

Effects of contrasting types of marine protected areas on seagrass- and coral communities

– are community-based reserves an important complement to government-managed protected areas?

Angelica Chirico

Department of Ecology, Environment and Plant Sciences
Licentiate in Philosophy Thesis 2013:3
Marine ecotoxicology
October 2013
ISSN 1401-4106



Stockholms
universitet

Abstract

Coastal ecosystems, including seagrass beds and coral reefs, are among the most ecological and economical important ecosystems on Earth. At the same time as these ecosystems support livelihoods of coastal communities they are being highly degraded worldwide. Government-managed marine protected areas (MPAs) are a common tool in marine conservation and have been demonstrated to successfully protect natural resources. At the same time, they are increasingly criticized for excluding and marginalizing local communities. Therefore, alternative types of management that are managed by the communities themselves (community-based reserves; CBRs) constitute a promising alternative since they have a much higher acceptance among local people. However, the scientific knowledge on protection effects of CBRs on these critical habitats are scarce, and most research on the effects of place-based management has largely focused on coral reefs. The aim of this thesis was therefore to investigate how MPAs and CBRs affect corals and seagrasses, and their associated communities, using coastal Kenya (East Africa) as a case study. Paper I examines effects from CBRs and MPAs on benthic community composition, and cover and diversity of seagrasses, hard corals and associated benthic organisms. Paper II examines the effects of CBRs and MPAs on the density, size, biomass and potential monetary value of fish; the basis for coastal fisheries that are a particularly important ecosystem service in the study area. The results demonstrate that the small and recently protected Kenyan CBRs can increase the diversity of benthic organisms, protect important functional groups, increase structural complexity, and additionally increase fish size, biomass and monetary value. The results also show that protection from MPAs can result in shifts in both seagrass beds and coral reef communities (from a dominance of stress-tolerant species in fished areas, to stress-sensitive species in protected areas), and that the two habitats were similarly affected by protection. In summary, this thesis suggest that i) locally-managed CBRs may be an important complement to MPAs, and ii) that seagrass beds should more often be included in management plans.

Contents

- List of papers 2
- Introduction 3
- Aim of thesis 6
- Study region 6
- Methods 7
- General findings 8
 - Paper I: Effects of contrasting types of marine protected areas on marine foundation species across a tropical seascape gradient..... 8
 - Paper II: Recently established community-based reserves disproportionately increase potential monetary value of fish stocks 9
- General discussion..... 9
- Conclusions 13
- Future studies 13
- Acknowledgements 14
- References 14

List of papers

This thesis is based on the following two papers, which hereafter will be referred to as paper I and paper II.

Paper I (*Manuscript*)

Effects of contrasting types of marine protected areas on marine foundation species across a tropical seascape gradient

Angelica A. D. Chirico, Jacqueline N. Uku and Johan S. Eklöf

Personal contribution: Responsible for all fieldwork and data sampling, analyzed all data and lead author.

Paper II (*Manuscript*)

Recently established community-based reserves disproportionately increase potential monetary value of fish stocks

Angelica A. D. Chirico and Johan S. Eklöf

Personal contribution: Responsible for all fieldwork and most data sampling, analyzed all data and lead author.

Introduction

Humans change ecosystems to meet growing needs for natural resources; for example most of the world's terrestrial landscapes have been altered by agricultural activities in favor for crop production (Power 2010). Overexploitation and extensive degradation of natural ecosystems are globally threatening biological diversity and future delivery of ecosystem services; the benefits that humans obtain from nature (Daily 1997; MEA 2005; Worm *et al.* 2006). Both ecosystem services and biological diversity are central targets in conservation, since they contribute to human wellbeing (MEA 2005; Spalding *et al.* 2013). Biological diversity is important because it can affect delivery of ecosystem services and the stability of an ecosystem (Johnson *et al.* 1996). Studies from both terrestrial and marine ecosystems have demonstrated that an increase in biological diversity can enhance delivery of ecosystem services (Worm *et al.* 2006; Quijas *et al.* 2010). Furthermore, high biological diversity may increase resistance to a system if an extinct species can be replaced by a species with comparable functional traits, thereby preventing or delaying a collapse of the system (Walker 1992).

Although ecosystem services are generated by ecosystems, they often depend on species that play a disproportionately important role for their creation; so called 'ecosystem service providers' (Kremen 2005). The identity, abundance and diversity of these organisms, such as habitat-forming corals in the sea or pollinating insects on land, are important factors influencing the diversity of, and the rate at which ecosystem services are generated. For example, tropical reefs with high cover of structurally complex hard corals are often associated with higher diversity and production of fish (Jones *et al.* 2004). Moreover, if diversity of pollinating bees decreases slightly it can still have a major negative effect on ecological functions that support ecosystem services, such as crop production (Larsen *et al.* 2005). It is therefore of great importance to identify factors that influence the identity, abundance and diversity of 'ecosystem service providers'.

Coastal marine ecosystems are highly productive and comprise a wide range of 'ecosystem service providers' (Bruno & Bertness 2001; UNEP 2006). In tropical coastal regions, three ecosystems – coral reefs, seagrass beds and mangrove forests – form the 'tropical seascape' (Ogden & Gladfelter 1983; Ogden 1988). These interconnected systems are recognized as vital for generating multiple ecosystem services such as fish production, shoreline protection, water purification, erosion control and tourism revenues (Martínez *et al.* 2007; Barbier *et al.* 2011). However, increasing demand on natural resources combined with poor management has resulted in overexploitation and degradation of these important habitats (Burke *et al.* 2001; Jackson *et al.* 2001; Lotze *et al.* 2006). A major part of the seascape is used as fishing grounds and because of poorly managed fisheries overfishing is one of the most important threats (Burke *et al.* 2001; Jackson *et al.* 2001; Orth *et al.* 2006). For example, the fisheries

have experienced reduced revenues through declining fish catches (Hsieh *et al.* 2006; Srinivasan *et al.* 2010). Mechanical disturbances from destructive fishing gears can greatly reduce coral cover and the structural complexity of coral reefs (McClanahan & Shafir 1990; Alvarez-Filip *et al.* 2009); factors that regulate the reef's ability to provide feeding and nursery grounds to many marine taxa (Graham & Nash 2013). Indirectly, fishing may result in sea urchin outbreaks caused by removal of sea urchin predators (McClanahan *et al.* 1996; Eklöf *et al.* 2008). This has caused overgrazing of seagrass beds and hence shoreline erosion and decreased sediment stabilization (Rose *et al.* 1999; Peterson *et al.* 2002). Also other human activities, such as aquaculture, impact the ecosystems in the tropical seascape. Shrimp and fish aquaculture is well-known to have resulted in significant loss of mangroves (Naylor *et al.* 2000) and their crucial shoreline protection service (Olwig *et al.* 2007). Additionally, disturbances have been suggested to cause shifts in entire communities. For example, on coral reefs overfishing in combination with other stressors, such as global warming, has resulted in a shift towards coral communities that are less diverse (Alvarez-Filip *et al.* 2009; Riegl *et al.* 2012). These communities are dominated by stress-tolerant and structurally simple species which has reduced reef habitat complexity and thereby likely diminished tourism revenues and shoreline protection (Graham & Nash 2013). Altogether, these disturbances have – by impacting 'ecosystem service providers' like seagrasses and corals – strongly impacted the creation of vital ecosystem services. Humans in coastal regions, particularly in developing countries, are often directly dependent of natural resources and their ecosystem services for their livelihoods. For example, fishing generates a daily income and acts as vital food source and coastal societies therefore become particularly vulnerable to marine resource degradation (Martínez *et al.* 2007; Barbier *et al.* 2011).

