priority for management actions (Edgar et al. 2010) where protection of a relatively small area will maximize conservation impact (Hamilton et al. 2011; Erisman et al. 2017) although few no-take fishing reserves specifically consider spawning sites (Sale et al. 2005). Closures have proved successful in the recovery of overfished spawning aggregations (Beets and Friedlander 1999) and such an approach could prove beneficial in South Sinai, where closures could be implemented based on the fish aggregations' primary locations or seasons.

Adaptive management approaches and the establishment of stronger social systems to enable management are essential needs for the governance of the *Mzeina* fishery (Cinner et al. 2006; Aswani et al. 2007; Anthony et al. 2015). Feedback and co-learning are essential components of any multi-stakeholder management approach and data collected must be analysed and presented back to the Bedouin community. Such information can help communities develop a more informed understanding of the drivers of marine resource harvest and the state of the resources (Semitiel-García and Noguera-Méndez 2019). This, in turn, helps inform effective, sustainable community-based fisheries management such as community-managed subsistence fishing areas (Ayers and Kittinger 2014). Currently, the relationship between reef users is poor, and Bedouin fishers are regularly shamed on social media by diver groups for fishing on local reefs, because it is widely believed that it is illegal (Lucrezi et al. 2017). Several dive schools have established reef monitoring programmes and Bedu and their fisheries knowledge should be integrated into these programmes with widespread dissemination of results among and collaboration with the Bedouin community.

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Appendix 5.1. Length-Length relationships for fish species where n≥3

Family	Species	FL=a+bTL (n)	SL=a+bTL (n)	SL=a+bFL (n)
Acanthuridae	*Ctenochaetus striatus*	FL=0.8822+0.816TL (4)	SL=-0.5277+0.7129TL (4)	SL=-1.256+0.8713FL (4)
	Caesio suevica	FL=-1.1413+0.9139TL (3)	SL=-1.3205+0.8666TL (3)	SL=-0.2359+0.9481FL (3)
Epinephelidae	*Cephalopholis argus*	-	SL=-13.856+1.3839TL (3)	-
	Cephalopholis miniata	-	SL=4.3272+0.6748TL (3)	-
	Epinephelus tauvina	-	SL=-5.9222+1.0207TL (3)	-
	Variola louti	FL=3.8048+0.7005TL (3)	SL=3.6281+0.6063TL (3)	SL=0.3483+0.8652FL (3)
Haemulidae	*Plectorhinchus gaterinus*	FL=2.7342+0.8585TL (4)	SL=6.209+0.6046TL (5)	SL=8.7943+0.534FL (4)
Holocentridae	*Myripristis murdjan*	FL=-0.8162+0.9673TL (8)	SL=-0.0565+0.824TL (8)	SL=0.7125+0.8488FL (8)
	Sargocentron spiniferum	FL=1.6773+0.8609TL (3)	SL=0.416+0.8265TL (3)	SL=-1.1964+0.9602FL (3)
Labridae	*Cheilinus lunatus*	-	SL=6.0232+0.5305TL (3)	-
Lethrinidae	*Lethrinus mahsena*	FL=-1.1036+0.968TL (7)	SL=-0.5114+0.8442TL (7)	SL=0.4589+0.8718FL (7)
Mullidae	*Mulloides vanicolensis*	FL=3.8175+0.6912TL (6)	SL=3.202+0.6694TL (6)	SL=0.7255+0.9FL (6)
	Parupeneus cyclostomus	FL=-0.2231+0.8868TL (4)	SL=-0.299+0.8239TL (4)	SL=-0.0241+0.9266FL (4)
	Parupeneus macronema	FL=0.3531+0.8783TL (8)	SL=-1.1067+0.8958TL (8)	SL=-0.0294+0.938FL (8)
Priacanthidae	*Priacanthus hamrur*	FL=1.3313+0.8963TL (3)	SL=2.1413+0.7557TL (3)	SL=1.0147+0.8433FL (3)
Scaridae	*Calotomus viridescens*	-	SL=1.6989+0.7732TL (7)	-
	Chlororus sordidus	-	SL=-1.8495+0.9258TL (8)	-
	Hipposcarus harid	FL=7.2123+0.7032TL (8)	SL=4.9936+0.6997TL (10)	SL=-1.9603+0.9828FL (8)
	Scarus ferrugineus	FL=-6.3086+1.1495TL (6)	SL=0.1093+0.8606TL (22)	SL=1.7909+0.836FL (8)
	Scarus gibbus	-	SL=-0.9827+0.8771TL (4)	-
	Scarus niger	FL=1.1305+0.921TL (6)	SL=0.9982+0.807TL (17)	SL=3.1844+0.7735FL (6)
Siganidae	*Siganus luridus*	FL=-3.7443+1.1078TL (4)	SL=0.3854+0.8304TL (17)	SL=2.7389+0.7418FL (4)
	Siganus rivulutus	FL=-3.4413+1.0179TL (4)	SL=-2.9108+0.9244TL (4)	SL=0.8025+0.8864FL (4)

CHAPTER 6. LONG TERM TRENDS IN CORAL REEF ECOSYSTEM DEGRADATION IN SOUTH SINAI

Abstract

Coral reefs are complex but heavily threatened marine ecosystems, essential to the subsistence and livelihood of millions of the poorest people worldwide. Reef ecosystems operate at a wide range of temporal and spatial scales and the scale of negative impacts often outspans the scale of scientific monitoring programmes in space and time and thus long-term monitoring has been noted as a management priority. The coral reefs of Dahab, South Sinai in the Gulf of Aqaba, Red Sea have been subject to increased negative impacts in recent years due to uncontrolled expansion of international tourism in the region. Several studies have attempted to document the coral reefs and associated biota in the Dahab area, but almost none have extended beyond a single year. Reef Check monitoring has been conducted by dive volunteers for 15 years, although there were gaps in the dataset because some sites were not surveyed in some years. I employed a log-linear Poisson regression model (RTrim) to imput missing data points and recreate long term trends for Reef Check fish, invertebrate and substrate indicators. The significant trends that were found painted a picture of long-term decline in ecological function of Dahab's reefs. Fish at all trophic levels showed declines (Serranidae: Wald=13.14, df=1, p<0.001; Lutjanidae: Wald=7.94, df=1, p< 0.01; Scaridae: Wald=10.05, df=1, p< 0.01; Chaetodontidae: Wald=24.64, df=1, p< 0.001) and top predators were absent, indicating overfishing. *Tridacna* giant clams showed an increase in abundance at depth (Wald=4.97, df=1, p< 0.05) even though they are heavily fished on the reef flat. Other invertebrate species had declined due to collection for food and the curio trade (Diademids: Wald=34.44, df=1, p< 0.001; *Tripneustes spp.*: Wald=23.91, df=1, p< 0.001; edible sea cucumbers: Wald=11.44, df=1, p< 0.001; triton: Wald=4.32, df=1, p< 0.05). Nutrient indicator algae had substantially increased in abundance (Wald=11.37, df=1, p< 0.001), likely as a result of eutrophication due to pollution from coastal development and reductions in herbivory. Hard coral cover had remained relatively stable, although it is likely to decline if the current environmental impacts from coastal development and fishing remain unmitigated. Tourism development should be more closely regulated and coastal areas must be designated for fishing closures or, at a minimum, gear and species restrictions, ideally in close collaboration with the local *Mzeina* population.

