Institutional Analysis of Marine Reserves and Fisheries Governance Policy Experiments

A Case Study of Nassau Grouper Conservation in the Turks and Caicos Islands

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ABSTRACT

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Many tropical fisheries around the world are in crisis because of the depletion of valuable reef species and the destruction of habitat upon which they depend. The complexity of reef fisheries and lack of management resources in southern nations limit the potential effectiveness of policies that focus on single species. As a result, ecosystem-based fisheries management is increasingly viewed as the only real alternative for managing these tropical reef fisheries. There is a widely held view that the devolution of management power from central government managers to local communities is central to the ecosystem-based fisheries management process and that marine reserves are the primary tool by which to implement ecosystem-based fisheries management. Marine reserves can protect or enhance multiple ecosystem services simultaneously and arguments are often made that they are more cost-effective than other management options because they are easy to monitor and enforce.

The first theoretically-oriented part of this research emphasizes the role that social capital – the norms, networks and governance infrastructure that facilitates mutually advantageous collective action – plays in ecosystem-based fisheries management. In the second part of the research, I illustrate the utility of taking an institutional analysis approach to ecosystem-based fisheries management policy by examining the case of Nassau grouper (*Epinephelus striatus*) conservation and fisheries management in the Turks and Caicos Islands. While the focus of this case study is a single, small island nation, I believe that the results – that there are substantial incentives for private sector and government actors to oppose implementation of marine reserves – have broader relevance in the debate over the use of marine reserves for tropical fisheries management and conservation. Marine reserves are widely viewed as cost-effective, all-purpose tools for fisheries enhancement and conservation, yet my results suggest that there are policy alternatives – in this case, a commercial trade ban on Nassau grouper in tourist-oriented restaurants – that are much more likely to be effectively implemented and that should be substantially more cost-effective than marine reserves. Market-oriented policy tools should not be under-emphasized in ecosystem-based fisheries management. In instances where local social capital is lacking, they may actually have a higher likelihood of

achieving conservation objectives and be more cost-effective than poorly supported marine reserves or 'paper parks'.

Keywords: Ecosystem-based fisheries management; marine reserves; marine protected areas; social capital; institutional analysis; Turks and Caicos Islands; Nassau grouper

PREFACE

This dissertation is the result of work over the past five years in the field of institutional analysis and tropical artisanal fisheries policy analysis. Needless to say, many people have been very supportive during this period, sharing their expertise, opinions, and time to help me develop my ideas, viewpoints and analyses.

There are a few people who I'd like to thank specifically. First and foremost, my promoter, Kees van Kooten (who was my Master's supervisor while at UBC) deserves credit for proposing that I consider Wageningen University when I went to work in the Caribbean and for motivating me to keep writing. Thanks also to my other co-promotor Dr. Prof. Henk Folmer, who not only provided prudent advice on how to approach this program of study, but was vigilent that the dissertation retained a high quality and that it was completed in reasonable time.

I also owe a special thanks to Elinor Ostrom at the Workshop in Political Theory and Policy Analysis, Indiana University. I actually started my Ph.D. studies with Lin in 1998 but was enticed away to the Caribbean after a year (in what was truly one of the most difficult decisions I've ever faced). Despite the fact that I didn't finish up my Ph.D. at Indiana, the year was instrumental in setting the direction that I have taken in this dissertation and in my career. Lin was understanding of the decision to leave, left the door open to return, and, even though that didn't happen, has acted as an external reader for this dissertation and has provided [amazingly quick] advice and encouragement in this work when needed.

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INTRODUCTION

CHAPTER 1

ECOSYSTEM-BASED FISHERIES MANAGEMENT, MARINE RESERVES AND FISHERIES GOVERNANCE

It is evident that efforts to govern fisheries using ‘traditional’ fisheries management models (*i.e*, optimization of the biological or economic yield of single species by using tools that seek to limit fishing effort and/or power) have often failed (Botsford *et al.*, 1997; Pauly *et al.*, 2002). Commercially important species are being depleted (FAO, 2000), resource rents are being dissipated (Milazzo, 1998), and ecosystem structure and function are being compromised (Pauly *et al.,* 1998). Nowhere is the failure of fisheries management more apparent than in tropical inshore reef fisheries (Johannes, 1998) where vulnerable species are often intensely exploited by artisanal fishers who use diverse, and sometimes destructive, fishing methods.

There is a growing consensus amongst fisheries scientists and managers that a new ecosystem-based fisheries management paradigm is needed for achieving fisheries sustainability (NMFS, 1999; NRC, 1999; Gislason *et al.,* 2000). General principles of ecosystem-based management (Christensen *et al.*, 1996) have been adopted in many fisheries and have been institutionalized at international (*e.g.*, *United Nations Convention on the Law of the Sea*), regional (*e.g.*, Commission for the Conservation of Atlantic Tuna) and national levels (*e.g.*, Canada’s *Ocean Act* and USA’s *Magnuson-Stevens Fishery Conservation and Management Act*). In some developing countries, where customary marine tenure is established or undergoing resurgence, community-based fisheries management efforts have often long taken an implicit ecosystem-based fisheries management approach (Ruddle, 1998; Johannes, 2002).

Under ecosystem-based fisheries management, experimental management [1] (Walters, 1986, 1997) is needed to build further understanding of the relationships between ecological processes, fisheries regulations, the behavior of resource users, and ecological and socio-economic outcomes in complex fishery systems. Experimental management goes beyond just implementing better ecological monitoring and engaging in *ad hoc* ‘adaptive’ responses to unexpected management challenges, replacing evolutionary learning by trial and error with learning by a process of directed policy selection (Walters, 1997). Human values, incentives and behaviors need to be considered when

[1] Some people consider ‘adaptive’ and ‘experimental’ management as synonymous. I use the term ‘experimental’ throughout the dissertation due to my emphasis on policy experiments.

selecting ecosystem-based fisheries management policies because they impact the feasibility, effectiveness and efficiency of meeting outcome-oriented conservation objectives (Costanza *et al.*, 1998; Rudd *et al.*, 2003). Institutions – the human-crafted rules and norms that infuse social order – shape human incentives and behavior (Ostrom, 1999) and, therefore, play a central role in experimental ecosystem-based fisheries management.

It is widely recognized that a variety of institutions (means) might be crafted to achieve any particular societal objectives (ends). Even small-scale artisanal fisheries use a plethora of rules to govern when and how resources are harvested and used by particular users (Schlager, 1990; Ruddle, 1998; Johannes, 2002). Furthermore, the array of options may vary greatly in costs, making it necessary to design, monitor and evaluate ecosystem-based fisheries management policy experiments that strategically test the effectiveness and efficiency of various policy options (Rudd *et al.*, 2003).

How can high-level strategic ecosystem-based fisheries management principles be implemented in practice? Two approaches are particularly popular in the current literature and are currently being widely tested in the field: (1) the use of marine reserves or marine protected areas for fisheries conservation and management purposes (Murray *et al.*, 1999; Dayton *et al.*, 2000; Palumbi, 2002; Halpern, 2003); and (2) the devolution of fisheries management responsibilities to communities or co-management organizations (in which communities and government formally share management responsibilities) (Pomeroy and Berkes, 1997; Pomeroy, 2001). These approaches are often advocated in tandem.[2]

Marine protected areas (MPAs) are viewed as a central policy tool for implementing marine ecosystem-based management (Costanza *et al.,* 1998; Murray *et al.,* 1999; Palumbi, 2002). For the purposes of this dissertation, I define MPAs as no-take reserves and refer to marine reserves and MPAs synonymously.[3] MPAs are thought to address the main themes of ecosystem-based fisheries management by contributing to the

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[2] For example, there are two projects currently underway that are both oriented towards the use of highly devolved community-based MPAs: the U.K.-funded project on *Institutional Arrangements for Caribbean MPAs and Opportunities for Pro-Poor Management* (Esteban *et al.*, 2002) and the WCPA-Marine/WWF-funded project on *International MPA Management Effectiveness Initiative* (Pomeroy *et al.*, 2002).

[3] More generally, a marine protected area is an area that is designated by law or other authority and restricts some activities so that the habitat, species and other resources within the MPA are afforded some level of protection. A marine reserve can be viewed as a special type of MPA that affords full protection for all resources within its boundaries.

maintenance of biodiversity and ecological processes that maintain resilience while enhancing fisheries, increasing opportunities for non-consumptive activities and building knowledge for improving coastal management (Costanza *et al.,* 1998; Conover *et al.,* 2000; Dayton *et al.,* 2000). It is well known that marine reserves can protect essential fish habitat and vulnerable fish species, leading to increased animal size and/or abundance within reserve boundaries (see Halpern, 2003). Theoretically, MPAs can also induce larval 'export' and density-dependent emigration ('spillover') across reserve boundaries (Murray *et al.*, 1999) although empirical evidence of these phenomena is limited (McClanahan and Mangi, 2000; Roberts *et al.*, 2001; Tupper and Rudd, 2002). MPA advocates argue that a key advantage of using MPAs is that they can simultaneously address multiple threats to ocean ecosystems (Palumbi, 2002), making them attractive multi-purpose tools for ecosystem-based fisheries management.

Economic arguments for marine reserves center on their pragmatic advantages relating to transaction costs: MPAs are thought to reduce management costs because of their simplicity and ease of enforcement, and posess a comparative cost advantage relative to traditional management tools (Roberts and Polunin, 1993; Murray *et al.,* 1999; Roberts, 2000). To date, however, there have been too few rigorous policy analyses of marine reserves to assess if reserves are, in fact, efficient policy tools for tropical inshore fisheries conservation and management (Rudd *et al.*, 2003). If credible analyses are not undertaken, there is a danger that current enthusiasm for marine reserves may wane as economic performance, in particular, fails to meet presumed potential.

Devolution of fisheries governance involves the transfer of decision-making authority to local communities (Pomeroy, 2001), implying that they share in decisions about fisheries management rules and MPA design. Decentralization of power, on the other hand, involves shifting decision-making to local branches of government. Local government agents, who may be more aware of the unique local social and natural environment, might be able to craft policies that are more consistent with local norms than a central government agency would be able to. The degree of devolution/decentralization that minimizes the transaction costs of ecosystem-based fisheries management depends on a number of factors.

Williamson (1985) developed the 'discriminating alignment hypothesis', which postulates that transactions have certain attributes that affect transaction costs and that governance systems have certain competencies and costs. Minimizing societal transaction costs requires that these two factors be aligned. The transaction of interest in tropical inshore fisheries is the production and maintenance of reef environmental quality that provides a flow of various ecological amenities. The transaction costs of ecosystem-based fisheries management include: the costs of environmental monitoring of reef conditions and fish stocks; discussing, developing and reaching agreement on rules (*i.e*, reserve design and access) to control resource use; the legal costs for implementing management solutions; ongoing monitoring costs to ensure compliance with the final policy package; and the costs associated with enforcing penalties for those who violate the rules. Policy experiments will play a central role in assessing whether, and at what cost, ecosystem-based fisheries management objectives are met under various governance systems.

Transaction cost economics provides the link between the devolution of fisheries governance and the implementation of MPAs under ecosystem-based fisheries management. MPA proponents (often ecologists) argue that MPAs reduce the transaction costs of ecosystem-based fisheries management, thus helping to achieve multiple societal objectives relating to conservation, economic efficiency and, quite often, social equity. There is a growing body of knowledge in the field of the 'new institutional economics' that provides the theory necessary for examining the transaction-cost economizing devolution or decentralization of governance. Using insights from transaction cost economics, it should be possible to assess when, and under what conditions, MPAs are likely to be efficient tools for achieving ecosystem-based fisheries management objectives. This link between fisheries management and the new institutional economics could provide the basis for designing and assessing a variety of MPA policy experiments.

Dissertation Goals and Objectives

Ecosystem-based fisheries management is transdisciplinary in nature, considering broad ecological, social, economic and governance objectives. From an ecological perspective, ecosystem-based fisheries management policies need to be able to meet conservation and fisheries management objectives of maintaining biodiversity and ecosystem structure/function in addition to traditional objectives of maintaining the productivity of target species. From an economic perspective, ecosystem-based fisheries management must consider the total benefits that people derive from ecosystem goods and services, the distribution of those benefits and the costs of implementing various policies that might be used to achieve overall ecosystem-based fisheries management objectives. This requires accounting for the transaction costs of management as well as the nonmarket values of ecosystem goods and services. From a governance perspective, the role of institutions – the 'rules-of-the-game' – is critical if we want to understand how particular bundles of rules and norms influence patterns of behavior and aggregate outcomes that may ultimately threaten the sustainability of a fishery system.

My goal in this dissertation is to integrate insights from the disciplines of ecology, economics and political science in a way that facilitates the design and assessment of ecosystem-based fisheries management experiments involving MPAs and/or devolution of fisheries governance. To do this, I modify the Institutional Analysis and Development (IAD) framework developed by Elinor Ostrom and colleagues (Ostrom, 1990; Ostrom *et al.,* 1994; Ostrom, 1999) to develop an integrated ecological-economic framework for ecosystem-based fisheries management policy analysis. The first theoretically-oriented part of the dissertation emphasizes the role of social capital – the norms, networks and governance infrastructure that facilitates mutually advantageous collective action and the reduction of transaction costs – in ecosystem-based fisheries management experimental design and assessment.

In the second part of the dissertation, I illustrate the utility of taking an integrated institutional ecosystem-based fisheries management approach by examining the case of Nassau grouper (*Epinephelus striatus*) conservation and fisheries management in the Turks and Caicos Islands. My goal in this section is to assess what policy or policies are most likely to lead to Nassau grouper conservation given the realities of the ecological, socioeconomic and political environments within the TCI.

The Nassau grouper is a predatory reef fish that fills an important ecological niche near the top of the food chain, is a highly desirable target species in commercial and subsistence fisheries, and is an 'icon' species that is widely recognized by dive tourists in the Caribbean. Nassau grouper stocks are inherently vulnerable and have already been extirpated by intense fishing pressure in several parts of the Caribbean (Sadovy and Eklund, 1999). Recent survey results indicate, for example, that Nassau grouper has been depleted to such an extent that fish were absent from 82% of shallow Caribbean reefs surveyed during a 5-year period (Hodgson and Liebeler, 2002).

While Nassau grouper abundance in the Turks and Caicos Islands (TCI) is still relatively high (Tupper and Rudd, 2002), the overall situation in the TCI is similar in many respects to that in a number of other tropical island nations with important reef fisheries. Data on the nature of the Nassau grouper resource and fishery is virtually non-existent despite the fact that it is important for subsistence purposes, as a component of a commercial artisanal fishery, and as an attraction for the dive tourism industry. Government capacity to effectively manage the resource is limited, as is the capacity of communities and the fishing industry to participate in co-management. The Nassau grouper fishery is currently a *de facto* open access fishery except for controls provided by a handful of small MPAs that were created in the 1990s; these reserves were not, however, designed to protect reef fish, nor are they closely monitored or enforced. Given the increasing numbers of tourists coming to the islands and the extreme vulnerability of the species itself, the issue of Nassau grouper conservation is of utmost urgency, especially considering that the TCI has some of the few remaining stocks of Nassau grouper in the tropical western Atlantic that are relatively healthy and potentially capable of exporting larvae to 'downstream' ecosystems in the Bahamas and Florida.

In the second section of the dissertation, I take a high-level ecosystem-based fisheries management goal – conservation of the Nassau grouper resource – and break the problem into a series of shorter, more focused research questions:

- Does the Admiral Cockburn Land and Sea National Park (ACLSNP) (an important MPA in the TCI that is potentially subject to both artisanal commercial fishing pressure and increasing dive tourism) effectively protect Nassau grouper within the reserve, thus enhancing non-extractive economic value for recreational divers?

- Does ACLSNP provide density-dependent emmigration (spillover) of Nassau grouper out of the reserve, thus increasing extractive economic values for commercial fishers?

- Who are the relevant actors in the TCI that may have an important direct or indirect impact on Nassau grouper conservation and what are their incentives for supporting or opposing ACLSNP or other MPAs designed for Nassau grouper conservation?

- Are there other policy tools that are more effective and efficient policy tools than MPAs for Nassau grouper conservation?

While the focus of this case study is a single, small island nation, I believe that the results – that there are substantial incentives for private sector and government actors to oppose MPA implementation – have broader relevance in the debate over the use of MPAs for tropical fisheries management and conservation. MPAs are widely viewed as an all-purpose tool for fisheries enhancement and conservation, yet my results suggest that there are policy alternatives – in this case, a commercial trade ban on Nassau grouper in tourist-oriented restaurants – that are much more likely to be effectively implemented and that should be substantially more cost-effective than MPAs.

A large part of the reason that MPAs are unlikely to be efficient conservation tools in the TCI relates to social capital and the capacity of the local community to engage in meaningful management of the local fishery. Where there is distrust and community apathy, as in the TCI, there are few internal social sanctions against individual opportunism. Where there is the potential for opportunism, there is a particularly serious threat to conservation. Local fishermen could reap financial benefits by fishing illegally in the expanded MPAs that would be needed to provide adequate protection for Nassau grouper in the TCI. Fishing would pose a threat to Nassau grouper stocks unless MPAs were very strictly enforced, fishers were compensated for their loss fishing income, and/or social norms shifted to such an extent that there were substantial social constraints on opportunism.

In many areas where opportunities for tourism or other employment are limited, poverty is widespread, or the possibilities for financial transfers from the international beneficiaries of conservation to those local fishers bearing the costs are low (*i.e*, the majority of the tropical developing countries that are most in need of marine conservation programs), the only real 'selling' point for MPAs may be the possibility of their local density-dependent spillover benefits for extractive fisheries. Thus, we sometimes see proponents of MPAs emphasize their benefits as fishery management tools even though it is clear that the primary reason they are being proposed is as a tool for conservation (*e.g.*, Palumbi, 2002). While it is clear that MPAs can be a very effective tool for conservation by providing 'insurance' against catostrophic stock collapses and, theoretically, can provide fishery benefits outside the reserve itself, it may be a dangerous strategy to promote MPAs on any grounds other than for conservation reasons because of the potential for longer-term disillusionment with MPAs should fisheries benefits fail to materialize.

Returning to the broader issue of devolution of fisheries governance, it is evident that the level of community capacity (the ability of people and communities to use social capital for mutually beneficial collective action) and government capacity (the ability of the government to provide public goods and legal infrastructure that can assure that citizen's rights are honored) in a region will substantially influence marine reserve effectiveness and efficiency. Based on insights from transaction cost economics, common property theory, and the results of the TCI case study, I submit that both community and state capacity are necessary for deriving the full benefits from marine reserves. Where one or both components of the equation are missing, policy tools other than marine reserves will likely be more efficient. While I cannot yet provide specific guidance as to which policy tools are likely to be preferred under certain sets of conditions in this dissertation, I do provide an institutional framework for addressing that question. A focus on social capital, institutions and transaction costs should permit rigorous examination of the devolution of fisheries governance under ecosystem-based fisheries management. Ascertaining the proper scope of governance – the efficiency-maximizing balance of the 'State' and the 'Market' – is currently a central issue in the transaction cost economics research agenda (*e.g.,* Williamson, 1999). A string of highly publicized fishery collapses has already moved fishery management research – and particularly research on MPA experiments – from a narrow subdivision of applied ecology to the forefront of the debate over resource management and policy (Conover *et al.,* 2000). An opportunity now exists to move tropical MPA research forward by rigorous analysis of the efficiency-maximizing boundaries between the 'State' and 'Community'.

Organization of Dissertation

Theory and Methods

In the first part of the dissertation, four chapters introduce and develop the key theoretical concepts that are needed to design and assess ecosystem-based fisheries management policy experiments regarding MPAs and the devolution of fisheries governance. This section focuses on bridging individual disciplines and integrating insights and approaches from each discipline into a policy analysis framework that is broader in nature than those currently used to evaluate strategies (devolution) and tools (MPAs) for experimental ecosystem-based fisheries management.

In Chapter 2, I outline the principles of economic cost-benefit analysis of market and nonmarket values for recreational fisheries using examples from various jurisdictions[4]. Recreational fisheries around the world provide humans with important economic benefits because people derive well being from participating in the act of fishing. Many of these benefits are difficult to value, however, because they are nonmarket in nature and depend on 'free' ecological services. Other sectors of society may also depend on these public goods. It is difficult to exclude people from using public goods and there is, therefore, a tendency for them to be under-produced by the private sector. Thus, there is often a need for government policy intervention to ensure the adequate production of

[4] This chapter was written specifically for a volume on recreational fisheries, but outlines general economic principles that are germane for tropical MPAs with recreational tourism potential as well as those without.

public ecological services and resolve conflicts over their use. Policies that affect recreational fisheries have costs and benefits, both for anglers and people in other sectors of society, that must be accounted for if social well being is to be maximized. Economics can be used to quantify the costs and benefits of various policy options available to society, and make recommendations that improve overall economic efficiency. Overall well being (welfare) consists of the sum of 'surpluses' accruing to producers and consumers. I also consider how economic analysis can be used to account for the transaction costs of fisheries management – costs often borne by society as a whole – for different forms of governance.

Chapter 3 expands on the issue of economic analysis of MPAs and develops the links between social capital and ecosystem-based fisheries management tools and approaches in further detail. MPAs are considered to be a central tool for ecosystem-based fisheries management in tropical inshore fisheries. The arguments supporting marine reserves are often based on both the nonmarket values of ecological amenities marine reserves provide and the pragmatic cost saving advantages relating to reserve monitoring and enforcement. Marine reserves are, however, only one of a suite of possible policy options that might be used to achieve conservation and fisheries management objectives and have rarely been the focus of rigorous policy analyses that consider a full range of economic costs and benefits, including the transaction costs of management. If credible analyses are not undertaken, there is a danger that current enthusiasm for marine reserves may wane as economic performance fails to meet presumed potential. Fully accounting for the value of ecological services flowing from marine reserves requires consideration of increased size and abundance of focal species within reserve boundaries, emigration of target species from reserves to adjacent fishing grounds, changes in ecological resilience, and behavioral responses of fishers to spatially explicit closures. Expanding policy assessments beyond standard cost-benefit analysis also requires considering the impact of social capital on the costs of managing fisheries. In the short term, the amount of social capital that communities possess and the capacity of the State to support the rights of individuals and communities will affect the relative efficiency of marine reserves. Reserves may be the most efficient policy option when both Community and State capacity is high, but may not be when one and/or the other is weak. In the longer term, the level of social capital that a society possesses and the level of uncertainty in ecological and social systems will also impact the appropriate level of devolution or decentralization of fisheries governance. Determining the proper balance of the 'State' and the ‘Community’ in tropical fisheries governance will require broad comparative studies of marine reserves and alternative policy tools.

To understand the social driving forces that lead to environmental change, we must account for the role of social interactions, the development of norms of behavior and the institutionalization of rules and norms – the development of ‘social capital’. Chapter 4 demonstrates the utility of social capital theory by articulating linkages between human decision making at individual and collective levels and social vision, an important research focus within the emerging ecological economics research tradition. Social capital theory clarifies relationships between social interactions and outcomes that contribute to the production of environmental quality, public peace and economic prosperity, necessary factors for long-term social and ecological sustainability.

Indicator systems are seen as central tools for ecosystem-based fisheries management, helping to steer fisheries towards sustainability by providing timely and useful information to decision-makers. Without testing hypotheses about the links between policies and outcomes, however, indicator systems may do little more than promote ad hoc policies, possibly even prolonging the transition to sustainable fisheries. The Institutional Analysis and Development (IAD) framework is a robust framework that has been used extensively to design policy experiments and empirically test theories and models linking ecological-economic systems, institutions and the sustainability of common pool resource systems. In Chapter 5, I develop a modified IAD framework that transparently encompasses both process-oriented Pressure-State-Response and structurally-oriented Sustainable Livelihood indicator frameworks, thus providing a platform for ecosystem-based fisheries management policy experiment design and monitoring. An institutional approach to fisheries management facilitates critical examination of important cross-cutting issues, including assumptions regarding what comprises sustainability and how market, government and civil society organizations use strategic investments in capital assets and institutions to achieve sustainability objectives. The emphasis on capital assets keeps attention on the relative merits of alternative investment options in policy experiments.

Turks and Caicos Islands Case Study

The second part of the dissertation addresses a specific case study, a policy analysis of Nassau grouper conservation and fisheries management options in the TCI. The development of the case proceeds in sections, each of which works to solve a small piece of the larger question regarding the effectiveness and efficiency of alternative policy options for Nassau grouper conservation.

The Turks and Caicos Islands is a sparsely populated, island nation located at the southern end of the Bahamian archipelago. The Caicos Bank has supported an export-oriented queen conch fishery for over 100 years. More recently, an export-oriented spiny lobster fishery developed and a burgeoning domestic market for reef fishes is currently developing as local tourism grows. Chapter 6 provides an overview of fishery production and trade in the Turks and Caicos over the past century.

Chapter 7 examines local market demand for reef fishes from the artisanal inshore fishery in the TCI. The rapid growth of tourism in the TCI has dramatically increased the demand for seafood but, as yet, the reef fish fishery is relatively undeveloped. Large carnivorous reef fish such as Nassau grouper are particularly vulnerable to overfishing because of their biology and their popularity in restaurants. The local fishing sector is protected by tariffs up to 40% on imported seafood products: theoretically, this should increase demand for local fishes as it makes imported products comparatively more expensive. This study uses a paired comparison conjoint survey of TCI restaurants to assess the effects of changes in the import tariff rate on market demand for fresh domestic and frozen imported grouper, and potential substitute products. I find that the import tariff significantly increases demand for local Nassau grouper and, hence, could place increasing fishing pressure on these vulnerable reef fish. The policy implications and alternatives for Nassau grouper conservation are briefly examined.

Understanding fisher incentives and behavior is crucial if we are to understand how various policy tools affect critical conservation and economic development objectives in tropical artisanal fisheries. This is a challenge in most tropical reef fisheries due to poor data and complex fishing patterns. In multi-species artisanal fisheries, fishers may use multiple gear types, move rapidly between sites and switch target species on an intra-day basis. Some fishers seem to allocate effort based on revenue targets rather than profit or revenue maximization criteria. It has also been argued that social standing and prestige cause fishers to allocate fishing effort in ways that would not be predicted economically (Béné and Tewfik, 2001). Chapter 8 briefly examines this argument using Béné and Tewfik's TCI data. An important conclusion of their research – that lobster fishing is preferred over conch fishing because of prestige – does not follow when reef fish landings are factored into their calculation. This provides indirect, but important, evidence that TCI fishers do respond to price signals in local markets and that increasing market demand for Nassau grouper, fueled by both increasing tourist arrivals and the government import tariff, will induce increasing fishing effort targeting Nassau grouper.

Since many fisheries are size-selective, the establishment of MPAs is expected to increase both the average size and abundance of exploited species such as the valuable, but vulnerable, Nassau grouper. Increases in mean size and/or abundance of protected species within MPAs may also provide non-extractive economic value to recreationalists. In Chapter 9, I assess scuba diver preferences for viewing Nassau grouper and the marginal trade-off's that divers exhibit between fish size and abundance, and dive group size and price in the TCI. I use results from a paired comparison conjoint survey to develop market share simulations of dive site choice. Market shares increased significantly for sites with increased Nassau grouper abundance and mean size. This implies that Nassau groupers provide non-extractive economic value to divers. The results suggest that accounting for the non-extractive value of increased fish abundance and size may influence the economic viability of MPAs.

If Nassau grouper stocks are at increasing risk, will the current TCI system of MPAs protect them? Chapter 10 reports on the results of an ecological survey of the Admiral Cockburn Land and Sea National Park. The effects of fishing pressure and habitat on biomass and catch per unit effort (CPUE) of three species of exploited reef fish were studied at South Caicos. Distribution and abundance of hogfish (*Lachnolaimus maximus*) and white margate (*Haemulon album*) were inversely correlated with cover of fleshy macroalgae. Nassau grouper were positively associated with vertical relief, but were unaffected by algal cover. Mean size, density, and biomass of hogfish were higher in a small (4 km^2) marine reserve than on fished reefs, as was biomass of white margate. CPUE of hogfish was inversely related to distance from the center of the reserve, suggesting that spillover of this species from the reserve to adjacent reefs may enhance local yields, possibly providing economic incentives for fishers to comply with reserve regulations. Fishing pressure, however, had no apparent effect on Nassau grouper. Larger fishes and those that migrate to spawn, such as economically valuable Nassau grouper, may move over too large a range to be effectively protected by small marine reserves. Small reserves may not protect all fish, but we conclude that they can increase the biomass of smaller or more sedentary reef fishes and may be a useful tool for the conservation or management of species such as hogfish. Other policy options, such as seasonal spawning closures or total allowable catches, need to be considered for larger, more mobile fishes in the Turks and Caicos Islands.

Nassau grouper have been reduced to a fraction of their previous abundance over much of their range. In the TCI, Nassau grouper stocks are relatively healthy but need to be insulated from growing fishing pressure caused by tourism-driven restaurant demand. Chapter 11 uses the Institutional Analysis and Development framework to organize a comparative assessment of policy options for Nassau grouper conservation in the TCI. Results suggest that a commercial trade ban for tourist-oriented restaurants offers the best chance to protect Nassau grouper as it has the highest probability of (1) being successfully implemented and (2) of being efficiently sustained over time. Fisheries managers may often neglect to consider consumer-oriented trade restrictions as part of their toolkit for ecosystem-based fisheries management but this case demonstrates that there are situations where trade restrictions may be the most promising option for protecting vulnerable species from market-driven fishing pressure.

Finally, in Chapter 12 I summarize my findings from the TCI case study, examine the broader implications of the case study for MPA research, and outline the significance of developing an institutional framework for designing, monitoring and assessing ecosystem-based fisheries management policy experiments.

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PART 1 – THEORY AND METHODS FOR THE INSTITUTIONAL ANALYSIS OF ECOSYSTEM-BASED FISHERIES MANAGEMENT

CHAPTER 2

ECONOMIC EVALUATION OF RECREATIONAL FISHERY POLICIES[1]

Marine and freshwater ecosystems around the world provide humans with important recreational fisheries that generate revenue and employment, particularly in rural and developing regions. They provide economic benefits to people who derive pleasure from participating in the act of fishing, enjoying the natural environment, viewing marine wildlife, consuming the fish they catch, and/or engaging in social interactions with fishing companions. Economics is the study of how individuals and societies make choices about the use of scarce resources to maximize overall well being. It plays a key role in efforts to evaluate the merits of policies that directly or indirectly impact recreational fisheries throughout the world. Using economics, the costs and benefits that accrue to recreational fishers and the suppliers that provide recreational fishing equipment and services can be compared under various policy options.

The healthier an ecosystem, the more that ecosystem can provide a ‘flow’ of fish that yields fishers increased satisfaction and well being. Ecosystem services typically have public good characteristics that provide benefits to those who use the resource but do not bear the full cost of providing healthy ecosystems. The private sector generally does not produce enough public goods from an economic efficiency, or social well being, standpoint, because it is not in any economic agent’s best interest to supply such goods. Individuals have an incentive to ‘free-ride’ by letting others bear the costs of providing the public good. In response to shortfalls in the production of public goods and conflicts between various interest groups, governments can respond with policy initiatives that prescribe or influence private behavior by setting rules that prohibit, require, or permit specified actions designed to increase the supply of public goods.

Several types of policies have been implemented to improve recreational fisheries. One consists of putting caps on commercial fishing in order to improve the quantity and quality of stocks available for recreational fishing. Another is aimed at improving the ecological conditions for the recreational fish stocks, such as improving water quality. Both types of policies imply benefits and costs for both recreational fishers and others. In

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the former case, the costs may include income reductions and employment losses in commercial fisheries. In the latter case, the costs relate to such issues as water purification. A prerequisite for implementing a given policy should be that its benefits outweigh its costs. One of the methodologies that have been developed in economics to assess policies is cost-benefit analysis (CBA).

Predicting the impacts of alternative policies on overall well being requires that policy makers consider the incentives individuals and organizations face, and how policy initiatives affect the costs and benefits associated with alternative behaviors. Economic considerations play an important role in policy design, with CBA routinely used to account for the economic costs and benefits of particular projects or policy options (Johansson, 1993; van Kooten and Bulte, 2000). Beyond fulfilling an accounting function, economic analysis can play an additional role in helping decision-makers understand how incentives affect different agents and lead to political mobilization and lobbying.

The general principle of CBA is that a policy is worthwhile and should be pursued if the discounted benefits of a particular policy exceed its discounted costs. Implementing this broad principle is problematic because governments, like individuals, face budget constraints. Priority should be given to projects and programs that contribute more to social welfare, thus enhancing overall economic efficiency. Further, market prices do not always reflect economic values where natural resources are concerned; prices often underestimate a resource's true value or fail entirely to take the natural amenity into account. This leads to the misallocation of resources. This is the case, for example, with the systematic over-use of a marine ecosystem for commercial fishing in regions with ecosystems that are important for recreational fisheries.

The main objective of this chapter is to outline the basic principles of CBA for applications related to recreational fishery policy analysis, using examples from various jurisdictions. When considering the demand side of the economic analysis, we emphasize the need to consider nonmarket values in CBA. For both producers and consumers, it is important to differentiate between financial and economic analyses: our basic contention is that narrow financial assessments do not give an accurate accounting of the real economic benefits of recreational fishing, nor the true costs of providing the public ecosystem services needed to support recreational fisheries. There is a pressing need for economic concepts to be applied in environmental policy analyses if countries around the world are to reap the rewards of developing or maintaining vibrant recreational fisheries.

Economic Policy Analysis

Changes in the economic well being of producers and consumers are the focus of economic analysis. Economic well being (welfare) includes financial measures but is more encompassing. For example, it includes nonmarket benefits related to the camaraderie experienced while participating in recreational fishing, plus benefits from knowing that an ecosystem is protected (existence value). Measuring the nonmarket values of recreational fishing can be difficult, but this does not mean that such benefits should be ignored in policy analyses as these values can be substantial. In Scandinavia, for instance, Kristrom and Johansson (2001) estimated that the recreational benefits from recreational salmon fishing outweigh the net benefits from commercial salmon fishing at sea. Taking account of all components of economic value can have important policy implications for recreational fisheries and the ecological systems that support them.

Economic efficiency refers to the maximization of social welfare, which is the sum of the “surpluses” accruing to producers and consumers. A surplus is simply the difference between total benefits and total costs, appropriately discounted to take into account the fact that surpluses accrue at different time periods. Producer surplus is simply given by the area above the supply curve and below price. Price multiplied by quantity gives total return, while the area under the supply (or marginal cost) curve for that quantity gives total (variable) cost. The difference is the producer surplus or quasi-rent (see van Kooten and Bulte, 2000).