To reduce the effects of anthropogenic disturbances, establishment of government-managed marine protected areas (hereafter referred to as 'MPAs') has become increasingly common as a tool in marine conservation (Spalding *et al.* 2013). The purposes with MPAs vary from preserving biological diversity and sensitive habitats, to replenishing depleted fish stocks and generate tourism revenues. Ideally, they should all be designed to simultaneously accomplish as many conservation objects as possible (Salm *et al.* 2000). A large body of literature has shown that MPAs can successfully protect biological diversity, critical habitats and fish stocks (Bohnsack 1998; Roberts & Hawkins 2000; Gell & Roberts 2002; UNEP 2006; Graham *et al.* 2011). Importantly, these effects strongly depend on how long an MPA has existed (Roberts & Hawkins 2000; McClanahan *et al.* 2007; Claudet *et al.* 2008; McClanahan *et al.* 2009). The reason is that there is a time-lag from the start of protection to recovery of protected populations that depend on e.g. re-colonization rates and reproduction cycles (Roberts & Hawkins 2000). Also the size of MPAs can influence protection effects, by for example influencing the population size of non-exploited populations (Claudet *et al.* 2008; McClanahan *et al.* 2009). On the other hand however, MPAs have become heavily criticized on a global scale, primarily because they exclude resource use and therefore can marginalize coastal communities (e.g. Gell & Roberts 2002;

Christie 2004; McClanahan *et al.* 2005). For example, the presence of MPAs have been shown to cause loss of fisheries revenues and community level values like involvement in community organizations and confidence in management of resources (Hicks *et al.* 2009). As a consequence, MPAs have become increasingly associated with increased resource user conflicts that decrease socio-economic stability and threaten MPA goals (McClanahan 1999; Christie 2004). A promising alternative to government-managed MPAs are community-based or 'bottom-up'-driven reserves (hereafter referred to as 'CBRs'). A small but increasing body of literature shows that CBRs are generally more accepted by local communities, since the communities themselves participate in and/or lead decision-making and management (Gray *et al.* 2001; Pollnac *et al.* 2001; Gell & Roberts 2002; Redford & Fearn 2007; Hicks *et al.* 2009). However, knowledge about ecological protection effects from CBRs is scarce compared to the numerous studies done on effects of MPAs. Studies on coral reefs demonstrate that CBRs can increase hard coral cover and diversity (e.g. White & Vogt 2000) and fish biomass and size (e.g. McClanahan *et al.* 2006; Aburto-Oropeza *et al.* 2011; Clements *et al.* 2012). If these effects are general, CBRs may be a good complement or possible alternative, to already existing MPAs, since they are likely to fulfill socio-economic community goals (Hicks *et al.* 2009) and at the same time protect natural resources.

Much marine research and management in the tropics target coral reefs, whereas other habitats in the seascape may be equally or even more important for delivering ecosystem services (Wells *et al.* 2007; Duarte *et al.* 2008). Seagrass ecosystems have received the least attention (Duarte *et al.* 2008), despite the fact that seagrass beds generate production of economically important finfish and shellfish, sequester carbon, stabilize sediment and protect coast-lines worldwide (Fonseca 1989; Koch 2001; Heck *et al.* 2003; Björk *et al.* 2008; Barbier *et al.* 2011). Seagrass beds have also experienced serious decline globally, due to disturbances from e.g. aquaculture, nutrient and sediment runoff, and destructive fishing practices (Orth *et al.* 2006; Waycott *et al.* 2009). Together, these facts highlight the importance of research and management that also target other habitats than coral reefs, and assesses how place-based protection affects these ecosystems. Another important factor in studies assessing effects of protection is the variables surveyed (Moffitt *et al.* 2013). Many studies and monitoring programs are based on a few and relatively 'coarse' ecological variables, such as fish biomass (e.g. Roberts & Polunin 1991; Halpern 2003) and total % cover of benthic organisms, e.g. corals (Wilkinson 2008). These variables, and the way in which they are affected by management, are of great importance since they are used when evaluating management effects. Yet, some effects may only be detected when using more detailed variables, such as community composition of taxonomic groups that may account for life-history traits (e.g. stress-tolerance) (e.g. Darling *et al.* 2012). It is therefore of great importance to assess and evaluate protection effects using a combination of coarse and more fine-scale variables.

Aim of thesis

The aim of this licentiate thesis was to investigate how contrasting types of place-based management – locally-managed CBRs and government-managed MPAs – affects coral and seagrass communities. Using coastal Kenya (East Africa) as a case study, I investigated how MPAs and CBRs affect two important ecosystem service providers; corals and seagrasses, and their associated benthic and fish communities. In paper I effects of protection from MPAs and CBRs (in relation to fished open-access areas) were compared in terms of community composition, and cover and diversity of seagrasses, hard corals and associated benthic sessile organisms. In paper II I assessed the effects of the two forms of contrasting management on the density, size, biomass and potential monetary value of seagrass- and coral associated fish stocks; the basis for coastal fisheries, which is a particularly important ecosystem service in these regions. Overall, this thesis aims to test if:

- i) CBRs can generate protection effects on these critical habitats and their associated communities,
- ii) effects differ between seagrass and coral habitats, and
- iii) different variables are affected differently by protection, e.g. the coarser variable total coral cover vs. coral taxonomic composition.

Study region

This research has been carried out in the tropical seascape of the southern coast of Kenya, East Africa. The marine environment along the Kenyan coastline is typical for that found along many shallow coasts in the Western Indian Ocean; the intertidal zone is covered by small seagrasses like *Halophila* and *Cymodocea spp.*, while larger species e.g. *Thalassodendron ciliatum* are more abundant in the mid and subtidal lagoon (Gullstrom *et al.* 2002). From the mid lagoon and outwards the bottom becomes gradually more dominated by hard bottom coral interspersed with sand, macroalgae and seagrasses.