6.1. Introduction

Coral reefs are complex, diverse and productive tropical ecosystems incorporating multiple physical and biological processes that interact over a variety of temporal and spatial scales (Connell 1978; Sorokin 1993). Many of the world's most vulnerable people are reliant on the continued function of these ecosystems for income, tourism, food and coastal protection, creating an economic value in the order of tens of billions of dollars per year globally (Moberg and Folke 1999; Cesar et al. 2003; Spalding et al. 2017). Coral reef ecology has only been studied in detail in relatively recent times, following the development of SCUBA diving equipment in the 1950s (Marx 1990; Jackson 1997). However, environmental pressures, the majority of which are anthropogenic in origin, have exceeded coral reefs' natural regenerative capacity and caused rapid deterioration, particularly over the last 50 years (Johannes 1975; Dustan 1977; McClenahan 1987; Hughes 1994; Bryant et al. 1998; Pandolfi et al. 2003; Bellwood et al. 2004; Birkeland 2019). It has been estimated that 19 % of coral reef area has been lost and at least 35 % more will be lost by 2050 (Wilkinson 2008). If such rates of degradation continue, it is likely that reefs will soon fail to fulfil their ecologically necessary and economically valuable functional roles resulting in food security problems for coastal populations worldwide (Hoegh-Guldberg et al. 2007).

A key issue for the environmental monitoring and management of coral reefs is the scope of the wide temporal and spatial scales at which they operate. At the global level, the most significant threat by far to reefs is climate change (Bruno and Valdivia 2016) which results in coral bleaching (Hughes et al. 2003) and ocean acidification (Doney et al. 2009). Regional threats include disease outbreaks (Mohamed and Sweet 2019), severe weather events (Beeden et al. 2015), sea urchin die-off (Lessios 2016), crown of thorns starfish *(Acanthaster planci)* infestations (Babcock et al. 2016) and pollution (Abdallah et al. 2016). At the local level, reefs are affected by fishing activity (Friedlander et al. 2018), sedimentation (Maina et al. 2013), eutrophication (Mwaura et al. 2017) and recreational diver impacts (Poonian et al. 2010). Sometimes these impacts take decades to manifest themselves in terms

of measurable ecosystem disruption (Riegl et al. 2009), and must be teased apart from natural cycles. Thus short-term, local studies are insufficient to fully understand the ecological processes occurring and thereby generate reliable management recommendations. Around the world, many reefs have shown consistent downward trends (Gardner et al. 2003; Pandolfi et al. 2003; Bellwood et al. 2004; Bruno and Selig 2007; De'ath et al. 2012; Cecarelli 2019); while others show recovery from impacts, at least at the local level (Glynn and Fong 2006; Baker et al. 2008; Graham et al. 2015; Holbrook et al. 2018; Edmunds et al. 2019). The most resilient reefs have been characterized by high levels of local engagement in management as well as beneficial environmental conditions such as deep water refugia (Cinner et al. 2016).

The long-term monitoring of coral reefs over time scales of more than 5 years and at multiple spatial scales has widely been noted as a priority (Magurran et al. 2010; Brown et al. 2016). However, evaluating long term trends on coral reefs can be complicated by disparities in survey methodologies between subsequent researchers and data missing for years or sites because of lack of funding or manpower (Vallès 2019; Peterson et al. 2020). Consequently, long-term trends can be overlooked, resulting in an unrecognized slow trajectory towards ecological decline (Pandolfi et al 2003). Reliable time series data are therefore essential to quantify the rate of change of reefs and inform projections of future conditions (Pandolfi et al. 2011). Furthermore, the variables that should be monitored can be difficult to determine, since the potential causes of community change on reefs are so diverse and dynamic (Bellwood et al. 2004).

Reef Check was founded in 1996, as a solution to overcome some of the issues associated with the long-term monitoring of reefs, and to date, Reef Check data are the only field-based long-term coral reef data collected on a global scale using a standardized protocol (Hodgson 1999, 2001; Hodgson et al. 2003; Obura et al. 2019). Reef Check's approach overcomes limitations of manpower and funding by using trained volunteer divers and the validity and resolution of data collected by these non-specialists has been well-documented (Darwall and Dulvy 1996; Goffredo et al. 2004; Forrester et al.

2015; Done et al. 2017; Gouraguine et al. 2019). Similar citizen scientist programmes have also been used to successfully monitor other taxa, including mammals (Barlow et al. 2015), birds (Jiguet et al. 2012), reptiles (Hof et al. 2017) and amphibians (Petrovan and Schmidt 2016). The Reef Check protocol focuses on a limited number of species, based on their economic and ecological value, to improve the likelihood of accurate data collection (Lincoln Smith 1988). Only a small number of selected indicators are required to diagnose stressors on coral reefs and guide management responses (Flower et al. 2017) and the high density of recreational divers in the Red Sea has created great potential for citizen science approaches (Branchini et al. 2015; Mancini and Elsadek 2019).

Although the Red Sea has been noted as one of the most important oceans for coral reef ecosystems in the world, the region lacks scientific research and data are currently inadequate for informed management (Berumen et al. 2013). To date, research has been disproportionately focussed on the Israeli coast, Ras Mohamed National Park and the Southern Red Sea. Long term (c. 50 y) studies in Eilat at the northern end of the Gulf of Aqaba have described widespread coral reef ecosystem collapse due to eutrophication from sewage discharge and intensive mariculture (Loya 2004; Gilad et al. 2018). Although corals in the Red Sea, particularly in the north, are believed to be particularly resistant to thermal stress (Fine et al. 2013; Sawall et al. 2015; Osman et al. 2018), they have been affected to varying extents by the three recent major bleaching events: 1998 (Kotb 2004), 2010 (Furby et al. 2013), and 2015 (Monroe et al. 2018).

The study area examined here includes the Gulf of Aqaba coastline surrounding Dahab, South Sinai from Gabr el Bint in the South to Ras Abu Gallum in the North (Figure 6.1). South Sinai has experienced rapid tourism development since the 1980s, with a focus on scuba diving (Shackley 1999; Hasler and Ott 2008). The associated domestic and international immigration has increased local populations, with a corresponding increase in impacts on the fringing coral reef that runs the length of the coastline (Bryant et al. 1998; Jobbins 2006). State-led efforts in the area to develop effective marine protected areas (MPAs) have shown some promise (Galal et al. 2002; Ashworth

2004; Ashworth et al. 2004; Ashworth and Ormond 2005), but have ultimately failed to fulfil their objectives (Galal et al. 2012; Samy et al. 2011). Previous research has suggested deterioration of reefs as a result of small-scale fishing, eutrophication and uncontrolled recreational diving and snorkelling (Table 6.1). However, these changes have not been quantified at larger scales beyond the level of individual study sites, specific ecological variables, or over longer time periods. Most studies had a narrow focus and did not prioritize the integration of their findings into long-term reef management strategies for the region.