Likewise, the area under the demand (or marginal benefit) curve represents the total benefits to consumers of consuming a given quantity, while price multiplied by quantity purchased represents total cost or expenditure. The difference is referred to as consumer surplus. Consumer surplus can readily be calculated for market goods when data are available (*i.e*, the price of the good, the prices of substitutes, consumer income levels). This is not true, however, for amenity services such as outdoor recreation and ecosystem functions that are nonmarket in nature and not traded in established markets. Consumer surplus for recreational services can be estimated in situations where the recreational activity impacts a market transaction. There are travel cost techniques, for example, that enable the analyst to estimate a demand curve for some recreational activities (*e.g.*, hunting, fishing, camping), and thus consumer surplus. The hedonic pricing method does the same, but there are difficult statistical problems to overcome (see Freeman, 1993).

Willingness to pay (WTP) for increases in the availability of an amenity or willingness to accept (WTA) compensation to forgo an increase (or decrease) in the availability of a natural resource are more appropriate concepts, compared to consumer surplus, when addressing consumer well being arising from certain nonmarket amenities. The theoretical foundation for these concepts is illustrated with the aid of Figure 2-1. Suppose an individual has total income M which is allocated to the purchase of market goods and services q_i ($i = 1, \ldots, n$) at prices p_i ($i = 1, \ldots, n$). Amount $q(0)$ of the public good (the environmental amenity) is initially available, so the individual is located on her indifference curve, $V(0)$, at the point where she is consuming M amount of market goods and $q(0)$ of the environmental amenity. Suppose a policy proposes to increase the availability of the environmental amenity from $q(0)$ to $q(1)$ in Figure 2-1.

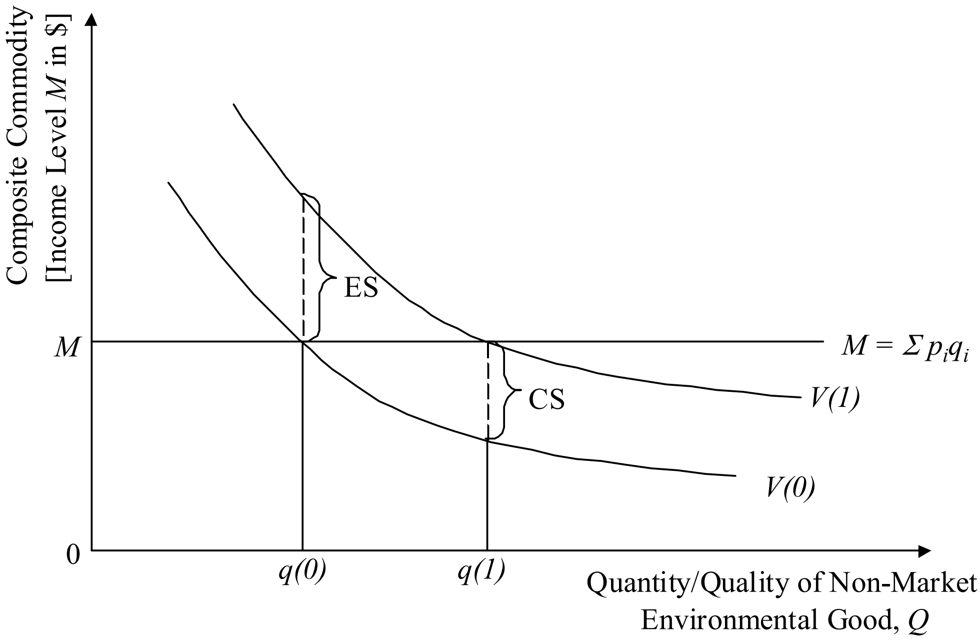

Figure 2-1 – Consumer welfare measures, compensating surplus (CS) and equivalent surplus (ES), for a nonmarket environmental good, Q. Consumption of the nonmarket good is determined by consumer income, M, the prices (p_i) of market commodities q_i, the indifference curves indicating trade-off's for market, $V(0)$, and nonmarket, $V(1)$, consumption, and property rights.

There are two candidates for the measure of consumer well being, depending on the presumed property rights. The compensating surplus (CS) assumes the 'consumer' of the environmental amenity has the right only to $V(0)$ – that is, $q(0)$. An increase in Q to $q(1)$ will enable the consumer to reach a higher level of utility on indifference curve $V(1)$. The CS is the change in income required to restore the original level of utility, $V(0)$, but with $q(1)$ rather than $q(0)$ (see Figure 2-1). It is the maximum amount the person is WTP for the opportunity to face $q(1)$ rather than $q(0)$.

The equivalent surplus (ES) assumes the consumer has a right to *q*(*1*) in Figure 2-1. Then ES is the minimum income that would have to be given to the consumer for her to forgo the increase in the public good – the WTA compensation. In theory, WTA should be approximately equal to WTP, but in practice WTA has been shown to be substantially larger than WTP (Mansfield, 1999). Of course, this has implications for CBA – benefits of a policy to enhance environmental values are higher if WTA is used rather than WTP.

Cost-benefit analysis requires that all costs and benefits be accounted for in each period over a relevant time horizon. Future costs and benefits are discounted a rate r to reflect time preferences for income (*e.g.,* $100 received 5 years in the future is worth less than $100 received today; the higher the interest rate, r, the less the current worth of $100 received in the future). The costs and benefits used in CBA are marginal values in the sense that a policy change does not have economy-wide price effects – that the project is small relative to the overall economy. If this is not true, more complicated general equilibrium analysis is required. Further, when distortions exist in other markets, adjustments should be made to costs and benefits so that they reflect true economic value (see van Kooten and Bulte, 2000 for a further discussion of 'shadow prices'). The streams of costs (C_t) and benefits (B_t), discounted at rate r over T periods, can be summed, respectively, into the present value of costs (PVC) and benefits (PVB):

$$PVC = C_0 + \frac{C_1}{(1+r)} + \frac{C_2}{(1+r)^2} + \ldots + \frac{C_T}{(1+r)^T} = \sum_{t=0}^{T} \frac{C_t}{(1+r)^t}$$

$$PVB = B_0 + \frac{B_1}{(1+r)} + \frac{B_2}{(1+r)^2} + \ldots + \frac{B_T}{(1+r)^T} = \sum_{t=0}^{T} \frac{B_t}{(1+r)^t}$$

To address government budget constraints (not every project with PVB > PVC can be implemented) and achieve maximum economic efficiency, it is useful to rank projects by their benefit-cost ratios:

$$\frac{B}{C} = \frac{PVB}{PVC}$$

Projects ranked by their benefit-cost ratios are chosen in order until the budget is exhausted. There are a number of philosophical and pragmatic difficulties that arise when using CBA, but these are beyond the scope of this chapter. For general introductions or reviews of key issues, see, for example, Arrow *et al. (*1993), Freeman (1993), Hausman (1993), Hausman and McPherson (1996), Foster (1997), Dixon and Pagiola (1998), Pearce (1998), Weimer and Vining (1998), and van Kooten and Bulte (2000). Suffice to say that there are difficulties in (1) equating personal satisfaction with welfare and income, (2) putting economic values on public goods such as ecosystem services, (3) choosing an appropriate discount rate, and (4) addressing income redistributional effects of policies where there are identifiable losers in the policy implementation process. The last point is problematic because the generic goal in economics is Pareto efficiency – no one person can be made better off without making at least one person worse off. Pragmatically this is usually impossible, so the Kaldor-Hicks compensation principle is generally invoked: if the 'winners' from a policy change gain more than the 'losers' lose, they could compensate the losers and still be better off. Society as a whole is left better off but, because compensation rarely occurs, some individuals are left worse off than they were prior to the policy (Hausman and MacPherson, 1996).

Economic efficiency is only one criterion by which a policy or project can be judged. Other possible criteria include equity (redistribution of income according to social goals), fiscal equivalence (the beneficiaries of policies are the ones who bear the costs), administrative efficiency and accountability (of elected officials and bureaucrats), conformance to general norms and social values, and the adaptability (to changing social or ecological conditions) of institutions themselves (*e.g.*, Ostrom *et al.,* 1993; Weimer and Vining, 1998). But, as Pearce (1998) points out, alternative decision rules appear to suffer as many, if not more, shortcomings as those faced by CBA.

Applying Economic Concepts to Recreational Fisheries Policy Analysis

Standard CBA calculates the economic costs and benefits of proposed policy alternatives for producers and consumers. Using recreational fisheries examples, we briefly illustrate the concepts that arise in undertaking research on supply-side producer surplus benefits and demand-side consumer surplus benefits. Recall that the value of a nonmarket amenity to a consumer depends on property rights and on a policy intervention to change the level of the environmental amenity being supplied. In some cases, the cost of a policy intervention can be calculated in a straightforward manner, but in other cases the costs of implementing policies that achieve the same ends can vary greatly, depending on the type of government or market organization providing the amenity. All three components –

consumer benefits, producer benefits and the costs of policy implementation – should be accounted for in economic policy analyses if social well being is to be maximized.

The Economic Value of Recreational Fisheries to Producers

Although producer surplus is theoretically easier to measure than consumer surplus, there has been very little economic research on recreational fishery service providers. Almost all work done in this field constitutes financial analyses of overall expenditures on fishing services – calculation of the price times quantity for services such as charter trips, boat rentals, accommodation, fishing supplies, and so on. Expenditures are not a surplus, and thus are not a benefit measure. Therefore, unless industry costs are quantified for specific recreational fisheries, estimates of producer surplus are not available.

Some research in the USA has gone beyond simple studies of expenditures and has assessed the operating costs of charter operators. An ongoing study of Lake Erie sport charter captains in Ohio (Lichtkoppler and Hushak, 2001) examines average operating costs, cash flow and net profitability for walleye (*Stizostedion vitreum,* Percidae) charter fishing. The researchers do not explicitly calculate producer surplus but the information they collect could be used for that purpose. They found that many charter captains, particularly those that conducted less than 41 trips per year, encountered financial difficulties. This implies low producer surplus in the sector because many charter operators are likely just covering variable costs, if at all.

Another effort is currently underway by the US National Marine Fisheries Service and the Pacific States Marine Fisheries Commission (PSMFC) to remedy some of the shortcomings in the analysis of charter boat economics. New research (D. Colpo, PSFMC, personal communication) is being implemented that will survey 500 to 600 active charter boat operators on the US West Coast. Data about randomly selected day trips and annual costs will be collected from the business owners; this information can then be used to develop further insights about the financial impacts of the charter industry and economic estimates of producer surplus based on industry operating costs.

The Economic Value of Recreational Fisheries to Consumers

Any particular fishing trip can be viewed as a bundle of physical and service attributes (*e.g.*, target species size and abundance, experience of the guide, type of boat used, number of other people on the trip, accommodations, price of trip). Each attribute may have an influence on the value that a person holds for the fishing trip, but in different ways because individuals have different preferences and incomes. Individuals, including those who do not engage directly in recreational fishing activities, may also derive well being and economic value from policies that directly or indirectly benefit recreational fisheries. For instance, fishers and non-fishers alike might experience an increase in well being if they know policy is adopted that preserves essential fish habitat needed to protect an endangered species. The main variants of economic value that ideally should be considered in CBA include (see Dixon and Pagiola, 1998 for an overview):

- Direct extractive use value (*e.g.*, fish taken as food or trophies by recreational fishers);
- Direct non-extractive use value (*e.g.*, non-lethal catch-and-release, wildlife viewing)
- Indirect use value (*e.g.*, the value provided by preserving key predatory fish that maintain overall ecosystem balance);
- Option and quasi-option value (*i.e*, the value of future direct and indirect use value and information);
- Bequest non-use value (*i.e*, the value derived from knowing that future generations will be able to fish even if the person holding this value does not fish themselves); and
- Existence non-use value (*i.e*, the value that people derive from knowing fish exist even if they have no plans to fish themselves).

Valuing Extractive Direct Use for Recreational Fisheries

There is a substantial market demand for fish as food in some recreational fisheries. In 1999, for example, marine recreational fishers in the United States kept almost 135 million fish, about 41% of the 329 million pounds (90,000 tonnes) landed in total (NMFS, 2000). For some high-quality species, such as dolphin (*Coryphaena hippurus,* Coryphaenidae), as much as 90% of the fish landed (1.85 million pieces) was kept by anglers, and total recreational landings exceeded those from the commercial fleet.

In Florida, anglers on multi-day trips with fishing charters where dolphin were popular each paid an average of US$246 per day for the charter and $96 per day for travel and lodging (NMFS, 2000). Up to 11% of Florida fishers on multi-day trips also take some time off work, which costs them, on average, another $195. The opportunity cost of fishing is calculated as the foregone or shadow wage anglers could have earned if they stayed at work; in recreational economics, opportunity cost is sometimes charged at half the wage rate in recognition of a less than one-to-one correspondence between recreational and work opportunities (Freeman, 1993, pp. 451-52).

Clearly, dolphins landed in the sport fishery are not 'cheap' fish. The average price of frozen dolphin fillets imported into the USA during 2000, by contrast, was about $4.35 per kg (or about $2.50 per kg for whole fish, allowing for processing costs and waste). The difference between the sport and commercial fishery illustrates that other non-extractive values strongly influence the value of dolphin to sport fishers, and that a fish landed by a recreational fisher is often 'more valuable' than the same fish caught in the commercial fishery (see also Kristom and Johansson, 2001). This example is based on a financial analysis of expenditures, however, and does not give an accurate estimate of economic welfare. Expenditures by anglers for hotels, restaurants, charter operators and so on provide revenue to firms supplying the recreational fishing industry; the actual surplus accruing to these suppliers depends on their cost structures and the opportunity cost of labor.

To expand on the contrast between expenditures and economic value, consider a study of the Costa Rican billfish recreational fishery during the 1993-94 season (Ditton and Grimes, 1995). Total direct expenditures by foreign fishers in Costa Rica were calculated at $3,446 per trip (average length of 7-days with 4-days fishing). A total of 15,970 billfish were landed during an estimated 5,219 charter trips (approximately 97% of these fish were released live and could be caught again). The financial impact on the national economy was almost $17.8 million as a result of the recreational fishery. This value, however, captures neither consumer nor producer welfare. Fishers were willing to pay even higher than market prices for the high-quality fishing experience in Costa Rica: a survey indicated that anglers had a consumer surplus of $1,777 per trip, or total consumer surplus of $3.99 million. This consumer surplus would need to be added to earnings over and above variable costs (*i.e*, producer surplus or quasi-rent) to get total benefits. The sum of consumer and producer surplus (or total welfare) would likely be only slightly above $4.0 million because producer surplus is probably small. Much of the $17.7 million in revenue for suppliers would likely be used to cover the variable costs of business operations in the competitive tourism and charter sectors in Costa Rica.

Valuing Non-Extractive Direct Use for Recreational Fisheries

How do we put an economic value on the increased well being that fishers derive from the fishing experience? There are several ways to infer this value: observing changes in the value of property rights; examination of actual behavior (often travel cost expenditures) when fishing sites vary by some key attribute(s); and conducting surveys of fishers' stated preferences – that is, elicitation of WTP or WTA directly. The valuation of non-use goods and services (*i.e*, bequest and existence values) can also be approached using stated preference methods (see Freeman, 1993 and Shechter, 2000 for general overviews of nonmarket valuation methodologies).

Where there are private property rights to provide recreational services, the value of the property right should reflect the sum of discounted future net economic benefits accruing to the owner of the property right. In Australia, for example, there is a licensing system for tourism operators in the Great Barrier Reef Marine Park (Davis and Gartside, 2001). The operators pay a nominal license fee for a 6-year permit to use the park, but the number of licenses issued is limited and the licenses are transferable among operators. Thus, the license has become a valuable property right that is reportedly worth up to A $100,000 in prime tourism areas. Like commercial fishing quota, changes in the anticipated revenue stream or interest rates affect the value of the license. Likewise, environmental degradation within the park would be expected to affect adversely the trading price for the license. The magnitude of the change in property right value is an indicator of the economic value of environmental quality.

A more common approach to evaluate market demand for recreational fisheries is based on the travel cost method (TCM). The TCM uses surveys to elicit revealed preferences of fishers and estimate the value of recreational experiences not priced in markets. Essentially, the number of visits to a particular fishing site serves as a proxy for quantity, and costs (travel and opportunity cost of time) as a proxy for price, in the statistical analysis. Economic benefits are then calculated as an area under the appropriate estimated demand function. For example, Layman *et al. (*1996) estimated that consumer surplus for salmon fishing in the Gulkana River, Alaska was between $15 and $104 per person per day using TCM in combination with other valuation methodologies. While TCM is commonly used to value outdoor recreation, there are difficulties in apportioning costs and benefits for multiple-purpose holiday fishing trips, and in valuing the opportunity cost of fishers' time.

Alternatively, fishers can be queried regarding their 'stated preferences' and asked directly about their willingness to pay for certain attributes; this approach is usually employed to value preservation and other attributes that leave no trace in markets. The contingent valuation method (CVM) is used to elicit WTP or WTA compensation for changes in the availability of an amenity (Hanemann, 1984). An alternative is the choice experiment (Adamowicz *et al.,* 1998), which seeks to value different attributes of the fishing experience by querying people directly about preferences using surveys. Choice experiments are a variant on conjoint analysis, which was developed for marketing research. Whereas CVM asks respondents whether they are willing to pay a fee to improve environmental quality, conjoint surveys ask respondents to make a choice between, or rate their relative preference for, two different product profiles. Price is one of the attributes included in each profile; this provides the basis by which to assess WTP.

CVM has been used extensively to value environmental goods and services, including increases in the populations of fish stocks and environmental quality important for recreational fisheries. For example, CVM has recently been used to evaluate recreational fishers' WTP for five popular fishes, snapper (*Pagrus autratus,* Sparidae), kingfish (*Seriola lalandi,* Carangidae), kahawai (*Arripis trutta,* Arripidae), blue cod (*Parapercis colias,* Odacidae) and rock lobster (*Jasus edwardsii* and *Jasus verreauxi,* Palinuridae). These fish have both consumptive use and non-consumptive use values, although snapper and blue cod are primarily valued as food fish and kingfish are valued most highly as sport fish. Table 2.1 summarizes the catch and value of the five species based on a survey by the South Australian Centre for Economic Studies (Williamson, 2000). Marginal WTP for kingfish, kahawai and rock lobster are based on fish caught (fishers indicated that the species were just as valuable for sport purposes as for eating), while figures for snapper and blue cod were based on fish kept (their primary value was for eating – marginal WTP for a fish released to the wild was assumed to be zero).

Species	**Number Caught (thousands)**	**Biomass Caught (tonnes)**	**Total Value (US million)**	**Average WTP (US $/fish)**	**Marginal WTP (US $/fish)**
Blue Cod	1,200	1,518	13.80	11.50	0.76
Kahawai	1,100	729	30.53	27.75	1.62
Snapper	4,300	3,229	62.35	14.50	2.70
Rock Lobster	534	313	12.12	22.70	3.07
Kingfish	74	382	6.30	85.12	9.29

Table 2-1 – Recreational landings and fishers' willingness to pay (WTP in US $) for catching five species of recreation fish in New Zealand (from Williamson, 2000). Marginal WTP is the value of an additional animal caught (kahawai, lobster and kingfish) or kept (blue cod and snapper).

The use of various conjoint methods to value environmental quality is a more recent development in environmental economics and applications to recreational fisheries are limited to date. A study in New England by Roe *et al.* (1996) estimated the compensating surplus for a variety of Atlantic salmon management alternatives. They found that recreational salmon license holders in Maine were willing to pay between $30 and $178 per day for policy initiatives that improved fishing quality by increasing run size from approximately 3,000 fish to either 6,000 or 10,000 fish. In the Turks and Caicos Islands, paired-comparison conjoint analysis was used to assess the impact of the abundance of spiny lobster, sea turtles and reef sharks on the willingness of dive tourists to pay for dive charters (Rudd, 2001). Market share for dive charters increased significantly in simulations when divers observed more of all three types of macrofauna, demonstrating that divers held non-extractive economic value for viewing marine wildlife.

Farber and Griner (2000) used conjoint analysis in conjunction with a random utility model to estimate welfare for stream quality improvements near Pittsburgh. They found that households were willing to pay between $44 and $122 per annum for a variety of stream improvements that varied according to baseline and post-recovery conditions, and that the multi-attribute context permitted the joint valuation of two substitute goods (environmental improvements on two streams).

CVM studies use the stated preferences of survey respondents to estimate measures of consumer surplus. They can be applied creatively and used to value public goods that are not traded in the marketplace. CVM surveys are not without their difficulties however and must be applied using guidelines (Hausman, 1993; Arrow *et al.*, 1993). While the valuation results are context-specific and there are challenges in applying the values derived in one situation to another (a practice known as 'benefits transfer'), stated preference methodologies offer the only real hope for valuing non-use values.

Potential Impacts of Externalities on Recreational Fisheries

Externalities involving recreational fishing species are common and will distort the true value of recreational fishing in society. These can arise when upland activities harm recreational fisheries (*e.g.*, siltation from logging decreases the survival of salmon eggs in Pacific Northwest streams), commercial fishing reduces available stocks of recreational fish, or when market prices are distorted due to subsidies or other government policies (*e.g.*, subsidizing agricultural fertilizers results in excessive nutrient runoff and eutrophication of lakes used for recreational fisheries). Consider the case of recreational billfish angling in Costa Rica. Fishers are concerned about the effects of domestic longlining, a commercial fishery with revenues under US$1 million, on billfish stock abundance (Ditton and Grimes, 1995). Over 80% of visiting anglers surveyed indicated that they would go to other countries if billfish stocks declined due to overfishing. The loss of surplus in the Costa Rican tourism industry might only partially be compensated for by relatively minor gains in producer surplus in the commercial longlining sector. In economic terms, the commercial longlining fishery may impose external costs on the recreational fishery, making it more expensive to find less abundant fish and/or reducing market demand for the Costa Rican fishing experience by making it less attractive for anglers, compared to countries with healthier stocks.

The effect of externality can be illustrated with the aid of Figure 2-2. Let S_{com}^{1} in Figure 2-2(a) represent the private marginal cost (supply) of fishing by the commercial sector if long-run sustainability of the stock is ignored (or discounted at a higher than socially optimal rate). Then S_{com}^{0} represents the true costs to society. Assume a constant world price, $p_{com}*$, for Costa Rican billfish because the small production has no impact on overall world prices. By externalizing the effects of commercial fishing on stock sustainability, producers gain area $[q_{com}^{1} - q_{com}^{0}] \times p_{com}*$ in revenue, but only area *abcd* in producer surplus. Thus, the economic benefit to the commercial longline fishery depends on the shape of the supply curve and will always be less that the increase in revenue so long as the marginal cost of catching more fish is positive.

Because the commercial fishery operates at S_{com}^{1} rather than S_{com}^{0}, there are increased costs in the sport charter industry (*e.g.*, charter boats must spend more time and fuel to land the same amount of fish as stock's decline). The supply curve in the sport-fishing sector shifts inwards from S_{rec}^{0} in Figure 2-2b to S_{rec}^{1}. The decline in billfish stocks would also cause a drop in demand for Costa Rican recreational fishing, from D_{rec}^{0} to D_{rec}^{1}, as anglers seek out alternative destinations where stocks are more abundant and the fishing experience more rewarding. Expenditures in the sport-fishing sector would fall from $[p_{rec}^{0} \times q_{rec}^{0}]$ to $[p_{rec}^{1} \times q_{rec}^{1}]$. Note that the effect of the changes in supply and demand indicate an unambiguous decline in the angler-days of sport fishing, but the effect on the price of a billfish charter trip (p_{rec}^{1}) is ambiguous, depending on the shape and extent of the shifts in the supply and demand curves. The combined shifts leave area *rsv* as consumer surplus and *rvy* as producer surplus. The net decrease in overall social welfare is given by area [*stwx* + *yuwz* – *uvxw*] = [*stwx* + *yvxz*], which is different from the decline in revenue.

Loss in social well being may be greater or less than the decline in revenue, depending upon the nature of supply and demand in the recreational sector.

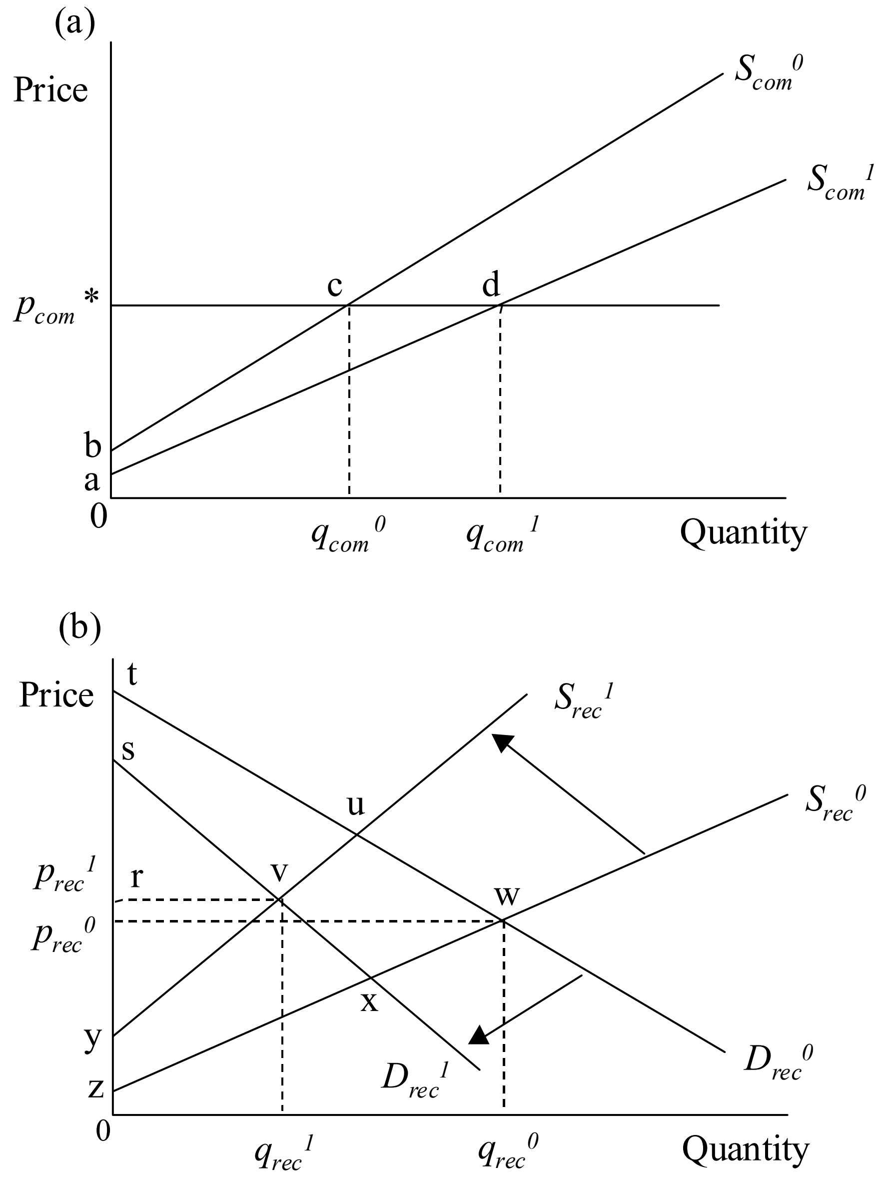

Figure 2-2 – Changes in (a) producer surplus in the commercial billfish longline fishery and (b) producer and consumer surplus in the billfish sport fishery in the face of overfishing in the commercial sector. Commercial fishing expands from q_{com}^0, to q_{com}^1 when supply expands to S_{com}^1 due to an externality. Sport fishery supply (S_{rec}) and demand (D_{re}) contracts to q_{rec}^1. The net change in social welfare is given by *abcd* (the gain to commercial sector) less (*stwx* + *yvxz*) (the loss to the sport fishery sector).

From an economic perspective, there could be an overall gain in aggregate social welfare if the commercial fishing industry bore the full costs of billfish stock depletion (operated along S_{com}^{0} rather than S_{com}^{1} in Fig. 2-2a), with any resulting increase in billfish abundance more than compensated for by the increased well being of recreational fishers and suppliers. If curtailing stock depletion was prohibitively expensive, however, and welfare in the recreational sector was only marginally enhanced by increased abundance of billfish, it could be socially optimal to increase stock depletion. The optimal level of fishing and the allocation between recreational and commercial fishing interests is a question that would require detailed empirical analysis.

Costs, Benefits, and Incentives

It is also useful to examine changes in the well being of various interest groups to develop an understanding of industry and interest group (actor) incentives, how particular policy proposals differentially affect actors, and how policy implementation outcomes might be affected. In Costa Rica, national longliners would be adversely affected by policy initiatives that increased billfish conservation if they already made substantial profits under the status quo. The threat of losing producer surplus could lead to lobbying if this group could mobilize politically, and there could be adverse implications for local politicians as a result. On the other hand, if there was no surplus in the industry now (boats just covered variable costs) and an increase in billfish conservation would only result in greater capitalization and a race to capture the extra fish, there may be few incentives for fishers to organize and oppose conservation policies. Understanding economic well being can shed light on political activities and the chances for successful policy implementation. Likewise, observing the political influence of different actors can provide strong hints at the magnitude of the surpluses those actors might enjoy (*e.g.*, consider the political power and organization of farmers in countries that have marketing boards or agricultural subsidization).

Questions arise about the distributions of expenditures and economic benefits when dealing with recreational fisheries. While the ‘winners’ from conservation (the sport sector) in the hypothetical Costa Rican example may gain more than the ‘losers’ (the longliners) lose, there is no assurance that compensation will be made or that any benefits from the sport fishing sector will flow to commercial fishers. The amount of surplus that is actually captured by Costa Rican interests in the government and the tourist industry may not be high, especially if foreigners own many of the charter boats. Politically, it may be hard to justify domestic policies where the beneficiaries of increases in overall welfare are non-residents and the costs are borne by local citizens. Ditton and Grimes (1995) estimated a local multiplier effect of 2.0 for Costa Rica (*i.e*, every dollar spent by sport fishers generated two dollars of local economic activity), but Ditton and Clark (1994) estimated the billfish recreational fishery multiplier at only 1.07 in Puerto Rico. This suggests that almost all revenue that came into Puerto Rico as a result of billfish sport fishing expenditures leaked out of the economy due to Puerto Rican dependence on foreign imports. In general, only a small portion of tourism revenues remains in developing countries (Gössling, 1999) and, as a result, policy analysts must pay careful attention to the regional welfare effects of policy alternatives on national actors, especially in developing countries and rural coastal areas. This is an area of research where estimates derived by economic analysis may be fruitfully employed in input-output analyses of regional economic impacts (see Hamel *et al.*, 2000 for an example of work proceeding in this direction). Again it is important to note that economic activity is not the same as economic benefits, or surpluses. Only if the shadow price of labor and capital are zero – there is significant unemployment, a common situation in rural areas in many developing countries – can input-output analysis be used to address benefits (van Kooten, 1993).

Transaction Costs and the Provision of Public Goods

In the past, government intervention was advocated in response to market failures arising from the over-exploitation of public goods, as was the case in the fishery. As it became evident that the 'command and control' approach had its limits, economists advocated a move to privatization of some public goods in the face of 'government failure' and an increased reliance on economic incentives to alter human behavior. Now there is evidence that, when dealing with the physical and cultural reality of many renewable resource systems, neither alternative – free markets or government regulation – is robust enough to alone ensure successful management of complex economic and ecological systems (Ostrom, 1990), and that local resource users need to play an important role in the management of public goods. Ostrom's empirical study of robust and durable renewable resource institutions led to the identification of eight 'design principles' that explained the success of institutions in sustaining a resource over time:

- There were clearly defined boundaries for the resource and withdrawal rights;
- There was congruence between appropriation/provision rules and local conditions;

- Collective choice arrangements existed, and in which local resource users participated;
- There were effective monitors of resource use who were accountable to resource users;
- There was a system of graduated sanctions for cheaters;
- There were low-cost, local conflict resolution mechanisms;
- There was recognition by government of rights of resource users to organize; and
- Resource management activities that require regional coordination were nested in a hierarchical fashion.

The transaction costs of producing public goods – protecting ecosystem services necessary for sustaining recreational fisheries – are often ignored in CBA even though the role of fisheries governance is to shape behavior so that management is conducted at minimum cost (Hanna, 1999). Transaction costs need to be considered if overall social welfare is to be maximized. These include the costs of developing initial contracts between parties responsible for the production of the ecosystem services necessary for recreational fisheries (*e.g.*, agreements to zone areas that protect essential fish habitat), the costs of information needed to manage the resource, the costs of monitoring and surveillance of 'cheaters', and the costs of imposing penalties or sanctions. These costs are usually not explicitly taken into consideration as part of a supplier's cost of production and are effectively externalized, with the costs most often borne by society as a whole via costly government management activities. Each management approach – the State, Market and Community – has transaction costs that will vary and depend on culture and social norms, property rights and institutional infrastructure, and the ecological nature of the fishery. The transaction costs of fisheries management can be prohibitive if done poorly and negate any surplus captured by producers and consumers For example, Schwindt *et al.* (2000) showed that the transaction costs of management exceed the total economic benefits of the fishery for the British Columbia commercial salmon fishery. It is now widely accepted that a top-down regulatory approach will be inefficient if the policies of the government do not conform to some degree to the norms of the local resource users (Costanza *et al.*, 1998). Without reasonable congruence between rules and norms, there will be widespread abuse by resource users, and government monitoring and enforcement costs will escalate (Ostrom *et al.*, 1993).

Economic analysis can be used to quantify the transaction costs for different types of resource management policies and regimes. While most data available are for government-managed commercial and recreational fisheries, there are modern examples of community-based and privately managed recreational fisheries. Innovations in property rights for access to recreational fish and fishing grounds are currently being considered and it is likely there will be new forms of market-based management for recreational fisheries in the near future. Individual fishing quota for sport charter vessel operators in Alaska has, for instance, been proposed for the recreational halibut fishery. Under this plan, transferable quota could be exchanged between the commercial and recreational fisheries, allowing increased economic efficiency for the combined fishing sectors (Hamel *et al.*, 2000). The North Pacific Fishery Management Council took a step in this direction in April 2001, voting to award individual catch shares to roughly 1,100 halibut charter boats that operate each summer in Southeast and Southcentral Alaska (Loy, 2001). If approved, the individual quota plan could take effect in two or three years.

Government management of recreational fisheries has encountered varying levels of success around the world. In Florida, for example, licensing of local and visiting anglers has generated substantial government revenue that has been re-allocated specifically towards education, research and enforcement activities in the recreational fishery. Thus, there have been generally high levels of compliance with fishing regulations and a high level of public investment in scientific research. A suite of regulatory and management tools is used in Florida, including seasonal and gear restrictions, bag and size limits, seasonal closures, and marine reserves. Government-managed recreational fisheries should be expected to be most successful in minimizing the transaction costs of management when target species are widely dispersed, requiring regional management (*i.e*, there is congruence between institutional and ecological scale), and where citizens have input into the policy planning and implementation process.