A majority of the households in coastal Kenya depend on fishing and other benefits associated with coral reefs and seagrass beds for their daily income and food security (Martínez *et al.* 2007). At the same time these habitats are severely threatened by resource use activities (such as tourism, shipping and fisheries) that have resulted in overfishing and habitat destruction (McClanahan *et al.* 2005). For example, in some tourist areas in Kenya hotel owners clear intertidal areas from seagrasses, resulting in loss of associated organisms and significant beach erosion (Green & Short 2003). Additionally, overfishing have triggered sea urchin population outbreaks caused by removal of sea urchin predators (McClanahan *et al.* 1996; Eklöf *et al.* 2008), which has increased bioerosion of coral reefs and loss of

crustose coralline algae (CCA) that are essential for coral recruits (McClanahan & Muthiga 1988; McClanahan & Muthiga 1989; O'Leary *et al.* 2012). As a result, corals and seagrasses have declined in cover and distribution, and overfishing have resulted in reduced fishery yields for local communities (Obura 2001).

A number of MPAs have been established in Kenya during the last four decades (Wells *et al.* 2007) to prevent further environmental degradation from overfishing and destructive gear use (McClanahan 1987; McClanahan & Muthiga 1988; Knowlton & Jackson 2008; McClanahan *et al.* 2008). These MPAs have been shown to successfully protect natural resources, but at the same time they are poorly accepted among local communities (Obura 2001; McClanahan *et al.* 2005; Hicks *et al.* 2009). As a result infected conflicts between fishing communities and authorities have arisen. This has resulted in the establishment of locally-managed CBRs, which have gained a much higher acceptance among local communities compared to MPAs (Hicks *et al.* 2009). Because of the presence of both MPAs and CBRs, southern Kenya constitutes a good study area to investigate effects of contrasting management on coral and seagrass communities. However, since the CBRs in Kenya are all recently established (between 1 and 6 years of protection), and the MPAs have been protected for >20 years, they will likely not generate the same protection effects. Therefore, protection effects of CBRs relative to fished areas are more relevant.

Methods

Paper I

Benthic inter- and sub-tidal communities were surveyed in twelve study areas (four CBRs, four MPAs and four fished areas) on the southern Kenyan coast during October to December 2012. To evaluate how management affects different groups of important 'ecosystem service providers', the percentage cover of all sessile benthic organisms were assessed in 0.5 m² quadrats, in three habitat zones along a seascape gradient; i) intertidal and ii) subtidal seagrass areas, and iii) subtidal coral reefs. Seagrasses were identified to species, and macroalgae and hard corals mostly to genus. Remaining benthic substrates were categorized into one of the following groups: soft coral, sponge, cyanobacteria, dead coral, stone and sand. On the coral reef we also estimated how structurally complex the habitat was, using a standard rugosity estimation (McClanahan & Shafir 1990), where a ten meter long chain was placed along the reef contour, and its length from the start to end was compared to that of the straight line distance (10 m). Effects of management type, habitat type and time since closure (age) on hard coral and benthic communities composition (based on taxonomic groups) were analyzed using PRIMER (v 6.1.15) (PERMANOVA and DISTLM) (following Anderson *et al.* 2008). Effects of management type, habitat type and time since closure on diversity of benthic organisms, reef

topographic complexity, total hard coral and seagrass cover, and additionally the two ecological important calcareous algae; crustose coralline algae and *Halimeda* were analyzed using different types of general linear mixed models in R (v. 3.0.1).

Paper II

To investigate effects of management on fish communities, fishes were surveyed in coral and seagrass habitats in September to December 2011 in two CBRs, two MPAs and two fished areas. Fishes were visually recorded in 25m² point transects, that are commonly used when comparing fish communities in contrasting habitats (Dorenbosch *et al.* 2005, 2006). Fishes were identified to species level, counted and size estimated (standard length, SL). Fish biomass was calculated using species-specific length-weight relationships from FishBase (Froese & Pauly 2012). Potential monetary value of the fish stocks were estimated by combining our fish survey data with 12 years of fish market data from the study area (McClanahan 2010). Since fish communities are usually affected by local cover of benthic habitats (Bell & Galzin 1984; Gell & Whittington 2002; Coker *et al.* 2012), the local cover of habitat-forming coral and seagrasses were simultaneously sampled in the quadrats. Effects of management type, habitat type and time since closure (age) on fish density, size, biomass and value were analyzed using different types of general linear mixed models in R (v. 3.0.1). Effects on fish group composition, based on how valuable the fishes are on the market (i.e. price per kg) (see McClanahan 2010), were analyzed using PRIMER (v 6.1.15) (PERMANOVA and DISTLM) (following Anderson *et al.* 2008).

General findings

Paper I: Effects of contrasting types of marine protected areas on marine foundation species across a tropical seascape gradient

Paper I examines how cover of four types of 'ecosystem service providers' (hard corals, seagrasses, crustose coralline algae and *Halimeda* spp.), and topographic reef complexity, diversity and composition of the benthic community are affected by CBRs and MPAs. The results show that total cover of hard corals was unaffected by management, whereas total seagrass cover was positively affected by MPAs (but not by CBRs) in intertidal and mid-lagoon areas, but negatively affected in reef areas. Protection effects from CBRs and MPAs increased cover of coral reef-associated crustose coralline algae (CCA), and cover of *Halimeda* algae in both coral and seagrass habitats. In terms of community composition, the MPAs caused major shifts in both coral and seagrass communities compared to fished areas and CBRs. Late-successional and disturbance-sensitive species replaced more stress-tolerant species that dominated in fished areas and CBRs. Importantly, these shifts had not occurred in any of the CBRs. However, CBRs (as well as MPAs) increased the overall diversity of

benthic organisms in both seagrass and coral habitats, and there was also a strong trend suggesting that CBRs – just like MPAs – increased coral reef structural complexity. Importantly, seagrass beds and coral reefs, which differed in terms of diversity and composition of taxonomic groups, were largely affected by CBRs and MPAs in the same ways (few interactions between the factors 'management type' and 'habitat zone').

Paper II: Recently established community-based reserves disproportionately increase potential monetary value of fish stocks

Paper II narrows the focus and investigates the effects of locally-managed CBRs and government-managed MPAs on the density, size, biomass and potential monetary value of seagrass- and coral reef-associated fish. Interestingly, we found that protection effects from young and small CBRs, just as the older and larger MPAs, increased total biomass, individual size and potential market value of fish, in both seagrass beds and coral reefs. Moreover, the potential market value increased much more strongly with time since closure (age), than did standard monitoring variables such as fish biomass or density. There are mainly two explanations for this finding. First, the increase in fish size likely generates a higher price per kg, because fishes can then be sold on the tourism market where prices are much higher than on local fish markets. Second, we found that high-value fish taxa (e.g. rabbitfish and goatfish) correlated strongest with time since closure, than did low-value taxa. Protection did not interact with habitat type (coral and seagrass) in our models, which indicate that seagrass- and coral reef-associated fish communities are affected by protection in largely the same way.