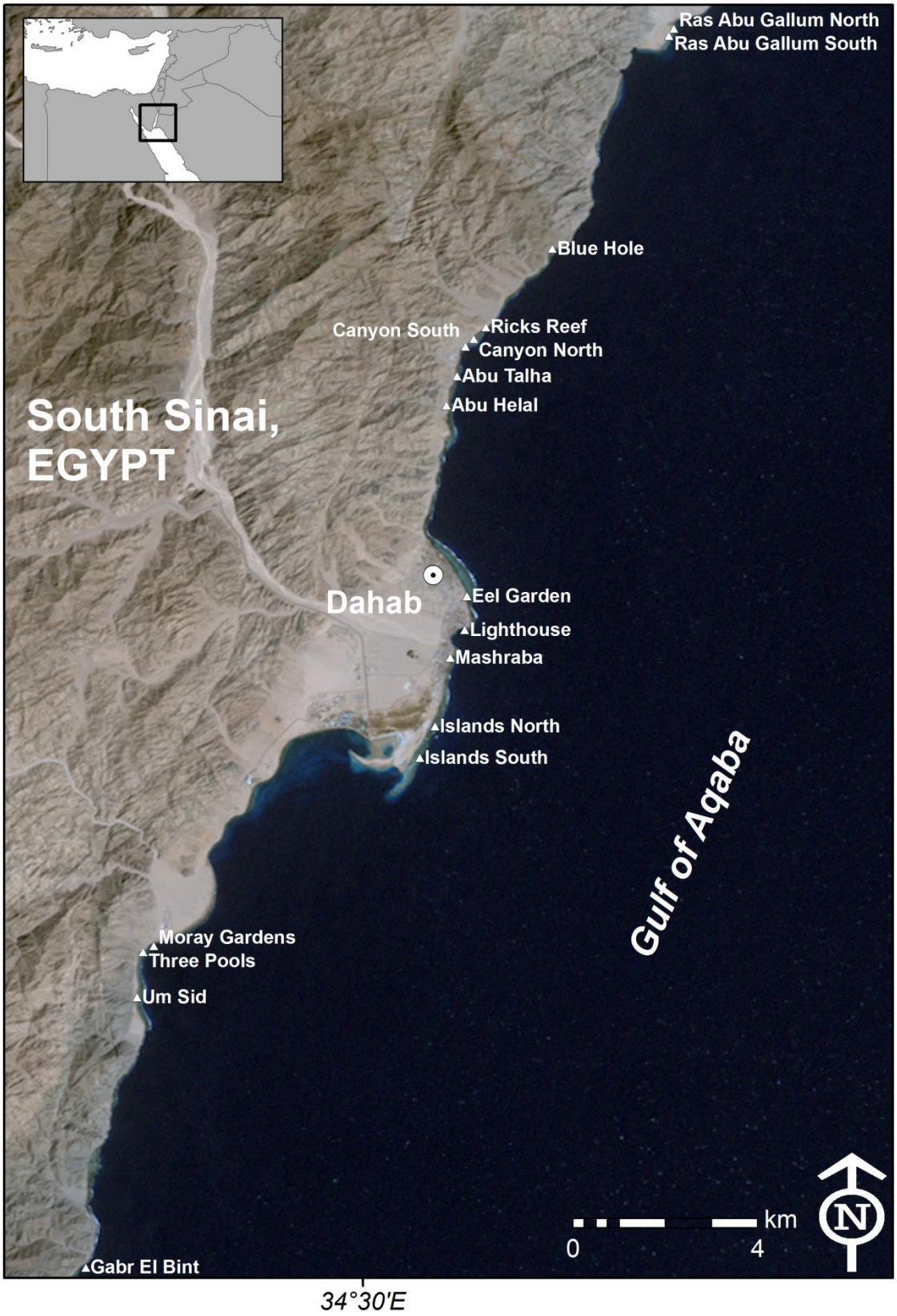


Figure 6.1. The study area showing Reef Check sites

Table 6.1. Previous surveys of coral reefs and associated organisms in the vicinity of Dahab, South Sinai

Time scale	Number of repeated surveys	Number of sites within study area	Summary of findings and/or subject studied	Citation
1986	4 intra-annual	1	Evaluation of population structure and growth parameters of *Panulirus penicillatus*	Plaut and Fishelson 1991
2000	1	2	Parrotfish density estimates and feeding behaviour study	Alwany et al. 2009
2002 and 2003	2	2	Moderate decline in coral cover	Tilot et al. 2008
2002 and 2012	2	7	Underwater Visual Census (UVC) of reef fish showing increases in abundance of overall assemblage but reductions in size of target species	Mabrouk 2015
2003	1	6	UVC of reef fish, showing that fish assemblages around Dahab were similar to unfished Nabq and lightly fished Ras Mohammed sites	Ashworth 2004
2003–2018	na	na	1,066 molluscan species confirmed in the Dahab area, including 781 Gastropoda, 250 Bivalvia, 16 Polyplacophora, 12 Cephalopoda and 7 Scaphopoda	Blatterer 2019
2004	1	6	Survey of reef community diversity and abundance: 107 hard and 19 soft coral species identified	Ammar et al. 2006
2006	1	8	Survey of benthic communities and coral condition to evaluate diver impacts	Hasler and Ott 2008
2006	1	2	Sea cucumber decline when compared with previous data	Hasan and El-Rady 2012
2006–2007	1	1	Survey of coral reef benthic community composition	Ali and Emara 2007
2007	1	1	Survey of coral cover, morbidity and mortality	Ammar 2009
2007	1	1	Survey of benthic communities and coral condition on the reef flat	Kompatscher 2009; Hannak et al. 2011
2007	1	1	Survey of coral diversity and *Drupella cornus* ecology	Schoepf et al. 2010

Time scale	Number of repeated surveys	Number of sites within study area	Summary of findings and/or subject studied	Citation
2007–2008	1	7	Survey of abundance and species diversity of sponges and % cover of coral	Wagler and Brümmer 2009
2008	1	1	Relatively low heavy metal contamination and high coral cover for the Northern Red Sea	Ali et al. 2011
2008–2010	3 intra-annual	7	Survey of Labridae abundance and diversity	Reininger 2012
2009	1	2	Coral diversity study	Ammar 2011; Ammar and Madkour 2011
2009– 2010	1	1	Presence of coral diseases	Mohamed et al. 2012
2010	1	1	Estimation of bivalve and gastropod abundance	Zuschin and Ebner 2015
2010	1	6	Evaluation of molluscan biodiversity on coral reefs	Westenberg 2012
2010	1	1	Intertidal habitat mapping and molluscan assemblage zonation	Gützer 2011
2010	1	12	Evaluation of *Conus spp.* diversity	Zauner and Zuschin 2016
2010–2011	1	1	Survey of *Pterois spp.* density	McTee and Grubich 2014
2010–2013	3	3	Rapid change in benthic community composition as a result of eutrophication from coastal development	Naumann et al. 2015
2011	1	2	Evaluation of damselfish abundance and distribution	Nadler et al. 2014
2014	1	3	Survey of Mullidae abundance and diversity	Sarhan et al. 2014
2016	4 intra-annual	1	Survey of benthic macroinvertebrate community	El Naggar 2017
Unknown	1	4	Shrimp goby diversity abundance and distribution	Freinschlag and Patzner 2012
Unknown	1	1	Declining water quality causing overgrowth of corals by macroalgae	Ali and Hamed 2006
Unknown	1	2	Coral disease study	Ammar et al. 2013

The aim of this analysis was to use publically-available historical Reef Check data from the coastline around Dahab, South Sinai to fill in some knowledge gaps and to evaluate long-term trends in densities of Reef Check 'indicator' species and assess their potential causes. To accomplish this, I employed a log-linear Poisson regression model to imput the missing data points and recreate general long-term trends to understand the most urgent reef management needs in the region.

6.2. Materials and Methods

Reef Check is a monitoring protocol developed in 1997 for the global assessment of coral reef status and trends (Hodgson 2001). The protocol consists of transect surveys conducted by volunteer divers at various depth contours along reef slopes to evaluate substrate composition and populations of fish and invertebrate 'indicator' species (Hodgson et al. 2003). Reef Check surveys collect four types of data: (1) a description of each reef site based on over 30 measures of environmental and socio-economic conditions and ratings of human impacts; (2) a measure of the percentage of the seabed covered by different substrate types, including live and dead coral, along four 20 m sections of a 100 m shallow reef transect; (3) invertebrate counts over four, 20 m x 5 m belts along the transect; and (4) fish counts, up to 5 m above the same belt. The Reef Check 'indicator' species recorded during each survey are given in Table 6.2; coral disease and bleaching, trash and coral damage are also noted.