Privately operated recreational fisheries, by contrast, are most likely to minimize transaction costs when the fishery is geographically concentrated and when private property rights can be effectively enforced using existing legal institutions. That is, non-authorized users can be effectively excluded from gaining benefits arising from the use of the recreational property. In this setting, long-term stewardship of the resource is economically sound because there are direct links between private investments in ecosystem quality and the benefits derived from selling access to the private fishing property. Leal (1996) outlines the Scottish example of salmon stream management, where privately held and transferable salmon fishing rights have existed since feudal times and are currently held in watersheds by individuals, fishing clubs, companies and non-governmental organizations. District management boards, created by the property owners within a watershed, are responsible for monitoring, habitat improvement, restocking, and management of salmon stocks within the district. They can charge anglers for the right to fish on inland rivers and gain revenue from net fisheries at the river mouth. To accommodate growth in sport fishing demand, the Atlantic Salmon Conservation Trust (Scotland) Ltd. purchased commercial fishing rights and let them go unused in order to increase the returns to anglers in the valuable inland salmon fishing streams.

The transaction costs for community-based recreational fisheries management are likely lower when relatively local in scale (Ostrom's second principle), but not easily privatized due to difficulties in excluding unauthorized resource users. Community management will result in lower monitoring costs when it is in the economic interests of community members, who are regularly on the water, to engage in monitoring and self-policing (Ostrom, 1990). This tends to occur in situations when community members have a direct economic stake in the fish guiding industry and will be adversely affected by deterioration in stock abundance or habitat quality.

Consider the case of the emerging bonefish (*Albula vulpes,* Albulidae) sport fishing industry in the Turks and Caicos Islands (TCI). The fish migrate around the Caicos Bank, a shallow area of about 3,000 km^2 that is also home to productive commercial queen conch and lobster fisheries. Privatization of the fishery or fishing grounds is not possible because commercial fishers cannot be excluded from their traditional fishing grounds. Government managers have very limited information about bonefish stock size or population dynamics, and limited resources with which they could implement an effective bonefish management plan. Community management would thus seem to be a potentially viable option for the bonefish resource.

While bonefish are only worth about US$2.20 per kg as food in the local market (M. Rudd, personal observation), there are a number of guiding operations catering to foreign fishers who place a much higher value on the fish, which are prized for their fighting ability. Visiting fly fishers typically pay between $350 and $500 per person per day for local guiding services, all of which operate under self-imposed catch and release norms. There are poachers on the Caicos Bank who place gillnets at the mouths of mangrove creeks to snare bonefish. The value of the bonefish resource to the sport industry has created economic incentives for resource protection – some fishing guides regularly cut away any illegal nets that they now find (A. Danylchuk, Center for Marine Resource Studies, personal communication). The bonefish guides are actively participating in tag and release studies designed to improve scientific understanding of stock dynamics and the potential effects of marine protected areas on bonefish conservation. The charter companies, all of which are controlled by TCI 'Belongers', also offer alternative employment opportunities for former commercial fishers in outlying rural islands. While community-based management has not been officially implemented for the TCI bonefish sport fishery, conditions are favorable for bottom-up participation in the management of the local recreational fishery. A challenge for a community-based TCI sport fishery management agency would be to limit the size of the industry, as congestion could rapidly lead to a degradation of the fishing experience (and angler WTP) even if bonefish stocks remain healthy.

Summing Up the Economic Value of Recreational Fisheries for Society

The overall changes in societal well being from any proposed policy that affects recreational fisheries can, in theory, be derived by summing the appropriately discounted costs and benefits. Many narrow CBA studies in the past have only considered extractive direct use values on the demand side and narrowly defined project costs on the supply side. While this may be appropriate in some sectors, recreational fisheries depend upon ecological services that are public goods. As such, they tend to be under-produced and require either government policy intervention or collective action at the local level to increase production to socially optimal levels. This causes three main complications that must be addressed in CBA: non-extractive and non-use values should be accounted for in order to derive all consumer welfare benefits; market distortions that indirectly affect estimates of both producer and consumer surplus in recreational fisheries should be accounted for; and the transaction costs of alternative forms of supplying public goods should be considered.

Recall from Figure 2-1 that people are willing to pay some amount, CS, for an increase in a public good. This surplus needs to be included in CBA calculations. The costs of policy implementation need to be considered as well. Many economists consider this to be a straightforward calculation, but this may not the case for public goods and the analyst must consider the costs for alternative forms of provision. Many policies may achieve a desired result (*i.e*, there can be many sufficient conditions for the policy outcome), but the costs could vary greatly depending on the social, ecological and institutional context. Hybrid forms of governance ('co-management') are possible and likely desirable for many recreational fisheries. A particular challenge in recreational fishery policy analysis will be to evaluate the efficient 'scope of government' (*e.g.*, Hart *et al.*, 1997). That is, what combination of State, Market and Community organizations that can minimize the overall costs of producing and allocating quasi-public ecosystem services necessary for the health of recreational fisheries?

Conclusions

Economic analysis can be a useful tool for evaluating recreational fisheries policies. It can be used to assess the changes in social welfare resulting from changes in government policy (or from changes in exogenous factors such as market prices or environmental change). In addition, economic analysis can be used effectively to identify key policy actors and provide valuable insights on incentives and the patterns of behavior that are likely to result from specific policy changes. When properly conducted – accounting for market and nonmarket values of consumers, the costs of production of firms, and the transaction costs of management – economic analysis can provide important information that can help policy makers improve economic efficiency and the effective allocation of resources to investments that benefit society as a whole.

When conducting economic analyses of recreational fisheries, it is important to consider the many nonmarket values that arise from non-consumptive use of the fish resources and the ecosystems that sustain them. While non-consumptive use value, indirect use value and non-use values have been considered extensively in other fields of environmental valuation, recreational fisheries seem to be under-studied by comparison. On the supply-side, there is a dire shortage of research on the cost structure of suppliers. As a result of these shortcomings, the true economic benefits to society of recreational fisheries around the world have not been adequately assessed. Given the prominence of recreational fishing as a form of leisure in many parts of the world, there is need for more detailed economic analyses of economically and socially important recreational fisheries. Finally, it is important to consider transaction costs in the economic analysis, an item often neglected in cost-benefit analysis. The costs of maintaining and managing ecosystems needed for recreational fisheries varies greatly over situations that differ in terms of ecology, social context and institutions. Only when all factors are considered will the effects of alternative policies on incentives, efficiency and social welfare be adequately understood. Without this understanding, societies will be hampered in efforts to develop effective policies and governance systems that ensure ecological and social sustainability of recreational fisheries.

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CHAPTER 3

POLICY ANALYSIS FOR TROPICAL MARINE RESERVES: DIRECTIONS AND CHALLENGES[1]

Despite the long-term importance of marine ecological goods and services to humans, it is evident that efforts to govern fisheries have often failed (Botsford *et al., 1*997; NRC, 1999; FAO, 2000). Commercially important species under increasing fishing pressure are being depleted (FAO, 2000), resource rents are dissipated under open access and through subsidization (Milazzo, 1998), and ecosystem structure and function are being compromised (Jennings and Kaiser, 1998; Pauly *et al., 1*998). Nowhere is this more apparent that in tropical inshore fisheries where vulnerable species are often intensely exploited by artisanal fishers who use diverse, and sometimes destructive, fishing methods (Dalzell *et al., 1*996; Munro, 1996; Johannes, 1998).

The reasons for this breakdown in fisheries management are complicated. Marine ecosystems are poorly understood and there are fundamental uncertainties regarding the linkages between fishing and stock depletion (*e.g.* Jennings and Polunin, 1996; Lipcius *et al., 1*997). In addition, social systems in tropical coastal areas can be very complex (Ruddle, 1998; World Bank, 2000) with intense competition over scarce marine resources. Even where the political will exists to manage coastal fisheries, governments and communities often lack the skills and resources necessary for effective management (Roberts, 1997a). The conventional approach of optimizing biological or economic yield of a single species is clearly unsuited for tropical inshore fisheries, where target species are often highly vulnerable, ecological data are incomplete, fish landings undocumented, and/or property rights ill-defined (Roberts and Polunin, 1991; Roberts, 1997a; Johannes, 1998; Coleman *et al., 2*000). This makes economically efficient policy tools such as transferable quotas impossible to implement for most tropical multi-species artisanal fisheries.

Ecosystem-based management (Christensen *et al., 1*996) has emerged as the primary alternative to the conventional fisheries management paradigm (Costanza *et al., 1*998;

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NRC, 1999; Gislason *et al., 2*000), and marine reserves are viewed as a central policy tool for implementing marine ecosystem-based management (Bohnsack, 1993; Roberts and Polunin, 1993; Costanza *et al., 1*998; Murray *et al., 1*999; Halpern, in press). For the purposes of this paper, we define marine reserves as no-take fishing reserves. Marine reserves address the main themes of marine ecosystem-based management by contributing to the maintenance of biodiversity and ecological processes that maintain resilience while enhancing fisheries, increasing opportunities for non-consumptive activities and building knowledge for improving coastal management (Costanza *et al., 1*998; Conover *et al., 2*000; Dayton *et al., 2*000).

Surprisingly, there is a scarcity of rigorous policy analyses of marine reserves (Carr, 2000; Milon, 2000) that demonstrate whether marine reserves lead to sustained socioeconomic benefits. Economic analysis is a well-established field and cost-benefit analysis (CBA) is commonly used to evaluate a wide variety of renewable resource policies and programs (Folmer and Gabel, 2000; van Kooten and Bulte, 2000). The guiding principles of CBA are simple: if the economic benefits (calculated by summing producer and consumer surpluses) of a policy exceed the costs, the policy is worthwhile and should be pursued, and that policies with higher ratios of benefits to costs should be pursued first to improve societal welfare. But to understand fully the economic impacts of marine reserves also requires accounting for transaction costs (*e.g.*, planning and reaching agreements about marine reserve configuration and rules, monitoring, enforcement and *ex post* opportunism such as free riding and rent seeking) that are often ignored in standard economic analyses (Rudd *et al., 2*002). For tropical inshore fisheries, transaction costs may comprise a large proportion of total fishery management costs.

The challenges for policy analysts are twofold. First, from an empirical perspective, evidence quantifying the ecological benefits of marine reserves is relatively sparse because of the *ad hoc* design of many reserves (Alder, 1996; McClanahan, 1999) and the inherent difficulties of ascertaining causal linkages in the marine environment (*e.g.* Jennings and Polunin, 1996; Paddack and Estes, 2000). In addition, many of the ecosystem services that reserves provide are nonmarket in nature (Moberg and Folke, 1999) and difficult to value, especially in developing countries where funds for economic studies are limited. Data on appropriately disaggregated costs of fisheries management and governance are also unavailable in many developing countries.

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challenges and opportunities facing fisheries managers, ecologists and economists collaborating on economic assessments of tropical reef fisheries.

Second, from a theoretical perspective, there is a need to develop theory that causally relates marine ecosystem-based 'best practice' management, which incorporates local ecological knowledge and increases stakeholder stewardship through comanagement of local resources, to socioeconomic outcomes. This requires theoretically sound policy experiments that test and screen ecological and socioeconomic hypotheses (Walters, 1997) and comparative studies that address causal complexity (Ragin, 2000) and its impact on ecological and economic outcomes.

Our paper addresses both issues, although the main focus is on the second, linking tropical ecosystem-based fisheries management and recent developments in transaction cost economics and social capital theory. The role of social capital is particularly important for the development of research hypotheses about the efficiency and effectiveness of marine reserves versus other policy options under varying ecological and social conditions. Our goal is not to provide a guide to conducting marine reserve policy analyses *per se* but rather to facilitate policy analyses by identifying economic and social theory needed to conduct integrated ecological-economic analyses of marine reserves and the challenges in putting the theory to practice.

Ecological Services and the Value of Marine Reserves

How can we know which of many possible policy packages to implement, given multi-faceted – and sometimes conflicting – societal goals? Whatever the ultimate goal, we do know that economic efficiency plays an important role in the evaluation of all policy options because a misallocation of society's resources will reduce overall social well being. If society chooses to put a high priority on marine conservation, implementing a high-cost policy that achieves the same end result as an alternative low-cost policy takes away resources from other social priorities. Each policy option has an opportunity cost: this implies that economic efficiency will always be an important evaluation criterion even when it is not the ultimate policy goal.

CBA involves tallying consumer and producer surpluses. Consumer surplus is the difference between what people are willing to pay and what they actually have to pay for an economic commodity. Likewise, producer surplus is simply the difference between a producer's total revenue and total variable cost or, at the margin, between price and marginal cost (with fixed costs considered sunk). Ecological amenities provided by marine reserves are also economic commodities and have a 'price', based on their marginal use and non-use values. Various types of nonmarket services generate consumer surplus even though they are not traded in established markets. Table 3.1 outlines the main types of economic values that need to be considered in economic analyses of marine reserves (see Dixon and Pagiola, 1998; Cesar, 2000).

Research on the producer surplus of those who use marine reserve services as inputs in their businesses (*e.g.*, artisanal fishers, sport fishing guides, dive charter operators) is complicated in developing countries and has been subject to even less examination than valuation of the flow of marine reserve services to consumers. Cost and earnings surveys would be needed to quantify producer surplus for specific situations. For many fisheries, producer surplus may be near zero as fishing revenue just covers the variable costs of fishing (Rudd *et al., 2*002). Thus, for many tropical artisanal fisheries, the majority of economic surplus is likely to consist of consumer surplus from various nonmarket and nonuse amenities.

Type of Economic Value	**Example**
Use Value	
Extractive Direct Use	Fish captured for food
Non-Extractive Direct Use	Wildlife viewing by divers
Indirect Use	Maintenance of ecosystem resilience
Non-Use Value	
Option Value	Value of conserving reefs for bioprospecting
Quasi-Option Value	Information for future fisheries managers
Bequest Value	Fishing lifestyle is available for children
Existence Value	Icon species preservation value to non-users

Table 3-1 – Types of economic values to consider in marine reserve analyses

Marine reserve costs can be substantial (Milon, 2000) and are largely ‘up-front’ costs (*e.g.*, planning, lost income) incurred before the long-run beneficial impacts of the marine reserve can be realized. By contrast, other policy options such as total allowable catch limits (TACs) that are mandated by government fisheries managers may have relatively low up-front costs but higher long-term costs (*e.g.*, monitoring, enforcement). The choice of discount rate used in an economic analysis will, therefore, have an impact of the economic viability of conservation-oriented marine reserve policies.

In this section, we examine several ecological services that marine reserves might provide, how they can be valued, and some potential costs of marine reserves. We concentrate on three potential services: (1) larger and/or more abundant fauna within marine reserves boundaries; (2) emigration or ‘spillover’ of animals from reserves to adjacent fishing grounds; and (3) increased ecological resilience. Other ecological services are certainly possible (*e.g.*, biodiversity preservation) and might provide a range of economic values outside the scope of the current discussion (see Dixon and Pagiola, 1998; Moberg and Folke, 1999; Cesar, 2000). The three services we focus on have relatively local impacts and are possible – with varying degrees of difficulty – to assess empirically or with computer models. The nonuse economic value of biodiversity (existence value) is potentially large, but has limited policy relevance for most tropical marine reserves until compensation mechanisms are developed to transfer income from global beneficiaries of increased biodiversity to the local citizens who bear the costs of conservation (van Kooten and Bulte, 2000).

Increased Size and Abundance within MPAs

Both commercial and subsistence fisheries tend to be size selective, often targeting large piscivores at the top of the trophic chain (Roberts, 1997a). A reduction in fishing pressure within marine reserves is hypothesized to lead to an increased abundance of fish that are, on average, older and larger than would be the case in the presence of fishing (Polunin and Roberts, 1993; Jennings, 2001). This in turn would lead to greater average fecundity and therefore greater gamete production within marine reserves than within similar areas of unprotected habitat (Roberts and Polunin, 1991). Empirical evidence supporting the hypothesis that marine reserves increase the size and/or abundance of many valuable species within marine reserves is substantial (Roberts *et al., 2*001; Halpern, in press). In general, reserves tend to lead to increases in size and/or abundance for benthic shellfish and carnivorous finfish that are relatively sedentary (Conover *et al., 2*000; Tupper and Rudd, 2002). For fishes lower on the trophic chain, there can be complications because the protection of top-level piscivores might cause decreases in abundance of some prey species (Tupper and Juanes, 1999). The evidence for indirect effects of fishing on community structure is inconclusive (Jennings and Polunin, 1997; Jennings, 2001), although some evidence exists for "second-order effects" where removal of top-level carnivores, such as large groupers, results in a proliferation of smaller species (Watson and Ormond, 1994; Chiappone *et al., 2*000).

Undisturbed marine reserve environments may provide valuable information (quasi-option value) for commercial fishery managers by allowing scientists to better estimate life history parameters (*e.g.*, growth and natural mortality rates) necessary for accurately modeling fisheries and developing appropriate policies in unprotected areas. Given the potentially large economic benefits that could result from better management of capture fisheries, the value of information could be substantial although this issue remains essentially unexplored.

A second economic benefit of having larger or more abundant animals within marine reserves derives from non-extractive use values. It is known that snorkelers and divers often prefer viewing 'big stuff' and 'icon' species (Williams and Polunin, 2000). Research to has demonstrated that divers significantly prefer viewing larger and/or more abundant spiny lobster (Rudd, 2001) and Nassau grouper (Rudd and Tupper, 2002), species that might benefit from marine reserves. Similarly, economists have found that 'flagship' or icon species such as elephants and rhinoceros have value in attracting tourists (see van Kooten and Bulte, 2000).

On the production side of the equation, effects of changes in animal sizes or abundance will likely change the behavior and profitability of recreational service providers and may increase producer surplus (see Rudd *et al., 2*002). Cost and earnings data required to assess industry profitability is usually not available for business sectors using tropical marine reserves but must be collected in the future to better quantify this portion of the economic value of ecological services. Marine reserves may induce changes in visitation, potentially leading to congestion effects (Davis and Tisdell, 1996) where people's willingness to pay for wildlife viewing declines due to the number of other people partaking in the same activity (Rudd and Tupper, 2002).

Spillover from MPAs to Adjacent Commercial Fishing Grounds

As the density of fish within a marine reserve rises due to protection, fish should theoretically 'spillover' reserve boundaries in density-dependent emigration to adjacent areas that are open to fishing, thus increasing yields for fishers in those areas. While the basic intuition may be simple, the habitat carrying capacity must be reached as a result of protection before density-dependent spillover is likely to occur (Lizaso *et al., 2*000). There is also a balance to strike between reserve boundary porosity and protection. When boundaries are too porous, there may be no advantage to protecting an area; conversely, when boundaries are impermeable, spillover to adjacent fisheries will not be possible. Contiguous habitat is most conducive for spillover (Roberts, 2000), but in practice marine reserve design has often been based on geographic features that limit spillover (*e.g.*, Rudd *et al., 2*001).

Alternatively, spillover may occur via random movements, independent of resident fish density, if the home range size of the fish is larger than the area of the marine reserve (Kramer and Chapman, 1999; Tupper and Juanes, 1999). In general, larger fishes have larger home range sizes (Zeller, 1997) and are more likely to cross reserve boundaries, while smaller species tend to have small home ranges and may spend all their time within marine reserve boundaries (Holland *et al., 1*993). However, if the home range of the species is much larger than the reserve, the reserve may provide limited benefits to that species at best (Tupper and Rudd, 2002).

Evidence for spillover is limited (Russ and Alcala, 1996, 1998; McClanahan and Mangi, 2000; Roberts *et al., 2*001; Tupper and Rudd, 2002), although there is anecdotal evidence of fishers concentrating effort near reserve boundaries (Lizaso *et al., 2*000). It is generally unknown whether spillover from marine reserves will be sufficient to offset lost catches from closed fishing grounds. The market price for fish emigrating out of marine reserves is easily documented, although analysts must be cognizant of possible market price distortions in tropical fisheries. For example, tariffs on imported seafood products distort local markets for Nassau grouper in the Turks and Caicos Islands, leading to artificially high prices (Rudd, in press).

As a result of the difficulty in experimentally quantifying spillover, there has been substantial modeling effort directed at this phenomenon (see Rodwell and Roberts, 2000, Sumaila *et al., 2*000). The results are mixed and depend upon assumptions about dispersal, life stage, boundary porosity, fishing mortality, and fishing effort reallocation. Sladek Nowlis and Roberts (1999) found that marine reserves could increase and stabilize landings in adjacent commercial grounds if properly designed. In a model of Hong Kong marine fisheries, Pitcher *et al. (*2000) showed that boundary porosity (and fish migration speed) had a strong influence on the success of marine reserves. Reserves of up to 20% of the fishing ground area could significantly increase returns to commercial fishers. Stockhausen *et al. (*2000) and Sanchirico and Wilen (2001) examine the possibility that marine reserves can provide a 'double dividend', where spawning biomass increases within reserves while harvest increases adjacent to reserves. In both models, the success of marine reserves in providing a double dividend depends on a number of specific model parameters, including boundary porosity.

Several important considerations will influence the producer surplus of fishers utilizing spillover from marine reserves. First, have costs of fishing increased or decreased as a result of the marine reserve? The implementation of a reserve can increase producer costs if fewer fish are available due to access restrictions, if there are higher fuel or labor costs to fish farther afield, or there is increased congestion on the fishing grounds. Alternatively, it might lower costs if there is a steady and reliable outflow of fish from a reserve to adjacent fishing grounds. The economic value of spillover depends not only on biological factors, but as much or more so on the behavior of fishers and the cost of fishing, neither of which is adequately considered in many biological models that incorporate *ad hoc* fisher behavior (Wilen *et al., 2*002). Further, in density-dependent systems, the cost of fishing may influence whether a single marine reserve is a 'source' or 'sink' relative to adjacent fishing areas (Sanchirico and Wilen, 2001). High intrinsic growth rates in areas with abundant essential fish habitat may also make it more expensive to close a productive fishing ground due to higher opportunity costs for fishers.

The models demonstrate that calculating the economic costs of marine reserves is not a simple matter. Recent advances in spatial econometrics and spatially explicit modeling (*e.g.*, Smith 2000; Walters 2000; Folmer *et al., 2*002; Wilen *et al., 2*002) should spur further research in the various approaches to this problem.

Increased Ecological Resilience due to Marine Reserves

Trophic Effects

Fishers usually target commercially valuable piscivores that are high on the trophic chain. As valuable piscivores are depleted, there can be increasing pressure to harvest species lower on the food chain (Jennings and Polunin, 1996; Pauly *et al., 1*998) and to engage in more destructive fishing measures. As fishing pressure increases, herbivores and algae may come to dominate degraded tropical inshore systems (McManus *et al., 2*000), reducing ecosystem resilience and leaving them even more vulnerable to anthropogenic and natural stresses and shocks.

Marine reserves are thought to help preserve resilience by protecting key species and habitat, thus leaving ecosystems with more capacity to resist stresses (Green *et al., 1*999). The field evidence for this assertion is ambiguous, however, due to the complexity of tropical inshore ecosystems. Fine-scale experimental evidence supports the hypothesis that predator removal will impact reef ecosystems (Carr and Hixon, 1995), but broader scale research has shown that tropical systems can be quite resilient because of the complexity of the ecosystem and overlapping niches for many species (Jennings and Polunin, 1997; Russ and Alcala, 1998). Marine reserve modeling, on the other hand, provides support for the hypothesis that marine reserves lead to increased functional resilience within and outside of the reserves (Guénette and Pitcher, 1999; Pitcher *et al., 2*000; Walters, 2000).

Effects of Connectivity

In addition to increasing ecological resilience locally by maintaining ecosystem structure and function, marine reserves may contribute to the maintenance of ecosystem services in downstream 'sinks'. Due to the broadcast nature of spawning for most tropical species, which disperse larvae on ocean currents, there is potential for widespread larval export beyond reserve boundaries (Roberts, 1997b). In theory, more larval dispersion from marine reserve 'sources' should lead to more juveniles and adults in the future in downstream 'sinks', thus helping to maintain species balance and ecosystem resilience. Whether or not this is, in fact, the case depends on assumptions about larval behavior, mortality and post-settlement factors. If larvae suffer high mortality and are widely dispersed, larval supply may only rarely saturate reefs and leave enough recruits that density-dependent growth or mortality occurs (Doherty and Fowler, 1994). As a result, adult year-class strength will depend mainly on larval input. If, on the other hand, most of the mortality occurring in recruits is due to post-settlement processes, larval and adult production may be spatially and/or temporally decoupled (Hixon and Carr, 1997; Lipcius *et al., 1*997; Tupper and Boutilier, 1997).

In addition to local oceanographic conditions that may favor retention of larvae near their natal reef (Sammarco and Andrews, 1998; Jones *et al.,*1999), recent field studies suggest stronger than expected self-recruitment due to the swimming capacities and vertical migrations of larvae (Leis and Carson-Ewert, 1998; Bellwood and Fisher, 2001). Mora and Sale (2002) conclude, however, that the extent to which coral reef fish populations are closed or open is currently unknown, but that open populations are likely in many environments. The extent to which marine reserves can protect self-recruiting populations or enhance downstream fisheries thus will likely be highly site and species specific. Even low levels of larval export should greatly reduce the probability of extinction of some marine species.

Valuing Resilience

Due to the complexities of field studies, core insights into the economic value of resilience for marine systems may come from modeling. Larval export is a very important marine reserve benefit in a number of models (Man *et al., 1*995; Stockhausen *et al., 2*000) but depends strongly on modeling assumptions. Mass-balance models developed using Ecopath (Walters *et al., 1*997; Pitcher *et al., 2*000; Walters, 2000) show promise for transdisciplinary analyses of ecosystem resilience and valuation of future ecosystem production under various degrees of risk and ecosystem robustness. Pitcher *et al. (*2000), for example, used biomass and catch data for 152 species in 7 gear sectors in a quasi-spatial mass-balance model and simulated the effects of a wide range of fishing pressures. They found that implementation of effective marine reserves led to a shift in species composition and a recovery in larger, more valuable demersal fishes at higher trophic levels.

How can the economic consequences of changes in resilience be valued? Marine reserves should be viewed as an insurance policy to protect basic ecosystem services and maintain resilience in the face of declining ecosystem capacity outside of protected areas. This marine service would be akin to that provided by terrestrial protected areas, where a range of studies demonstrate the importance of risk diversification to avoid pitfalls associated with spatially correlated catastrophic events (Gilpin, 1987). Economic value is derived from the ability of a system to maintain the flow of ecological amenities given natural and anthropogenic shocks.

Perrings (1998) describes how the evolution of ecological-economic systems can be modeled as a Markov process in which the resilience of a system in any locally stable domain is measured by the probability of transition to some other state. Different states will have differing capacities for the generation of ecological goods and services. Theoretically, use and nonuse values in the various states can be estimated and the change in expected value calculated for feasible transitions, which may be relatively few in number from step to step. Analyses of this sort have not yet been applied to marine reserves.

Perrings (1998) points out that the matrix of transition probabilities depends on initial conditions (properties of the system in a given state, including number of species), the disturbance regime and the social constructs that prescribe the possible uses for resources and structure feedback that determines how people react to environmental change. Because it is important to reduce the probability of transitions to undesirable states and/or increase the probability of transitions to desirable states, managers of tropical inshore systems must consider factors underpinning the resilience of the system. Valuing ecosystem resilience therefore requires establishing the role of species and processes and the social opportunity costs of those species and processes (Perrings, 1998). Considering transition probabilities as policy targets then encourages the development of institutions that respond to environmental change through adaptive management.

Transaction Costs and Marine Reserve Policy Analysis

Marine reserves are clearly an inferior policy alternative from an economic theory perspective: without concurrent effort controls, marine reserves can lead to rent dissipation if effort merely shifts farther afield or conflicts erupt over access to remaining fishing grounds. Economic arguments for marine reserves hinge, instead, on pragmatic advantages relating to transaction costs – the costs of collecting information, negotiating and reaching agreements regarding the design and implementation of reserves, monitoring compliance, enforcing marine reserves rules, and *ex post* costs due to strategic behavior (*e.g.*, 'free riding' and rent seeking). These factors – especially low monitoring and enforcement costs – are hypothesized to provide marine reserves with their comparative cost advantage relative to traditional management tools (Bohnsack, 1993; Roberts and Polunin, 1993; Murray *et al., 1*999; Roberts, 2000).

In this section we first provide a brief overview of key concepts and terminology needed to address the issue of the transaction costs of marine reserve planning and management. Our focus is on institutions – the social norms and formal rules that shape and constrain opportunistic human behavior – and their impact on marine reserve management and, more broadly, tropical fisheries governance regimes.

We illustrate the utility of taking an institutional focus in the two subsequent sub-sections. First we consider the situation in which the formal governance system can be considered exogenous. The relevant question is: What is the most efficient policy package that can be used to achieve ecological, economic and social objectives in tropical inshore fisheries? The suitability of particular alternatives depends on matching the capacity of the policy package with the fisheries management problem. Policy analysts need to consider if/when marine reserves have a comparative advantage relative to other alternative policy options for fisheries management and conservation.

Next, we consider the situation where the formal governance system itself is endogenous. Here we consider the longer-term economic implications arising due to the governance regimes themselves. There is now widespread consensus that some form of comanagement (Pomeroy, 1995; McCay and Jentoft, 1998) will be needed for most, if not all, fisheries governance regimes. When the governance system is considered endogenous, the real policy question becomes: To what degree should tropical inshore fisheries management be decentralized or devolved from State to Community in order to most efficiently meet societal goals using marine reserves or other policy tools suited to local conditions?

Theoretical Background

Common Pool Resources and Common Property

Maintenance of the flow of ecological goods and services from tropical reef systems pose a classic social dilemma. Society would be better off if more ecosystem services were produced, but it can be in an individual's self-interest to 'free ride' and shirk on investing in ecosystem services and/or engage in unsustainable use of services (*i.e*, the "Tragedy of the Commons"). Reef ecosystem goods and services usually have the characteristics of public goods or common pool resources (Figure 3-1).

Excludability \ Subtractability	Low	High
Difficult	Public Goods	Common Pool Resources
Easy	Toll Goods	Private Goods

Figure 3-1 – A classification of resources based on the ease of exclusion of non-authorized users and the degree of subtractability (after Ostrom *et al.*, 1994).

Public goods are those for which use of the resource by one person or group does not leave others with less available to them (low subtractability) and for which there is difficulty in excluding non-authorized users from harvesting the resource. Increases in ecosystem resilience, for instance, clearly can be characterized as a public good because one person's use of resilience does not diminish opportunities for others to share in the same benefits. If one group invests in measures to increase ecosystem resilience, it is virtually impossible to prevent others from benefiting from the investment. As a result, there is a general tendency for under-investment in public goods (Olsen, 1965).

Common pool resources are those for which the use of the resource by one person or group subtracts from the stock of the resource so that others are left with less available to them (high subtractability) and by the difficulty of excluding non-authorized users from harvesting the resource. Most fish stocks would be characterized as common pool resources as harvesting would leave less fish available for others.

It is important to differentiate between a type of resource (common pool resource) described in terms of physical characteristics and a type of property right (common property) described in terms of access and harvest rules (Ostrom, 2000a). Common property is a type of property right in which a group of owners of a resource has a specified bundle of access and use rights. Common pool resources need not be managed using common property rights regimes; they may be effectively managed using private property rights or by the State. Much of the fisheries economics literature continues to conflate types of resources (public, private, common pool, toll goods), types of property rights regimes (individual, State, common property and open access – a situation where nobody holds property rights), and types of management organizations (firms, communal organizations and government agencies).

Transaction Cost Economics

There has been increasing recognition in the discipline of economics that social context is an important variable affecting economic efficiency. The field of 'transaction cost economics' explicitly considers the attributes of human actors and social embeddedness in research on the economic efficiency of different forms of governance organizations (Williamson, 1999). While many ecologists (and sociologists) may find it surprising that economics has only recently recognized the importance of social context, it is important to understand that the discipline of economics has been built on rational choice theory. In its purest form, rational choice theory assumes individual decision makers have full information, unlimited cognitive capacity and choose to optimize their production or consumption activities given their production cost structure or preferences, respectively. This is relaxed in transaction cost economics; individual decision makers are assumed to exhibit 'bounded rationality', where their calculation skills are bounded by uncertainty, limited cognitive capacity, and spatially explicit local interactions. Thus, models of human decision-making tend to rely on heuristics and learning over time, rather than optimization.

Strategies, Norms and Rules

Under a transaction cost economics approach, individuals exhibit goal-oriented and somewhat opportunistic behavior while going about their everyday lives within complex ecological and socio-economic environments (Ostrom, 1998). Because they have imperfect mental models of the world, people use strategies or heuristics based on prudence and experience to achieve personal goals. A smart person will adopt a behavior because that behavior is rewarded in their particular ecological or social context.

Johannes (1980), for instance, documents how South Pacific reef fish often form annual multi-species spawning aggregations. Fishers can take advantage of the predictable geographical and temporal concentrations of fish, and their unusual docility during spawning, to land large numbers of fish with minimal effort. For an individual fisher, it is a prudent strategy to fish an aggregation because the reward is high for a given level of effort. That effective fishing strategy can, however, be based on an incomplete or incorrect model of the world. Veitayaki (1998) points out that there is a common misconception amongst rural Fijians that fishery resources are not in danger because they have always been sufficient for subsistence in the past. Reef fish replenishment is believed to occur supernaturally, so there is no need for conservation. While a strategy of fishing spawning aggregations may be effective for a few individual fishers, the result can prove disastrous if too much fishing pressure is brought to bear on the aggregation (Coleman *et al., 2*000). The outcome is a social dilemma: all fishers and their community could derive greater long-run economic benefits by cooperating and exercising restraint.

Social norms often evolve to address social dilemmas (Ostrom, 1999). A norm is "a pattern of behavior that is customary, expected and self-enforcing. Everyone conforms, everyone expects others to conform, and everyone has good reason to conform because conforming is in each person's best interest when everyone else plans to conform" (Young, 1996: 105). Norms encourage or discourage certain actions (*i.e*, they alter the incentives for resource users) but do not rely on formal government enforcement mechanisms (North, 1990). In some regions, norms are expressed in terms of local taboos on resource use (see Colding and Folke, 2000).

Three of the most important broad-based human norms include equality, equity and reciprocity (Ostrom, 1998). Under norms of equality, gains from a transaction should be shared equally in the absence of objective differences between individuals even though one party may have the opportunity to take more than their portion (Ostrom, 1999). Norms of equity imply that a greater contribution by an individual to a social exchange should lead to greater returns. Reciprocity norms have an evolutionary basis (Hoffman *et al., 1*998; Ostrom, 1998). An offer to share or assist by one person is accompanied by an expectation for reciprocity in a reasonable time frame. Thus, reciprocity is an investment having a short-term cost for an expected long-term gain. If the 'generous' behavior is not reciprocated, there are implicit or explicit threats of punishment. These norms are present across cultures to varying degrees; they do not require formal enforcement under law but form an important part of the *de facto* ‘rules-of-the-game’ by which fisheries are managed and governed.