General discussion

The results from paper I and II suggests that small and recently established locally-managed CBRs can clearly benefit functionally important foundation species and their associated communities, similar to older and larger government-managed MPAs.

Paper I demonstrates that CBRs, just like MPAs, positively affected the cover of the important ecosystem service providers; crustose coralline algae and *Halimeda* spp., as well as community diversity of benthic organisms and coral reef structural complexity. Overexploitation has resulted in reduction in structural complexity of reefs (McClanahan & Shafir 1990; Alvarez-Filip *et al.* 2009) and sea urchin population 'outbreaks' (McClanahan *et al.* 1996; Eklöf *et al.* 2008). This has in turn resulted in increased bioerosion of corals and loss of crustose coralline algae (McClanahan & Muthiga 1988; McClanahan & Muthiga 1989). The increase of crustose coralline algae and reef complexity from effects of CBRs and MPAs was likely an effect of the lack of stressors associated with fishing. Earlier

studies have in the same way demonstrated greater cover of crustose coralline algae and *Halimeda* spp., as well as increased reef complexity in MPAs compared to fished areas (McClanahan & Shafir 1990; Alvarez-Filip *et al.* 2009; Graham *et al.* 2011). The positive effects of protection on cover of crustose coralline algae and the calcareous algae *Halimeda* spp. suggests that both CBRs and MPAs likely sustain higher rates of reef accretion than fished areas. This is because these two algae are the main producers of carbonate sediments in many reef systems (Vroom 2011) and hence are important in maintaining coastal protection (Koch *et al.* 2009). Additionally, higher cover of crustose coralline algae is critical for coral settlement (Anthony *et al.* 2008). These results suggest that CBRs, similar to MPAs, provide better opportunities for coral recruitment than fished areas, and hence ecological stability of reefs. Complex and diverse habitats are likely to support higher diversity of associated species, e.g. fish species through providing more habitats (Jones *et al.* 2004), and also likely sustain more ecosystem services such as tourism revenues (Worm *et al.* 2006; Graham & Nash 2013).

We also found that MPAs generated distinct shifts in coral and seagrass communities, from dominance of 'weedy' and stress-resistant taxa in fished areas, to structurally more complex and stress-sensitive species within MPAs. For example the grazing-sensitive seagrass species *Thalassodendron ciliatum* (Alcoverro & Mariani 2004), that has a complex morphology and has been shown to harbor more fish biomass than more simple seagrasses (Gullstrom *et al.* 2008), dominated in the MPAs. These effects were, however, not generated by the more recently established CBRs alone, suggesting that community shift need longer time of recovery before communities shifts into more late-successional species. Likewise, Chirico *et al.* (in preparation) demonstrates that recently established CBRs still have high abundances of sea urchins, possible due to slow recovery of sea urchin predatory fishes. Therefore, CBRs are unlikely to exclude stressors to grazing- and stress-sensitive coral and seagrass species, which explain why the detected community-shifts have not yet occurred in the CBRs.

Paper II demonstrated an increase in fish biomass, size and, particularly, potential monetary value in CBRs and MPAs. These results are consistent with those from other studies that also show that CBRs can generate positive effects on biomass and size of coral reef fishes (e.g. McClanahan *et al.* 2006; Aburto-Oropeza *et al.* 2011; Clements *et al.* 2012). The increased abundance, biomass and size of valuable fishes inside CBRs (and MPAs) may result in density-dependent 'spill-over' (Bohnsack 1998) of fishes that possess a high value and hence that may benefit fisheries outside the closure (Unsworth *et al.* 2010). Since fishing is an important source of generating fisheries revenue and sustain daily food requirements (Roberts & Hawkins 2000; UNEP 2006), these possible effects are of great importance for coastal livelihoods.

It is important to study protection effects from alternative management types such as CBRs, since they from a socio-economic perspective often have higher acceptance from local communities and more

likely fulfill community goals than MPAs (e.g. Pollnac *et al.* 2001; Gell & Roberts 2002; Hicks *et al.* 2009). Hence, if CBRs also protect natural resources, they have the potential to deliver both socio-economic and ecological benefits, and may therefore constitute a good complement to MPAs. Together, paper I and paper II demonstrate that CBRs generated positive effects on natural resources; they increased the cover of key functional groups (crustose coralline algae and *Halimeda* spp.), and increased reef complexity, diversity and value of fish communities. The shorter time of protection of the CBRs, compared to the MPAs is a likely explanation to why the CBRs have not yet generated the community-shifts observed in the MPAs. Since the time of protection covaries with the type of protection (CBR vs. MPA), it is difficult to assess if the observed differences in effects are due to the differences in time since closure, or differences in how the areas are managed. Size of closure is also known to affect the performance of protected areas (e.g. Claudet *et al.* 2008; McClanahan *et al.* 2009). Small reserves can successfully protect sedentary organisms, while large reserves may be better in protecting larger and more mobile organisms, as well as containing different types of habitats and therefore likely sustain higher biological diversity (Hilborn *et al.* 2004; Nardi *et al.* 2004). CBRs are usually small and established close to populated areas, and therefore these areas are likely easier to control (McClanahan *et al.* 2006). The smaller size of CBRs likely support some conservation goals e.g. sustain local fisheries through 'spillover' to nearby areas. While the MPAs due to their larger sizes are more likely to maintain several habitat types and protect highly mobile organisms, they are more likely to fulfill other conservation goals such as maintaining high biological diversity (Weeks *et al.* 2010). Since CBRs may fulfill socio-economic community goals (e.g. Pollnac *et al.* 2001; Gell & Roberts 2002; Hicks *et al.* 2009) and even after just a few years can generate protection effects on natural resources (Paper I and II), this thesis suggest that CBRs may constitute a “golden mean” that can deliver both ecological and social benefits and hence be an important complement to MPAs. A combination of different types of management is likely to fulfill goals on both regional and community level (Weeks *et al.* 2010). Multiple-use systems have turned out to be successful in the Great Barrier Reef Marine Park, Australia, where environmental conservation and human resource use have been successfully integrated (Day 2008).

In general, coral and seagrass habitats were similarly affected by protection from CBRs and MPAs. For example, effects on diversity of benthic organisms, benthic community composition and fish community variables did not differ between the two habitats. This indicate that even though conservation efforts in the tropics are primarily targeting coral reefs (Wells *et al.* 2007), protected areas also generate positive effects on seagrass ecosystems. The need to also protect seagrass habitats was made even clearer with the fact that MPAs had a positive effect on seagrass cover in the seagrass habitat, but a negative effect on seagrass cover in the coral habitat. A likely explanation to this effect is that seagrasses within reef habitats experience increased competition for space from corals that spread as an effect of protection (McClanahan *et al.* 1994). Seagrasses may also be exposed to high grazing

rates from herbivorous fishes in the MPAs. Studies from Kenya, as well as other parts of the world, demonstrates that herbivores fishes consume substantial amounts of seagrass and that grazing pressure from herbivorous fishes is greatest within MPAs (McClanahan *et al.* 1994; Alcoverro & Mariani 2004; Mumby *et al.* 2006; McClanahan 2008), which support this hypothesis.