Table 6.2. Species recorded during Reef Check surveys in the Red Sea

Reef fish counts	Invertebrate counts	Substrate % cover
• Grouper >30cm (Serranidae) • Butterflyfish (Chaetodontidae) • Sweetlips (Haemulidae) • Parrotfish >20 cm (Scaridae) • Broomtail wrasse *(Cheilinus lunulatus)* • Snapper (Lutjanidae) • Humphead wrasse *(Cheilinus undulatus)* • Moray eel *(Muraenidae)* • Bumphead parrotfish *(Bolbometopon muricatum)*	• Long-spined black sea urchin (*Diadema* and *Echinothrix diadema)* • Banded coral shrimp *(Stenopus hispidus)* • Lobster (spiny and slipper/rock) (Malacostraca - Decapod) • Sea Egg/Collector urchin *(Tripneustes spp.)* • Crown-of-thorns star *(Acanthaster planci)* • Triton *(Charonia tritonis)* • Giant clam *(Tridacna spp.)* • Sea cucumber (edible only) • Pencil urchin *(Heterocentrotus mammilatus)*	• Hard Coral • Soft Coral • Recently Killed Coral • Nutrient Indicator Algae • Sponge Rock • Rubble • Sand • Silt/Clay • Other

Reef Check surveys have been conducted by the Red Sea Environmental Centre (RSEC) in Dahab from Ras Abu Gallum to Gabr el Bint since 2002, generating a fifteen-year timeline of data, with the latest data available online from 2016. An earlier one-off survey had been carried out in 1997 at Gabr El Bint, but was not included since it would not be acceptable to the statistical model owing to the large time gap. I downloaded the historical, publically available dataset from Reef Check's online Global Reef Tracker (http://data.Reef Check.us/) in order to assess long-term trends in coral reef ecosystems along the South Sinai coastline. However, the dataset was patchy, with some sites not surveyed every year and others done multiple times in the same year. A total of 32 survey sites were identified with deep (8–11 m) and shallow (4–6.5 m) areas regarded as independent sites. Over the 15 years, a total of 168 complete surveys were conducted out of a possible 480 with between 1 and 20 sites being surveyed each year (Table 6.3). All survey sites coincided with popular dive sites (Figure 6.1). If multiple surveys occurred in the same year, then extra surveys were disregarded, based firstly on the presence of any errors in the survey process and secondly on the temporal spread when compared to previous and subsequent surveys at that site.

The gaps in these datasets were dealt with through use of the rTrim-package in R; the TRIM model is used to estimate species populations based on annual counts at multiple sites (Bogaart et al 2018). The model takes account of missing data by imputing prior to estimation of the population totals. The rTrim package uses a log-linear Poisson regression model, accounting for missing counts, serial autocorrelation and overdispersion to estimate temporal trends and variability of population counts (Van Strien et al. 2004; Pannekoek and van Strien 2005). Indices were plotted to illustrate the change, since the counts themselves only covered the sampled area and the aim was to elucidate the overall trend. The effect of depth as a covariate was also evaluated. The significance of temporal trends and influence of covariates were tested with a Wald $\chi 2$ test (Pannekoek and van Strien 2005; Bogaart et al 2018).

Table 6.3. Survey sites and years of previous Reef Check surveys in Dahab

*only fish survey; ** only fish and invert survey; ***only substrate survey

Site		2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	TOTAL
Abu Helal	Deep						✓		✓		✓		✓	✓		✓	**6**
Abu Helal	Shallow						✓		✓		✓		✓	✓	✓		**6**
Abu Talha	Deep			✓	✓**	✓											**3**
Abu Talha	Shallow		✓	✓*	✓												**3**
Blue Hole	Deep						✓		✓	✓	✓	✓	✓	✓	✓	✓	**9**
Blue Hole	Shallow						✓		✓	✓	✓	✓	✓	✓			**7**
Canyon North	Deep						✓		✓	✓	✓			✓	✓	✓	**7**
Canyon North	Shallow								✓					✓			**2**
Canyon South	Deep				✓		✓		✓	✓	✓			✓	✓	✓	**8**
Canyon South	Shallow			✓							✓						**2**
Eel Garden	Shallow			✓	✓***												**2**
Gabr El Bint	Deep			✓	✓	✓	✓		✓	✓		✓		✓	✓		**9**
Gabr El Bint	Shallow			✓			✓		✓	✓	✓						**5**
Islands North	Deep			✓					✓	✓	✓	✓	✓				**6**
Islands North	Shallow			✓				✓					✓				**3**
Islands South	Deep								✓	✓	✓		✓	✓			**5**
Islands South	Shallow								✓	✓	✓		✓	✓	✓		**6**
Lighthouse	Deep	✓			✓	✓			✓	✓	✓	✓		✓	✓	✓	**10**
Lighthouse	Shallow								✓	✓	✓	✓		✓	✓		**6**
Mashraba	Shallow						✓										**1**
Moray Gardens	Deep	✓		✓	✓	✓			✓	✓	✓	✓	✓	✓	✓	✓	**12**
Moray Gardens	Shallow			✓					✓	✓	✓	✓	✓	✓	✓	✓	**9**
Ras Abu Gallum North	Deep			✓													**1**
Ras Abu Gallum North	Shallow			✓													**1**
Ras Abu Gallum South	Deep			✓													**1**
Ras Abu Gallum South	Shallow			✓													**1**
Ricks Reef	Deep				✓	✓			✓	✓	✓	✓	✓	✓	✓	✓	**10**
Ricks Reef	Shallow			✓	✓	✓			✓	✓	✓	✓	✓	✓	✓		**10**
Three Pools	Deep				✓	✓				✓							**3**
Three Pools	Shallow			✓		✓											**2**
Um Sid	Deep					✓	✓		✓	✓	✓			✓	✓	✓	**8**
Um Sid	Shallow								✓	✓	✓			✓			**4**
TOTAL		**2**	**1**	**15**	**8**	**9**	**10**	**1**	**20**	**18**	**19**	**10**	**12**	**18**	**13**	**9**	**168**

6.3. Results

During the 15 years of Reef Check surveys conducted in the Dahab area, higher predators, endangered and exploited species were rarely recorded. Manta rays *(Manta spp.)* were absent, and surveys only recorded one shark, nine *Bolbometopon muricatum*, 27 *Tridacna* larger than 50 cm, three Serranidae longer than 60 cm, a single lobster and five sea turtles. However, indications of a healthy reef ecosystem noted over the period included few (7) *Acanthaster planci*, no white band disease, only three incidents of anchor damage to corals and no evidence of dynamite fishing.

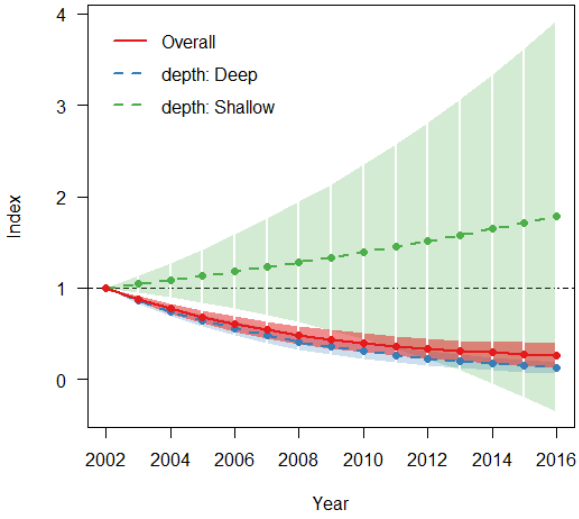
All reef fish species at a variety of trophic levels that showed a significant trend were found to have declined in number around Dahab by at least 50% between 2002 and 2016 (Figure 6.2). The most substantial declines were apparent in high trophic-level species, the Serranidae (Wald=13.14, df=1, $p<0.001$) and Lutjanidae (Wald=7.94, df=1, $p< 0.01$); no trend was found for the Haemulidae. There was also an effect of depth on the decline of the Serranidae, with a greater decline in deeper compared to shallower transects (Wald=4.21, df=1, $p<0.05$)). The herbivorous family, the Scaridae (Wald=10.05, df=1, $p< 0.01$) and the primarily corallivorous family, the Chaetodontidae (Wald=24.64, df=1, $p< 0.001$) had also declined albeit to a lesser extent.