Rules are prescriptions about behaviors that are required, permitted or prohibited by society that help solve social dilemmas by altering the expected payoffs for various individual goal-directed behaviors (Ostrom, 1990; Ostrom *et al., 1*994). Formal rules go further than norms, explicitly specifying a sanction that is backed by a formal governance organization (whether at the village level or through an international treaty). Effective rules need monitoring and established sanctioning procedures, as they must provide a credible threat to alter the costs and benefits that individuals perceive for various actions. In extreme cases, individuals may also derive utility from breaking rules under governance regimes that are perceived as illegitimate (Crawford and Ostrom, 1995).

When social norms do not reward constraint or when monitoring and credible threats of sanctions are ineffective, we see ‘paper parks’ where users violate marine reserve access rules with impunity (Alder, 1996; McClanahan, 1999; Mascia, 2000; Rudd *et al., 2*001). Compliance with rules is most probable – and hence the transaction costs of management reduced – when there is congruence between norms and formal rules (Ostrom, 1990; Mascia, 2000). For example, the 1988 Samoan Fisheries Act recognizes local village fishery management by-laws (Zann, 1999) and enhances the capacity of local communities to develop and enforce fishing regulations that are consistent with local norms and applied to both community residents and outsiders. This initiative has resulted in more security over resource use for villages and increased incentives for local conservation. As a consequence, almost 40 villages had chosen to adopt marine reserves as part of their local fisheries management plan by 1999 (King and Fa’asili, 1998).

Institutions

Institutions are simply the interrelated rules and social norms that govern social relationships or, more informally, the ‘rules-of-the-game’ (North, 1990). They are human constructs that craft mutual interdependencies, constrain the choice set that individuals face, and, thus, provide increased predictability for, and minimize conflicts between, individuals engaged in social transactions (Ostrom, 1998). It is important to note that the definition of institutions held across the social sciences differs substantially from the everyday definition of institutions likely to be held by most fishery biologists, ecologists and managers. Institutions are not management organizations: organizations – firms, governments, civil society – are the ‘players’ that develop rules and shape norms in a democratic society so that social dilemmas can be solved in an effective and efficient manner.

Social Capital

The broad complex of social interactions, norms, rules and institutions are known as social capital (Woolcock, 1998, 2001; Rudd, 2000; Ostrom, 2000b; Woolcock and Narayan, 2000). Increasing levels of social capital increase the likelihood of successful coordination and cooperation needed to solve social dilemmas and, hence, reduce transaction costs.

Consider the well-known case of Apo Island, Philippines (Russ and Alcala, 1996, 1998, 1999) where the local community successfully implemented a marine reserve. Fish density and biomass increased within the reserve and fishing improved outside it. The ecological services flowing from the marine reserve resulted in a long-term stream of economic benefits to local residents that they would not have otherwise enjoyed. Without the community norms and social networks that prevented opportunism on Apo, it is virtually certain that all economic rents would have been dissipated under *de facto* open access conditions (*i.e*, no rules). At nearby Sumilon Island, breakdowns in management – caused in part by a lack of trust between the community and outsiders, and in part by local politicians engaging in opportunism – led to depletion of fish stocks and dissipation of resource rents that might have been collected had there been cooperation.

Social relationships themselves can be viewed as assets that contribute to the production of human well being. In a study in rural Tanzania, Narayan and Pritchart (1999) found that the effects of group or association membership of village residents were large, with one standard deviation increase in membership leading to household expenditure increasing by 20-30% (as large as the tripling of education levels). Internationally, Knack and Keefer (1997) examined 29 market economies and found that investment and growth rates were higher in countries where interpersonal trust and norms of civic cooperation were greater. They also found that trust and cooperation are stronger in countries with well-developed legal and institutional infrastructure protecting property and contract rights, and in countries that are relatively homogenous in terms of income level and social standing. More recently, Uphoff and Wijayaratna (2000) documented how social capital helped residents capture gains of up to US $20 million by facilitating water sharing arrangements that led to increased productivity during a 1997 drought in Sri Lanka.

It is useful to consider two forms of social capital (Uphoff and Wijayaratna, 2000; Woolcock, 2001). Structural social capital consists of the rules, procedures, and protocols that make it easier for people to work together to achieve mutually beneficial collective action. Cognitive social capital consists of the norms and values that people hold and which predispose them to cooperate with each other for mutually beneficial collective action. Social capital can function on three levels, as an asset that can be used for 'linking' (Woolcock, 2001), 'bonding' or 'bridging' (Woolcock and Narayan, 2000). Linking results when local residents have increased access to decision makers. For instance, dense kin and social networks in the Turks and Caicos Islands (Bennett *et al., 2*001) allow rural fishers access to senior government decision makers that would otherwise be unavailable in regions with less interconnectedness. Bonding results when strong intracommunity ties give kin and communities a sense of identity and common purpose. Bonding social capital is especially important for the rural poor because it serves as a substitute for the State when citizens are deprived of basic services.

Bridging results when communities endowed with diverse intercommunity ties are in a stronger position to confront problems and take advantage of economic opportunities. In Samoa, the government has worked closely with village councils to develop national legislation that supports local fisheries management (Zann, 1999) and has provided extension officers to assist councils develop local management plans (King and Fa'asili, 1998). The rapid adoption of village management plans and a surprisingly high number of village marine reserves is indicative of bridging social capital, where ideas and information have flowed between villages. All villages that are part of the network may benefit, increasing their capacity for solving local social dilemmas by learning about their marine resources and about how to monitor and enforce their village rules.

Comparative Policy Analysis of Marine Reserves

Different policy instruments may be used to achieve any given social goal. The suitability of particular alternatives depends on matching the capacity of the policy package with the fisheries management problem that is, in part, defined by its social and institutional context (Ostrom, 1990). Current marine reserve management research consists largely of single-case or small-*n* comparative studies (*e.g.*, Fiske, 1992; Polunin and Roberts, 1993; Russ and Alcala, 1999; Chiappone and Sulivan Sealy, 2000; Mascia, 2000). This is typical of common property research on fisheries, forestry, irrigation and agriculture, where case studies play an important role in furthering our understanding of mechanisms that link specific ecological and social systems.

Despite their individual importance, many case studies of renewable resource systems exhibit two potential weaknesses. McCay and Jentoft (1998) and Agrawal (2001) emphasize that some studies have not sufficiently accounted for aspects of the ecological, cultural and institutional context, or power relationships. Such ‘thin’ studies often end up broadly advocating simplistic solutions for renewable resource management. Secondly, awareness of the theoretical relevance of causal variables or systematic tests of causality are lacking in many studies (Agrawal, 2001). Conclusions about the universality of local findings (*i.e*, the effectiveness and efficiency of protected areas) may, therefore, be overstated or overly simplistic. Agrawal (2001) concludes his review of common property management research with a plea that researchers should emphasize “multiple tests of several theories [in] carefully conducted, deeply engaged comparative studies” (p. 1665). Marine reserve analysts should heed this advice.

When conducting a comparative policy analysis, the goal is to ascertain how (1) various policies will differentially impact individual incentives and behavior and (2) how aggregate patterns of behavior arising from various policies impact ecological and socioeconomic well being. While a range of socioeconomic variables may be important for a comparative analyses, two facets of social capital are of particular interest for marine reserve policy analyses: micro-level ‘community capacity’ (*i.e*, the ability of the community to use social networks and norms for mutually beneficial collective action); and macro-level ‘institutional capacity’ of the government (*i.e*, the ability of the government to provide public goods and assure that property rights are honored).

An argument can be made that both community and institutional capacity are needed if marine reserves are to be successful. Without adequate community capacity, the norms that constrain opportunism are absent and widespread cheating is likely to negate any benefits that marine reserves might provide (Fiske, 1992; Russ and Alcala, 1999; Mascia, 2000; Rudd *et al., 2*001). Without institutional capacity, communities that wish to initiate reserves to solve local social dilemmas will be unable to legally exclude outsiders from free-riding on their efforts (Cooke *et al., 2*000).

Community-level social capital and formal State -level institutions can act as substitutes to some extent. When government and community are both strong, social capital can play a bridging role and help community residents achieve higher levels of well being. In tropical developing countries, where formal institutions are often relatively weak, social networks remain important for solving social dilemmas such as inshore fisheries management (Sutherland, 1986; Ruddle, 1998; Bennett *et al., 2*000; Cooke *et al., 2*000; World Bank, 2000). If the State lacks capacity, high levels of local social capital can compensate to some degree, helping communities cope with fisheries self-governance even in the absence of formal state support. If the State is strong but social capital at the community level low, citizens may become disengaged and expect that government fisheries departments will solve fishery management problems because 'that is their job'. When both the government and community have low capacity levels, resource destruction is virtually guaranteed under *de facto* open access.

Social Capital and Fisheries Management – Three Cases

Relatively capable communities and government have co-existed in Belize for several decades (Sutherland, 1986; Mascia, 2000). Fishers have a history of collective action going back to the 1960 formation of the Northern Fishermen Cooperative Society, an organization formed to contest the market power of foreign lobster buyers. Government is relatively capable by Caribbean standards and has been generally supportive of the cooperative movement. Local lobster and conch fishers have been able to collect substantial economic rents, investing in their own processing plants and diversifying their fishing activities.

Coastal Belize is not pristine, but compared with much of the Caribbean is ecologically intact despite export-oriented commercial fisheries. In general, however, Belizean fishers have been successful in maintaining commercially viable valuable conch and lobster stocks and capturing substantial resource rents. Marine reserves have enjoyed strong support from local residents (Mascia, 2000) and the backing of government; as a result, marine reserves such as Hol Chan are held up as an example of the ecological and economic benefits reserves are capable of providing (Polunin and Roberts, 1993; Mascia, 2000).

Contrast this to the situation in the Turks and Caicos Islands. There is a national system of marine reserves in the Turks and Caicos, but there was little community input in the design process (Rudd *et al., 2*001). A government fisheries department manages export-oriented spiny lobster and queen conch fisheries using conventional fishery management models. Although government resources are limited, historical fisheries landing records are extensive and institutional capacity is relatively high by Caribbean standards due to British technical support. Community capacity in the islands is low, however, and there is no evidence of cooperation and self-restraint to solve local social dilemmas. Just the opposite, in fact, appears true (Rudd *et al.,* 2001) and is amply illustrated by "The Big Grab" (the local name for the opening of lobster season). Economic rents are dissipated due to the widespread illegal harvest of undersize, immature lobsters (up to 95% undersize from some popular fishing grounds early in the fishing season). Overall economic returns could be increased greatly if fishers could cooperate and exercise restraint in the early season and spread the lobster harvest out over an extended season.

In Fiji, some strong traditional fisheries management systems are still intact (Veitayaki, 1998; World Bank, 2000) but the Fijian government plays a relatively limited role in the management of inshore reef fisheries in many parts of Fiji due to their limited resources and inter-governmental jurisdictional conflicts (Cooke *et al.,* 2000; Virdin, 2000). Many communities in Fiji are left more or less on their own; even though they possess high levels of social capital, their asset is used for bonding purposes, helping to cope and manage local Customary Fishing Rights Areas (CFRAs) without strong government support. Veitayaki (1998) provides an illustration of how structural and cognitive social capital co-exist in traditional Fijian fishery management and help communities cope when the State is weak:

> "Traditional management arrangements are enforced through traditional authority, which means that there are protocols to be followed. The social structure and close-knit units in Fijian communities demand that people strictly follow tradition and respect each other. Decisions made by the group are often conveyed through the social channels of communication, which ensures that all those involved are made aware of the group's decisions. Consequently, the traditional system of retribution is an effective way of ensuring compliance. Nonconformists are treated harshly, and this is an effective deterrent to others…" (p. 52).

The Votua CFRA provides an example of a community with high management capacity, but limited support from government (Cooke *et al.,* 2000). Three clans share management of the region and are highly motivated to manage the fishery effectively because the CFRA is their only source of revenue. Community-based management and enforcement have helped these clans cope while they function in relative isolation; threats of physical violence may have to be used to ensure local compliance with the CFRA.

Poaching is more common in other parts of Fiji where community management is less aggressive, fishing grounds are close to urban areas or the clan with ownership rights to a CFRA does not live adjacent to the fishing grounds (Veitayaki, 1998). Rural Fijians perceive increased overfishing, poaching and destructive fishing to be major threats to inshore fisheries viability (World Bank, 2000) but some CFRA owners feel powerless to stop the poaching because of the lack of formal institutional backing (Cooke *et al., 2*000). As a result, community members in some parts of Fiji are passive observers of government fishery managers (Veitayaki, 1998) who possess neither the resources nor local knowledge to manage inshore fisheries.

The Importance of Social Capital in Analyses of Marine Reserves

Why does social capital matter for marine reserve implementation and management? We maintain that it matters because conclusions about the suitability of marine reserves as an efficient policy tool will only hold under a certain set of community and institutional conditions. When there is a high level of local social capital and an institutional backstop that provides legally binding sanctions that enable exclusion of non-authorized users, community-based marine reserves may well hold a comparative advantage over other policy tools. While locally managed marine reserves may be a preferred policy tool in Belize, what about the Turks and Caicos where community capacity is low, Fiji where institutional capacity is limited, or other regions where both community and institutional capacity is lacking?

Where there is community apathy, as in the Turks and Caicos, we would hypothesize that a community-based marine reserve management strategy is likely to fail, due to widespread cheating because there are few internal social sanctions against individual opportunism. We might expect to see more efficient policy tools that rely less on local community capacity and more on enforcement by a stronger central authority in the short term. In such an environment, there is an important role for education programs and norm-seeding activities by government and, perhaps more importantly, by civil society organizations. Without first building community capacity and increasing levels of trust to such an extent that mutually beneficial collective action is feasible, investments in marine reserves – that depend on local norms of compliance – are likely to provide lower returns than investments in community capacity building.

When institutional capacity is limited, as in Fiji, local leaders may feel powerless trying to use traditional sanctions on fishers from outside their own community. Hence, we might hypothesize that community-based marine reserves would be successful where the local fishery is relatively isolated from potential poachers or where fisheries income is important to the community, justifying independent monitoring and informal enforcement measures. Policy interventions might best focus on institutional strengthening in such circumstances (*e.g.*, educating decision-makers, coordinating policies between government agencies and increasing the capacity of the court system to uphold rules), or increasing the salience of fisheries conservation and management to the local community.

When both institutional and community capacity are lacking, external governments and environmental NGOs may face a daunting task to even reach a point where marine reserves can be considered as a policy tool for fisheries management and conservation.

Causal Complexity in Comparative Analyses of Marine Reserves

As important as social capital is in the comparative analysis of policy options for fishery management and conservation, it is by no means the only causal variable influencing the potential viability and economic efficiency of marine reserves. As additional causal variables are considered in a comparative analysis, the number of distinct case types rises exponentially. Ragin (2000) points out that case types that vary in only one attribute can be completely different in nature.

The research challenge in comparative policy analysis arises because there will be no simple blanket policy prescriptions for all situations. A policy prescription for a case in one fishing ground with attributes *x x x y x* may be different than for a neighboring area with attributes *x x y x x*. The power of the comparative institutional approach derives from the fact that there may be other fisheries, perhaps in different parts of the world, that share the same set of attributes *x x x y x* with the first fishing ground. If these are considered as a group, a theoretically-based comparative approach has the potential to identify robust kernels of 'truth' about the effectiveness of policy prescriptions for particular case types, independent of the geographical location of the particular case instances (*e.g.*, Hellström, 2001). Broad-based comparative analyses that incorporate explicit links between causal variables chosen on the basis of ecological and social theory, on the one hand, and ecological and economic outcomes on the other hand, are needed for marine reserves.

In summary, we should expect that there are situations when marine reserves are preferred policy tools but that there are also situations where they will not be the most efficient policy option for fisheries conservation and management. Analysts must be willing and able to explore a full range of options and recommend those that most efficiently achieves social objectives. In some cases, the most efficient policy option will be education or other forms of community and/or institutional capacity building that enable marine reserves to eventually be more effectively implemented. This process is a long-term effort and leads us to consider the endogeneity of the system of governance itself.

Economically Efficient Fisheries Governance Regimes

For longer-term marine conservation and management initiatives, analysis at the constitutional choice level becomes important. At the constitutional level, choices are made about whom is entitled to make lower level rules and how the rule-making process itself is governed (Ostrom, 1990). Thus, the structure of governance – the private sector, public sector and civil society organizations and institutions that help a society to steer itself (Hubbard, 2000) – can be viewed as endogenous.

Governance regimes that are based primarily on individual property rights are likely inappropriate for virtually all inshore tropical systems due to the importance of the suite of nonmarket ecological amenities they provide and the difficulty of matching ecological and management scale under private property rights. The importance of traditional community-based property rights in tropical artisanal fisheries is, however, widely recognized (Graham and Idechong, 1998; Ruddle, 1998; Veitayaki, 1998; Zann, 1999; World Bank, 2000). Fishing most often requires consideration of large physical areas suited to communal property rights because of the fluidity of the environment and economies of scale in management. For instance, in Fiji, most CFRAs are managed at the aggregate clan level (*vanua* or *yavusa*) rather than at the sub-clan level (*mataqali*) (Cooke *et al., 2*000), and there is often cooperation where *yavusa* cooperatively manage larger CFRAs (Ruddle, 1995).

Ruddle (1998: 107) notes that "it is commonly asserted, although still largely undemonstrated, that traditional community-based systems of inshore fisheries management offer a modern management alternative [to State regulation]… [and] the devolution of control over natural resources would likely reduce the social, political, legal, conservation-related and management cost problems to be addressed by central or provincial governments". That is, there may be broad transaction cost advantages for community-oriented comanagement of tropical inshore systems.

The Proper Scope of Governance

A major research focus within transaction cost economics addresses questions of the ‘proper scope of government’ (Picot and Wolff, 1995; Hart *et al., 1*997; Williamson, 1999) and the efficiency-maximizing governance balance between the ‘State’ and the ‘Market’. Because of the growing emphasis on comanagement and the use of marine reserves in tropical fisheries (*e.g.*, White *et al., 1*994; Pomeroy, 1995; Virdin, 2000), the focus of research on the proper scope of governance for fisheries systems should more properly examine the balance between the ‘State’ and ‘Community’.

Two options exist for incorporating local input in fisheries governance systems. Devolution of governance involves the transfer of decision-making authority to local communities, implying that they share in decisions about reserve design and fisheries management rules. Decentralization of power, on the other hand, involves shifting decision-making to local branches of government. Local government agents, who are presumably more aware of the unique local social and natural environment, may be able to tailor policies that are more consistent with local norms than a central government agency would be able to.

The degree of devolution/decentralization that minimizes transaction costs of governance depends on a number of factors. Williamson (1985) developed the ‘discriminating alignment hypothesis’, which postulates that transactions have certain attributes that affect transaction costs and that governance systems have certain competencies and costs. Minimizing societal transaction costs requires that these two factors be aligned. The transaction of interest in tropical inshore fisheries is the production and maintenance of reef environmental quality that provides a flow of ecological amenities. The transaction costs of producing or maintaining reef environmental quality include: the costs of environmental monitoring of reef conditions and fish stocks; discussing, developing and reaching agreement on rules (*i.e*, reserve design and access) to control resource use; the legal costs for implementing management solutions; ongoing monitoring costs to ensure compliance with the final policy package; and the costs associated with enforcing penalties for those who violate the rules.

Impacts of Uncertainty on Transaction Costs and Governance

For tropical artisanal fishery management, uncertainty is the key characteristic of the reef environmental quality transaction. Many tropical systems have been managed locally historically and fishers may have a very good idea of how fish behave, spawn and migrate (Johannes, 1980). If fish production is predictable in time and space, and there is a tight coupling between adult abundance and recruitment, then users and managers can know and predict what resource withdrawal can be permitted and management at the local level should be more effective than centralized governance. The orientation in this situation would tend to be more on economic optimization rather than conservation *per se* (Ruttan, 1998).

As uncertainty increases and managers are less able to predict either the causes or consequences of resource fluctuation and use, the goal of governance should tend more to insurance functions. Thus, we would hypothesize that the transaction cost minimizing balance in comanagement would shift increasingly towards the State as uncertainty rises, scientific research becomes more important, and the consequences of ecological degradation have external effects in other regions or other sectors of the economy. When demographic pressure increases, new markets open for non-traditional seafood products, or local communities have not been able to demonstrate that local social norms have controlled opportunism, then there may be a justification for a stronger public role in comanagement.

For instance, the maintenance of giant clam (Tridachnidae) stocks is a transaction that requires local knowledge because it is highly site specific, with subtleties that can only be appreciated with very detailed local knowledge. This raises an interesting policy puzzle: why, if the production of giant clam has characteristics that would at first seem to make it amenable to community-based management using marine reserves, are giant clam stocks so depleted (Dalzell *et al., 1*996) even in areas where community management is used to successfully manage vulnerable reef fish (Jennings, 1998). One possibility is that the very high market prices for giant clam have altered incentives for individuals to such an extent that even traditionally strong local management systems have simply been overwhelmed and social norms supporting conservation have broken down. In addition, giant clams are inherently vulnerable: they are stationary, easy to harvest, slow growing and have limited larval dispersal (Lucas, 1994), so the maintenance of stocks may be considered an unpredictable transaction. A counter argument could then be made for stronger precautionary State management as insurance against catastrophic stock decline. Of course, this argument requires a government with the capacity to provide strong management.

It is easy to see that there may be less than desirable compromises to be made and that neither State nor Community governance may be ideal without substantial capacity building efforts. One can also see that there are numerous feasible research hypotheses that emerge when the fisheries management problem is framed in terms of uncertainty, social capital and transaction cost economics.

In summary, the success of devolution for marine ecosystem-based management depends on a number of factors. Community-oriented comanagement is likely to be efficiency-maximizing when the maintenance of reef quality requires local knowledge, when uncertainty regarding the causes and consequences of environmental change is relatively low, and when social norms and reputation can be used effectively to counter opportunistic behavior. Decentralized State governance may be more appropriate, however, if local knowledge is required but Community capacity is low. Social capital is an appropriate indicator of the extent to which user-managers of fishery resources are able to overcome social dilemmas and of the potential of marine reserves to efficiently achieve ecological and economic objectives. Thus, social capital is an important consideration in first identifying economically efficient policy options using comparative policy analyses and, second, as an indicator of the likelihood of successful efficiency-maximizing devolution or decentralization of governance systems over the longer-term.

Concluding Remarks

Marine reserves are seen as a central tool for implementing marine ecosystem-based management (Bohnsack, 1993; Roberts and Polunin, 1993; Costanza *et al., 1*998; Murray *et al., 1*999). The rationale for marine reserves is usually based on their potential for conserving or enhancing ecological amenities that provide a suite of market and nonmarket values while simplifying and reducing the costs of management. To date, however, there have been too few rigorous policy analyses of marine reserves to assess if reserves are, in fact, efficient policy tools for tropical inshore fisheries conservation and management. If credible analyses are not undertaken, there is a danger that current enthusiasm for marine reserves may wane as economic performance fails to meet presumed potential.

In narrowly focused economic analyses, the costs and benefits of commercial and subsistence fisheries from which participants derive extractive direct use value are emphasized. The economic benefits of spillover from marine reserves are theoretically easy to measure because market prices exist but, in practice, the difficulties in ecologically quantifying spillover (Lizaso *et al., 2*000) make economic valuation a challenge. Further, the costs of marine reserves to fishers have been poorly quantified to date and there has been insufficient consideration of fisher behavior (Wilen *et al., 2*002). This is an area of increasing research and one in which economic theory will play a central role.

Marine reserves provide broader benefits than just increasing fish stocks for capture fisheries (Moberg and Folke, 1999; Cesar, 2000). There is widespread recognition that marine reserves can provide a range of nonmarket values but research quantifying these values is limited. Fully accounting for the range of nonmarket economic benefits of marine reserves may help counter short-run pressure favoring fisheries over conservation but that leads to the dissipation of long-run economic benefits if poorly managed. Where marine tourism potential exists, the non-extractive economic value provided by rapid buildup of icon species (Halpern, in press) within marine reserves may provide especially important short-run economic benefits for local communities and help bridge the gap until long-term benefits become apparent (Rudd, 2001).

Many of the arguments for marine reserves are based on the perception that the transaction costs of management – especially monitoring and enforcement – are lower for reserves than for other options. Policy analyses that go beyond the scope of standard CBA are needed to assess these claims. At the first level of analysis, when the existing governance regime is essentially fixed, the focus of a policy analysis is on the cost-minimizing policy package that achieves society's ecological and socioeconomic goals. Cost minimization requires that analysts consider recent theoretical developments showing how social capital can have a direct impact on the transaction costs of tropical fisheries management. Social capital makes it easier for individuals to trust others, increases the likelihood of cooperation, and thereby lowers the costs of maintaining the flows of ecosystem amenities from public goods and common pool resources.

The level of Community capacity (the ability to use social capital for mutually beneficial collective action) and State capacity (the ability of the government to provide public goods and assure that property rights are honored) in a region will substantially influence marine reserve effectiveness and efficiency. We hypothesize that both are necessary for deriving full benefits from marine reserves and that where one or both components of the equation are missing, policy tools other than marine reserves will be more efficient.

At a higher level of analysis, a focus on social capital and transaction costs also permits rigorous examination of the core marine ecosystem-based management tenet of comanagement. Ascertaining the proper scope of governance – the efficiency-maximizing balance of the 'State' and the 'Market' – is currently a central issue in the transaction cost economics research agenda (*e.g.* Hart *et al., 1*997; Williamson, 1999). A string of highly publicized fishery collapses has already moved fishery management research from a narrow subdivision of applied ecology to the forefront of the debate over resource management and policy (Conover *et al., 2*000). An opportunity now exists to move tropical marine reserve research into the mainstream of the transaction cost economics research agenda by rigorous analysis of the efficiency-maximizing boundaries between the 'State' and 'Community'.

There are analytical challenges in following this policy analysis strategy, of course. The questions regarding what causal variables to consider needs to be guided by theory and structured so that testable hypotheses are developed. Substantial guidance on which factors should be considered and their *a priori* effects is now available from the common property literature (Agrawal, 2001). Recent advances in survey methods for measuring social capital (Uphoff and Wijayaratna, 2000) and new methodologies such as Ragin's (2000) fuzzy qualitative comparative analysis hold promise as tools for integrating broad comparative analyses of marine reserve that address policy design and implementation issues that have regional or global relevance.

In conclusion, we believe that nonmarket valuation, transaction cost economics and social capital theory will play an important role in marine reserve research over the coming years. Fisheries managers and ecologists need to be aware of relevant economic theory, terminology and empirical research directions and challenges given the increasing need for transdisciplinary marine reserve planning and assessment in complex ecological and social environments.

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CHAPTER 4

LIVE LONG AND PROSPER: COLLECTIVE ACTION, SOCIAL CAPITAL AND SOCIAL VISION[1]

Increasing human impact on the environment has led to broad and unique global environmental challenges (Turner *et al.*, 1990) that are characterized by complex technological, social and ecological systems interacting at variable spatial and temporal scale (Folke *et al.*, 1998; Gibson *et al.*, 1998). An emerging scientific consensus is of the opinion that these human-induced environmental impacts are problems that could impose enormous economic and ecological costs on human society and the biosphere (Arrow *et al.*, 1995; Costanza *et al.*, 1997).

Folke *et al.* (1998) point out that it is vital to recognize that environmental problems are not only ecological in perspective but are also, at their root, social problems caused directly or indirectly by the aggregate effects of humans going about their everyday lives. This viewpoint strongly suggests that environmental problems are linked to issues of collective choice at local, regional, national, and international levels (Ostrom, 1999). Humans, as key instigators of environmental change, will need to act collectively to develop cooperative solutions that can help control further environmental degradation that threatens our quality of life and, possibly, our very life support system itself. The emerging research tradition[2] of Ecological Economics, in attempting to address issues of environmental sustainability, therefore finds itself drawing from the realm of classic collective choice theories developed at the intersection of political science and economics (Commons, 1950; Olson, 1965; North, 1990; Ostrom, 1990; Miller, 1992).

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[2] Laudan (1977) introduced the concept of "research tradition" which he defines as the set of general assumptions about the entities and processes in a domain of study and about appropriate methodology to be used for investigating problems in the domain and constructing theories about the domain. A research tradition influences constituent theories by (1) determining types of problems researchers address by delimiting domain and methodology, (2) constraining types of theories that can be developed, (3) providing heuristic control or guidelines about how theories can be modified and improved, and (4) providing a justification role for baseline assumptions about nature and causal processes whose existence and operation the specific theories take as given.

Within formal models of rational choice it is generally assumed that human and organizational behavior can be explained adequately in terms of internal economic cost-benefit calculations. While these models are powerful when narrowly focused, social context as a determinant of human choice is critically import for understanding and solving human problems of collective action and rarely considered in formal models of rational choice. This is a central theme of the 'new institutionalism' in economics (Coase, 1960; Williamson, 1985; North, 1990), political science (March and Olsen, 1984; Ostrom, 1990; Sandler, 1992) and sociology (Smelser and Swedberg, 1994; Nee and Ingram, 1998; Portes, 1998).

This recognition, that social context matters for collective action, has further led to the viewpoint that social relationships themselves can constitute resources which individuals can appropriate to assist them in increasing their well being . Beyond personal instrumental benefits, social relationships can also lead to the development of norms of trust and reciprocity that have spillover effects - positive social externalities - within society as a whole. At the most expansive level, the institutions of markets, property rights, the legal system and the structures of governance affect the social and cultural environment within which social relationships are constituted. This complex of social relationships, norms, and institutions is now often referred to as "social capital" (Coleman, 1987; Grootaert, 1998; Woolcock, 1998; Ostrom, 1999).

Why would such concepts matter? There are enormous implications for the 'production' of universal public goods such as knowledge and peace in a modern world characterized by complex, inter-related, and contestable political and economic relationships set within bio-physically constrained and uncertain environments. There is substantial empirical evidence to support the hypothesis that norms of reciprocity, trust, and institutions matter in the production of quasi-public and pure public goods (Putnam, 1993; Flora *et al.*, 1997; Narayan and Pritchart, 1997; Temple and Johnson, 1998). The production or enhancement of ‘nature’ itself can also legitimately be viewed as a public good. Without clear rules of responsibility and reciprocity, the maintenance and production of environmental quality can be jeopardized because it is otherwise in narrow individual self-interest to ‘free-ride’ on the contributions of others (Olson, 1965; Hardin, 1968). Social capital, as an input factor in the production of environmental quality, thus needs to become an important focus in the sustainability debate.

Elements of a social capital framework provide promising theoretical links between research themes in ecological economics and 'new institutionalism' in social science. A particularly important link can be drawn between behavioral theories of rational choice (Satz and Ferejohn, 1994; Ostrom, 1998), the structural variables that help create social capital, and the collective choice processes that generate shared vision. Shared social vision allows consideration of alternative futures and acts as a normative filter, leading to policy prescriptions and concrete human action (*e.g.*, Senge, 1990; Meppem and Gill, 1998; van den Belt *et al.*, 1998).

The purpose of this paper is to clarify elements of a social capital framework that can be used to develop theories of the effects of institutional structure on sustainability and to articulate the links between social interaction, collective action and social vision. Specifically I (1) outline the theoretical bases of social capital, (2) examine disciplinary distinctions in defining social capital, (3) develop a link between social capital theory and behavioral theories of rational choice and (4) discuss the importance of applying social capital theory in ecological economics research on social vision that can link individual-level behavioral rational choice and collective decision making.

Theoretical Background

There has been increasing academic interest in social networks, civil society and "social capital" over the last fifteen years within the disciplines of political science (Putnam, 1993; Fukuyama, 1995; Jackman and Miller, 1998; Ostrom, 1998; Woolcock, 1998), sociology (Granovetter, 1985; Coleman, 1987; Bourdieu and Wacquant, 1992; Portes, 1998; Portes and Sensenbrenner, 1998; Burt, 2000), and economics (North, 1990; Besley *et al.*, 1993; Robison and Siles, 1997; Castle, 1998). The focus of much of the work has been on the reasons for cooperation and how trust develops. Cooperation is fundamental where humans develop successful management institutions for public goods including natural resources (Ostrom, 1990). Because of the burgeoning interest in social capital and its potential impact for affecting collective action in sustainable renewable natural resource institutions, contributions from political science, sociology and economics will inevitably influence research conducted by ecological economists.

Why is Social Capital Important?

Economic problems involve making choices under conditions of uncertainty and scarcity. Factors of production are transformed to produce commodities including diverse quasi-public goods such as education and public health, and universal public goods such as environmental quality or international order. Public goods share two characteristics important to theories of collective choice: (1) they are under-produced; and (2) we would be better if more were produced (Coleman, 1987). A crucial cause of underproduction are incentives rewarding the maximization of short-term self-interest while leaving all participants worse off in aggregate than feasible alternatives (*i.e*, individuals tend to free ride).

If optimal levels of public goods are to be produced and trade gains be fully exploited, shared understandings and patterns of collective action develop beyond immediate kin. Research evolutionary psychology (Barkow *et al.*, 1992; Cosmides and Tooby, 1994) and experimental economics (Ostrom *et al.*, 1994; Hoffman *et al.*, 1998) suggests that humans are more predisposed to social exchange through reciprocity than is expected under narrowly defined rational choice models. Cognitive science holds that a variety of heuristics (Heiner, 1983; Simon, 1996) and reciprocity (Ostrom, 1998) are ‘hard-wired’ by evolutionary selection, resulting in a propensity to cooperate with others not perceived as foes, even if the functional mechanisms of reciprocity vary between societies. Satz and Ferejohn (1994) and Ostrom (1998) outline a behavioral approach to a ‘softer’ rational choice theory compatible with developments in evolutionary psychology and especially useful in developing concepts of social capital.

Collective action is facilitated through the inhibition of short-term self-interested behavior via a self-reinforcing cycle of trust and reciprocity. At the individual level, norms of trust and reciprocity lead to the formation of reputation, an important asset that can help reduce the transaction costs associated with exchange in situations of information asymmetry (Ostrom, 1998). In the aggregate, increased returns are achieved via increased levels of generalized social trust (Putnam, 1993; Fukuyama, 1995) and by institutionalizing mechanisms of trust, reputation and reciprocity (North, 1990), both of which reduce transaction costs.

Institutions are themselves human artifacts, crafted and revised over generations (Ostrom, 1997). They help humans cope with environmental complexity by providing the infrastructure that allows humans to develop and propagate formal and informal norms and rules across generations. Institutions allow people to interact with reasonable confidence in the predictability of their actions. When institutions contribute to successful achievement of aspirations, a feedback mechanism reinforces the norms and rules within the institution. Institutions thus evolve over time as humans modify the 'rules of the game' in response to changing physical conditions, technology and shifting value (North, 1990; Hodgson, 1993; Ostrom, 1997). Some institutions have proven to be very robust over time; Ostrom (1990) documents the characteristics of successful institutions for the management of common pool resources (CPRs) that have survived for generations.