There were also some clear and general differences between the two habitats, regardless of protection effects. Coral reefs had higher diversity of benthic organisms (paper I) and also sustained fish communities with higher density, biomass and potential market value than seagrass beds (paper II). This supports other studies showing that coral reefs generate fish production by e.g. providing shelter and feeding habitats through their complex three-dimensional structures (Bell & Galzin 1984; Moberg & Folke 1999; Coker *et al.* 2012). Based on the assumption that increased biological diversity can enhance delivery of ecosystem services (Worm *et al.* 2006; Quijas *et al.* 2010), my results may suggest that coral reefs generate more ecosystem services per unit area, than do seagrass beds. Meanwhile, we found that benthic community composition (based on taxonomic groups) (Paper I) and fish group composition (Paper II) clearly differed between seagrass and coral habitats. This supports the suggestion that these two habitats, by sustaining very different types of species and functions, are likely to support different ecosystem services (Duarte *et al.* 2008). For example, seagrass beds support carbon sequestration and alleviation of pH stress ("ocean acidification") on calcifying organisms (Semesi *et al.* 2009; McLeod *et al.* 2011). More importantly, even though seagrass beds per unit area may sustain fish communities with lower density, biomass and value, seagrass-associated fish have over the last decade become disproportionately important for artisanal fisheries, especially in Kenya (de la Torre-Castro & Rönnbäck 2004; Cinner *et al.* 2013). Two possible explanations could be that seagrasses cover much larger fishing areas than coral reefs (and therefore support greater fish stocks in total), and/or that coral reefs have been struck harder by overfishing and/or climate change. Combined with the results from Paper I and II, this highlights the need for management in the tropical seascape, which has primarily focused on protecting coral reefs (Wells *et al.* 2007; Duarte *et al.* 2008), also should include seagrass beds.

The two studies presented in this thesis also demonstrate that depending on what variable that is measured, we see different effects of protection. Total coral cover, which is a common variable in coral reef monitoring (Wilkinson 2008), was not affected by management, whereas community composition clearly shifted when comparing fished areas and MPAs (paper I). Second, potential market value of fish stocks was a more sensitive fish variable than fish density, biomass and individual size (paper II). These results show that from a management perspective it is important to carefully evaluate how different variables respond to different types of protection (Moffitt *et al.* 2013). Management effects, in terms of how successful they are in protecting natural resources, are often evaluated based on relatively 'coarse' variables like fish biomass and coral cover, which this thesis

shows are potentially not sensitive enough to identify actual effects. If ignoring the fact that different variables respond differently, the actual effects of protection may be over- or underestimated.

Conclusions

The results from paper I and paper II can be summarized into three main conclusions:

- i) Small and recently established CBRs can increase diversity and cover of benthic organisms and their associated communities, and enhance fish biomass and more so, the potential market value of fish stocks. CBRs may hence be an important complement to MPAs in marine conservation.
- ii) Seagrass- and coral reef habitats were largely affected by the different types of management in the same way, and protection effects can result in community shifts to dominance by structurally complex 'ecosystem service providers' like branching corals and seagrasses. Consequently, seagrass habitats should more often be included in management plans, just like coral reefs.
- iii) Protection effects strongly differed depending on what variable was examined, and traditional monitoring variables (e.g. total coral cover and fish density) were those variables that were the least sensitive to protection effects.

Future studies

My research will continue to study the effects of place-based protection on seagrass and coral habitats, and investigate some of the functions and ecosystem services associated with these important habitats. In 2011 I collected data on sea urchin abundance, predation pressure on tethered urchins, and herbivory rates on seagrasses. In the autumn of 2012 I also collected sediment and seagrass samples in the intertidal, subtidal and reef zones (see paper I). Based on these samples, the effects of management and cover of benthic communities will be assessed on variables that can be linked to the ecosystem services; sediment stabilization, erosion control and carbon sequestration. Additionally, I in Nov 2011 started a disturbance experiment (removal of seagrass) in two fished areas and one CBR in Kenya, to examine how recovery and re-colonization rates are influenced by fishing activities and the size of disturbance. Finally, I will in the autumn 2013 participate in a regional field study (Tanzania, Kenya and Mozambique), estimating the effects of MPAs and CBRs on habitat characteristics and fish communities. This will show the generality of the protection effects shown in this thesis on a larger spatial scale.

Acknowledgements

First of all I would like to thank my supervisor Johan Eklöf. I am very grateful for this opportunity and more importantly – from interesting discussions, support and guidance – you make me want to continue to explore this fascinating subject. I would also like to thank my co-supervisors Nils Kautsky and Beatrice Crona for guidance and expert advice. I sincerely thank my local co-supervisors Dr. Jacqueline Uku (Kenya Marine and Fisheries Research Institute) and Dr. Timothy McClanahan (Wildlife Conservation Society) for making my fieldwork possible. My further gratitude extends to my colleagues and co-workers Samuel Ndirangu and Masudi Zuma (Kenya Marine and Fisheries Research Institute), and to the communities in Kuruwitu and Kanamai in Kenya. Also, I would like to thank my supporting colleagues at the department – it means a lot to me.

Finally, to my extended family Chirico-Klaussén – without you I would not be here today. And Oscar, I am indebted for your patience. Nimeshukuru.

This work was financially supported by Sida (the Swedish International Development Cooperation Agency) and C. F. Liljevalch J:ors scholarship.

References

- Aburto-Oropeza, O., Erisman, B., Galland, G.R., Mascarenas-Osorio, I., Sala, E. & Ezcurra, E. (2011). Large recovery of fish biomass in a no-take marine reserve. *PloS one*, 6, e23601.
- Alcoverro, T. & Mariani, S. (2004). Patterns of fish and sea urchin grazing on tropical Indo-Pacific seagrass beds. *Ecography*, 27, 361-365.
- Alvarez-Filip, L., Dulvy, N.K., Gill, J.A., Cote, I.M. & Watkinson, A.R. (2009). Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. *Proceedings. Biological sciences / The Royal Society*, 276, 3019-3025.
- Anderson, M.J., Gorley, R.N. & Clarke, K.R. (2008). *PERMANOVA+ for PRIMER: Guide to software and statistical methods*. PRIMER-E: Plymouth, UK.
- Anthony, K.R., Kline, D.I., Diaz-Pulido, G., Dove, S. & Hoegh-Guldberg, O. (2008). Ocean acidification causes bleaching and productivity loss in coral reef builders. *Proceedings of the National Academy of Sciences of the United States of America*, 105, 17442-17446.
- Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C. & Silliman, B.R. (2011). The value of estuarine and coastal ecosystem services. *Ecol. Monogr.*, 81, 169-193.
- Bell, J.D. & Galzin, R. (1984). Influence of Live Coral Cover on Coral-Reef Fish Communities. *Marine Ecology Progress Series*, 15, 265-274.
- Björk, M., Short, F., Mcleod, E. & Beer, S. (2008). Managing seagrasses for resilience to climate change. IUCN. Gland, Switzerland, 56 pp.
- Bohnsack, J.A. (1998). Application of marine reserves to reef fisheries management. *Austral Ecology*, 23, 298-304.