All of the invertebrate species recorded that showed a significant trend had declined over time with the notable exception of the *Tridacna* giant clam (Figure 6.3). In particular, both urchin taxa showed major declines of 90 % or more since 2002 (Diademids: Wald=34.44, df=1, $p< 0.001$; *Tripneustes spp.*: Wald=23.91, df=1, $p< 0.001$). The edible sea cucumbers showed continued declines throughout the monitoring period (Wald=11.44, df=1, $p< 0.001$). The Triton population, a species occasionally collected by Bedu for the curio market has been almost completely decimated (Wald=4.32, df=1, $p< 0.05$) with a slight recovery in 2009 and 2010. Interestingly, the *Tridacna*, an important food species collected by Bedu have increased in abundance by over 50 % since 2002 (Wald=4.97, df=1, $p< 0.05$).

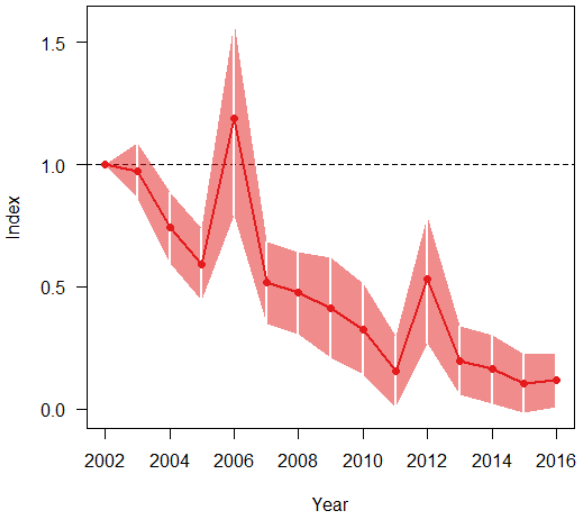
Significant trends in substrate cover are shown in Figure 6.4. Hard coral cover showed no overall change during the period of study. Nutrient indicator algae had substantially increased to around five times its original abundance over the 12-year study period (Wald=11.37, df=1, $p< 0.001$). Decreases in cover to around 50% of the original in 2002 were apparent in both rubble (Wald=4.00, df=1, $p<0.05$) and recently killed coral (Wald=5.03, df=1, $p< 0.05$).

Reef fishes

(a) Groupers (Serranidae - all sizes)



(b) Snappers (Lutjanidae)



(c) Parrotfishes (Scaridae)



(d) Butterflyfishes (Chaetodontidae)

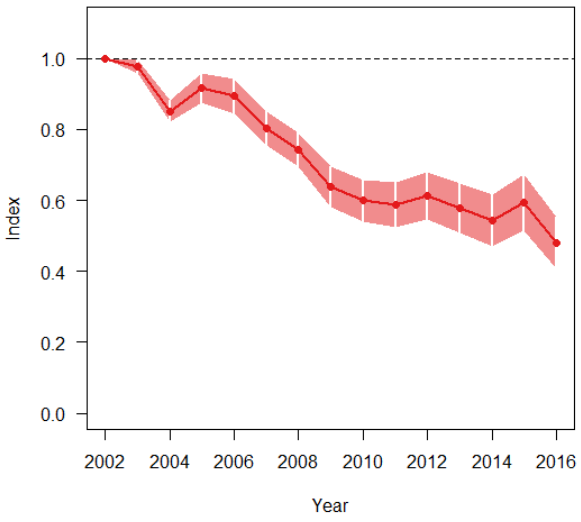


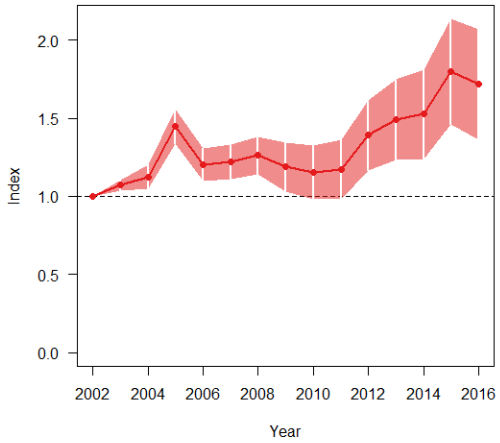
Figure 6.2. Indices of abundance of Reef Check reef fish groups in South Sinai

(estimated from a log-linear Poisson regression taking into account over-dispersion and serial correlation)

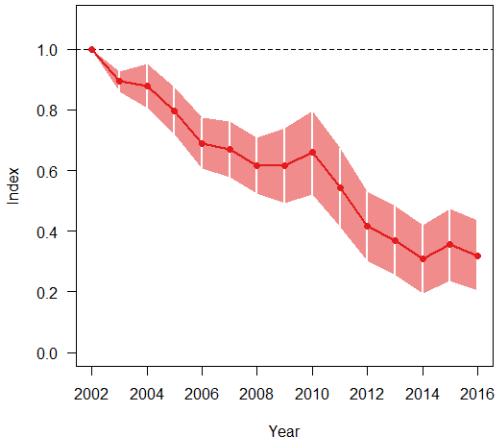
Standard errors for the indices are shown using a transparent band and white error-bars.

Invertebrates

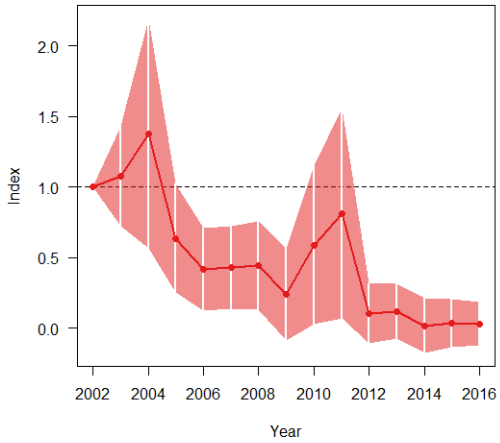
(a) Giant clam (*Tridacna* spp.)



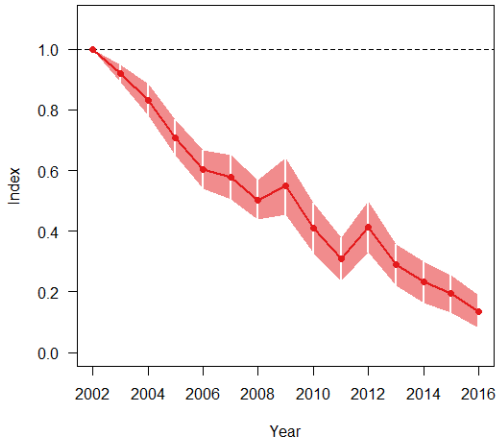
(b) Edible sea cucumbers



(c) Triton *(Charonia tritonis)*



(d) Long-spined black sea urchin (*Diadema* and *Echinothrix diadema)*



(e) Sea Egg/Collector urchin *(Tripneustes spp.)*

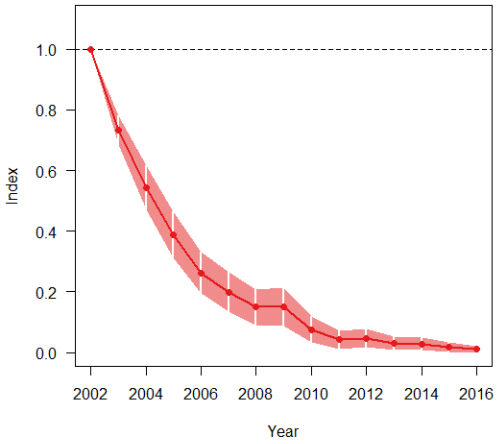


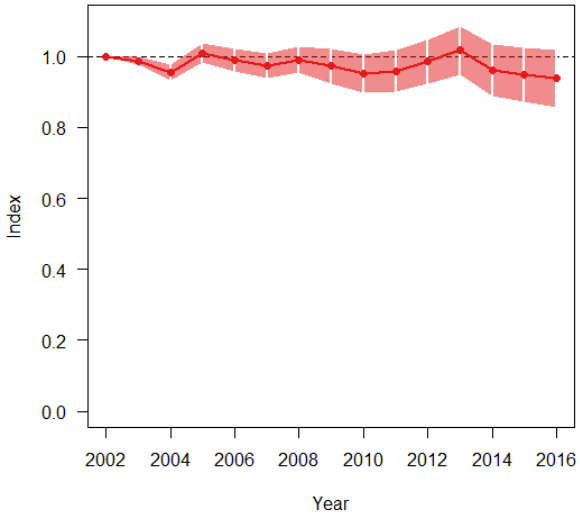
Figure 6.3. Indices of abundance of Reef Check invertebrate groups in South Sinai