Do societies that develop successful institutions, solve social dilemmas and produce increased levels of public goods gain tangible benefits that can provide them a competitive advantage and lead to increased levels of social or economic success? Yes, if the benefits include prosperity (internal economic development), enhanced political stability and increased education and societal knowledge about the world and/or transformation processes. Developing a social vision is one aspect of developing long-term sustainability and competitive advantage – this is also an area where ecological economics research might have a particularly important impact. I return to deal with this in Part 3 after briefly examining the evidence that social capital affects economic performance and political stability.

Within the discipline of economics, empirical studies show that returns to investments in land, labor and manufactured capital often do not account for economic performance at either micro- or macro-levels. Even when controlling for human capital (Becker, 1962; Schultz, 1963), there remain large unexplained differences in economic performance between communities, regions and countries (North, 1990; Putnam, 1993; Flora *et al.*, 1997; Narayan and Pritchart, 1997; Knack and Keefer, 1997; Portes and Sensenbrenner, 1998).

Social capital theory suggests that this residual performance is due to social and cultural factors (Collier, 1998) at both micro- and macro-levels. The development of individual norms of reciprocity and trust can lead to instrumental gain through the brokerage of information and further exploiting gains from trade through social specialization. Empirical evidence lends support to both propositions.

At the level of the individual, Burt (2000) summarizes results from extensive empirical studies on information networks that provide competitive advantage for business managers and entrepreneurs. Burt hypothesizes that information circulates more within groups than among groups of people. Consequently, those people maintaining weak ties between groups can exploit them and control the flow of information between non-redundant network contacts. Burt demonstrates instrumental returns for business managers (higher evaluations of manager's work performance, higher probability of early promotion, and higher compensation relative to peers) to be a function of network size, density, and hierarchy.

Also at the micro-level, Narayan and Pritchart (1997) found that household incomes in Tanzania depend on village level social capital in addition to household characteristics. The effects of community level social capital, measured by the frequency of group or association membership of village residents, are large: one standard deviation increase in village level social capital led to household expenditure increasing by 20-30% per person (this is as large as the tripling of education level). Villages with high levels of social interaction occurring through informal association were ones in which there were higher levels of use of credit in agriculture, for example, compared to villages with little social interaction.

At the macro-level, Knack and Keefer (1997) recently examined 29 market economies and found that investment and growth rates were higher in countries where interpersonal trust and norms of civic cooperation were greater. They also find that trust and cooperation are stronger in countries with well-developed legal and institutional infrastructure protecting property and contract rights and in countries that are relatively homogenous in terms of income level and social standing.

Is political stability a necessary condition for sustainability? Political performance is more problematic to define and measure compared to economic performance, yet is clearly related to social structure and economic performance (Coleman, 1987; Putnam, 1993; Nee and Ingram, 1998; Ostrom, 1999). The prospects for living long in a tension-filled, rapidly changing world filled with high technology instruments of destruction are greatly enhanced if nations can avoid war. Cooperation to exploit gains from trade are much more likely in stable societies where people have a higher degree of trust in each other and/or the legal infrastructure that allows parties unfamiliar with each other to enter into credible contracts (Williamson, 1985). Political stability can increase the odds of successful collective action by allowing increased levels of predictability about the social environment within which individuals engage in trading activities.

Theories about the 'democratic peace' (Russett, 1993) have recently been at the center of some heated academic debates (*e.g.*, Layne, 1994) and have exerted substantial influence in American foreign policy. The primary puzzle is the assertion that democracies don't go to war with each other. The democratic peace hypothesis posits that there are structural and/or normative explanations of the phenomena. The citizen-based normative explanation is based on the argument that citizens' values, beliefs, and norms - that we shouldn't fight illegitimate conflicts with other democracies - constrain government action in the international arena.

The evidence that citizen norms and values can constrain the behavior of national governments suggests the importance of norms and values to affect human behavior at all levels of decision making. The development of shared norms developed through societal-level discourse may also have policy impacts in other fields. Increased social interaction and strengthened social capital might influence social values and social vision over a relatively short time span, a key requirement for any shift to true ecological and social sustainability.

The Conceptual Basis for Social Capital

Social capital is a productive asset that enables individuals to better fulfill their aspirations through access to goods and services via their social network and through collective action (Castle, 1998). Social capital is deemed to increase production of quasi-public and public goods by increasing levels of knowledge about production, transformation processes and trading partners, and by exploiting the gains from trade through specialization. Transaction costs associated with trading are reduced via an increase in levels of trust between trading partners and the development of institutions that provide incentives for lasting cooperation (Coleman, 1988; North, 1990; Ostrom, 1990; Woolcock, 1998; Ostrom, 1999). Social capital, unlike other forms of capital, is not depleted with use but actually increases in value with use (Ostrom, 1999)

In social capital theory, norms and rule-ordered relationships are viewed as resources which individuals can use to reduce risk, access services, obtain information, and coordinate collective action (Grootaert, 1998). The overall capacity for coordination raises output in four potential ways according to Collier (1998): (1) social sanctions against opportunism by free-riding individuals reduce transaction costs; (2) common pool resources can be effectively managed on a sustainable basis; (3) public good provision can increase; and (4) society can take advantage of economies of scale in nonmarket activities. Social capital can act as an input to the production function for individuals and organizations by constraining opportunism and thereby increasing the probability of collective action to deal with social externalities.

The literature abounds with specific definitions of social capital[3] that can be cleaved into three broad categories: (1) the view that social capital is 'generalized trust', formed largely as a byproduct of the activities of individuals interacting with each other within voluntary or informal associations (*e.g.*, Putnam, 1993; Fukuyama, 1995; Inglehart, 1997; Stolle, 1998) in a manner originally emphasized by Tocqueville (1945 [1835, 40]); (2) the view that social capital consists of the norms and social networks that facilitate collective action for instrumental and collective benefit (*e.g.*, Granovetter, 1985; Coleman, 1987; Nee and Ingram, 1998; Ostrom, 1999; Portes and Sensenbrenner, 1998; Burt, 2000); and (3) the view that social capital consists of the institutional infrastructure that facilitates the development of trust, cooperation and trade between individuals who would otherwise remain socially isolated (*e.g.*, North 1990; Williamson, 1994; North, 1998).[4]

The question quickly arises: but is it really capital? The key characteristics of capital are that there is an opportunity cost required investing in it and that it permits people to become more productive in fulfilling human aspirations. In the case of social capital, both time and effort are indeed expended on transformations and transactions to build social assets - norms, rules and institutions - today that increases income in the future through increased productivity (Bourdieu, 1986; Narayan and Pritchart, 1997; Castle, 1998; North, 1998; Ostrom, 1999). Changes in the level of social capital have economic and political consequences.

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3 Specific definitions of social capital by some of the prominent disciplinary social capital theorists include: (1) "…those expectations for action within a collectivity that affect the economic goals and goal-seeking behavior of its members, even if these expectations are not oriented toward the economic sphere" (Portes and Sensenbrenner, 1998: 129); (2) "…the sum of resources, actual or virtual, that accrue to an individual or a group by virtue of possessing a durable network of more or less institutionalized relationships of mutual acquaintance and recognition" (Bourdieu and Wacquant, 1992: 119); (3) "…as one's sympathy (antipathy) for others, idealized self, and things" (Robison and Siles, 1997: 3); (4) "…a culture of trust and tolerance, in which extensive networks of voluntary associations emerge" (Inglehart, 1997: 188); (5) "…obligations and expectations, which depend on trustworthiness of the social environment, information-flow capability of the social structure, and norms accompanied by sanctions" (Coleman, 1988: S98); (6) "…features of social life - networks, norms, and trust - that enable participants to act together more effectively to pursue shared objectives" (Putnam, 1995: 664-665); (7) "…a capability that arises from the prevalence of trust in a society or certain parts of it" (Fukuyama, 1995: 26); (8) "…the quantity and quality of associational life and the related social norms" (Narayan and Pritchart, 1997: 2); and (9) "The social capital of a society includes the institutions, the relationships, the attitudes and values that govern interactions among people and contribute to economic and social development… It includes the shared values and rules for social conduct expressed in personal relationships, trust, and a common sense of 'civic' responsibility, that makes society more than a collection of individuals" (World Bank, 1998: 1)

4 Portes (1998) and Portes and Sensenbrenner (1998) provide useful overviews of different conceptions of social capital from the classic sociological perspectives of Durkheim, Simmel, Weber, Marx and Engles and how they relate to contemporary conceptions of social capital.

In economic terms, a change in the level of social capital can lead to an alteration in the terms of trade, individuals internalizing externalities, a change in the probability of successful collective action, a change in the number of opportunities for specialization and gains from trade, changed personal income levels, and a redistribution in income and social welfare (Robison and Siles, 1998). Additional evidence that social capital is indeed 'capital' comes from observing the tradeoffs between social and other forms of capital in the production of quasi-public goods. Frank (1992), Grootaert (1998) and Castle (1998) note that there are distinct tradeoffs between market and social network mechanisms of social provision. Services previously provided by more informal social capital mechanisms - care-giving, insurance, information on trustworthiness - are often provided by the market as economic affluence increases in a society.

What is the Essence of the Debate over Social Capital?

Many of the academic debates over social capital result from issues of (1) the narrow definition of social capital construed by a number political scientists (*e.g.*, Putnam, 1993; Fukuyama, 1995; Inglehart, 1997) and (2) the nature of endogeneity between social interactions, trust and economic and political performance. With regards to methodology, there have also been criticisms that research confuses causality and uses inherently deficient social survey data on generalized trust to draw broad conclusions about economic and political performance at aggregate levels (Foley and Edwards, 1998; Jackman and Miller, 1998).

Generally, the narrowest view of social capital is the most prevalent in the discipline of political science. The “civil society” perspective is characterized by the idea that a culture of trust and tolerance develops where networks of voluntary exchange emerge (Putnam, 1993; Fukuyama, 1995). Proponents of this view emphasize the importance of a vibrant civil society and exogenous 'generalized trust' - trust that extends beyond the boundaries of face-to-face interaction (Stolle, 1998) - as primary factors affecting economic and political performance. This perspective draws heavily on the insights that Tocqueville made regarding the importance of customs and habits of the American people in their participatory system of direct, democratic self-governance (Tocqueville 1945 [1835, 40]).

Levels of generalized trust in societies are obtained through surveys[5] and trust indices used as the independent variable in regressions with economic growth:

Economic Performance = *f*(generalized trust) = *f*(*g*(voluntary association)) (1)

In slightly broader terms, sociologists tend to focus on social capital as social relationships and networks that allow actors to meet objectives of influence and control (Granovetter, 1973; Granovetter, 1985; Coleman, 1987; Bourdieu and Wacquant, 1992; Portes, 1998; Portes and Sensenbrenner, 1998; Burt 2000). The key difference from the view of the 'civil society perspective' is that trust is endogenous, resulting from specific social structure. The basic question becomes: under what social conditions can 'narrow trust'[6] be generated? This leads researchers within this perspective to explore social-structural explanations of economic life that focus on the sources of social capital rather than the consequences.

Economic Performance = *f*(narrow trust) = *f*(*g*(social structure)) (2)

The final common perspective on social capital is the broadest of the three main groups. This view, commonly associated with New Institutional Economics, holds that the social and political environment is what enables norms to develop and shape social structure (North, 1990). Because of the influence formal institutions have over the actions that humans take at the individual level - through constraints and sanctions - an argument can be made that economic activity in society is affected by the complex of norms, rules and formal institutions that themselves comprise social capital. Transaction costs, fundamentally a problem of trust[7], change when there is a change in the credibility of commitment for either economic or political actors (Coase, 1960; Williamson, 1985). Institutions that reduce uncertainties or increase the credibility of commitment between contract parties (*e.g.*, markets, laws, legislatures, international treaties and agreements)

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5 For example, Stolle (1998) asked respondents in his study six questions: (1) would you think that one can trust other people, or should one be careful with others?; (2) in this society one does not need to be constantly concerned about being cheated; (3) no matter what they say, most people are not willing to stand up for others; and three scaled questions about how much the respondent would trust a foreigner, a stranger, and another fellow citizen.

6 Perhaps more properly thought of as experience-based prudence (Coleman, 1987) or as personalized or private trust which involves people personally known (Stolle, 1998).

7 If humans fully trusted each other, there would be no need for monitoring and enforcement of agreements. Conversely, the more distrust there is between contracting parties, the more resources must be spent on the monitoring, contract specification and enforcement mechanisms that comprise much of transaction costs.

are the substance of social capital because they allow joint production of social goods through collective action and maintain 'rules-in-use' to capture the gains from trade.

$$\text{Economic Performance} = f(\text{institutional infrastructure}) = f(g(\text{rules-in-use})) \quad (3)$$

The primary misunderstanding between the different camps arises over the issue of the endogeneity of trust. Putnam (1993) viewed trust as an exogenous variable independent of social structure while Coleman (1987) viewed it as endogenous, dependent upon social structure to create incentives for individuals to behave in a trustworthy manner. The nature of the micro- to macro-level linkages and the nature of causality from social interaction to trust, trust to social capital, and social capital to economic and political performance are readily obscured given the myriad perspectives on social capital. Some authors emphasize the link from social structure to trust only and downplay the second stage, from trust to the aggregate economic and political performance on which both Putnam (through informal associations) and new institutional economists (through formal institutions) focus (*e.g.*, Jackman and Miller, 1998). Similarly, a number of prominent political scientists have failed to recognize that voluntary associations are not, in and of themselves, social capital that can explain economic growth (for a critical discussion of recent work. see Foley and Edwards, 1998 and Edwards and Foley, 1998) but only a structural variable that can explain the development of both narrow and generalized trust which, in turn, can help increase outputs and reduce transaction costs.

Operationalizing Concepts of Social Capital

A Framework for Social Capital

The Social Capital Initiative Project of the World Bank has published a number of recent reports by scholars examining theoretical and methodological aspects of social capital formation and measurement (Collier, 1998; Grootaert, 1998; World Bank, 1998; Feldman and Assaf, 1999). Collier (1998) presents a useful framework for conceptualizing social capital based on social interaction; it can account for both the social network and generalized trust perspectives on social capital and help make sense of the endogeneity debate.

A traditional economic good, V, might be a good or service such that output $V = v(L, K)$, where L is labor input and K is traditional manufactured and financial capital input. If a capital purchase is made because of collective action, the output can be viewed as $V =$

$v(SI, L, K)$ where the additional input, *SI*, is the social interaction needed to initiate the purchase. Besley *et al.* (1993) provide an example of rotating savings and credit associations (ROSCAs) where funds are allocated amongst members via either random draws or bidding procedures. 'Winning' the pot of funds (obtaining a loan) permits individuals to make purchases of indivisible goods that they would normally not be able to afford; social relationships between ROSCA members and the threat of social sanction are the primary forms of collateral for individuals that would be viewed as credit risks by conventional lenders. In the case where the funds are used to purchase farm equipment, for example, agricultural output would be a function of farmer labor, the new capital equipment, and the social interaction that allowed a farmer to obtain credit otherwise unattainable through traditional sources. The net present value of the flow of income to the farmer after the costs of capital and labor are covered is the residual return to social capital.

The value of the social interaction is the net return to social interaction once the costs of the other inputs are met – returns which would not have been possible without the durable effects of social interaction. If the stream of benefits can be aggregated as present value then we should consider the relevant collective action as a form of social capital. In Putnam's (1993) case, social interaction between Italian choir members lead to increasing levels of trust beyond the immediate choir members and persisted even when the choir itself was disbanded. The economic intuition behind Putnam's work is that choir members are more likely to engage in trade with strangers and thereby increase their gains from trade into a present value significantly higher than that of a non-choir member.

Collier goes on to develop a framework for examining the effects of social interactions on social externalities. He suggests four types of information flow: (1) observation, in which information flow is informal and one-way; (2) hierarchies, in which information flow is formal and one-way; (3) networks, in which information flow is informal and reciprocal; and (4) clubs, in which information flow is formal and reciprocal.

The flow of information, as a knowledge-builder, has three effects: (1) increased knowledge of the behavior of others (assessment) reduces risk and transaction costs; increased knowledge about the non-behavioral environment (knowledge) improves productivity and reduces risk and transaction costs; and collective action (coordination) increases overall social benefits. When solving social dilemmas, coordination is unique because it also requires people to assess a potential trading partners' propensity for opportunism.

Finally, Collier states that for the effects of social interaction to be classified as social capital, either (1) the structure that produces the effects is durable or (2) the effects themselves are durable. In the case of ROSCAs, the formal organizational structure of the credit association is an example of a durable structure representing social capital. There are also durable effects resulting from ROSCAs – one could conceive of situations where a one-time contribution of credit from a ROSCA enabled a small farmer to break the cycle of poverty, invest in education and continue to improve his/her economic well being as a spin-off benefit arising from the original loan. In both cases, there are long-term economic benefits that can be aggregated into net present values that would be unattainable without the social interaction essential to ROSCA formation and operation.

Using this framework, informal or formal social interactions help solve problems of social externalities by reducing transaction costs and increasing knowledge about both the world and the trustworthiness of other individuals. There is thus an increase in net benefits available for humans through learning, trust, and rules. All three disciplinary perspectives – political science, sociology and economics – can be accounted for within this framework. Putnam's social capital is based on informal associations while Coleman's is either informally or formally based on obligations, norms and rules-in-use; Putnam's version of social capital can be seen to be a subset of the broader version of the social capital concept of sociologists. Both mechanisms produce social capital that can lead to the reduction of transaction costs on which new institutional economists focus.

$$\text{Economic Performance} = \phi(\text{transaction costs}) \tag{4}$$

$$\text{Transaction Costs} = \gamma(\text{social capital}) \tag{5}$$

$$\text{Social Capital} = \tau(\text{social interaction}) \tag{6}$$

Economic performance can be enhanced via quantity increasing measures (increased knowledge about the world and the transformation processes involved in production), cost-reducing measures (a reduction in production and transaction costs) and/or revenue enhancing measures (via gains from trade or increased knowledge about other trading partners). Transaction costs become a function of social capital – institutions specify the rules-of-the-game via informal rules-in-use or via formal rules embedded in the legal infrastructure. Social capital, in turn, is a function of social interactions and their structural components such as voluntary network structure and social contacts that may serve no explicit instrumental economic function.

Using Collier's framework, we can summarize the social interaction structures and mechanisms used to build social capital (Table 4.1). Focusing on the mechanisms with durable effects one can define the cost of copying to be the cost of observation (*e.g.*, the cost of a demonstration project for a new farm crop) and the cost of pooling to be the cost of facilitating two-way communication (*e.g.*, the cost improving phone service in rural areas). Information on trustworthiness can be obtained through one-way information flows of repeated transactions within hierarchies (*e.g.*, the payment record for a business customer) while reputation requires networks to update the reputation of agents known to the network (*e.g.*, the 'ecological economics research community 'grapevine' which formally or informally reviews the research of those within the community). For reputation to be an effective mechanism for building social capital, the agents in a network must be sufficiently similar to know the same mutual colleagues and have similar 'trading' relations.

	Durable effects	
	Reciprocal flow of information	**One-way flow of information**
Knowledge	Pooling	Copying
'Narrow' trust	Reputation	Assessment
Rules-in-use	Deference	Coordination

Table 4-1 – Social interaction structure and mechanisms used to build social capital

Coordination can be increased through the one-way flow of information associated with deference – the acknowledgement of the expertise and experience of mentors, for example. Finally, there are potential instrumental rewards for abiding by social rules-in-use –norms, rules, and behavioral constraints – imposed by institutions. Conversely, there may be heavy costs to bear for breaking societal norms (*e.g.*, fines, incarceration, and social ostracism at a minimum).

While Collier (1998) goes on to explore the empirical implications of the social interactions, my immediate concern is to link the social interactions that serve as structural variables in the creation of social capital to the second generation rational choice theory of collective action outlined by Elinor Ostrom (1998). She noted that research on the effects of structural variables on the likelihood of organizing for collective action was an immediate priority. Important new questions, she further speculates, will relate to the development of trust, selection of social exchange partners, and the effects of institutional structure on the development of norms.

Linking Social Capital and Collective Action

Ostrom (1998) developed a theory of behavioral rational choice which proposes the idea that a ‘core relationship' between trust, reputation and norms of reciprocity reinforce each other and can lead to increased levels of cooperation and, hence, net benefits. Theory can, as a result, be used to develop testable models for the laboratory or field. For any particular ‘action situation’ there might be a mix of structural variables, some of which would produce social capital via their enduring structure (*e.g.*, small group size) and some in which the effect itself is enduring (*e.g.*, information about past actions).

Using the typology of structures and effects based on Collier (1998) and presented in Table 4-1, one can refine ideas presented by Ostrom. In her theory, trust, reputation and reciprocity form a self-reinforcing triad. Here, each of the six components of social capital can be explicitly accounted for via the type of social interaction (Figure 4-1).

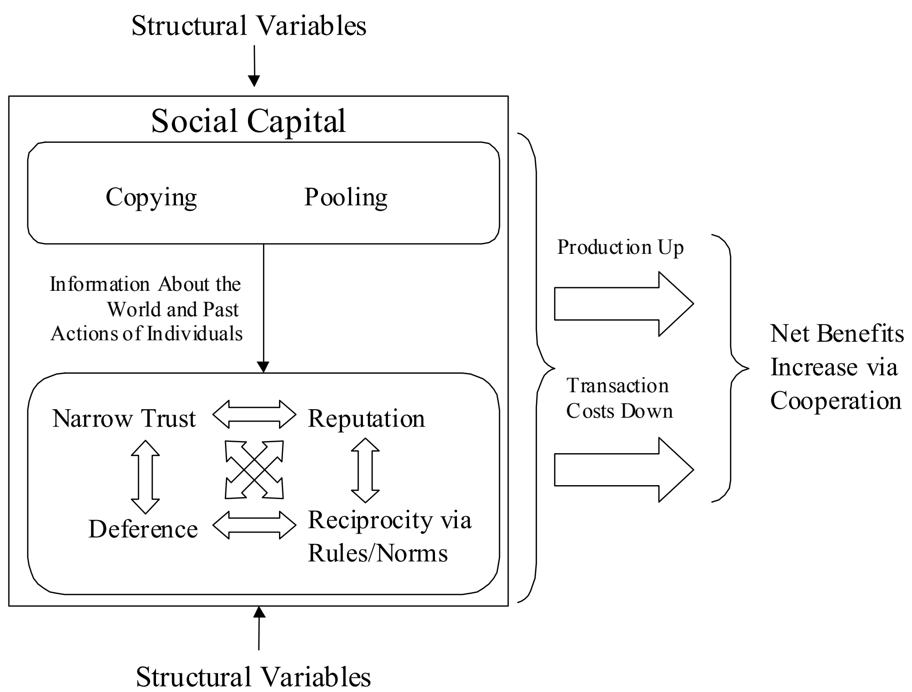

Figure 4-1 – Structural variables, social capital and net benefits

The four core elements are linked because they all increase the probability of successful collective action. Increasing one's knowledge about the reliability of other people can help increase output by itself, but it is also needed for successful coordination for collective action (Collier, 1998). Increasing one's knowledge of physical production processes is useful for increasing output but not necessary for successful collective action *per se*. Each social interaction can now be viewed as affecting the core relationships or knowledge about the world. Trust, norms of reciprocity, deference and reputation are the substance of social capital; increasing cooperation and production requires investment in, and maintenance of, social capital if advantages of trading are to be exploited.

This framework provides an important link between behavioral theories of rational choice and social capital: a link can be made from the development of human norms and values to the production of public environmental goods. By extension, this theoretically links social interaction to the issue of ecological and social sustainability. The pragmatic advantage of such a framework is that it provides a theoretical basis for developing hypotheses regarding the effects of structural variables on the level of and/or likelihood of cooperation between interdependent humans. Ostrom (1998) identified some potential structural variables of interest in her theoretical framework: group size; symmetry of interests of actors; the availability of face-to-face forums for discussion; availability of information; and long time horizons.

Social vision can now be seen as a key structural variable articulating linkages between human decision making at individual and collective levels – it affects the time horizon that individuals consider in their decision-making processes and also contributes to the development of shared norms. The process of undertaking the scoping exercise necessary to build a shared social vision also has positive externalities within social capital theory: the interactions that the participants in the vision building exercise have can lead to the development of generalized trust in the same way in which Putnam (1993) documents the development of generalized trust in Italian social organizations. In another study on the characteristics of successful community development, O'Brien *et al. (*1998) found that the horizontal linkages characteristic of successful communities led to benefits even if the specific project that volunteers worked on was a failure. The process of local people working together is more important than the accomplishment of any specific project of objective. The vision-building process thus can serve to both create shared understanding and build generalized trust.

What are the indicators of social vision? The purpose of this paper is not to catalogue all potential indicators but to suggest fruitful avenues of research that arise naturally when one works within the social capital framework in developing behavioral theories and models of social vision. In this case, questions arise as to how to measure vision, how to measure the structural effects of scoping exercises themselves (apart from any direct benefits they may impart on the planning process) and how community activism impacts community vision.

Social Vision and Self-Governance

What is the vision that we as a society have of our future quality of life, both in terms of social and environmental indicators? What alternative development paths are possible for humans, how do we choose amongst alternatives and how might we use our technology and social capital to reach more desirable paths? Given that there are biological and ecological limits to the amount of economic growth that we can sustain, and given that individuals and institutions are inherently more adaptable than ecological and cultural systems, what incentives can we use to encourage human behavior that is compatible with sustainability? These are the sorts of crucial questions that the emerging research tradition of Ecological Economics can address.

One implication of the evolutionary meta-paradigm (*e.g.*, Popper, 1972; Campbell, 1987; Hahlweg and Hooker, 1989) housing ecological economics is the normative dimension it introduces to policy development and collective action. Evolution is not progressive *per se* but evolution in the direction that satisfies criteria of value can be viewed as progressive. Commons (1950: 91) emphasizes the difference between "purposeless" natural selection and "artificial, or rather purposeful, selection [which] introduces ethical ideas of fitness - the ethical ideas of rights and duty, goodness and badness, justice and injustice." Hodgson (1993: 47) further argues that "given that socioeconomic behavior is purposeful, systems of values, visions of the future, economic expectations, can all guide and accelerate such a process [evolution]."

This situation is in sharp contrast to neoclassical economics. Consumer sovereignty, the principle upon which neoclassical economics is based, makes it difficult or impossible for social vision to be considered within the narrow confines of the voluntary market exchange institutional framework. Well being is defined by people's satisfaction of personal preferences. When dealing with complex environmental problems driven by human choice within imperfect institutional environments, there is no reason to believe that relying on market decisions based on the preferences of individuals will steer society in a desirable or sustainable direction.

The processes of developing a community or social vision requires that groups of individuals identify and raise issues, becoming a 'public' in Dewey's (1991[1927]) terminology. A community must discuss alternative resolutions, consider adverse impacts and amelioration, engage in the production and provision of solutions, and provide for sanctions, monitoring, conflict resolution and evaluation of outcomes. This interactive social vision process of identification of alternatives, discussion, contestation and decision-making builds social capital.

The process of building social vision really is one and the same as the process of direct democracy in which each person takes responsibility for their own actions and their role in collective decisions (*e.g.*, Tocqueville 1945 [1835, 40], Dewey, 1991[1927], and the modern work of Vincent Ostrom, 1997). If one takes a shallow view of democracy as merely universal suffrage and relatively uncorrupt elections, the implications for development of social vision are far different than if one considers deeper concepts of direct democracy and the role of citizen participation in self-governance. The crucial features of participatory democracies are face-to-face communication, dialogue, contestation, and critical assessment that serves to enlighten, educate, articulate community preferences and help resolve problems in an open society. Using the social capital framework allows ecological economists to study collective choice decision-making mechanisms of self-governing systems while maintaining the crucial link with individual-level behavior that is a primary driving force behind environmental change.

If citizen norms supporting a democratic peace contribute to sustainability of governance and economic institutions, is it also possible that norms of environmental stewardship can be fostered through certain social interactions and structure? It will be important for ecological economists to be able to explain the process of the evolution of pro-environmental norms through social capital theory. Social capital theory provides the theoretical basis for the pivotal role of social vision and can guide the construction of theories and models about effects of specific structural variables that are part of the vision-building process on the likelihood of cooperation and collective action.

Increasing social capital in society can help people avoid violent conflict, exploit gains from increased specialization, and increase knowledge about the physical and social factors important in the production and provision of public goods. It will be necessary for ecological economists to use insights from social capital theory in order to link theories of individual choice, collective choice, sustainability and the social forces driving environmental change. Only then may we be in a position to truly live long and prosper.

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CHAPTER 5

AN INSTITUTIONAL FRAMEWORK FOR DESIGNING AND MONITORING ECOSYSTEM-BASED FISHERIES MANAGEMENT POLICY EXPERIMENTS[1]

According to the American Fisheries Society, "sustainability of fisheries and other aquatic resources is a state in which these resources, and the ecosystems that support them, are managed in such a way that their long-term viability and productivity are maintained for the benefit of future generations" (Knuth *et al*., 1999). Achieving sustainability has proven elusive to date, but it is internationally recognized as a primary goal of fisheries management (FAO, 1995; NMFS, 1999; NRC, 1999; Garcia, 2000; Garcia and Staples, 2000).

There is a growing consensus that an ecosystem-based fisheries management paradigm is needed for achieving fisheries sustainability (Costanza *et al*., 1998; NRC 1999; Gislason *et al*., 2000). Under ecosystem-based fisheries management, experiments are needed to build further understanding about complex fishery system processes (Walters, 1997). This goes beyond just implementing ad hoc 'adaptive' responses to unexpected ecological or economic crises, replacing trial and error learning with a directed process of active policy selection. Policy selection is driven by societal objectives that are ultimately a reflection of the values, preferences and behaviors of individuals and organizations within that society.

Institutions, the human-crafted rules and norms that infuse social order, shape human incentives and behavior (Ostrom, 1990, 1999) and a variety of institutions (means) can be crafted to achieve any particular objectives (ends) envisioned under ecosystem-based fisheries management. Even small-scale, self-governing fisheries use a plethora of rules to govern when and how resources are harvested and used by particular users (Ostrom *et al*., 1994) and in more complicated fisheries the rule set may become very complex (Sinclair *et al*., 1999). Furthermore, the array of options may vary greatly in costs, making it be necessary to design and monitor policy experiments that strategically test

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the cost-effectiveness of policy bundles that can help achieve diverse societal objectives under ecosystem-based fisheries management (Rudd *et al*., 2003).

A variety of indicator frameworks have been proposed to monitor fisheries sustainability (Garcia and Staples, 2000; Garcia *et al*., 2000; Sutinen, 2000; Charles, 2001; Olsen, 2003), the sustainability of other common pool resources (Prabhu *et al*., 1999; Campbell *et al*., 2001), and for broader assessment purposes (Hammond *et al*., 1994; Ashley and Carney, 1999; Bossell, 1999; OECD, 2000; UN, 2001; World Bank, 2001; Segnestam, 2002; NRTREE, 2003). The OECD pressure-state-response (PSR) framework (OECD, 2000) and variants are process-oriented frameworks that are gaining exposure in the fisheries field (Garcia and Staples, 2000; Bowen and Riley, 2003). Exogenous driving forces and endogenous anthropogenic impacts exert pressure on the state of the environment; societies respond by attempting to mitigate the pressures. An alternative structurally-oriented framework, the Sustainable Livelihoods model (Ashley and Carney, 1999; Bebbington, 1999), is popular in the forestry and agricultural development fields (Prabhu *et al*., 1999; Campbell *et al*., 2001). An emphasis in the Sustainable Livelihoods framework is on the role of capital assets (natural, produced or physical, human, social and financial) on sustainability and human welfare.

The emphasis in both PSR and Sustainable Livelihoods frameworks has been largely on using indicator systems to communicate useful information to decision-makers (Ashley and Carney, 1999; Garcia and Staples, 2000; Garcia *et al*., 2000; Segnestam, 2002); relatively little explicit emphasis has been placed on the role of frameworks in developing policy experiments. Without testing hypotheses about the links between policies and outcomes, however, indicator systems may do little more than promote ad hoc policy responses, possibly even prolonging the transition to fisheries sustainability. There is, therefore, a need to use a framework that can be used for both the design and monitoring of fisheries policy experiments.

The Institutional Analysis and Development (IAD) framework (Ostrom, 1990, 1999) is a robust framework that has been used extensively to design policy experiments and empirically test theories and models linking institutions and the sustainability of common pool resource systems (Ostrom *et al*., 1994). The strength of the IAD framework is derived from its systematic theoretical focus on the impact of rules and norms on individual incentives in complex ecological-economic systems, its empirically-oriented focus on outcomes (including the transaction costs of management) and by its accounting for dynamic system interactions at multiple tiers of analysis (Ostrom, 1999). To date, however, the IAD framework has not been used to organize indicators of sustainability.

In this paper, I present a modified IAD framework that transparently encompasses both the PSR and Sustainable Livelihoods frameworks, thus providing a platform for designing, monitoring and communicating the results of ecosystem-based fisheries management policy experiments. The framework encourages analysts to organize indicators to take full account of the ecological, social and institutional variables that influence and shape the incentives and behavior of individuals and organizations. Further, there is a clear differentiation between aggregate patterns of behavior (*e.g.*, fishing effort), the impacts those behaviors have on capital assets (*e.g.*, species depletion, rent capture), and the threats that those impacts pose to capital assets (based on societal goals and fishery management objectives). Finally, societal responses to threats to capital assets are clearly differentiated through the investment choices that various sectors of society (private, public and civil society organizations) make in response to those threats. Investments can be made in the capital assets themselves or in institutions that influence that shape human behavior. While this paper focuses on fisheries management, the modified IAD framework can be applied to other renewable resource systems.

The Institutional Analysis and Development Framework

The IAD framework was developed by scholars at the Workshop in Political Theory and Policy Analysis, Indiana University (Ostrom, 1990; Ostrom *et al*., 1994) as a multidisciplinary tool to frame policy research on public goods and common pool resources at multiple levels of analysis. It does this by facilitating the organization and analysis of specific policy problems, and by identifying the universal elements that policy researchers need to consider. It was originally used for studies of metropolitan public services and later applied in a wide variety of fields, including the study of governance systems, donor-sponsored international development infrastructure projects, and international political order. A common theme running through the diverse research is that an institutional framework can be productively applied to the study of public and quasi-public goods that require cooperation if long-term sustainability is to be achieved.

The general elements of the IAD framework are illustrated in Figure 5-1. When conducting an institutional analysis, the analyst first identifies the ‘action arena’ or the focus of analysis that is of primary interest. In ecological-economic analyses, a geographically explicit action arena accounts for the behavioral linkage between contextual variables and rules-in-use, on the one hand, and ecological, social and economic outcomes on the other. For example, Yandle (2001) conducted an institutional analysis of the Individual Transferable Quota (ITQ) system in New Zealand in a broad analysis of multiple fisheries at a national scale. Rudd's (2003) analysis, on the other hand, focused on a single vulnerable species, Nassau grouper, in a geographically compact area in the Turks and Caicos Islands.