- Bruno, J.F. & Bertness, M.D. (2001). Habitat modification and facilitation in benthic marine communities. *Marine Community Ecology*, 201-218.
- Burke, L., Kura, Y., Kassem, K., Revenga, C., Spalding, M. & McAllister, D. (2001). Pilot analysis of global ecosystems: Coastal ecosystems. World Resources Institute, Washington DC, 77 pp.
- Chirico, A.D.A., Uku, J. & Eklöf, J.S. (in preparation). Effects of contrasting management on grazing and herbivory rates in coral and seagrass habitats.
- Christie, P. (2004). Marine protected areas as biological successes and social failures in southeast Asia. In: *Aquatic Protected Areas as Fisheries Management Tools* (ed. Shipley, JB). Amer Fisheries Soc Bethesda, pp. 155-164.
- Cinner, J., McClanahan, T., Wamukota, A., Darling, E., Humphries, A., Hicks, C. *et al.* (2013). Social-ecological vulnerability of coral reef fisheries to climatic shocks. Food and agriculture organization of the united nations (FAO) Fisheries and Aquaculture Circular No. 1082, 64 pp.
- Claudet, J., Osenberg, C.W., Benedetti-Cecchi, L., Domenici, P., Garcia-Charton, J.A., Perez-Ruzafa, A. *et al.* (2008). Marine reserves: size and age do matter. *Ecology letters*, 11, 481-489.
- Clements, C., Bonito, V., Grober-Dunsmore, R. & Sobey, M. (2012). Effects of small, Fijian community-based marine protected areas on exploited reef fishes. *Marine Ecology Progress Series*, 449, 233-243.
- Coker, D.J., Graham, N.A.J. & Pratchett, M.S. (2012). Interactive effects of live coral and structural complexity on the recruitment of reef fishes. *Coral Reefs*, 31, 919-927.
- Daily, G.C. (1997). Nature's services: societal dependence on natural ecosystems. Washington, DC: Island Press.
- Darling, E.S., Alvarez-Filip, L., Oliver, T.A., McClanahan, T.R., Cote, I.M. & Bellwood, D. (2012). Evaluating life-history strategies of reef corals from species traits. *Ecology letters*, 15, 1378-1386.
- Day, J. (2008). The need and practice of monitoring, evaluating and adapting marine planning and management—lessons from the Great Barrier Reef. *Marine Policy*, 32, 823-831.
- de la Torre-Castro, M. & Rönnbäck, P. (2004). Links between humans and seagrasses—an example from tropical East Africa. *Ocean & Coastal Management*, 47, 361-387.
- Dorenbosch, M., Grol, M.G.G., Nagelkerken, I. & van der Velde, G. (2005). Distribution of coral reef fishes along a coral reef-seagrass gradient: edge effects and habitat segregation. *Marine Ecology Progress Series*, 299, 277-288.
- Dorenbosch, M., Grol, M.G.G., Nagelkerken, I. & van der Velde, G. (2006). Different surrounding landscapes may result in different fish assemblages in east african seagrass beds. *Hydrobiologia*, 563, 45-60.
- Duarte, C.M., Dennison, W.C., Orth, R.J.W. & Carruthers, T.J.B. (2008). The Charisma of Coastal Ecosystems: Addressing the Imbalance. *Estuaries Coasts*, 31, 233-238.
- Eklöf, J.S., de la Torre-Castro, M., Gullström, M., Uku, J., Muthiga, N., Lyimo, T. *et al.* (2008). Sea urchin overgrazing of seagrasses: A review of current knowledge on causes, consequences, and management. *Estuarine, Coastal and Shelf Science*, 79, 569-580.
- Fonseca, M.S. (1989). Sediment stabilization by *Halophila decipiens* in comparison to other seagrasses. *Estuarine, Coastal and Shelf Science*, 29, 501-507.
- Froese, R. & Pauly, D. (2012). Fish Base. World Wide Web Electronic Publication.
- Gell, F.R. & Roberts, C.M. (2002). The fishery effects of marine reserves and fishery closures. WWF, Washington DC, USA, 89 pp.
- Gell, F.R. & Whittington, M.W. (2002). Diversity of fishes in seagrass beds in the Quirimba Archipelago, northern Mozambique. *Mar. Freshw. Res.*, 53, 115-121.
- Graham, N.A.J., Ainsworth, T.D., Baird, A.H., Ban, N.C., Bay, L.K., Cinner, J.E. *et al.* (2011). From microbes to people: tractable benefits of no-take areas for coral reefs. In: *Oceanography and Marine Biology: An Annual Review, Vol 49* (eds. Gibson, RN, Atkinson, RJA & Gordon, JDM). Crc Press-Taylor & Francis Group Boca Raton, pp. 105-135.
- Graham, N.A.J. & Nash, K.L. (2013). The importance of structural complexity in coral reef ecosystems. *Coral Reefs*, 32, 315-326.
- Gray, G.J., Enzer, M.J. & Kusel, J. (2001). Understanding community-based forest ecosystem management. *Journal of Sustainable Forestry*, 12, 1-23.