(estimated from a log-linear Poisson regression taking into account over-dispersion and serial correlation)

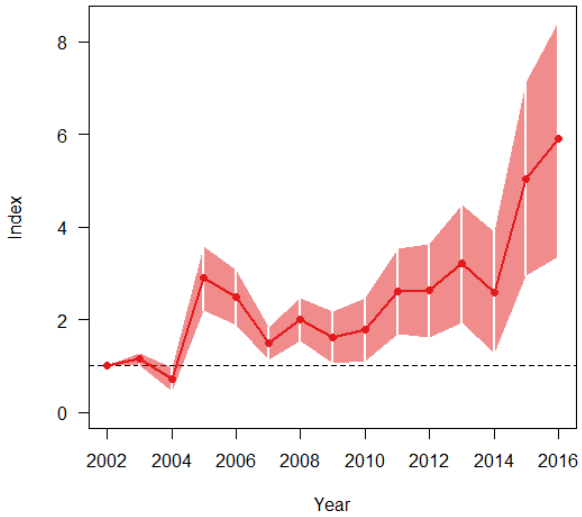
Standard errors for the indices are shown using a transparent band and white error-bars.

Substrate

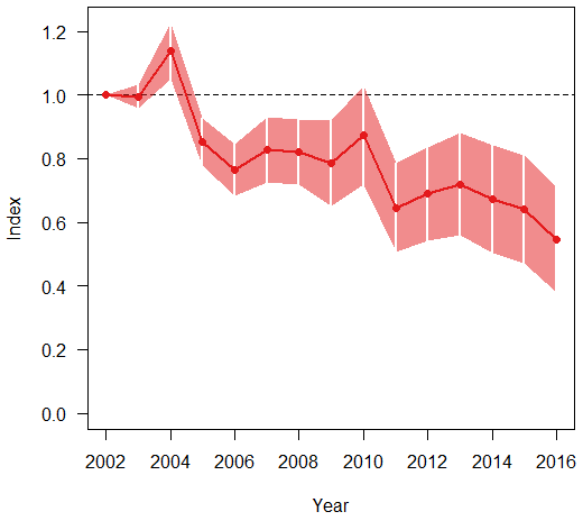
(a) Hard coral cover



(b) Nutrient Indicator Algae



(c) Rubble



(d) Recently killed coral



Figure 6.4. Indices of abundance of Reef Check substrate groups in South Sinai

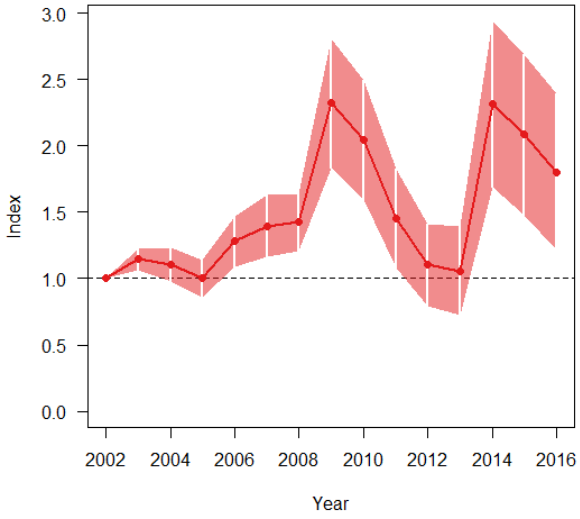
(estimated from a log-linear Poisson regression taking into account over-dispersion and serial correlation)

Standard errors for the indices are shown using a transparent band and white ‘error-bars’.

Trends in other impacts are shown in Figure 6.5. Coral damage has varied substantially over the study period, with an overall increase and peaks in 2009 and 2014 (Wald=4.42, df=1, p< 0.05). Abandoned fishing nets have rapidly become less frequent over the study period, and have rarely been noted since 2009 (Wald=26.12, df=1, p< 0.001).

Other impacts

(a) Coral damage (other)



(b) Fishing nets

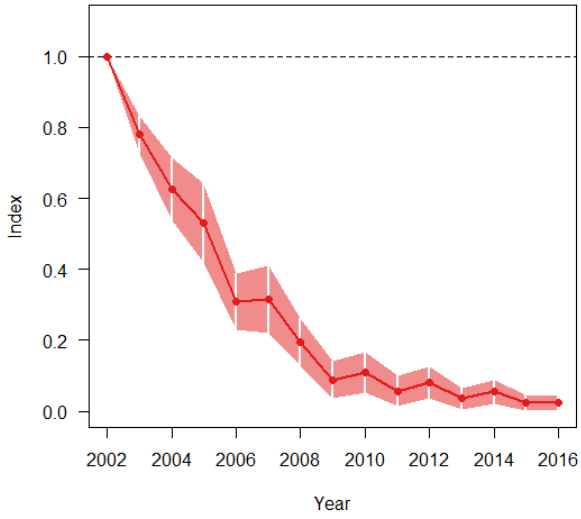


Figure 6.5. Indices of abundance of Reef Check impact indicators in South Sinai

(estimated from a log-linear Poisson regression taking into account over-dispersion and serial correlation)

Standard errors for the indices are shown using a transparent band and white 'error-bars'.

6.4. Discussion

The significant trends that were elucidated by this study in almost all cases indicated a decline in the ecological function of Dahab's reefs. Groups or species of conservation significance including sharks (Roff et al. 2016), sea turtles (Bjorndal and Jackson 2002), *Manta* spp. (Lawson et al. 2017), *Cheilinus undulatus* (Sadovy et al. 2003) and *Bolbometopon muricatum* (Mccauley et al. 2014) have all only rarely been recorded over the last 15 years, indicating that the reefs were likely already in a state of decline before the start of the monitoring in 2002; an inference that is supported strongly by local Bedouin historical knowledge (Chapter 2). These impacts have been primarily caused by a rapid increase in the human population of the area and degradation of the coastal environment because of rapid tourism expansion since the early 1990s (Hasler and Ott 2008; Gladstone et al. 2013; Naumann et al. 2015).

The near absence of marine megafauna such as sharks indicates that these top predators were likely removed by fishing decades ago. Although traditional Bedouin fishing methods are unlikely to be suitable to catch large sharks, they were caught occasionally up to the 70s (Melamid 1957; Baranes and Ben-Tuvia 1978) and overfishing of sharks is well-reported in the wider Red Sea (Tesfamichael et al. 2014; Spaet et al. 2016; Jabado and Spaet 2017). Both herbivorous and piscivorous fish showed declining population trends indicating that fishing at current levels is unsustainable. Notably, predatory Lutjanidae and Serranidae showed the greatest decline; this is likely because these larger-bodied species are widely preferred by fishers, however, they are also slower to reproduce and so are often among the first species to be extirpated under sustained fishing pressure (Jennings and Polunin 1996, 1997; Pauly et al. 1998; Pauly and Palomares 2005; Friedlander et al. 2018). As these preferred families have become less abundant, fishers have begun to target less desirable herbivores such as the more resilient Scaridae that are able to reproduce more rapidly. The effect of depth on the rate of decline observed in the Serranidae could potentially be because this predatory family is most often targeted at depth using hook-and-line. The significant decline of abandoned fishing nets over

time was surprising since it was expected that their abundance would correlate with fishing pressure (Advani et al. 2015; Williamson et al. 2015). It is possible that this may be a result of less fishing pressure as fish stocks have declined, making fishing at these sites unviable. Alternatively fishers may have moved away from the survey sites due to spatial conflict with SCUBA divers, since the Reef Check sites are all located at popular dive sites and diver numbers in Dahab have grown substantially over the last two decades. Additionally, disproportionately high numbers of abandoned nets during earlier years may have been a relic of more intense fishing in the past since nets can persist on the reef for several years (Link et al. 2019).