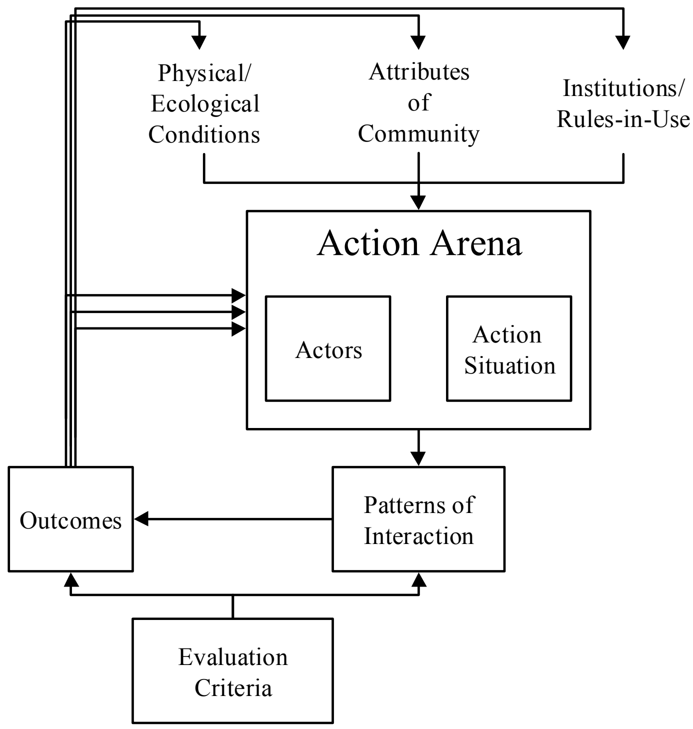

Figure 5-1 – General elements of the Institutional Analysis and Development (IAD) framework (modified from Ostrom *et al*., 1994)

In all institutional analyses, the contextual variables that frame and constrain the action arena need to be specified, including variables relating to the physical and material world within which the actors interact, the attributes of community, and the institutions or 'rules-in-use' that govern behavior. Institutions are crafted by humans to increase predictability and provide order in uncertain environments, thus increasing the propensity of people to cooperate and facilitating the production of public goods. Institutions are comprised of formal rules and/or informal prescriptions (*e.g.*, norms, taboos) that permit, prohibit or require certain actions or outcomes while specifying explicit material or implicit social sanctions for breaking rules (Crawford and Ostrom, 1995).

Given a set of external ecological, social and institutional constraints, actors (individuals and organizations) consider the costs and benefits of various behaviors, and act according to their perceived incentives. These incentives are based on their underlying values and preferences, the information they have about the state of the world and the intentions of other actors (which may be incomplete and/or imperfect), and the threat of material or social sanctions. Their aggregate patterns of interaction (*e.g.*, fishing effort) lead to outcomes (*e.g.*, landings) that can be evaluated according to socially relevant criteria (*e.g.*, are landings less than Total Allowable Catch limits designed to protect resource productivity?). Outcomes dynamically feed back to both the action arena and to higher levels.

IAD analyses can also be carried out at these higher levels of decision-making (Ostrom, 1999). In 'collective choice' situations (Figure 5-2), analyses focus on how rules regarding resources access and harvesting methods are formulated, rather on the day-to-day operational consequences of those rules. At an even higher level, in 'constitutional' situations, analyses address questions of whom is eligible to craft collective choice level rules and how their preferences are aggregated (*e.g.*, committee voting rules). Note that the term 'constitutional' refers to the process of articulating and aggregating the preferences of various members or sectors of society, not to the 'Constitutions' of various jurisdictions per se. Cultural factors (*i.e*, relatively stable long-run values and beliefs) shape and influence decisions at all levels.

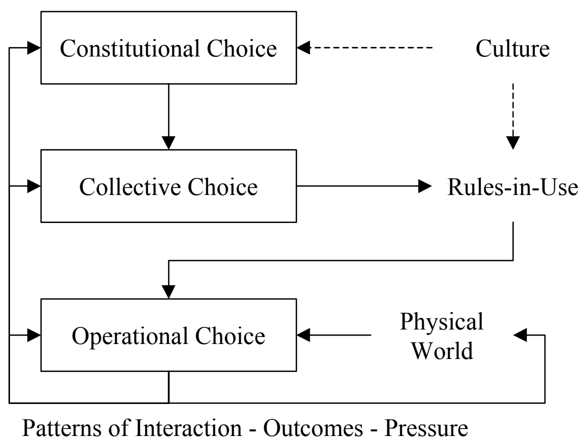

Figure 5-2 – Multiple levels of institutional analysis

Modified IAD Framework for Ecosystem-Based Fisheries Management

It is useful to recast the IAD framework in terms that are more familiar to fisheries scientists and managers, and that are useful for developing specific indicators of sustainability. Given the historical emphasis of IAD research on the production and appropriation of common pool resources, a modification of the original IAD framework to specifically include capital assets is reasonable. In the following section, I outline the structure and features of a modified IAD framework (Figure 5-3), showing the five types of capital assets used in the Sustainable Livelihoods framework and incorporating terminology commonly used in PSR and related frameworks.

Capital Assets

Each capital asset is composed of a 'stock' that provides a 'flow' of goods or services that people can use to help them meet objectives and achieve their aspirations. All types of capital share two fundamental distinguishing characteristics: each capital investment entails an opportunity cost (savings or consumption foregone) and each can be used by people to help them increase their well-being. For indicator systems, it is important to consider what comprises the capital stock, what comprises the flow, and what quantity of the flow can be used sustainably. For a simple single-species fishery this may be easy but for coastal ecosystems that provide important nonmarket services (*e.g.*, life support, resilience, maintenance of coastal lifestyles) the issue of identifying stocks and usable flows is a much more complex matter.

Natural Capital

Ekins (2003) summarizes how natural capital provides four classes of fundamental environmental functions: source functions (*e.g.*, providing fish used for consumption); sink functions (*e.g.*, assimilation of upland wastes); life support functions; and human health and welfare functions, including the provision of non-use values.

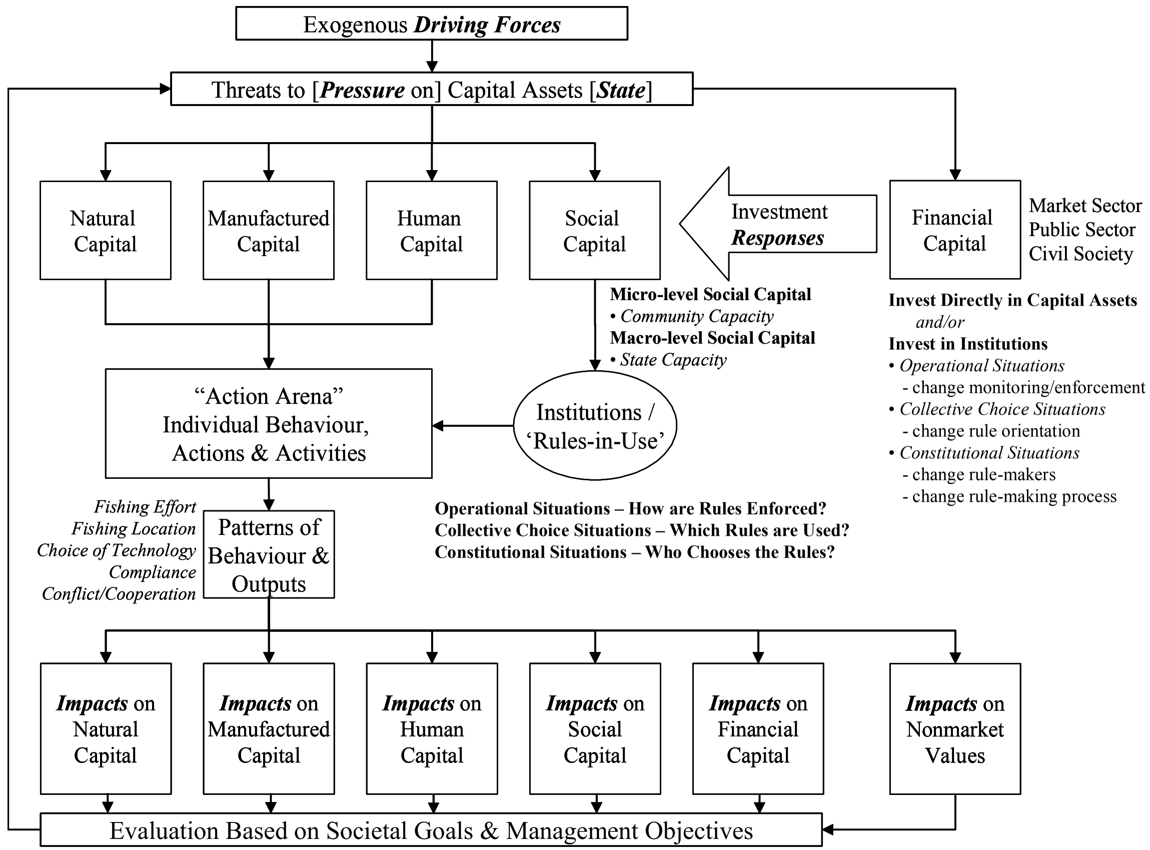

Figure 5-3 – A modified Institutional Analysis and Development Framework for ecosystem-based fisheries management

The strong sustainability principle is broadly accepted by fisheries managers and is institutionalized under UNCLOS and the FAO Code of Conduct for Responsible Fisheries (Garcia and Staples, 2000). It proposes that substitutability between natural and human-made capital is limited, implying that relative rates of resource depletion must be less to or equal the natural rate of renewal. In the 'simple' situation where only fish extraction (a source function) is considered, the problem is one of determining sustainable yield. In tropical Pacific inshore ecosystems, for example, lightly exploited or unexploited reefs typically have standing stock biomass of 50-100 t/km^2 and limited data suggests that finfish yields in the range of 5- 20 t/km^2/yr are probably sustainable in long-run (Dalzell *et al.*, 1996).

For pragmatic reasons, many fisheries managers still focus only on the extractive value of directed fisheries. Other conservation objectives beyond sustainable fisheries yield are, however, increasingly important in ecosystem-based fisheries management (Gislason *et al.*, 2000). A structurally sound ecosystem (*e.g.*, high biodiversity, undegraded habitat, balanced trophic structure) can provide indirect services for humans (*e.g.*, 'insurance' due to ecological resilience) as well as non-use values (*e.g.*, the value people derive from knowing that vulnerable species are preserved for future generations). Considering all four basic classes of natural capital functions implies that a suite of ecological indicators is needed for comprehensive ecosystem-based fisheries management, a much more challenging proposition than just developing indicators about the sustainability of fisheries production for select target species.

Using traditional fisheries management approaches (*i.e*, single species surplus production or dynamic pool models), the narrowly defined management focus tends to be on either maximum or target fishing mortality rates (*e.g.*, F_{MSY}, $F_{0.1}$, $F_{40\%}$ – see Caddy and Mahon, 1995). Biomass-based reference points, which depend on demographic parameters, are becoming more common and advances have been made to account for shifts in life history parameters such as natural mortality rate (Collie and Gislason, 2001). Trenkel and Rochet (2003) review a variety of fishery community indicators and found that they varied substantially in reliability and power, but that relatively simple indicators, such as the proportion of noncommercial species in the community, are promising indicators suitable for ecosystem-based fisheries management applications. Trophic level indicators, based on food web models, have also been applied by Pauly *et al.* (1998) and suggest that fishing industries have sequentially targeted species lower on the food chain after depleting higher-level piscivores.

Obviously the development of appropriate indicators for specific situations will need to consider data availability and cost, the relative importance of various ecosystem services, and the relative importance of exogenous driving forces in the system. This will be a substantial challenge in some regions, especially where historic data or resources for extensive scientific monitoring are limited.

Manufactured Capital

Manufactured (= 'human-made', = 'physical') capital is the stock of produced assets that people can use over time. It includes market sector equipment, facilities, technology and software devices that are designed to increase the effectiveness or efficiency of the process of transforming resources, including fish, into commodities that contribute to human well-being. It also includes physical infrastructure in rural communities (*e.g.*, wharves, roads, schools, communications) that are needed to support marine industries and supplied by public or market sector organizations.

In the case of fishing fleets, one measure of capacity is simply based on the physical assets used in the fishery (*e.g.*, the vessels, nets, electronics). This concept of manufactured capital is of limited use because it does not account for increases in under-utilized variable inputs (*e.g.*, labor, fuel) that can be used to compensate for reductions in the overall number of vessels fishing. A more appropriate measure of manufactured capital is capacity output, the maximum level of production that fixed inputs (*i.e*, vessels) can support when all the variable inputs (*i.e*, labor, fuel) are fully utilized, given the present state of technology and resource availability (Kirkley *et al*., 1999). The concept of capacity output can be applied equally well to other types of manufactured capital.

Developing indicators of manufactured capital stocks is relatively straightforward compared to other capital assets because they are closely tracked by many business and government agencies, largely using traditional accounting practices. Ideally, comprehensive cost and earnings surveys would be used to develop capacity-related indicators (as well as rent capture indicators). As this information in often unavailable, techniques such as Data Envelope Analysis (DEA) can be used to develop capacity-related indicators based on physical inputs and outputs when prices are unavailable (Kirkly *et al*., 1999). Dupont *et al*. (2002) have demonstrated the usefulness of this DEA approach in the management of multi-species fisheries by examining the impact of an individual transferable quota (ITQ) system on capacity utilization in the multi-species Scotia-Fundy groundfish fishery. They found evidence that institutional change –

implementation of an ITQ policy regime – did appear to reduce aggregate excess capacity.

Human Capital

Human capital is an asset that reflects the individually-possessed knowledge, competencies, education and skills embodied in individuals that help them increase personal, economic and social well-being (OECD, 2001). Individuals with higher levels of human capital can more effectively and efficiently use other capital assets to create goods or services that contribute to well-being.

Simple indicators of human capital, including educational attainment, job experience and on-the-job skills training, are often available from secondary data sources. Human capital is important for fishing because fishing requires a broad mix of navigation, maintenance and people skills as well as a thorough understanding of fish behavior, ocean conditions and risk management. The 'skipper effect' – a persistent differential in catch rates over time – has been shown to be related to human capital proxies such as age, education and fishing experience (Kirkley and Squires, 1999).

Social Capital

The broad complex of social networks, protocols, norms and values comprise social capital (Rudd, 2000; Woolcock and Narayan, 2000; Krishna, 2002). An increase in social capital can reduce the cost of fishery production by increasing the likelihood of successful coordination and collective action. If social networks and norms are viewed as the capital stock, the flow of services from social capital are information (about the world and the behavior of others) and trust, respectively.

At the micro-level of individuals and communities, social capital can serve three functions, as an asset that can be used for either 'bonding', 'bridging' or 'linking' (Krishna, 2002; Woolcock and Narayan, 2000). Bonding results when strong intra-community ties give kin and communities a sense of identity and common purpose based on trust. Bridging results when communities endowed with diverse inter-community ties are in a stronger position to confront problems and take advantage of economic opportunities. Linking refers to the capacity of community members to gain access to decision-makers in private and government organizations. At the macro-level, social capital helps facilitate trust and exchange amongst strangers through legal infrastructure and institutions such as property rights. Rudd *et al*. (2003) provide a more extensive discussion of social capital in a fisheries context.

At the community level, surveys about social structure, trust and access to decision-makers have recently been developed to assess the effectiveness of agriculture, forestry, health and community development programs (Grootaert and van Bastelaer, 2002; Krishna, 2002) and these types of indicators should be applicable in the fisheries field. Macro-level indicators of some broad aspects of governance capacity are commonly available at the national level (*e.g.*, O'Driscoll Jr. *et al*., 2003). These indicators typically consider issues such as legal frameworks, fiscal capacity, staff expertise, policy consistency, internal communications and the level of corruption in government. Other indicators of institutional capacity are being developed at the micro- (*e.g.*, Elher, 2003) and macro-level (*e.g.*, Spangenberg *et al*., 2002) but may require substantial work to customize for fisheries governance.

Financial Capital

Financial capital is of special interest because it is generated by the production process itself, can be re-invested in any other type of capital and is highly mobile. The stock of financial capital can be viewed as the level of wealth a society possesses and the flow as the amount that it available for other investments on an ongoing basis. Market, public and civil society organizations all have financial capital that may need to be considered in fisheries indicator systems. Indicators of the level of wealth in a society can be measured using standard accounting measures and census data although care must be taken to account for the underground economy in some regions.

Institutions

The institutions (rules-in-use) that influence actor incentives and behavior include both formal and informal rules. Formal rules specify actions or outcomes that are permitted, prohibited or required, and prescribe formal sanctions for rule violation (Crawford and Ostrom, 1995). Norms similarly prescribe acceptable and unacceptable behaviors but do not impose formal, legal sanctions on violators. The IAD framework recognizes three main situations in which institutions operate: operational, collective choice and constitutional.

Operational Situations

In operational situations, the focus is on the day-to-day impacts of existing rules and norms on the incentives of actors. The primary activities at this level – gathering information, monitoring fishing activities and enforcement of formal or informal rules – are costly and government, resource users or some combination of the two may bear these fishery management transaction costs (*e.g.*, Sinclair *et al*., 1999). From a monitoring perspective, we are interested in how changes in the quality of information, monitoring intensity and enforcement of existing institutions will change actor incentives and behavior and help achieve management objectives.

A wide range of management tools can be used in operational situations. In the complex, multi-species Scotia-Fundy fisheries (Sinclair *et al*., 1999), for example, fishers in different segments of the industry may be required to record and report landings, hail in (report by radio their estimated landings by species prior to docking), submit to dockside monitoring by third party monitors, carry on-board independent observers and/or install 'black box' vessel tracking gear. Depending on the fishery, some or all of these costs may be borne by the market sector while government agencies retain responsibility for land, sea and air patrols, data analysis, case building and prosecution of violators.

In general, if a threat of detection and punishment is not credible, individual behavior is unlikely to change significantly even with formal rules in place. It is also possible, however, that behavior will still not change if the formal rules do not coincide to some degree with social norms; if regulations are imposed from outside and/or fishers feel that the formal rules are illegitimate or unfair, there may actually be incentives to violate those rules (see Crawford and Ostrom, 1995). Indicators of operational level information gathering, monitoring and enforcement can be quite simple (*e.g.*, proportion of fishers reporting landings, dockside monitoring coverage, number of patrols, cases taken to court, type and amount of sanctions).

Collective Choice Situations

The focus in collective action situations is on the types of rules that are chosen for a fishery. Fishing rules can be cleaved according to a variety of factors, including whether they are input- or output-oriented (*e.g.*, do they focus on restricting fishing power or the quantity of fish caught?), whether they are production- or conservation-oriented (*e.g.*, do they focus on maximizing yield or on protecting vulnerable species and habitats?), and whether they are regulatory- or market-oriented (*e.g.*, do rules focus on directly controlling fishers' behaviors or on providing financial incentives that induce behavioral change?).

Adequate indicators of fisheries management orientation are lacking at the current time and need to be developed for comprehensive indicator systems. In theory it should be possible to develop these indicators based on analyses of fisheries management regulations, controls and legislation. A general approach is to use multidimensional scaling or other ordination method to construct a management orientation index based on a series of expert ratings for key qualitative characteristics of alternative management approaches.

Constitutional Situations

At the constitutional level, the primary focus is on whom has the rights and power to set lower level rules regarding access to, and utilization of, resources. The rules-in-use at this level are concerned with the aggregation and expression of stakeholder and societal preferences. This is largely a matter of property rights that define what level of control resource users and other actors have over resource access and appropriation. Pure regulatory approaches to achieve fisheries sustainability are today seen as untenable and, as a result, there has been an increasing move to rights-based fisheries or co-managed industrial and artisanal fisheries. In co-managed fisheries, there is some balance in shared decision-making between market, public and community organizations (*e.g.*, Yandle, 2001).

Schlager and Ostrom (1992) developed a matrix of property rights and categories of rights holders that are useful for developing indicators of co-management and property rights in fisheries. 'Claimants' hold rights of resource access, resource withdrawal, and participation in the management process, including decisions concerning harvest limits and production technologies. Many 'rights-based' fisheries currently operate are at this level. At the next higher level of property rights, 'proprietors' hold additional rights to determine who may access and harvest fish. At the highest level, 'owners' – who may be individuals, corporations, communal groups, or government – hold all lower level rights and are also allowed to transfer or sell their rights subject to specific conditions.

Actors

Fisheries managers need to consider how certain policy options alter human incentives. Is it necessary or desirable to monitor actor's values so that we can model the effect of policy alternatives on various incentives and the strategic interactions they imply? For 'corporate' actors (*e.g.*, market, public or non-governmental organizations), goals may be clearly spelled out in vision or policy statements, or implicit just by the nature of the business. Individual values can be more difficult to assess but are important for understanding markets and organizational performance (*i.e*, do the incentives of individuals clash with those of their organization?). Monitoring values and preferences over time may also be valuable in situations where the effectiveness of education and awareness campaigns needs to be assessed. That said, the micro-level research entailed in assessing the incentives of individuals may be infeasible for broad monitoring programs and implicit models of human behavior would then need to be adopted. It should be possible, however, to use the IAD monitoring system to test working hypotheses regarding underlying preferences using observations of aggregate behavioral patterns.

Patterns of Interaction

Actors make choices based on their own preferences, objectives, or mandates (in the case of government agencies), the costs and benefits that they assign to alternative actions and outcomes, and strategic considerations (*e.g.*, expectations of the behavior of others). These individual choices lead to aggregate patterns of interaction relevant for ecosystem-based fisheries management. Using the IAD framework, behavioral patterns of interaction are distinguished from impacts or threats. Patterns of interaction result directly from the aggregate effects of individuals going about their day-to-day decision-making and usually five main patterns will be of interest: fishing effort (or used of other resource users); fishing location; choice of technology; compliance with existing rules; and conflict or cooperation with other resource users.

Effort

Fishing effort can be readily monitored using existing indicators in many fisheries. Vessel-day or person-day measures of fishing activity are suitable and other similar measures can be applied for other types of resource use.

Location

Logbook data on fishing location is also available for many, but not all, fisheries. Indicators of fishing location can be as simple as effort directed at specific geographic areas to more complex measures of spatial extent and concentration (*e.g.*, Salthuag and Aanes, 2003).

Choice of Technology

If capacity output is used as a measure of fishing fleet manufactured capital, then capacity utilization, the proportion of overall capacity that is actually used, is a useful indicator of fleet effort (*i.e*, how much of the available 'flow' of services from manufactured capital is actually used). In some countries, the choice to fish using illegal technologies (*e.g.*, small-mesh nets, underwater lights, bleach, cyanide, dynamite) also needs to be factored into indicator systems. While illegal fishing practices may be difficult to track, market sales of some of the 'tools' used to fish illegally may sometimes be more readily available.

Compliance

Compliance with formal rules and regulations deserves special attention in ecosystem-based fisheries management. If there are strong incentives for fishers to cheat, then the likelihood of achieving fisheries sustainability can be severely compromised. The degree and type of monitoring, the types of sanctions for contravening rules, and local social norms all influence the likelihood of cheating. People violating rules will not usually be forthcoming about their activities and monitoring compliance is, as a result, one of the biggest challenges in developing effective monitoring systems for ecosystem-base fisheries management.

Simple indicators of compliance such as the number of violations noted, charges laid and successful prosecutions are possible but may be of limited value for tracking actual compliance levels. More sophisticated indicators are available for situations when sufficient fishery monitoring data is available. Allard and Chouinard (1997), for example, developed a non-parametric indicator of discarding based on the empirical length-frequency distribution of the catch. By comparing the indicator values derived using at-sea observer coverage and those derived using shore-based dockside monitors, it is possible to infer which vessels or industry segments engage in illegal discarding practices.

Cooperation/Conflict

Conflict and cooperation are also behavioral patterns resulting from individuals interacting. In many cases, opinion surveys of resource users about their subjective perceptions on the number of, and trend in, conflicts or cooperative acts in their fishery may be more useful than indicators based on officially recorded conflicts. For example, Bennett *et al*. (2001) used semi-structured interviews of fishers and processors to assess levels of conflict in a multi-country study of conflict management in artisanal fisheries.

Impacts and Pressure

The aggregate interactions of individual and corporate actors making day-to-day decisions lead to patterns of behavior that can be monitored, in many cases, using relatively simple indicators and with existing data. There should then be a testable causal link between these patterns of behavior and their impacts on various types of capital assets. For example, does increasing fishing effort lead to a decline in resource abundance (a relatively straightforward question), a loss in biodiversity or a loss of ecosystem resilience (a more difficult question)? In many cases, the simplest approach to monitoring human impacts on capital assets will be to simply track changes in the overall stock of capital over time rather than focus on more problematic indicators of difficult-to-measure flows. That is, it may be pragmatic to track changes in biodiversity rather than develop indicators of resilience, for example, if there are strong theoretical grounds for linking the two.

Impacts themselves do not necessarily pose threats to capital stocks; only when the appropriation of the flow of services from capital assets exceeds their rates of renewal does the impact become a “pressure”. Impacts therefore need to be defined in terms of relevant evaluative criteria. The choice of evaluative criteria depends on the vision, goals and objectives of actors within the fishery, broader societal interests and, for life support systems, some basic ecological limits.

It should be noted that unambiguous indicators of behavioral patterns could have multiple or conflicting impacts on capital assets. A high level of capacity utilization may, for instance, have impacts on natural capital (*e.g.*, target species depletion), manufactured capital (*e.g.*, depreciation) and financial capital (*e.g.*, resource rents are being generated).

Evaluative Criteria

Evaluative criteria define what impacts are viewed as acceptable or unacceptable by specific actors or society as a whole. The challenge in developing criteria that can be used in indicator systems is to specify them is such a way that there are measurable and meaningful indicators to evaluate them. Three broad dimensions – environmental, economic and social – are commonly viewed as salient for classification of sustainability outcomes (*e.g.*, UN, 2001). In the IAD-based framework, evaluative criteria can be developed more explicitly for each capital asset.

While criteria will vary from application to application, some general observations are possible. One of the most important is that developing objectives for each capital asset makes assumptions regarding sustainable development explicit. If non-declining natural capital is not adopted as a goal of a policy experiment, that decision will be transparent and can be assessed in light of the actors involved, their property rights and constitutional-level processes that shape resource access and utilization rules.

Wealth generation is an important criterion for all fisheries systems and evaluative criteria should consider three components: resource rent capture, the transaction costs of management, and retention of wealth within the focal system. Rent capture focuses largely on the economic efficiency of resource users while the transaction costs of management are potentially borne by market, public and civil society sectors. Many regions will also have specific objectives regarding the retention of economic benefits within their own region (*i.e*, a high multiplier effect). In addition, it will be important in many systems to consider the nonmarket value of one or more ecosystem services so that economic well-being as a whole is considered, rather than the narrow subset of financial impacts. Finally, some evaluative criteria will relate specifically to economic equity, often based on the principle of 'user pay' or to the explicit protection of vulnerable segments of society.

Further, there may be broader evaluative criteria appropriate for some situations, including institutional adaptability, conformance with general social norms and values, and various governance criteria such as bureaucratic accountability or transparency (*e.g.*, Ostrom *et al*., 1993). Adaptability is an evaluative criterion that deserves special attention in ecosystem-based fisheries management. Wilson (2002) argues that matching ecological and institutional scale in complex adaptive fisheries systems is the central element needed to ensure adaptability for ecosystem-based fisheries management. Management should focus on maintaining long-run system stability while the flow of ecosystem services cycle within normal bounds, thus allowing resource users to recognize resource abundance patterns and maintain sufficient flexibility to adjust to those cycles. Sproule-Jones (1999) notes that different actors with different bundles of property rights incur different transaction costs. The lower the transaction costs, the more adaptable actors will be and, thus, more willing and able to experiment and innovate in ecosystem-based fisheries management policy experiments. This suggests that monitoring constitutional-level institutions such as property rights may provide a reasonable proxy for adaptability of governance systems.

Driving Forces

It is also important to consider that not all pressures on capital stocks are endogenous but that exogenous driving forces (*e.g.*, demographic, environmental or technological change) also exert pressure on capital stocks. Jameson *et al*. (2002), for instance, argue that MPA managers often fail to meet their management objectives because of threats from exogenous environmental stressors. Macroeconomic market forces are also important in many fisheries because of the speed at which market prices, interest rates and other key variables change, and due to the importance of market price and input cost variables in influencing day-to-day resource use decisions at the operational level.

Investment Responses

When capital assets, and hence the productive capacity for humans to meet their objectives and fulfill aspirations, are threatened, society can respond in a number of ways. All responses, however, can fundamentally be viewed as investment decisions by market, public or civil society organizations.

Investments in Natural Capital

Projects or programs that protect or improve important habitat, directly enhance the production of target species, or engage in predator control are all types of investments in natural capital. They they may sometimes be problematic due to our limited understanding of ecosystem dynamics (*e.g.*, cod stock enhancement – Svåsand *et al*., 2000). Indicators of investments in these activities should account for time, in-kind contributions of supplies and equipment, and for direct financial contributions from market, public and civil society organizations.

Investments in Manufactured Capital

Manufactured capital assets depreciate and require investment to maintain. Investments by the market and public sectors in manufactured capital falls within the realm of traditional economic theory and there are many expenditure-based indicators that can be used to track these types of investment. Investments will not only be targeted at fishing fleets, but also at the development of new technologies (often a joint effort between market and public organizations) and the infrastructure needed to support industrial use of marine resources.

Investments in Human Capital

It is now widely recognized that human capital can be developed through investments in the training, education and health of workers (Helliwell, 2001). Data on many of these investments can be tracked using conventional sources such as census surveys. Other non-traditional investments such as the documentation and dissemination of local ecological knowledge are also clearly investments in human capital. Investments by civil society organizations in human capital may be as, or more, important than public sector investments for some fisheries.

Investments in Social Capital – Community Capacity

Public and civil society organizations have become increasingly aware of the importance of investments of social capital over the past decade, largely due to the influence of Putnam *et al*. (1993) and a plethora of more recent work (*e.g.*, Helliwell, 2001; Grootaert and van Bastelaer, 2002). Indicators of investments in social capital should take into account all three aspects of micro-level community capacity building (bonding, bridging, and linking). This implies developing indicators of investments in social structural variables (*e.g.*, communications, social networks, venues for dispute resolution), norm-seeding (*e.g.*, awareness and stewardship campaigns) and leadership-building activities (*e.g.*, mentoring programs). Accounting for civil society investments in community capacity is particularly important in some regions. The Pew Foundation, for instance, has allocated substantial funds for marine-related efforts and some of that funding has been channeled, via Pew Fellows, to community capacity building in artisanal fishing and aboriginal communities (Malakoff, 2002).

Investments in Social Capital – State Capacity

Indicators can also be used to track investments in the legal and institutional infrastructure that comprises macro-level social capital. For example, the Government of Canada has actively invested to increase fisheries governance capacity (DFO, 2000) by enhancing communications within and between departments ('horizontality' initiatives), aligning policies, building core capacity (investments in infrastructure and technology), and improving organizational effectiveness. International donor agencies also commonly target investments towards government management capacity in developing countries.

Investments in Institutions

Investments can also be made in institutions at the operational, collective choice and constitutional levels by (1) increasing the level of information gathering, monitoring and/or enforcement activity for existing rules, (2) changing the status quo rules governing the fishery, or (3) changing the fisheries governance rule-making process itself. These investments collectively comprise the primary transaction costs of fisheries management.

Investments in Institutions – Operational Situations

Investments in better information, monitoring and enforcement can be made at low cost relative to investments needed for higher-level institutional change. Enforcement costs can escalate rapidly and become prohibitively expensive, however, if local social norms are not congruent with formal rules (Ostrom, 1990). Thus, investments in monitoring and enforcing existing rules are important for successful resource management, but are not alone sufficient for long-run sustainability.

Most public agencies closely track expenses devoted to monitoring, compliance and enforcement. When publicly available, this data provides the basis for indicators of investment in operational-level institutions. Fishers and other resource users may, in addition, contribute significant in-kind and financial resources to self-monitoring. In some cases these investments may be well documented – for example, there has been a recent move in Canadian fisheries to management cost recovery by passing on certain well-known costs (*e.g.*, observer monitoring) to industry (Sinclair *et al*., 1999). In self-governing or artisanal fisheries, however, indicators of these investments may be more difficult to develop because much of the monitoring activity may be a 'byproduct' of routine fishing activities.

Investments in Institutions – Collective Choice Situations

Another response to threats against capital assets is to change the formal rules governing behaviors or outcomes that are required, prohibited or permitted by law. Because changing rules is a higher-level process, societal investments aimed at changing the formal rules-in-use will be more expensive than simply increasing monitoring and enforcement. Indicators at this level focus on the costs of activities such as the development of management plans, publication of rule changes and costs associated with legislative change. At the collective choice level, it is also possible for government, NGOs or other ‘norm entrepreneurs’ to effectively invest in norm-seeding activities that seek to change the informal rules-in-use (Sunstein, 1996).

Investments in Institutions – Constitutional Situations

If sustainability outcomes consistently fail to meet broad societal expectations, there may be increasing calls for political changes about the rule-setting process itself. Constitutional elvel rules about the articulation of stakeholder interests are those that refer to selecting and representing stakeholders for the governance process. Aggregation rules deal with the transformation of diverse stakeholder interests into actions, often specifying the timing or frequency of meetings and technical rules about voting procedures needed to resolve conflicts. Constitutional level change is more expensive again relative to lower level changes that simply devote more resources to enforcement or shift management orientation. At this level, appropriate indicators of investment relate to resources dedicated to litigation, political lobbying and core investments in strategic decision-making processes by public, private and civil society organizations. Data is likely to be much more difficult to come by at this level and in-kind contributions to the process very important.

Concluding Remarks

Successful implementation of ecosystem-based fisheries management policies requires that managers consider multiple ecological and socio-economic objectives in transdisciplinary policy experiments. A modified IAD framework encompasses both the structurally-oriented Sustainable Livelihoods framework and the process-oriented PSR framework and is thus well-suited for designing and monitoring policy experiments because of its multi-level causal linkages and flexibility.

Causality is a key consideration in ecosystem-based fisheries management policy experiments. There is clearly a need for directed policy selection in ecosystem-based fisheries management (Walters, 1997) and this implies that analysts and decision-makers must have some understanding of how potential control variables relate to anticipated policy impacts. In complex ecological-economic systems, our understanding will always be incomplete but directed policy experiments will help increase this understanding of fishery systems. The IAD framework is flexible in that it permits policy makers to not only test specific policy hypotheses, but to also more broadly test competing theories and underlying assumptions (Ostrom, 1999). It can also help frame studies that seek to identify necessary and sufficient conditions for multi-criteria sustainability (*e.g.*, Heikkila, 2001). Indicator systems that present only a snapshot of system status without fully considering causal connections are not nearly so useful.