- Green, E.P. & Short, F. (2003). *World Atlas of Seagrasses*. UNEP World Conservation Monitoring Centre. Berkeley: University of California Press.
- Gullstrom, M., Bodin, M., Nilsson, P.G. & Ohman, M.C. (2008). Seagrass structural complexity and landscape configuration as determinants of tropical fish assemblage composition. *Marine Ecology Progress Series*, 363, 241-255.
- Gullstrom, M., de la Torre-Castro, M., Bandeira, S.O., Bjork, M., Dahlberg, M., Kautsky, N. *et al.* (2002). Seagrass ecosystems in the Western Indian Ocean. *Ambio*, 31, 588-596.
- Halpern, B.S. (2003). The impact of marine reserves: Do reserves work and does reserve size matter? *Ecol. Appl.*, 13, S117-S137.
- Heck, K.L., Hays, G. & Orth, R.J. (2003). Critical evaluation of the nursery role hypothesis for seagrass meadows. *Marine Ecology Progress Series*, 253, 123-136.
- Hicks, C.C., McClanahan, T.R., Cinner, J.E. & Hills, J.M. (2009). Trade-Offs in Values Assigned to Ecological Goods and Services Associated with Different Coral Reef Management Strategies. *Ecology and Society*, 14, 18.
- Hilborn, R., Stokes, K., Maguire, J.-J., Smith, T., Botsford, L.W., Mangel, M. *et al.* (2004). When can marine reserves improve fisheries management? *Ocean & Coastal Management*, 47, 197-205.
- Hsieh, C.H., Reiss, C.S., Hunter, J.R., Beddington, J.R., May, R.M. & Sugihara, G. (2006). Fishing elevates variability in the abundance of exploited species. *Nature*, 443, 859-862.
- Jackson, J.B., Kirby, M.X., Berger, W.H., Bjorndal, K.A., Botsford, L.W., Bourque, B.J. *et al.* (2001). Historical overfishing and the recent collapse of coastal ecosystems. *Science*, 293, 629-637.
- Johnson, K.H., Vogt, K.A., Clark, H.J., Schmitz, O.J. & Vogt, D.J. (1996). Biodiversity and the productivity and stability of ecosystems. *Trends Ecol Evol*, 11, 372-377.
- Jones, G.P., McCormick, M.I., Srinivasan, M. & Eagle, J.V. (2004). Coral decline threatens fish biodiversity in marine reserves. *Proceedings of the National Academy of Sciences of the United States of America*, 101, 8251-8253.
- Knowlton, N. & Jackson, J.B. (2008). Shifting baselines, local impacts, and global change on coral reefs. *PLoS biology*, 6, e54.
- Koch, E.M. (2001). Beyond light: Physical, geological, and geochemical parameters as possible submersed aquatic vegetation habitat requirements. *Estuaries*, 24, 1-17.
- Koch, E.W., Barbier, E.B., Silliman, B.R., Reed, D.J., Perillo, G.M.E., Hacker, S.D. *et al.* (2009). Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. *Frontiers in Ecology and the Environment*, 7, 29-37.
- Kremen, C. (2005). Managing ecosystem services: what do we need to know about their ecology? *Ecology letters*, 8, 468-479.
- Larsen, T.H., Williams, N.M. & Kremen, C. (2005). Extinction order and altered community structure rapidly disrupt ecosystem functioning. *Ecology letters*, 8, 538-547.
- Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C. *et al.* (2006). Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science*, 312, 1806-1809.
- Martínez, M.L., Intralawan, A., Vázquez, G., Pérez-Maqueo, O., Sutton, P. & Landgrave, R. (2007). The coasts of our world: Ecological, economic and social importance. *Ecological Economics*, 63, 254-272.
- McClanahan, T.R. (1987). Overfishing and coral reef degradation: a preliminary report from East Africa. *Conserv. Biol.*, 1, 97-102.
- McClanahan, T.R. (1999). Is there a future for coral reef parks in poor tropical countries? *Coral Reefs*, 18, 321-325.
- McClanahan, T.R. (2008). Response of the coral reef benthos and herbivory to fishery closure management and the 1998 ENSO disturbance. *Oecologia*, 155, 169-177.
- McClanahan, T.R. (2010). Effects of fisheries closures and gear restrictions on fishing income in a Kenyan coral reef. *Conservation biology : the journal of the Society for Conservation Biology*, 24, 1519-1528.
- McClanahan, T.R., Graham, N.A., Calnan, J.M. & MacNeil, M.A. (2007). Toward pristine biomass: reef fish recovery in coral reef marine protected areas in Kenya. *Ecological applications : a publication of the Ecological Society of America*, 17, 1055-1067.

- McClanahan, T.R., Graham, N.A.J., Wilson, S.K., Letourneur, Y. & Fisher, R. (2009). Effects of fisheries closure size, age, and history of compliance on coral reef fish communities in the western Indian Ocean. *Marine Ecology Progress Series*, 396, 99-109.
- McClanahan, T.R., Hicks, C.C. & Darling, E.S. (2008). Malthusian overfishing and efforts to overcome it on Kenyan coral reefs. *Ecological applications : a publication of the Ecological Society of America*, 18, 1516-1529.
- McClanahan, T.R., Kamukuru, A.T., Muthiga, N.A., Yebio, M.G. & Obura, D. (1996). Effect of Sea Urchin Reductions on Algae, Coral, and Fish Populations. *Conserv. Biol.*, 10, 136-154.
- McClanahan, T.R., Marnane, M.J., Cinner, J.E. & Kiene, W.E. (2006). A comparison of marine protected areas and alternative approaches to coral-reef management. *Current biology : CB*, 16, 1408-1413.
- McClanahan, T.R. & Muthiga, N.A. (1988). Changes in Kenyan coral reef community structure and function due to exploitation. *Hydrobiologia*, 166, 269-276.
- McClanahan, T.R. & Muthiga, N.A. (1989). Patterns of predation on a sea urchin, *Echinometra mathaei* (de Blainville), on Kenyan coral reefs. *Journal of Experimental Marine Biology and Ecology*, 126, 77-94.
- McClanahan, T.R., Mwanguni, S. & Muthiga, N.A. (2005). Management of the Kenyan coast. *Ocean & Coastal Management*, 48, 901-931.
- McClanahan, T.R., Nugues, M. & Mwachireya, S. (1994). Fish and sea urchin herbivory and competition in Kenyan coral reef lagoons: the role of reef management. *Journal of Experimental Marine Biology and Ecology*, 184, 237-254.
- McClanahan, T.R. & Shafir, S.H. (1990). Causes and consequences of sea urchin abundance and diversity in Kenyan coral reef lagoons. *Oecologia*, 83, 362-370.
- McLeod, E., Chmura, G.L., Bouillon, S., Salm, R., Björk, M., Duarte, C.M. *et al.* (2011). A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO₂. *Frontiers in Ecology and the Environment*, 9, 552-560.
- MEA (2005). *Ecosystems and Human Well-Being: Synthesis*. Island Press, Washington, DC.
- Moberg, F. & Folke, C. (1999). Ecological goods and services of coral reef ecosystems. *Ecological Economics*, 29, 215-233.
- Moffitt, E.A., White, J.W. & Botsford, L.W. (2013). Accurate assessment of marine protected area success depends on metric and spatiotemporal scale of monitoring. *Marine Ecology Progress Series*, 489, 17-28.
- Mumby, P.J., Dahlgren, C.P., Harborne, A.R., Kappel, C.V., Micheli, F., Brumbaugh, D.R. *et al.* (2006). Fishing, trophic cascades, and the process of grazing on coral reefs. *Science*, 311, 98-101.
- Nardi, K., Jones, G.P., Moran, M.J. & Cheng, Y.W. (2004). Contrasting effects of marine protected areas on the abundance of two exploited reef fishes at the sub-tropical Houtman Abrolhos Islands, Western Australia. *Environmental Conservation*, 31, 160-168.
- Naylor, R.L., Goldburg, R.J., Primavera, J.H., Kautsky, N., Beveridge, M.C., Clay, J. *et al.* (2000). Effect of aquaculture on world fish supplies. *Nature*, 405, 1017-1024.
- O'Leary, J.K., Potts, D.C., Braga, J.C. & McClanahan, T.R. (2012). Indirect consequences of fishing: reduction of coralline algae suppresses juvenile coral abundance. *Coral Reefs*, 31, 547-559.
- Obura, D.O. (2001). Kenya. *Marine pollution bulletin*, 42, 1264-1278.
- Ogden, J.C. (1988). The influence of adjacent systems on the structure and function of coral reefs. Proceedings of the Sixth International Coral Reef Symposium, Townsville, Australia, p 123-129.
- Ogden, J.C. & Gladfelter, E.H. (1983). Coral reefs, seagrass beds and mangroves: their interactions in the coastal zones of the Caribbean. UNESCO Reports of Marine Science. 23, 133.
- Olwig, M.F., Sørensen, M.K., Rasmussen, M.S., Danielsen, F., Selvam, V., Hansen, L.B. *et al.* (2007). Using remote sensing to assess the protective role of coastal woody vegetation against tsunami waves. *International Journal of Remote Sensing*, 28, 3153-3169.
- Orth, R.J., Carruthers, T.J.B., Dennison, W.C., Duarte, C.M., Fourqurean, J.W., Heck, K.L. *et al.* (2006). A global crisis for seagrass ecosystems. *Bioscience*, 56, 987-996.