Tridacna are intensely harvested by Bedu along the South Sinai coastline, usually by women gleaning on the reef flat on foot and by hand or using basic tools (Ashworth et al. 2004; Richter et al. 2008; Chapter 4); so it was interesting that *Tridacna* showed an increasing trend at the study sites. Although *Tridacna* are usually more abundant in shallow areas (Junio et al. 1989), it is a possibility that the deeper areas of the reef where Reef Check surveys were conducted provide refugia from fishing pressure. However, it has been noted in the last few years that Bedu are more often using freediving techniques to collect *Tridacna* from deeper areas (Chapter 2) because they are now much reduced in density and size in the shallows (Chapter 4). Thus individuals inhabiting deeper areas of the reef may also soon start to decline. Alternatively, the *Tridacna* may simply be recovering from the major declines in their populations between 1997 and 2002, attributed to sedimentation from coastal development in the region (Hassan et al. 2002).

Charonia tritonis are primarily collected by Bedu as curios for the tourism industry, and as such have declined, probably through overexploitation, as in many other regions of the world (Nijman et al. 2016) although interestingly, their population appears to have recovered slightly from 2009 to 2011, which could be a result of reduced tourist arrivals in 2009 and 2011 (World Bank 2019) and associated reduced demand for souvenirs. Sea cucumbers were overexploited in the Gulf of Aqaba, resulting in severe reductions in both density and diversity and ultimately a fishing ban in 2006

(Hasan and El-Rady 2012), however it appears that thereafter they continued to decline. Lobsters were almost never observed, and while this is likely partly due to their cryptic and nocturnal nature (Pitcher et al 1992), lobsters have been almost extirpated in South Sinai in recent decades to supply demand from tourist restaurants (Chapter 2).

Excessive growth of algae is likely a result of eutrophication from continued unchecked coastal development (Naumann et al. 2015), with the compounding impacts of reduced herbivory due to declines in Scaridae and urchins (Jessen and Wild 2013; Jessen et al. 2013; Nash et al. 2016). In the absence of sufficient grazing pressure, algae smothers coral, preventing photosynthesis and can eventually result in a phase shift to algal dominated reefs (Hughes et al. 2007; Ainsworth and Mumby 2015). There was also a noticeable increase in algal cover following the Red Sea bleaching event of 2010 (Furby et al. 2013) where bleached corals were likely outcompeted and colonized by algae (McClanahan et al. 2018). Although Gulf of Aqaba corals are known to be resistant to bleaching (Osman et al. 2018), sub-lethal effects may still stress corals, thus increasing their susceptibility to disease and other disturbances and reducing their ability to compete with algae (Lesser et al. 2007). Both rubble and 'recently killed coral' cover decreased over the study period, it is likely that both of these substrates were generated by the major bleaching event in the Red Sea in 2002 (Kotb 2004) and over the years, the rubble has slowly been recolonized with new coral recruits. Likewise, there was an increase in both 'recently killed coral' and rubble around the 2010 bleaching event (Furby et al. 2013).

Urchin populations contribute heavily to herbivory and bioerosion on reef flats in the Indo Pacific (Peyrot-Clausade et al. 2000; McClanahan and Muthiga 2001; Mapstone et al. 2007) and so their decline is surprising considering the increase in cover of algae and decline of their main feeding competitor, herbivorous fish. However, urchins are generally less abundant on the Great Barrier Reef and Oceania than other areas in the Indo Pacific (Hutchings 1986; Done et al. 1991) and are therefore considered less functionally important in the former regions (Ogden 1987; Sammarco 1987).

Similarly, urchins may be of lower functional importance in the Red Sea, and their decline may simply be due to some other factor not recorded in the Reef Check monitoring such as the disease-induced mass-mortality in the Caribbean during the 1980s (Mumby et al. 2006).

Declines in hard coral cover have been reported globally, from, among other places, the Great Barrier Reef (De'ath et al. 2012), the Caribbean (Miller et al. 2009), and the Indo-Pacific (Bruno and Selig 2007). In the Red Sea, it has been noted that species diversity and larger coral colonies are decreasing (Riegl et al. 2012). Hard coral cover overall remained unchanged around Dahab between 2002 and 2016, despite a bleaching event in the Red Sea during 2010 (Furby et al. 2013). Hard corals in the Gulf of Aqaba are subjected naturally to extreme temperature regimes and so may be more resilient to bleaching (Fine et al. 2013; Shlesinger and Loya 2016); although this resilience may be compromised in the presence of multiple stressors (Hall et al. 2018). Chaetodontidae have been widely identified as an indicator of hard coral cover (Crosby and Reese 1996; Zekeria and Videler 2000; Khalaf and Crosby 2005); thus the dramatic declines in abundance of this family suggest that overall hard coral health is declining even if it has not manifested itself through a measurable decline in hard coral cover as yet. Additionally, the rapid growth in macroalgae previously mentioned, particularly where herbivorous fish populations have been reduced by fishing, does not bode well for the future of hard corals on Dahab's reefs.

Coral damage that was not caused by anchoring or dynamite is most likely the result of divers touching and breaking pieces of coral (Hawkins and Roberts 1992; Rouphael and Inglis 1997; Hasler and Ott 2008; Poonian et al. 2010) and has increased significantly during the study period. Noticeably, there was a decline in coral damage that starts around 2009 – 2010 and continued until 2013 which may be due to the decrease in international tourist arrivals previously mentioned. The subsequent rapid increase in damage may be from renewed growth in domestic tourism in 2012, resulting from subsidies provided by the Egyptian government to counteract the fall in international tourism because of the 2011 revolution.

In the continued absence of effective management actions for coral reefs in South Sinai (Samy et al. 2011; Galal et al. 2012), it is likely that the negative trends shown in this study will continue. However, if fishing pressure in the Dahab area was reduced immediately, recovery would most likely start with an increase in herbivorous fish biomass. This increase in herbivory would reduce the proliferation of nutrient indicator algae and would ultimately see a return of higher trophic level fish species such as the Serranidae and Lutjanidae as long as fishing pressure stayed under control. The return of apex predators such as sharks could potentially take over 20 years, since their ranges extend to a much larger scale than is practicable for most MPA initiatives (McClanahan et al. 2005, 2007; Stevenson et al. 2007; DeMartini et al. 2008). The recovery of reefs would further be dependent on minimizing the impacts of eutrophication effects, diver damage and climate-related effects such as coral bleaching events. Gear restrictions are often more readily accepted by local fishers than outright closures (Cinner et al. 2009), and so a potential starting point could be to collaborate with the *Mzeina* to develop restrictions on net fishing, which primarily targets herbivores (Dalzell 1996; Chapter 5) or create closed areas specifically for herbivorous species (Cox et al. 2013; O'Farrell et al. 2016; Williams et al. 2016). However, such precisely-defined approaches require increased cost and effort for enforcement, and ultimately the most efficient management method may be to simply introduce completely closed MPAs, where no fishing of any kind is permitted, although this top-down approach is unlikely to be acceptable to the *Mzeina* and compliance will probably be limited because of their distrust of the Egyptian state (Glaser at al. 2010; Lopes et al. 2013; Katikiro et al. 2015).