The data requirements for a full monitoring system will be extensive although the amount and type of data collected will vary greatly from site to site and according to monitoring objectives. In many cases secondary sources of data – including indicators developed for PSR and Sustainable Livelihood indicator systems – can be used directly in an IAD-based monitoring system and the framework can be used to identify any key information gaps. It should be noted that resources devoted to monitoring ecosystem, socioeconomic and institutional aspects of fisheries sustainability are often only a fraction of those allocated for monitoring ecological aspects of commercial fisheries and that increased resources could have a large impact on the quality and quantity of data available for monitoring purposes.

Communication of concise information that is based on large amounts of underlying data is a challenge for monitoring systems in general. Decision-makers often have very little time to consider key implications of their decisions and they are often called on to make decisions in fields in which they have limited expertise. Indicator systems must convey critical information simply and compactly as a result; lengthy narratives or series of tables are unlikely to be closely scrutinized and of limited value. A variety of communication methods have been proposed that can be implemented within an IAD-based monitoring system, including ‘orientor stars’ (Bossell, 1999), the ‘dashboard of sustainability’ (Hanson, 2003) and ‘traffic light’ systems (*e.g.*, Jamieson *et al*., 2001).

Financial resources are always scarce (*i.e*, individuals, corporations, non-governmental organizations and public managers all face budget constraints) and a variety of investment options may be available to help achieve sustainability objectives. An advantage of using the modified IAD framework is that all societal responses to pressures on capital assets can be viewed in terms of the investments that different segments of society make in those capital assets directly or in the development of institutions designed to protect or enhance them. For example, financial capital might be invested in habitat rehabilitation (natural capital) that increases the sustainable flow of fish from the ecosystem, in skills development programs for young fishers (human capital), in research that improves fishing technology (physical capital), in meetings that allows fishers to share ideas and build networks (social capital), in enforcement (an operational level institutional investment), in the development of a new marine protected area (a collective action level institutional investment), or in the development of participatory democratic processes needed to effectively govern coastal ecosystems (a constitutional level institutional investment).

The benefit-cost ratio of the different investments may, however, vary greatly. One important role for policy experiments is to generate information about the relative returns from different types of investments. With this type of information, a more informed 'business case' can be made for those investments that contribute most to achieving management objectives and societal goals. Investments based on a 'marginal benefit equals marginal cost' principle should be a goal for all fisheries management systems so that those initiatives that contribute most to meeting objectives are identified and funded in a cost-effective manner. Using the IAD framework helps analysts to systematically consider a broader range of investment options than is typically considered in most fisheries management systems (*e.g.*, investments in social capital, local knowledge, processes to devolve fisheries management).

In conclusion, the IAD framework is a useful tool for designing and monitoring an entire array of policy experiments that are defined in terms of societal investments in capital assets or institutions designed to alleviate pressure on capital assets in fishery systems. It is flexible, pragmatic and has a history as a base for rigorous empirical applications. These features will be crucial if ecosystem-based fisheries management policies are to be successfully developed, tested, implemented and monitored in support of sustainable fisheries governance.

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PART 2 – NASSAU GROUPER CONSERVATION AND FISHERIES MANAGEMENT IN THE TURKS AND CAICOS ISLANDS

CHAPTER 6

THE TURKS AND CAICOS ISLANDS STUDY SITE [1]

The Turks and Caicos Islands (TCI) are located at the southern end of the Bahamian archipelago (Figure 6-1) and are comprised of three platforms: the Caicos, Turks and Mouchoir Banks. Caicos Bank is a shallow, oolitic limestone platform covering an area of about 6,140 km^2 (Olsen, 1986) and is comprised of sand (64%), mixed coral and algae (18%), coral reefs (7%), and other habitats (11%) at depths typically 1-5 m. Extensive coral reefs fringe the shelf edge and are characterized by steep drop-offs. The smaller Turks Bank (about 324 km^2) is comprised mainly of sand (43%), mixed coral and algae (29%), coral reefs (26%), and other habitats (3%) (Olsen, 1986). Mouchoir Bank (N 20.6, W 70.4; 1,109 km^2) is located east of Turks Bank and consists largely of coral and sand.

The Caicos Bank supports export-oriented fisheries for queen conch (*Strombus gigas*) and spiny lobster (*Panulirus argus*), and a domestic fishery for 'scale-fish' (primarily reef fish, including groupers, snappers, grunts and hogfish), which are most often landed opportunistically by lobster fishers.

Virtually all commercial fishing takes place on the Caicos Bank. South Caicos is the traditional home of the artisanal fleet, but landings of conch and reef fishes on the island of Providenciales ('Provo') have increased over the last two decades as Provo has been developed for tourism. Limited subsistence fishing occurs on the Turks Bank, where fishers seek reef fishes and lobsters for local use. TCI fishers seldom visit Mouchoir Bank, although there are anecdotal reports of illegal fishing for lobsters and reef fish by boats from the Dominican Republic and Haiti.

There are currently about 60 commercial licenses operating from South Caicos, 75 from Provo, and 14 from Grand Turk (Halls *et al.*, 1999). Almost all lobster is landed in South Caicos, while the conch total allowable catch (TAC) is split evenly between processors (currently three in South Caicos and two in Provo). Small 14-ft fiberglass runabouts

[1] This is an abbreviated version of a paper that will appear as (tentative title): Rudd, M.A. Fisheries production and trade of the Turks and Caicos Islands. In: *Western Central Atlantic Fisheries Catches and Ecosystem Models in Space and Time* (Zeller, D., ed.). Fisheries Centre Research Reports. Vancouver: UBC Fisheries Centre. I would like to thank Wesley Clerveaux, Scientific Officer, DECR, for supplying recent TCI production data and CPUE data for the lobster and conch fisheries. I've benefited greatly from ongoing discussions with two former colleagues – Andy Danylchuk and Mark Tupper – from my time at the Center for Marine Resource Studies, South Caicos. A number of South Caicos fishers and residents (Gangar and Franklin Lockhart, Tony

equipped with 70- to 110-hp outboards are popular for fishing as they handle waves well, are maneuverable, and can be used to reach fishing grounds up to 40-km from home port.

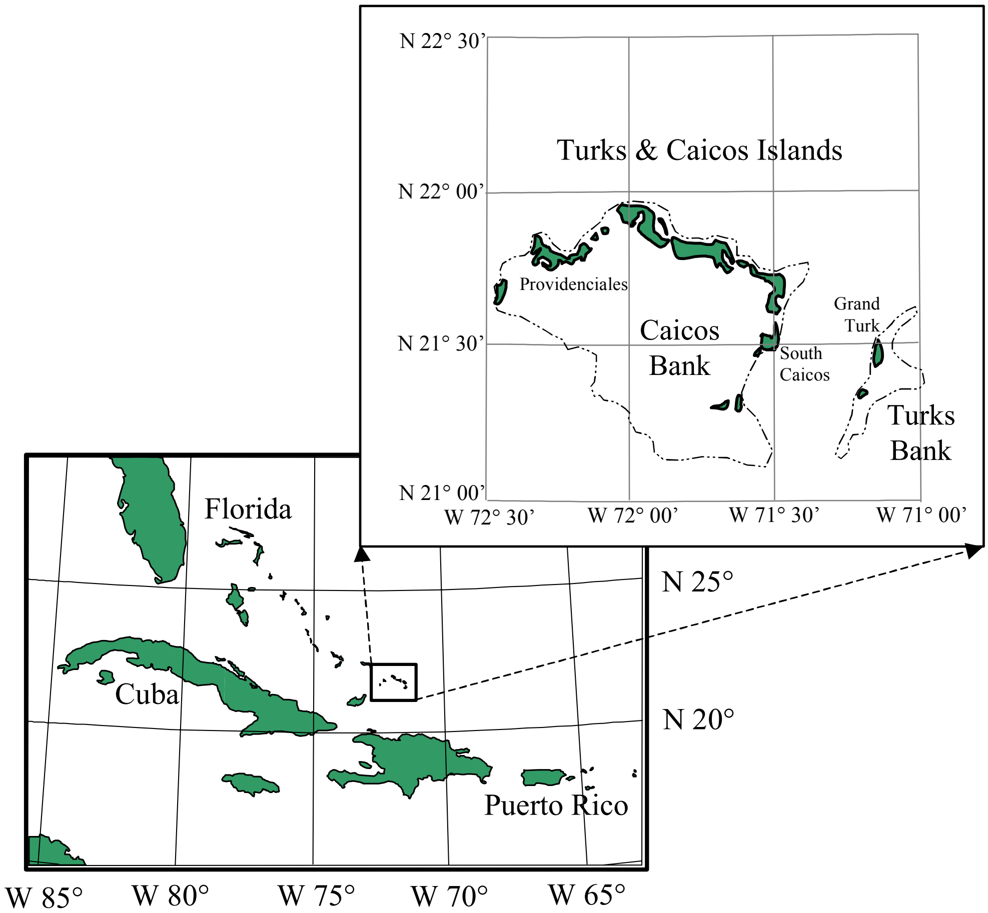

Figure 6-1 – Location of the Turks and Caicos Islands, showing approximate 100-m depth contour for Caicos and Turk Banks.

The Department of Environment and Coastal Resources (DECR) manages the conch and lobster fisheries using traditional tools. As an Overseas Dependency of the United Kingdom, the TCI has received technical support for fisheries management and has extensive [albeit imperfect] landings data. Resource assessments have been undertaken for conch and lobster (Medley and Ninnes, 1997, 1999). A combination of TACs, seasonal closures, gear restrictions (a prohibition on SCUBA being the most important), minimum size limits for conch and lobster, and other restrictions are used to manage export-oriented conch and lobster fisheries. Despite the regulations, compliance with

Morris, Sherlock Forbes, Muriel and Ed Hanchell, Cornelius Basden) have helped explain current and historical fishing in the TCI. Andy Danylchuk provided valuable comments on the manuscript.

rules has been poor since the 1960s (Olsen, 1986; Raven, 1994; Rudd *et al.*, 2001). Rampant drug smuggling from the 1980s also encouraged a culture of distrust and disregard for authority in the TCI[2].

Besides primary DECR data and reports (Homer, 2000a, 2000b, 2000c; Clerveaux and Danylchuk., 2001; Clerveaux and Vaughan, 2001), a number of reports and articles have been completed relating to fisheries and fisheries habitat in the TCI, including work sponsored by the British government (Raven, 1994; Medley and Ninnes, 1994, 1997, 1999; Ninnes, 1994; Ninnes and Medley, 1995; Medley, 1998; Halls *et al.*, 1999; Bennett and Clerveaux, 2001; Bennett *et al.*, 2001), by faculty at one time associated with Center for Marine Resource Studies, South Caicos (Green *et al.*, 1996, 1997, 1998a, 1998b; Steiner, 1999; Tewfik and Béné, 1999; Béné and Tewfik, 2001; Danylchuk *et al.*, 2001; Rudd, 2001; Rudd *et al.*, 2001; Rudd and Tupper, 2002; Rudd, in press a, b; Rudd *et al.*, in press; Tupper, 2002; Tupper and Rudd, 2002), and other miscellaneous books, reports and theses (Doran, 1958; Hesse, 1976, 1979; Nardi, 1982; Simon, 1983; Olsen, 1986; Sadler, 1997).

Driving Forces in TCI Fisheries

The introduction of snorkeling gear and freezing technology led to the development of the modern lobster fishery in the TCI in the 1950s and 1960s. The renewal of the conch fishery in the 1970s was driven by demand-side factors, as new export markets opened in Florida. More recently, tourist arrivals in the TCI have increased sharply (Figure 6-2). This has led, in turn, to an influx of permanent residents (Figure 6-3), as expatriate business owners and retirees settle in the islands. In addition, tourism development has spurred immigration from poorer neighboring countries (primarily Haiti and the Dominican Republic) as people seek service and construction jobs.

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2 According to the President's Commission on Organized Crime (1986): “Drug-related corruption has reached the highest offices of government in the British-held Turks and Caicos Islands, where in March 1985, that country's Chief Minister, Norman Saunders, was convicted of conspiracy to travel in furtherance of a drug plot and on five counts of traveling in furtherance of in illicit drug transaction. Saunders, the first foreign head of state to be convicted on drug charges, was found not guilty of more serious charges of conspiracy to smuggle marijuana and cocaine. Trial witnesses testified that Saunders accepted a total of $50,000 to allow drugs to move freely through his island chain. He planned to use the islands as a "safe-haven" for traffickers smuggling illicit drugs from Colombia to the United States.” Britain temporarily dissolved the government of the TCI in 1986 as a result of the scandal. After serving prison time in Miami, Saunders returned to the TCI and now serves as an elected representative of the Legislative Council (the TCI Government) from South Caicos.

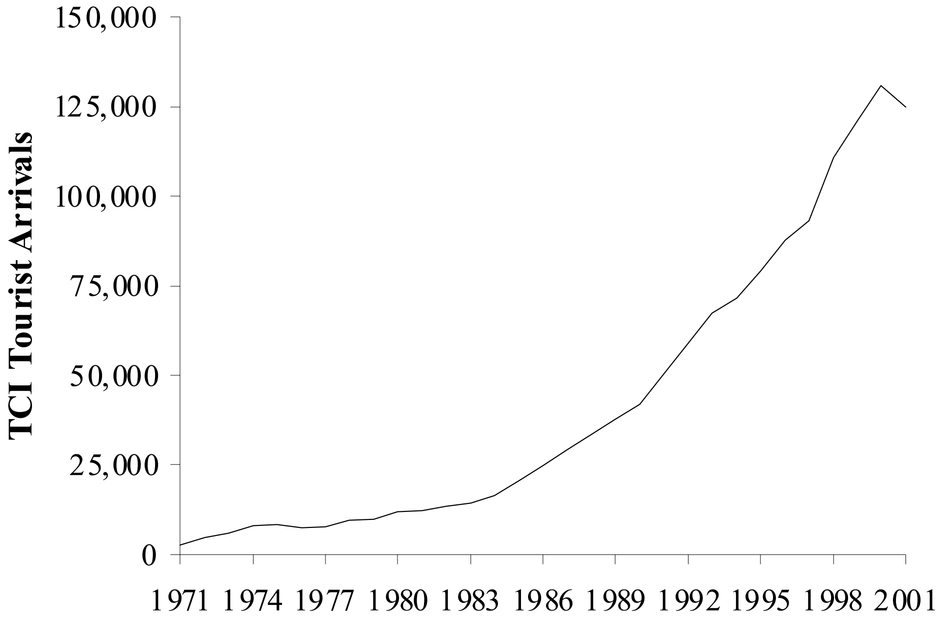

Figure 6-2 – TCI tourist arrivals, 1971-2001 (source: TCI Tourism Board, 2001 estimated)

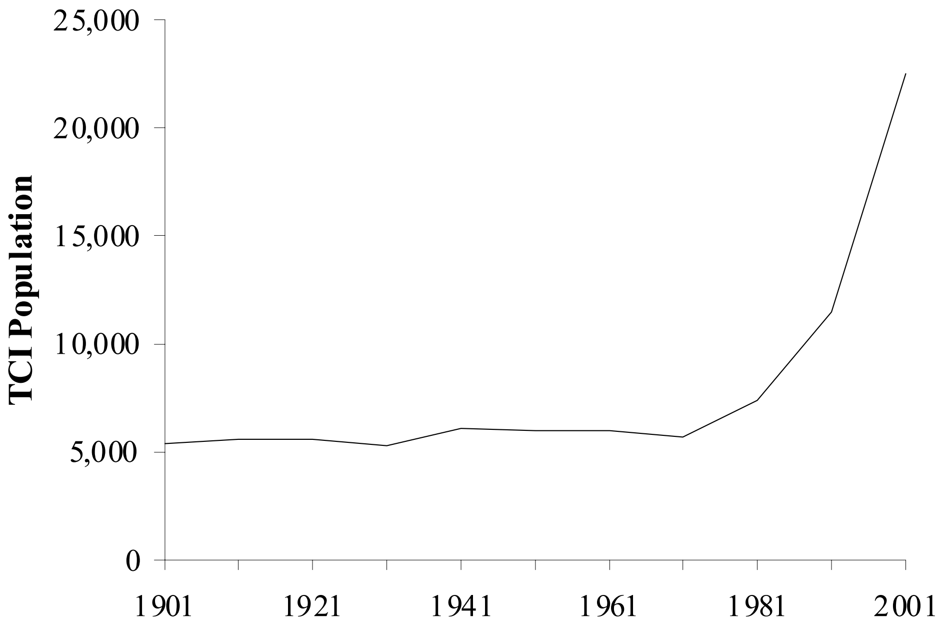

Figure 6-3 – Population of the Turks and Caicos Islands, 1901-2001

Fisheries Production

Queen Conch

In the TCI, dried conch have been traded with Haitians since the mid-1800s, when conch were bartered for fruit, sugar cane, vegetables and rum (there is very little farm production on the dry, barren limestone islands of the TCI) (Sadler, 1997). Doran (1958) documented trading records going back to 1904, and other records have been found going back to 1888 (Raven, 1994).

Wooden sailing sloops would act as collecting platforms for 2-man teams in 3 or 4 small wooden tenders (Doran, 1958). Conch were taken using waterglasses and a long conch rake. After cleaning ('knocking'), conch meat was dried for several days. Weekly expeditions took 75,000 to 125,000 conch per sloop, and each sloop made two or three such trips a year. As late as 1960, there were 60 sloops in operation, fishing from South Caicos to the outer conch grounds near Ambergris Cay (Raven, 1994). Conch hooks remained in use until the mid-1970s, but by the 1980s most conch fishing was conducted by free divers operating from fiberglass boats equipped with outboard engines (Nardi, 1982). The traditional East Harbour (Cockburn Harbour) grounds on South Caicos were closed to commercial fishing in 1993. The East Harbour Lobster and Conch Reserve was implemented in 1993 and currently provides protection for an important conch juvenile nursery ground (Danylchuk *et al.*, 2001).

Figure 6-4 shows total estimated conch production in the TCI for the period 1905-2001. Domestic consumption of conch (round weight in kg) was estimated using TCI population statistics and per capita consumption rates. Olsen (1986) estimated a per capita consumption rate of 35-kg conch per person in the early-1980s based in part on the fact that there is virtually no agriculture in the TCI. In Figure 6-4, I assume that historical per capita conch consumption is lower, at 20-kg per person, peaking at 30-kg during war years, because salt cod was readily available most of the first half of the 20th century. I assume consumption stayed at 20-kg per person in the 1950s and 60s, fell to 10-kg from the 70s to 90s, and has since fallen further to 5-kg per person. This decline is due to an influx of immigrants who do not eat much conch, the number of alternative imported foods available, and the demand (and cash payment) for export conch by the processing plants. A restaurant survey (Rudd, in press a, unpublished data – Table 6-1) estimated total TCI restaurant consumption to be about 160 t; total domestic consumption is estimated at 280 t.

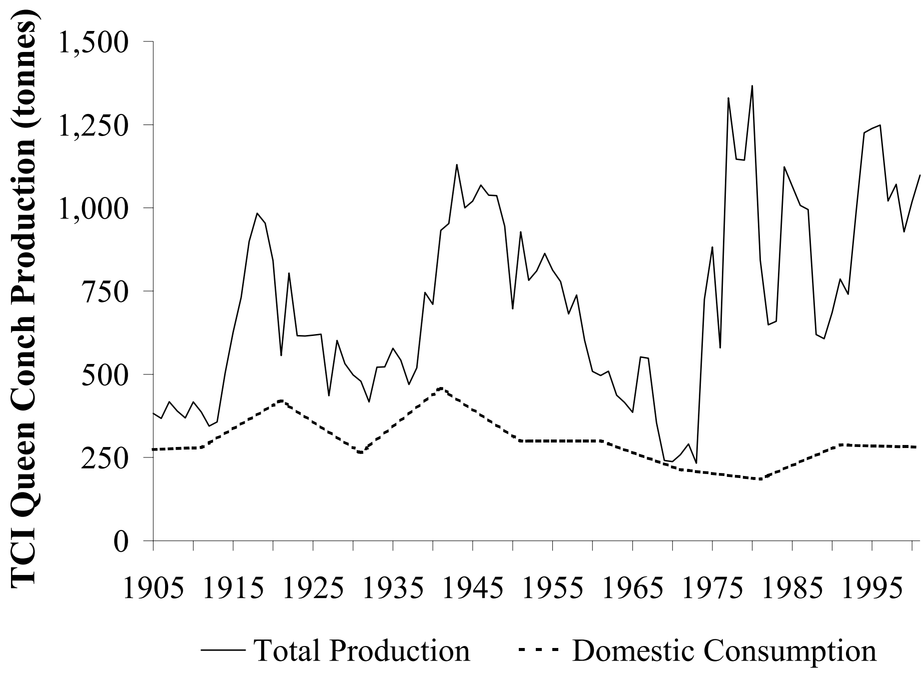

Figure 6-4 – Total conch production of the Turks and Caicos Islands, 1905-2001

	Restaurant Purchases (kg/week)	Locally Landed (%)	Tourist Restaurant Consumption (tonnes/year)	Native Restaurant Consumption (tonnes/year)	Total Dressed Weight Consumption (tonnes/year)	Meat Yield (%)	Total Round Weight Consumption (tonnes/year)
Grouper (Nassau)	725	60%	25.1	25.1	50.3	100%	50.3
Lobster	650	90%	33.8	33.8	67.6	36%	187.8
Conch	550	100%	31.8	31.8	63.6	40%	158.9
Snapper	440	50%	12.7	12.7	25.4	100%	25.4
Mahi Mahi	400	100%	23.1		23.1	100%	23.1
Wahoo	125	100%	7.2		7.2	100%	7.2
Other	265	0%	-	30.3	30.3	100%	30.3
TOTAL	3,155		133.8	133.8	267.5		483.0

Table 6-1 – Estimated annual restaurant consumption of locally landed seafood in the Turks and Caicos Islands (Rudd, in press a, unpublished data). Processing yield of 40% for conch and 34% for lobster. Local reef fishes are sold whole to restaurants. 'Tourist' restaurant consumption is based on weekly consumption and an estimated 90% survey coverage for tourist restaurants in the TCI. 'Native' restaurant consumption is approximately equal in volume to tourist restaurant consumption. Note that 30.3-tonnes of native restaurant consumption of other fish is locally landed 'small fish' (a variety of reef species) and was estimated as equaling tourist restaurant consumption of pelagics.

Commercial conch landing data were derived from a number of sources (Olsen, 1986; Raven, 1994; Medley and Ninnes, 1994, 1997, 1999; DECR unpublished data). Raven (1994) found that annual landing data from a variety of sources often conflicted. Landing slips are often not properly filled out in the TCI and commercial landing data is suspect, especially prior to the 1980s.

CPUE data for the conch fishery is available from 1975 (Figure 6-5), but early data is of questionable quality. There have been no discernable trends in landing size. Conch abundance has declined substantially close to the South Caicos harbor over time: Doran (1958) reported that a crew of two could land 1,000 conch per day in areas near South Caicos in the 1950s.

Figure 6-5 – Conch CPUE (kg per person-day) 1975-2001 (01 August to 31 July)

Spiny Lobster

Spiny lobster was harvested early in the 20th century by women undertaking 'torch walks', where lights were used at night to attract lobster in shallow, accessible waters (Raven, 1994). Lobsters were canned starting in the early-1930s and were rapidly depleted in accessible areas as fishers began using boats, waterglasses, and barbed poles or nets on poles ('bullying').

Free diving became more prevalent in the mid-1950s after the introduction of masks, snorkels and fins (Raven, 1994). Lobsters were captured by hooking (a hook on a flexible pole) or using the ‘toss’, a flexible spring noose on a stick. By the late-1960s, two man crews were landing as many as 1,000 lobster per day using the toss.

In 1958, lobster traps were introduced by Jamaicans (Raven, 1994). Lobster fishing remained an uncommon occupation in South Caicos during the 1950s and early-60s as the salt industry still employed most locals; only six lobster divers were operating as late as 1966, as salt production ended. Larger trap boats entered the fishery in 1972. Due to low capital costs (divers initially worked from canoes), free diving became the most prevalent fishing method and trapping has usually only accounted for 5-10% of lobster landings (Medley and Ninnes, 1997).

Fibreglass boats and outboard engines (2.5 to 6 hp) were first used for lobster fishing in 1952. As engine horsepower increased over time (for fishing and/or smuggling in the 1980s), distant parts of the Caicos Bank were opened for fishing. By 1983 all areas of the Bank had been exploited by fiberglass runabouts with 55- to 70-hp outboards. As late as the 1960s, productive grounds close to South Caicos (The Bank, Six Hills, South Caicos) still yielded large lobster (Raven, 1994). By the mid-1970s, fishers complained that these areas had only barely legal and sublegal lobster. Deeper water grounds (The Lake, South of Ambergris Cay, Seal Cays, Bush Cay, White Cay, East Side, North Side of East Caicos, Phillips Reef – see Rudd *et al.*, 2001) were progressively exploited as lobster fishers ventured farther afield and into deeper waters.

Despite a hook ban until the late-1970s, it was – and remains, despite periodic bans – the lobster fishing tool of choice. The use of bleach and detergent (to flush lobsters out of dens) has become widespread despite the damage it causes to important coral habitats, possibly leading to increases in macroalgal coverage on coral reefs in heavily fished areas (Tupper and Rudd, 2002).

Divers from Provo have always fished conservatively in relatively shallow waters compared to fishers from South Caicos (Raven, 1994). When lobster inhabiting shallow water (<13-m) became scarce, Provo divers tend to switch to conch (or more recently, finfish) while South Caicos lobster divers have tended to go farther afield and dive deeper for lobster.

The lobster fishery is regulated using minimum size limits (3.25” carapace), a closed season (April 1 to July 31) and prohibitions on the use of scuba (high compliance) and noxious chemicals (low compliance). Capture of mature females is prohibited, but compliance is relatively low and there has been a major problem with minimum size limit compliance in the TCI. The beginning of the lobster season is known locally as the ‘Big Grab’. As many as 95% of lobsters landed from some accessible fishing grounds fall below the legal minimum size (Rudd *et al.*, 2001).

Production statistics from DECR were available until the 2000-01 fishing season (01 August 2000 to 31 July 2001). USA import data is available to 1977 and appears to be a reasonable proxy for TCI landings in recent years. Lobster landings were therefore calculated as the maximum of DECR production figures or USA imports, converted to round weight equivalent at 34% recovery. Domestic consumption was estimated using per capita consumption of 5-kg per person-year for the periods 1948 to 1971, and 1991 to 2002. Per capita consumption of 10-kg per person was used for the period 1971 to 1991 as reliance on local food products was likely higher this time than earlier (when local fishing activity was minimal) or later (when more imported food was available).

The restaurant survey conducted in 2000 (Rudd, in press a, unpublished data – Table 6-1) estimated restaurant consumption of 188-tonnes per year in the TCI. Total domestic consumption for 2000 is estimated at 331-tonnes based on per capita consumption. This seems to be reasonable, as many people in the TCI buy lobster directly from fishers and store them in home freezers for consumption throughout the year. Total estimated spiny lobster production is shown in Figure 6-6.

Raven (1994) reported anecdotal information that mean lobster size in the range of 3-kg had been reported by the early trap fishers. Since the 1970s, average sizes remained quite constant around 0.70-kg. After starting at high levels in the early-1970s, CPUE has fallen and leveled off in the 20- to 30-kg per person-day from the 1980s (Figure 6-7). Early CPUE data is likely not very reliable.

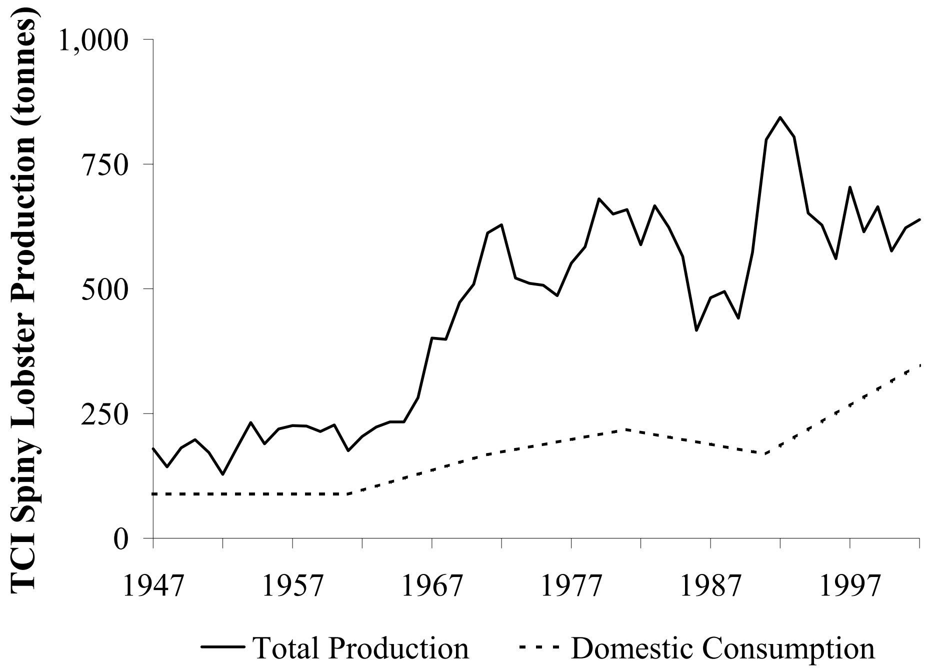

Figure 6-6 – Total spiny lobster landings in the Turks and Caicos Islands, 1947-2002 (based on fishing season, 01 August to 31 July)

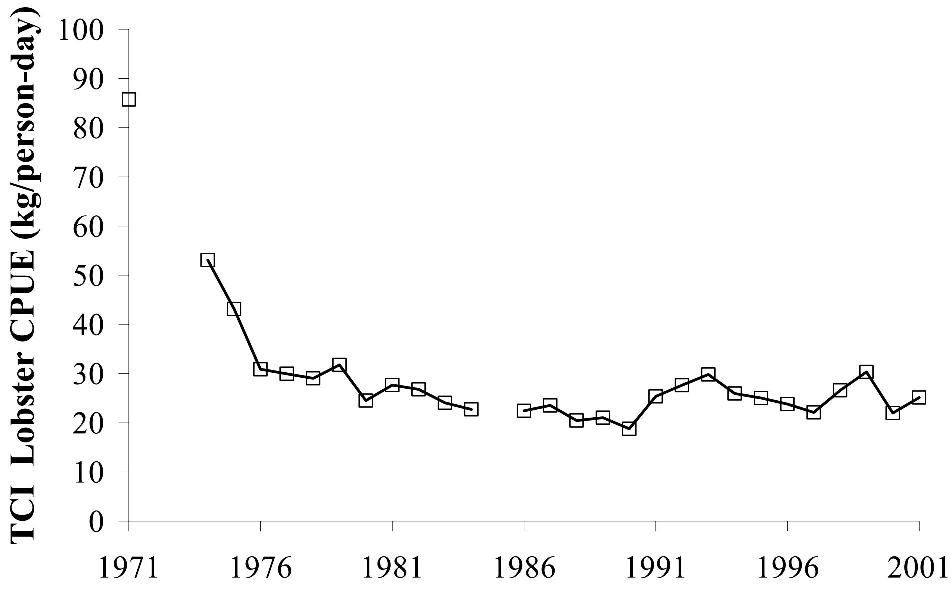

Figure 6-7 – Spiny lobster CPUE (kg per person-day), 1971-2001

Lobster landings are highly seasonal, with most effort and landings occurring during the first month of the season (August). For the period 1989-1998, almost 40% of annual production was landed during the August 'Big Grab' (Béné and Tewfik, 2001). Effort and landings decrease rapidly during the autumn.

The Big Grab is a phenomenon driven by visiting 'Belonger' (a term referring to native islanders) fishers (Rudd *et al.*, 2001). Many Belongers take leave from other employment to travel to South Caicos for several weeks in August. All Belongers have a right to fish lobster and, although they are supposed to abide by regular fisheries rules, there are widespread violations. Visiting fishers tend to be less skilled than resident fishers (many cannot free dive more than 10-m). As a result, they tend to target shallow areas, intercepting young lobsters migrating from the shallow Caicos Bank to deeper fringing reefs. The result is severe growth overfishing as well as indirect effects on the conch fishery (*i.e*, causing fulltime fishers to shift effort to conch sooner than would be normal) (Béné and Tewfik, 2001). Processing plants are complicit in the illegal harvest, as they regularly receive and process undersize lobster tails as 'head meat' (author's personal observation). Other undersize lobster is used locally in Native restaurants or sold to individuals.

Recent research (Rudd, 2001) has shown that spiny lobster have non-extractive economic value, increasing divers' willingness to pay for dive charters in which lobster are observed.

Finfish

The production and use of finfish in the TCI is poorly documented. It is likely, however, that some species of fish have been important for subsistence purposes back through the 20th century and beyond. There are three types of finfish resource that have been exploited in the TCI at different times: demersal reef fishes (*e.g.*, groupers, snappers, grunts, hogfish), pelagics (*e.g.*, mahi mahi, tuna, wahoo, swordfish) and bonefish.

Bonefish (*Albula vulpes*) and Nassau grouper (*Ephinephelus striatus*) are the historically favored species for local consumption (Olsen, 1986). Bonefish are shallow-water bottom feeders that school on the Caicos Bank. They were historically important for subsistence, but consumption has fallen in recent decades as old-time "haulers" retire. Bonefish is also regarded as a 'poor man's' food to some extent and is not as popular with islanders as it was historically. Bonefish is a highly regarded sport fish and several companies now offer catch-and-release fishing charters on the flats of the Caicos Bank.

Reef fishes are primarily caught opportunistically by commercial lobster fishers (Rudd, in press b). Nassau grouper is the preferred species, due to size and flesh quality, but a number of other fishes are also taken. Nassau groupers are often speared by lobster fishers as they follow close to free divers, waiting for opportunities to snatch lobsters (Tony Morris, personal communication, South Caicos, 2000). Sometimes lobster boats will take a day to target reef fishes exclusively. In dockside samples, Tupper and Rudd (2002) found CPUE for reef fish was 3.2-kg per hour for the 456 hours fishing effort (*i.e*, lobster was primary target) in regularly fished grounds. In lightly fished lobster grounds, reef fish CPUE rose to 17.8-kg/hour.

Anecdotal evidence suggests that a large multi-species spawning event occurs annually around the January full moon at Phillip's Reef, off East Caicos. A spawning aggregation site at Shark Bay, just outside the South Caicos harbor, is smaller but Nassau grouper sometimes congregate here as well (Andy Danylchuk, Center for Marine Resource Studies, personal communication, 2002). Dive charter operators from Provo have also reported seeing spawning aggregations near West Caicos (Rudd and Tupper, 2002).

The aggregations do not yet appear to be regularly targeted by artisanal fishers. The Shark Bay aggregation has been targeted specifically in the past, but the presence of large sharks acts as a deterrent to dive fishermen. Weather conditions from December to February often prevent fishers from reaching spawning grounds on the full moon, although fishing activity in the area is relatively high for a month or two preceding spawning time (despite the fact that the area is mostly inside a marine national park).