- Peterson, B.J., Rose, C.D., Rutten, L.M. & Fourqurean, J.W. (2002). Disturbance and recovery following catastrophic grazing: studies of a successional chronosequence in a seagrass bed. *Oikos*, 97, 361-370.
- Pollnac, R.B., Crawford, B.R. & Gorospe, M.L.G. (2001). Discovering factors that influence the success of community-based marine protected areas in the Visayas, Philippines. *Ocean & Coastal Management*, 44, 683-710.
- Power, A.G. (2010). Ecosystem services and agriculture: tradeoffs and synergies. *Philos. Trans. R. Soc. B-Biol. Sci.*, 365, 2959-2971.
- Quijas, S., Schmid, B. & Balvanera, P. (2010). Plant diversity enhances provision of ecosystem services: A new synthesis. *Basic Appl. Ecol.*, 11, 582-593.
- Redford, K.H. & Fearn, E. (2007). Protected areas and human livelihoods. Wildlife Conservation Society. Working Paper no. 32, 199 pp.
- Riegl, B.M., Bruckner, A.W., Rowlands, G.P., Purkis, S.J. & Renaud, P. (2012). Red Sea coral reef trajectories over 2 decades suggest increasing community homogenization and decline in coral size. *PLoS one*, 7, e38396.
- Roberts, C.M. & Hawkins, J.P. (2000). Fully-protected marine reserves: a guide. WWF, Washington DC, USA and Environment Department, York, UK, 133 pp.
- Roberts, C.M. & Polunin, N.V.C. (1991). Are marine reserves effective in management of reef fisheries? , 1, 65-91.
- Rose, C.D., Sharp, W.C., Kenworthy, W.J., Hunt, J.H., Lyons, W.G., Prager, E.J. *et al.* (1999). Overgrazing of a large seagrass bed by the sea urchin *Lytechinus variegatus* in Outer Florida Bay. *Marine Ecology Progress Series*, 190, 211-222.
- Salm, R.V., Clark, J. & Siirila, E. (2000). *Marine and Coastal Protected Areas: A guide for planners and managers*, IUCN. Washington DC. xxi + 371 pp.
- Semesi, I.S., Beer, S. & Björk, M. (2009). Seagrass photosynthesis controls rates of calcification and photosynthesis of calcareous macroalgae in a tropical seagrass meadow. *Marine Ecology Progress Series*, 382, 41-48.
- Spalding, M.D., Meliane, I., Fitzgerald, C. & Hale, L.Z. (2013). Coastal and Marine Spatial Planning. Protecting marine spaces: global targets and changing approaches. In OCEAN yearbook 27, Leiden Boston.
- Srinivasan, U.T., Cheung, W.L., Watson, R. & Sumaila, U.R. (2010). Food security implications of global marine catch losses due to overfishing. *J Bioecon*, 12, 183-200.
- UNEP (2006). Marine and coastal ecosystems and human well-being: A synthesis report based on the findings of the Millennium Ecosystem Assessment. UNEP, 76 pp.
- Unsworth, R.K.F., Cullen, L.C., Pretty, J.N., Smith, D.J. & Bell, J.J. (2010). Economic and subsistence values of the standing stocks of seagrass fisheries: Potential benefits of no-fishing marine protected area management. *Ocean & Coastal Management*, 53, 218-224.
- Walker, B.H. (1992). Biodiversity and Ecological Redundancy. *Conserv. Biol.*, 6, 18-23.
- Waycott, M., Duarte, C.M., Carruthers, T.J., Orth, R.J., Dennison, W.C., Olyarnik, S. *et al.* (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences of the United States of America*, 106, 12377-12381.
- Weeks, R., Russ, G.R., Alcala, A.C. & White, A.T. (2010). Effectiveness of marine protected areas in the Philippines for biodiversity conservation. *Conservation biology : the journal of the Society for Conservation Biology*, 24, 531-540.
- Wells, S., Burgess, N. & Ngusuru, A. (2007). Towards the 2012 marine protected area targets in Eastern Africa. *Ocean & Coastal Management*, 50, 67-83.
- White, A.T. & Vogt, H.P. (2000). Philippine coral reefs under threat: Lessons learned after 25 years of community-based reef conservation. *Marine pollution bulletin*, 40, 537-550.
- Wilkinson, C. (2008). Status of coral reefs of the world: 2008. Global coral reef monitoring network and reef and rainforest research centre, Townsville, Australia, 296 pp.
- Worm, B., Barbier, E.B., Beaumont, N., Duffy, J.E., Folke, C., Halpern, B.S. *et al.* (2006). Impacts of biodiversity loss on ocean ecosystem services. *Science*, 314, 787-790.
- Vroom, P.S. (2011). "Coral Dominance": A Dangerous Ecosystem Misnomer? *Journal of Marine Biology*, 2011, 1-8.

Stockholms universitet /Stockholm University
SE – 106 91 Stockholm
Telefon/Phone: 08-16 20 00
www.su.se



**Stockholms
universitet**