In this analysis of the most accurate long-term dataset on the status of coral reefs around Dahab, sampling error and intra-annual variation are likely significant, and the lack of replicates at the site level restricts the statistical power to identify trends and differences between sites (Thompson and Mapstone 2002). The process through which these data were obtained means that seasonal variation has to be accepted, with likely significantly lower biomass at sites during the winter season (van

Hoytema et al. 2016, Chapter 3), although this is unlikely to influence the overall 15-year trend as elucidated by the TRIM model. RSEC is dependent on the availability of volunteer divers, many of whom are university students and so they are not always able to focus their efforts in the same season each year. Additionally, the Reef Check surveys focussed on established dive sites, partly because RSEC is reliant on a local dive centre to organize logistics and to some extent is dependent on the preferences of recreational divers to share survey costs. Thus, more inaccessible sites were not covered by the Reef Check monitoring and it is likely that as the more accessible sites have declined ecologically, fishers have moved progressively further from the urban centre of Dahab in order to maintain their catches. Furthermore, with the increasing availability of cost-effective underwater camera and video equipment, it would be useful to record transects to enable future reference and review by taxonomic experts (Leujak and Ormond 2007).

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GENERAL CONCLUSIONS

It is clear from this work that the coral reef ecosystems in South Sinai are degraded and have been in decline for at least the last 15 years. In general, negative effects are reduced with distance from the urban centre of Dahab and so it is highly probable that they are anthropogenic in origin. This deterioration is likely to be further compounded by the impacts of climate change in the coming decades (Furby et al. 2013). The root cause of this decline is the rapid, uncontrolled development of coastal tourism in the region, which has degraded reef habitat through pollution and high densities of uncontrolled divers and snorkellers accompanied by a several-fold increase in demand for marine fishery resources (Jobbins 2006; Gladstone et al. 2013).

The coral reef fish assemblage is substantially reduced in diversity, biomass and trophic function; invertebrate fishery stocks have been reduced in both size and abundance due to overexploitation. Historically, the reef fishery may have been sustainable, but sedenterization of the Bedu, migration to South Sinai from other areas of Egypt and increased demand from the tourism market has raised the demand for fisheries products beyond a sustainable level (Goodman 2013). This decline in the environmental state of coastal ecosystems is a grave threat to national income from tourism and threatens the socioeconomic security of both the Bedu, Nile Valley Egyptians, and international residents of South Sinai. Furthermore, the loss of coral reef fisheries threatens an important enduring cultural resource of the *Mzeina* (Lavie 1990).

To date, coral reef management efforts have proved ineffective due to a lack of resources and enforcement capacity (Samy et al 2011) and most importantly, failure to include the Bedu in decision making, monitoring and self-regulation (Mascia 2003; Dias et al. 2020). The Bedu have an unrivalled understanding of the reef fishery resources of South Sinai and the information that they hold could prove the key to socially-acceptable, yet effective fishery management action. Management approaches that have shown promise in other regions include community-managed

closed areas, fishing restrictions and seasonal closures for invertebrate fisheries (Oliver et al. 2015) alongside strict enforcement of coastal development regulations (Gladstone et al. 2013) and carrying capacity of dive sites (Hawkins and Roberts 1997). A complementary environmental education programme for all stakeholders (Lucrezi et al. 2019) and livelihood-focussed interventions to increase socioeconomic resilience (Wright et al. 2016) would also be vital components of a successful integrated management programme. Effective management of coastal ecosystems and fishery resources must be implemented with urgency. Fortunately, much of the legal and policy structure already exists but a diplomatic approach is essential to overcome historical distrust between the Bedu and the Egyptian state (Lavie 1989).

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SCIENTIFIC OUTPUTS

1. Chapter 2 was presented at the 12th International Coral Reef Symposium, 9–13 July 2012, Cairns.

Traditional reef fisheries management practices of the South Sinai Bedouin

Poonian, Chris, University of Nottingham, UK

Having lived for centuries under the harsh conditions of the Sinai desert, the Bedouin have developed a sophisticated understanding of their natural environment and have developed strict laws and traditions governing the use of its resources. However, Bedouin fishing practices and traditional rights to coastal resources have been restricted over recent years. During the Israeli occupation (1967–1979), large-scale commercial fishing by Bedouin fishermen was prohibited and individual permits were required. In more recent times, tourism and National Park development along the coastline, particularly in Sharm el Sheikh, has largely limited Bedouin access to fishery resources. Tourism has also brought social complications including prostitution, sexual relations and intermarriage with foreigners, drug dealing and addiction, and materialism. Until the early 1980s, 80 % of coastal Bedouin communities relied on fishing as their main source of livelihood; fish was not only used for household consumption, but was also dried and traded with inland Bedouin from Saint Katherine and the surrounding area. Bedouin are still permitted to fish within the National Parks, using traditional gears from the shore and, although these concessions have contributed to the Bedouin community's acceptance of protected areas, there still remain substantial opportunities to improve the management of coastal resources. If studied carefully and extensively, incorporating traditional management practices into National Park policy may provide an opportunity to ensure tangible and effective inclusion of Bedouin stakeholders and could provide useful ideas to more effectively manage the coastline.

Poonian C (2012) Traditional reef fisheries management practices of the South Sinai Bedouin. Oral presentation at the 12th International Coral Reef Symposium, 9–13 July 2012, Cairns

2. Chapter 2 was also accepted for a poster presentation at the International Conference on the Marine Environment of the Red Sea, KAUST, Thuwal Saudi Arabia, 14–16 November 2016, but I was unable to attend due to visa issues.

Incorporating unwritten tribal laws of the Bedouin into contemporary MPA management in the Egyptian Red Sea

Poonian, Christopher, University of Nottingham, UK

Bedouin people have lived in harmony with their natural environment in the South Sinai for centuries but are now finding themselves increasingly marginalized by top-down conservation policy, unregulated tourism development and influxes of Egyptian migrant workers. Social complications including prostitution, sexual relations and intermarriage with foreigners, drug dealing and addiction, and a new materialism plague the Bedouin in the growing Red Sea tourist hotspots that were once their seasonal fishing villages. However, having lived for centuries under the harsh conditions of the Sinai desert, the Bedouin have developed a sophisticated understanding of their natural environment and have developed strict laws and traditions governing the use of its resources. The Bedouin live by an unwritten honour code known as Sharaf, and rarely need to involve national law enforcement to manage disputes among their communities. This code is implemented by every member of the Bedouin community collectively, with respected elders appointed to deal with any disputes. This code of living is thus strongly respected and widely enforced, with serious social punishments for anyone found breaking the code. This work explores the potential for incorporating these tribal laws into national marine conservation policy and looks at options to ensure that this knowledge and rules are not lost forever as younger Bedouin more and more commonly choose to pursue non-traditional livelihoods.

3. Chapter 3 was accepted for a speed talk at the International Marine Conservation Congress, 24–29 June, Kuching, and for a poster presentation at the Arabian Coral Reefs: Insights from Extremes Conference, 11–13 November 2019, New York University, Abu Dhabi, but I was unable to attend either due to lack of funding.

Bedouin clam fisherwomen of South Sinai, Egypt - a culturally significant fishery in decline

Poonian, Chris, University of Nottingham, UK

Although overfishing in tropical finfish fisheries has been studied in some depth, invertebrate fisheries are rarely monitored and often unregulated. Invertebrates have been gathered on foot in the intertidal zone throughout the tropics since prehistoric times, primarily by women. In South Sinai, Mzeina Bedouin women traditionally harvest Tridacna spp. at low tide on shallow reef flats using metal spears. I evaluated Tridacna density and size at eight intertidal fishing sites in South Sinai and discussed details of fishery with Bedouin women. It was apparent that the fishery had significantly declined in recent years, and was also subjected to negative impacts from unrestricted coastal development. There was a noticeable gradient of improved habitat and larger clam density and size with increasing distance from the main Bedouin settlement. In areas where the Bedu gained alternative income from tourism, clams were also larger and more abundant. The decline in these resources may have the dual effect of altering women's roles and importance within the household and the community on the one hand and causing a decline in household levels of food security and income on the other. Currently, the Bedu are exempt from fishing regulations within MPAs in the area. However, enforcement of fishing restrictions, particularly at remote fishing sites as those fished by the Bedu would be complicated. As such, the most realistic option could be to develop a locally-supported, self-imposed management regime among Bedouin women, building awareness of the biology of the target species and links between resource management and sustainability.