Reef fish fishing is essentially open access in the TCI. There is a prohibition on the use of scuba gear, but there are no size limits, seasonal closures or TAC. A small marine reserve near South Caicos provides some protection for smaller hogfish and white margate, but there are no differences density inside and outside the reserve for the larger Nassau grouper (Tupper and Rudd, 2002). While finfish densities are high in the TCI relative to other countries in the region, the historic focus on Nassau grouper has almost certainly reduced their abundance substantially from pristine conditions (Tupper and Rudd, 2002). Nassau grouper is a high-profile species in the dive tourism industry and divers in the TCI are willing to pay more for dive packages on which they observe more and/or larger fish (Rudd and Tupper, 2002). Lack of effective management of fisheries may thus impose significant economic externalities on the dive tourism industry.

Pelagic fishes (*e.g.*, tuna, wahoo, swordfish, mahi mahi, marlin) have rarely been targeted in the TCI. A Japanese company leased 24 Taiwanese vessels and was granted licenses to fish in the TCI from 1980 to 1992 (Halls *et al.*, 1999). The vessels used longlines, targeting swordfish and tuna (and some red snapper) near the Gentry Banks. The licenses were not renewed after 1992 due to fears that fishing would adversely impact the sport fishery. Small amounts of pelagics are landed by sport charter boats from Provo and sold to local restaurants (Rudd, in press). It is estimated that about 30-tonnes per year are consumed locally, all in tourist-oriented restaurants. The mortality rates for catch-and-release fish in the sport fishery are unknown.

Total finfish production is shown in Figure 6-8. I assume the domestic per capita finfish consumption was 20-kg per person-year from the 1951 to 1981 and then decreased to 15-kg per person-year. This is below the estimate of 35-kg per year by Olsen (1986). Total restaurant consumption of all finfish in 2000 was about 135-tonnes. Using per capita consumption of 15-kg, this translates to landings of about 360-tonnes. The difference between these figures is substantial, indicating the estimate may be somewhat high or, alternatively, suggesting that native restaurant consumption may be under-estimated.

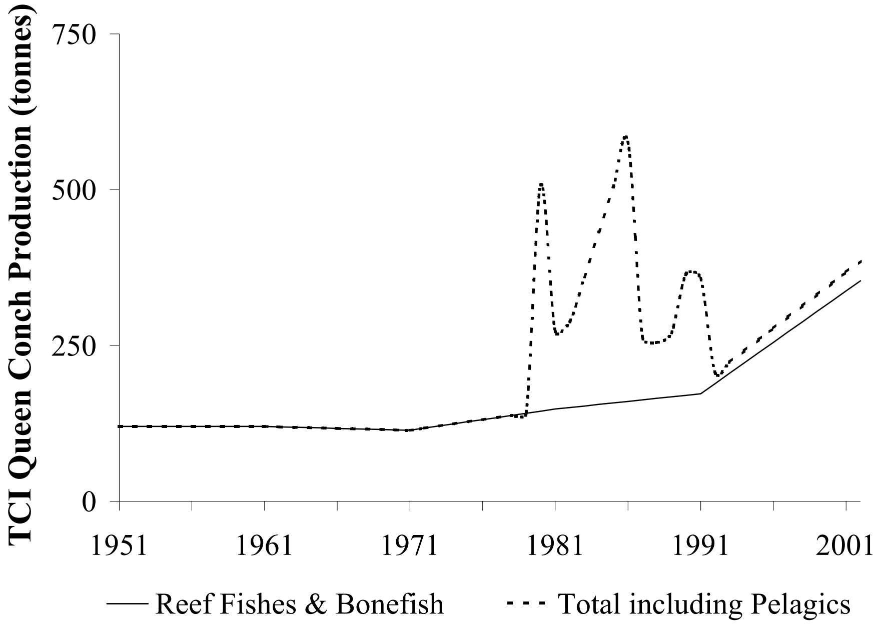

Figure 6-8 – Finfish Production in the TCI

Conclusions

Based on current estimates, total seafood production in the TCI is just reaching the 1,000-tonne per year mark (Figure 6-9). While conch landings have remained relatively steady, there have been increases in lobster and finfish landings to satisfy growing local demand by hotels and restaurants that cater to tourists.

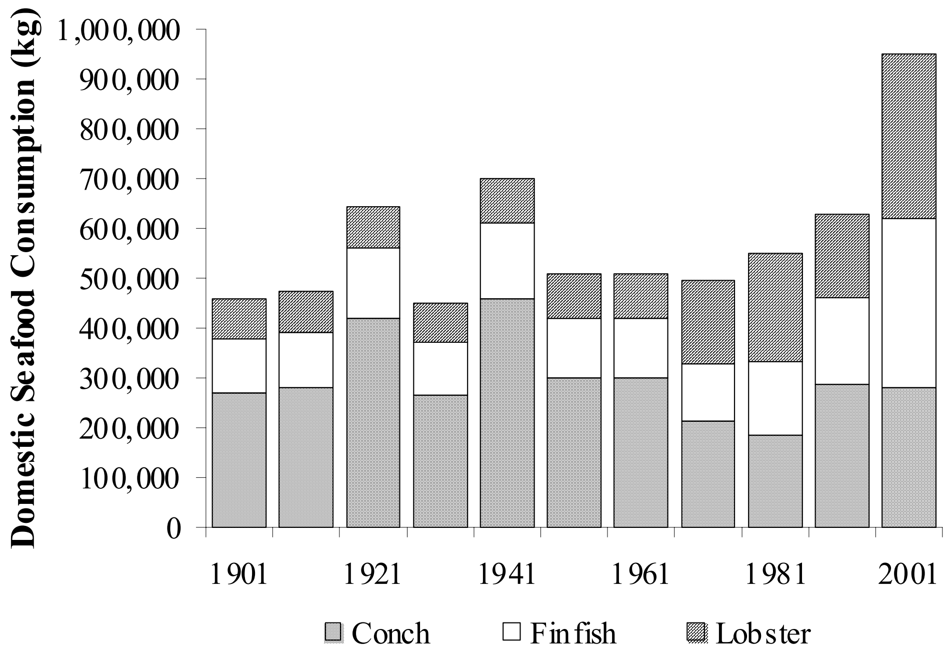

Figure 6-9 – Total Seafood Production from TCI (not including pelagic fishes)

Although data for the TCI goes back in time farther than most other countries in the region, data quality is poor, especially prior to the late-1970s. Estimates of current consumption should be viewed as such. There are many landing sites in the TCI (Halls *et al.*, 1999), and a comprehensive seafood consumption survey would be the only way to accurately assess domestic seafood consumption.

Recent congruence between USA import and TCI production figures is promising. The USA statistics are available online from the U.S. National Marine Fisheries Service with a delay of only about three months. This should allow accurate monitoring of TCI conch and lobster production. When products are misclassified, it is quite easy to sort out proper classifications in USA imports because the TCI ships such a limited variety of products.

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CHAPTER 7

THE EFFECTS OF SEAFOOD IMPORT TARIFFS ON MARKET DEMAND FOR NASSAU GROUPER IN THE TURKS AND CAICOS ISLANDS [1]

Rapid growth of tourism in the Turks and Caicos Islands (TCI) has dramatically increased the demand for seafood but, as yet, the reef fish fishery is relatively undeveloped. This situation is changing, however, as increasing numbers of fishers from South Caicos, the center of the commercial lobster and queen conch fisheries, are increasingly willing to make the 60-km trip across the Caicos Bank to sell reef fishes to restaurants and hotels. Fishers receive prices around US \$3.25 per kg for grouper and snapper species from the South Caicos processing plants, which are primarily set up for handling export-oriented queen conch and spiny lobster landings (author's personal observation). On the resort island of Providenciales, on the other hand, fishers receive up to \$15.00 per kg for the same fish.

Imported seafood products in the TCI are subject to tariffs of up to 40%. This policy generates revenue for government and should, theoretically, increase income in the artisanal fishing sector (the *de facto* social safety net for rural TCI 'Belongers'). An import tariff makes local reef fish more competitive relative to expensive imported products. While an import tariff causes some economic welfare losses (deadweight losses), local governments gain tariff revenue and fishers' producer surplus (PS) increases as a result of the tariff. Hence, local actors capture increased resource rents at the expense of consumers – largely foreign tourists in the TCI – and there are economic incentives for both government and fishers to support the maintenance or expansion of import tariffs.

Figure 7-1 illustrates the general effects of a seafood import tariff. In the absence of trade, the domestic supply (S^0) and demand (D^0) curves would determine a market-clearing equilibrium at their intersection. When international supply is not subject to tariffs, the supply curve is kinked, forming supply S^1 at the world market price p^1. Total

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[1] In press as: Rudd, M.A. 2003. The effects of seafood import tariffs on market demand for Nassau grouper in the Turks and Caicos Islands. *Proceedings of the Gulf Caribbean Fisheries Institute* 55. This research was conducted while employed at the SFS Center for Marine Resource Studies (CMRS), South Caicos. Thanks are due to 'TCI seafood critics' Laurie Darian and Juliet Christian-Smith for assisting in survey design and conducting interviews while students at CMRS, and Mark Tupper for input on survey design. Special thanks to the many restaurant buyers and managers who

consumption is q^{t1}, of which q^{d1} is supplied domestically and the balance (q^{t1} - q^{d1}) imported. Total economic welfare under free trade is consumer surplus (CS), area *acd*, plus producer surplus (PS), area *def*. When an import tariff is imposed, raising the price of imports to p^2, the supply curve is given by S^2. Total consumption falls to q^{t2}, of which q^{d2} is supplied domestically and the balance imported. PS increases by area *dfgh* (dark gray), government captures tariff revenues, area *bghi* (medium gray), and CS falls to area *abh*. Deadweight losses under the import tariff are the two lightly shaded triangles, areas *bci* plus *fgh*.

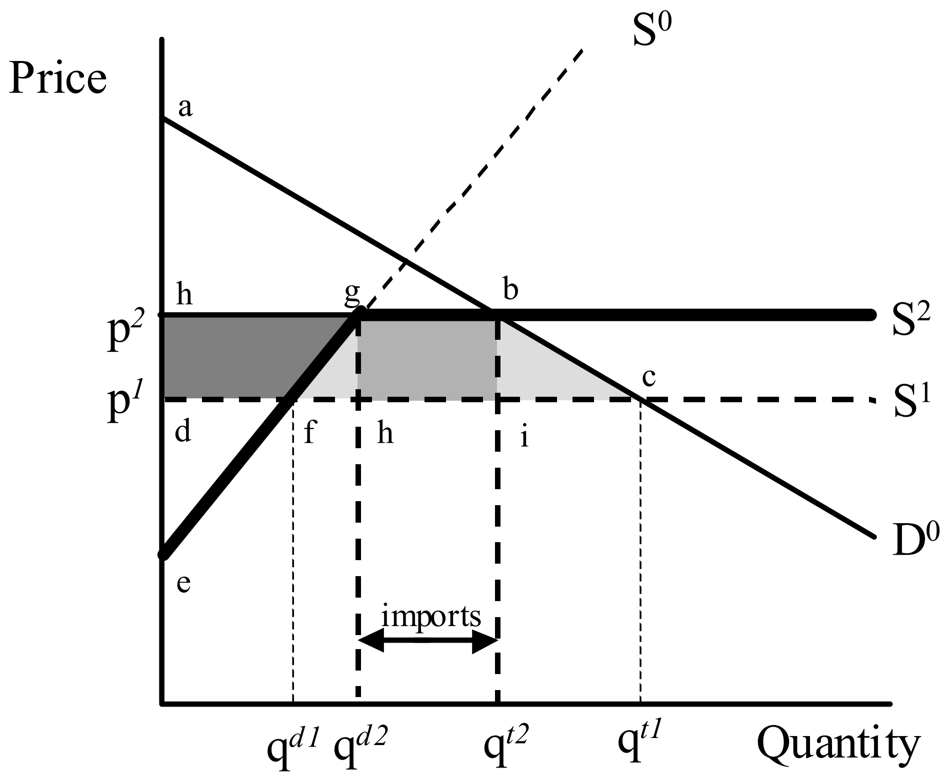

Figure 7-1 – Welfare Impacts of an Import Tariff

Whether the import tariff contributes to reef fish depletion depends on three factors. First, what is the market demand response to the tariff? Second, how responsive are artisanal fishers to price signals in the local marketplace? Finally, how inherently vulnerable are

gave valuable time for the survey. Daniel Pauly and participants of a UBC Fisheries Centre seminar provided valuable comments on a previous draft.

the target stocks?[2] This research addresses the first question and assesses whether the seafood import tariff increases domestic market demand for Nassau grouper, *Epinephelus striatus,* a vulnerable reef fish (Sadovy, 1994; Coleman *et al., 2*000) that also holds considerable value for the dive tourism industry (Rudd and Tupper, 2002).

Methods

A two-part survey was developed for restaurant managers and buyers in the TCI. In the first section, respondents were presented with open-ended questions regarding what seafood products they used, whether their buying decisions were influenced by the import tariff, and their general observations on trends in the availability of local fishes. This research focused on the 'tourist' restaurants of Providenciales and Grand Turk, but the qualitative survey was also used to interview a number of 'native' restaurants catering to TCI Belongers.

In the second part, a paired comparison conjoint survey was used to assess the marginal trade-offs that restaurant buyers make regarding key product attributes for reef fishes and potential substitute products. Paired comparisons of this type are designed to elicit maximum information about subtle preference trade-offs and have long been used in market research (Green and Srinivasan, 1978). The survey was administered in person by trained student researchers during April 2000. Respondents were assured that all information collected would remain confidential.

Each seafood product is composed of a bundle of attributes that provide value for restaurant buyers (*i.e*, contribute to profitability) but that vary in level between two profiles. After a pilot survey of selected restaurant owners and managers, experimental design was finalized. Key attributes and levels that were included in the final survey instrument included: product form (fresh or frozen); product source (local or imported); product type (grouper - Serranidae, snapper - Lutjanidae, mahi or dolphin fish – *Coryphaena hippurus*, and wahoo – *Acanthocybium solandri*); and purchase price (US $5.00, 6.00, 7.00, 7.20, 8.00, 8.40, 9.60 or 11.20 per lb). Price is one of the relevant

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[2] When fish are vulnerable, the supply curve can bend backwards because more fishing effort leads to decreasing, not just diminishing, returns. For highly vulnerable reef fish, fishers may end up chasing fewer, but increasingly valuable, fish. In the TCI, as in other areas, the fishery is essentially open access. There are no catch limits, no gear restrictions, and few other means to meaningfully control reef fish fishing effort. Rather than fishermen using MC = MR as a criterion for fishing effort decisions in these types of open access situations, we would expect that fishers will expend effort to the point where TR =TC. This Gordon-Schaefer open access equilibrium

product attributes, allowing the assessment of marginal pricing trade-off's for other product attributes. It should be noted that almost all local grouper landed in the TCI and sold to restaurants is Nassau grouper. Some other grouper species (*e.g.*, tiger grouper, *Mycteroperca tigris*) are occasionally consumed, but *Ciguatera* toxicity limits sales of other groupers in restaurants.

Paired comparison conjoint surveys are cognitively challenging. The design space for paired comparison questions rises exponentially with the number of attributes and levels considered. This survey used two attributes with two levels, one with four levels, and one with eight levels, yielding a potential design space of $4^2 2^4 8^2$ = 16,384 possible paired comparisons. The experimental design challenge was to select a limited number of questions from the design space in such a way that useful information about attribute trade-off's is maximized. This was done using the Sawtooth Software Conjoint Value Analysis (CVA) software (Sawtooth Software, 1996).

A nearly orthogonal and balanced experimental design consisting of 20 survey questions was constructed by choosing the design with the highest D-efficiency from a pool of 500 candidate surveys. The final survey instrument had D = 0.932 (where a score of 1.0 is fully orthogonal and balanced). Each of the twenty survey questions (*e.g.*, Figure 7-2) asked respondents to express their preferences for one profile relative to another using a rating scale.

Category	Option A	Option B
Product Form	Frozen	Fresh
Type of Seafood	Mahi Mahi	Grouper
Purchase Price (US $/lb)	$9.60 per lb	$8.00 per lb
Source of Seafood	Imported	Local Fishery

1	2	3	4	5	6	7	8	9
A is much better		A is somewhat better		A & B are about equal		B is somewhat better		B is much better
Please circle a number from 1 to 9 that reflects your rating								

Figure 7-2 – A Paired Comparison Question Rating Two Seafood Products

reduces rent capture from the industry profit maximizing equilibrium, where MR = MC, or the point of maximum economic yield, where AR = MC.

The results were collated and analyzed using the CVA software. An ordinary least square (OLS) regression was conducted for each survey respondent. The regression coefficients, known in the marketing literature as part-worth's (the marginal valuations of choice variables), were then available for use in market simulations. See Rudd (2001) for a more detailed explanation of the CVA paired comparison analysis and simulation.[3]

The CVA market simulation module was used to model the market share for various hypothetical seafood products. In these simulations, total utility for each alternative product was calculated based on the part-worth's for each individual survey respondent. Each respondent was assumed to choose the seafood product with the highest overall utility in the simulation. The individual choices were aggregated to determine market share (% of respondents choosing the option) for each seafood product.

The simulations included six potentially competitive seafood products and used average prices reported in the qualitative survey: (1) imported frozen grouper at US $5.20 per lb; (2) local fresh grouper at $5.75 per lb; (3) local fresh snapper at $6.54 per lb; (4) imported frozen snapper at $6.88 per lb; (5) local fresh mahi at $7.60 per lb; and (6) local fresh wahoo at $8.00 per lb. Market shares were calculated for two simulations in which the price of either frozen imported or local fresh grouper fluctuated. Other prices and products were held constant across simulations.

The first simulation, which varied the price of imported frozen grouper, simulates the effects of a change in the TCI seafood import tariff. The second simulation, which varied the price of local fresh grouper, simulates the effects of a change in local seafood prices (*e.g.*, due to local depletion or by policies that make fishing more expensive). Two-tailed paired *t*-tests (using Bonferroni adjustment, total α-level of 0.05) were used to test the hypotheses that the market shares of the survey respondents for different products were significantly different (*e.g.*, was market share for imported frozen grouper at $5 per lb different from that of local fresh grouper at $6 per lb?).

Results

Of 25 restaurants surveyed, 24 completed the qualitative surveys and 20 completed full conjoint surveys. These restaurants likely account for > 90% of TCI tourist restaurant consumption and approximately 50% of total TCI seafood consumption.

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3 Details of the analysis are included in Appendix 9A.

General Market Observations

Approximately 3,200 kg of fish were consumed weekly during the study period; grouper (725 kg) was the single most popular seafood. This implies the annual consumption of grouper is about 85 t (52 * [725 kg / 0.90] / 0.5), of which approximately 50 t (60%) is landed locally. Weekly consumption of other types of seafood was 650 kg spiny lobster, 550 kg queen conch, 440 kg snapper, 400 kg mahi, 125 kg wahoo, and 265 kg of other products (primarily salmon, tilapia and tuna).

Most local grouper consumed was Nassau grouper. Imported grouper consisted primarily of frozen filets imported from Southeast Asia or Central America via Florida. Most mahi and all wahoo was landed by local sport fishing charter boats and sold directly to local restaurants.

The product mix in native restaurants was much different than in tourist restaurants. The native restaurants tended to use many more species of local fish interchangeably – several restaurants differentiated their purchases only as small or large fish. Native restaurant sales volume was variable, but was very high in some cases (*e.g.*, one 'small' native restaurant used as much fish in one week as a large all-inclusive resort with over 1000 guests).

Several restaurant buyers commented on the declining availability of reef fish in the Providenciales area. Buyers in Providenciales also noted that inconsistent quality and delivery from local fishers was the main reason that they purchased imported seafood, despite the import tariffs. Most buyers indicated that the import tariff did have an impact on their purchase decisions and that a reduction of the tariff, from 40% to 20% on finfish, would alter their purchasing behavior.

A cluster analysis characterizing market niches for the 20 restaurants that completed the full survey was conducted using Datadesk (Velleman, 1997). Four distinct clusters were identified: seven restaurants for which price was the most important factor (price sensitive 'casual dining' operations); two restaurants for which product form (fresh) was the dominant factor; four restaurants that placed equal importance on product form, type, price and source; and seven restaurants that put a low priority on source and that shared a characteristic of high entrée price ('white tablecloth' restaurants that source quality product from wherever they can find it).

Conjoint Survey and Simulation Results

Regressions were conducted for 20 individual respondents and part-worth's were calculated for use in simulations. R^2 for high (> 0.88) for all respondents, indicating internal consistency in decision making.

In Simulation 1, the price of frozen imported grouper was varied, simulating the effect of changes in the seafood import tariff. Market shares for various seafood products in the face of varying imported grouper prices are shown in Table 7-1. Frozen imported grouper might be available from U.S. wholesalers for as low as $3.15 per lb, so $4.00 per lb may already represent an import tariff in the 25% range. As the price of the imported grouper rises to $5.00 per lb, market share for that product falls from 30% to 15%. As price rises to $6.00 per lb, market share falls further to 5% and frozen imported grouper is priced totally out of the market at $8.00 per lb.

As frozen imported grouper market share falls from 30% to 0%, three other products gain market share equally: fresh local grouper, fresh local snapper, and frozen imported snapper. The average price of frozen imported grouper during the survey period (April 2000) was $5.20 per lb. Reducing the price of imported product from the $5.00 range, by reducing the import tariff, would lead to reduced market demand for local grouper and snapper, potentially reducing fishing pressure on local stocks.

		Market Share (%) for Six Seafood Products					
		Fresh Local Grouper	Frozen Imported Grouper	Fresh Local Snapper	Frozen Imported Snapper	Fresh Local Mahi	Fresh Local Wahoo
Price of Frozen, Imported Grouper (US $/lb)	$4.00	40%	30%	10%	0%	10%	10%
	$5.00	45%	15%	20%	0%	10%	10%
	$6.00	50%	5%	20%	5%	10%	10%
	$7.00	50%	5%	20%	5%	10%	10%
	$8.00	50%	0%	20%	10%	10%	10%

Table 7-1 – Market share for seafood products when tariff on frozen imported grouper varies

Simulation 2 models the effects of changes in the price of local fresh grouper. This could happen if fishing pressure started to deplete local stocks, driving up prices. It could also happen if the costs of fishing changed, or if the value of grouper for the dive industry were taken into account, correcting for market distortions resulting from fishing externalities. Table 7-2 shows the market shares for various seafood products when the price of fresh local grouper varies.

Price of Fresh, Local Grouper (US $/lb)	Market Share (%) for Six Seafood Products Fresh Local Grouper	Frozen Imported Grouper	Fresh Local Snapper	Frozen Imported Snapper	Fresh Local Mahi	Fresh Local Wahoo
$5.00	70%	5%	20%	0%	0%	5%
$6.00	45%	15%	20%	0%	10%	10%
$7.00	15%	30%	35%	0%	10%	10%
$8.00	0%	30%	40%	0%	20%	10%

Table 7-2 – Market share for seafood products when fresh local grouper price varies

Table 7-2 demonstrates that there is a more complex reaction in the market to changes in the price of fresh local grouper. Buyers are very price sensitive, with market share falling from 70% at $5.00 per lb to 0% at $8.00 per lb. All other fresh local products – snapper, mahi, and wahoo – gain market share as the price of fresh local grouper rises. Market share for frozen imported grouper also rises.

Two-tailed paired *t*-tests were used to test the equivalence of market shares (*i.e*, restaurant buyer indifference between two products) for different seafood products under different pricing conditions. For instance, the null hypothesis that market shares were equal for frozen imported grouper at $5.00 per lb ($S_0$) and fresh local snapper at $6.54 per lb ($S_1$) could not be rejected (Table 7-3 – fail to reject H_0 at total $\alpha = 0.05$, $p > 0.05$). Conversely, market share equivalence for frozen imported grouper at $6.00 per lb ($S_0$) and fresh local grouper at $5.75 per lb ($S_1$) was rejected ($p = 0.004$) (*i.e*, cheaper fresh local grouper was significantly preferred).

Table 7-3 summarizes tests for frozen imported grouper, which has a statistically indistinguishable market share from that for fresh local snapper at all frozen imported grouper prices from $5.00 and $8.00 per lb and fresh local grouper at a frozen imported grouper price of $5.00 only. As frozen imported grouper price falls well below $5.00 we should expect to see restaurant buyers increasingly prefer the import product, perhaps to the extent that the equivalence of market share with fresh local grouper could again be rejected (*i.e*, when frozen imports are at $6.00, fresh local grouper is significantly preferred by buyers, when imports are at $5.00, buyers are indifferent, and when imports fall under $4.00, buyers may significantly prefer the imported product).

		Market Share Test p-value (H_0: $S_0 = S_1$)				
		Fresh Local Grouper ($5.75/lb)	Fresh Local Snapper ($6.54/lb)	Frozen Imported Snapper ($6.88/lb)	Fresh Local Mahi ($8.00/lb)	Fresh Local Wahoo ($8.00/lb)
Price of Frozen, Imported Grouper (US $/lb)	$5.00	0.082	0.171	0.001	0.031	0.031
	$6.00	0.004	0.110	0.004	0.017	0.017
	$7.00	0.004	0.110	0.004	0.017	0.017
	$8.00	0.004	0.110	0.017	0.017	0.017

Table 7-3 – Tests of equality of profile market shares for frozen imported grouper (S_0) and other products (S_1). Two-tailed paired t-test p-values are for total $\alpha = 0.05$, individual $\alpha = 0.001$.

Similarly, Table 7-4 shows the two-tailed paired t-test results when fresh local grouper (S_0) at various prices of is compared with other seafood products at current prices. The patterns are more complex than in the previous case. For example, restaurant buyers exhibit indifference between fresh local grouper at $7.00 per lb and fresh local mahi at $8.00 per lb ($p$ = 0.666). However, the equivalence of market shares is rejected when the price of fresh local grouper falls to $6.00 per lb ($p$ = 0.031, indicating that the cheaper grouper is significantly preferred) or rises to $8.00 per lb ($p$ = 0.042, indicating that the mahi is now significantly preferred to more expensive grouper).

Table 7-4 demonstrates that fresh local grouper is a substitute for each of the other five products. At a low price of $5.00 per lb, it is significantly preferred to all other seafood products. At intermediate price levels, market shares are not significantly different for fresh local grouper and other products. When the price of fresh local grouper rises to $8.00 per lb, all products except fresh local wahoo are preferred to expensive grouper. By contrast, frozen imported grouper can only be viewed as a substitute for fresh local snapper and grouper.

Discussion

A variety of market niches exist in the TCI restaurant sector. Preferences for seafood products varies substantially, but total annual seafood consumption in the TCI probably is in the 325- to 425-tonne range, assuming that consumption in restaurants during the study period (April) is average, our sample accounted for 90% of tourist restaurant purchases, and that tourist restaurants account for about 50% of total seafood consumption in the TCI.

		Market Share Test p-value (H_0: $S_0 = S_1$)				
		Frozen Imported Grouper ($5.75/lb)	Fresh Local Snapper ($6.54/lb)	Frozen Imported Snapper ($6.88/lb)	Fresh Local Mahi ($8.00/lb)	Fresh Local Wahoo ($8.00/lb)
Price of Fresh, Local Grouper (US $/lb)	$5.00	0.000	0.014	0.000	0.000	0.000
	$6.00	0.083	0.171	0.001	0.031	0.031
	$7.00	0.330	0.214	0.083	0.666	0.666
	$8.00	0.010	0.002	0.000	0.042	0.163

Table 7-4 – Tests of equality of profile market shares for fresh local grouper (S_0) and other products (S_1). Two-tailed paired *t*-test *p*-values are for total $\alpha = 0.05$, individual $\alpha = 0.001$.

Characteristics of Market Demand

Frozen imported grouper and fresh local grouper are substitute products. In Simulation 1, a 50% change in the import price (from $4.00 to $6.00 per lb) induced a 10% increase in market share for both fresh local grouper and snapper. Frozen grouper is generally viewed as a low-quality product; changes in prices had no impact on high-end local products (mahi and wahoo), and the main impact of frozen imported grouper on fresh local grouper market share occurred at prices less than the current import price ($5.20 per lb).

Market demand for fresh local grouper is very price sensitive. A 60% increase in price, from $5.00 to $8.00 per lb, resulted in market share falling from 70% to 0%. The biggest market share gainer was frozen imported grouper (+ 30%), but fresh local snapper and mahi also increased 20% in market share as fresh local grouper price rose over this range. Fresh local grouper is viewed as a high-quality product and can substitute for high-end mahi and wahoo.

The results of both simulations imply that market demand for fresh local Nassau grouper in the TCI is quite elastic (*i.e*, the demand curve is relatively flat), and that the curve will shift up or down based on the price of substitute imported grouper, which is partially determined by the import tariff.

Characteristics of Market Supply

What are the next steps that need to be taken in an analysis of Nassau grouper management options in the TCI? The second step is to assess the response of local fishers to changes in market price signals. If fishers were profit maximizers, we would expect to see increased effort allocated to reef fish capture as demand rises. Fisher behavior and motivations in tropical artisanal fisheries are likely more complex, however, and factors such as risk preferences, alternative fishing opportunities, and revenue goals (rather than profit) may come into play. Many Nassau grouper caught on the South Caicos fishing grounds, for example, are taken by lobster divers who opportunistically spear fish. Few data are available from fisher logbooks regarding the allocation of effort between different fishing activities, fishing locations or landing volumes. Further work on the supply-side of the market is clearly required.

Policy Implications

Further complications arise in economic analyses of reef fish management options. First, large reef fish such as Nassau grouper provide non-extractive economic value for the dive tourism industry (Williams and Polunin, 2000; Rudd and Tupper, 2002) as well as being popular in restaurants. This makes policy decisions based on economic maximization criterion difficult because of nonmarket valuation challenges. This may also lead to conflicts between policy actors with different goals. For instance, the finance division of government may have a goal of revenue generation (in the TCI, there are no income, property or business taxes – substantial government revenue is raised by license fees and import tariffs) while other departments might have goals of efficient resource utilization, promotion of tourism, or conservation. Fishers and the dive tourism operators may also have conflicts over reef fish utilization.

Secondly, the biology and ecology of many reef fishes, including Nassau grouper, make them extremely vulnerable to overfishing (Coleman *et al.,* 2000). They will have the classic backward-bending supply curves of the Gordon-Schaefer model that, in the absence of effective property rights, lead to an open access equilibrium where average, not marginal, cost just equals demand (*i.e*, total rent dissipation).

When an import tariff is imposed, consumer surplus falls. When harvest levels are relatively low, the tariff may increase producer surplus as local fishers increase production and get higher prices (recall Figure 7-1). Government gains tariff revenue. Once harvests rise above MSY, however, further increases in fisher revenue can be completely offset by rising costs. Strong incentives may still exist, however, for government to maintain or increase import tariffs if they are having difficulties meeting revenue generation goals. Thus, at low levels of fishing, goals of increasing fisher income and generating government revenue may coincide, but this will not necessarily remain the case as tariffs rise.

Nassau grouper stocks in the TCI are still in relatively healthy condition. In fact, densities in the TCI are the highest observed in the Caribbean region (Tupper, 2002; Tupper and Rudd, 2002). However, given the particular vulnerability of grouper to even low levels of fishing (Sadovy, 1994; Coleman *et al., 2*000) and their value to the dive tourism industry (Rudd and Tupper, 2002), it would seem prudent to implement pro-active policies that effectively protect this valuable resource as well as meeting other social and government revenue generation goals.

The optimal policy mix must account for species ecology, fisher behavior, government revenue generation, and market demand in both the restaurant and dive tourism sectors. Setting a total allowable catch (TAC) for Nassau grouper may not be feasible because of problems monitoring catch. Similarly, minimum size limits are likely of limited usefulness because of monitoring difficulties and of the importance of old, large Nassau grouper to overall reproductive capacity.

Marine protected areas (MPAs) have been advocated for groupers (*e.g.*, Coleman *et al., 2*000). Tupper (2002) and Tupper and Rudd (2002) have not found Nassau grouper to be more abundant within a small MPA near South Caicos, however. This is likely due to the relatively small size of the MPA relative to the comparatively large home range of Nassau grouper (the MPA had significant positive effects on the size, abundance and biomass of smaller, more sedentary hogfish, *Lachnolaimus maximus*). Full protection of essential Nassau grouper habitat and spawning migration corridors on the very narrow fringe of the Caicos Bank would impose economic hardship on local fishers who depend on those areas for commercial species (spiny lobster) and subsistence fishing. Enforcement, again, would be problematic and a large MPA could degenerate into another ‘paper park’.

An alternative option may be a commercial trade ban (*i.e*, no purchase of, or trade in, Nassau grouper by restaurants, but no restrictions on subsistence fishing). This policy could have a number of pragmatic advantages: local fishers would maintain access to fishing grounds for lobster and reef fishes other than Nassau grouper; government would maintains tariff revenues (a trade ban may even increase imports of substitute frozen grouper, increasing overall revenue); enforcement efforts could focus on shore-based restaurant buyers, reducing more expensive fisheries field enforcement costs; and Nassau grouper conservation would continue to provide valuable non-extractive economic value for the dive tourism industry.

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CHAPTER 8

A COMMENT ON FISHING EFFORT ALLOCATION IN THE TURKS AND CAICOS ISLANDS[1]

Marine ecosystem-based management is gaining acceptance as a management paradigm and will be especially important for multi-species artisanal fisheries where fishers use multiple gear types, move rapidly between sites and may switch target species on a intra-day basis. Understanding fisher incentives and behavior is crucial if we are to understand how various policy tools can simultaneously address critical conservation and economic development objectives in tropical island nations.

Béné and Tewfik (2001) recently analyzed the fishing effort allocation between queen conch and spiny lobster fishery for artisanal fishers in the South Caicos region of the Turks and Caicos Islands (TCI). South Caicos is the primary commercial fishing area, accounting for 60% of TCI lobster landings in 1998 (Halls *et al.*, 1999). In their attempt to identify factors that affect switching behavior between the two export-oriented species, Béné and Tewfik found that fishers allocated more effort to fishing lobster than would be expected based on strictly bioeconomic rationale. Based on their revenue per unit effort (RPUE) analysis, they assert that "when the lobster fishery stops being more attractive than the conch fishery (on the basis of the bioeconomic indicator $RPUE_{ij}$), the TCI fishermen still allocate more than 63% of their fishing effort on lobster" (p. 176). They point out that lobster fishing requires higher skill relative to conch fishing and state that peer pressure "reinforces and extends their [fishers'] preferences for lobster beyond the incentives induced by the bioeconomic (*i.e*, RPUE) condition of the two stocks" (p. 178-179). By extension, the authors further postulate that switching from lobster to conch fishing causes a 'statutory' loss: "By switching to a more labor intensive, less skilled, less remunerative, and above all else less socially valued species, these fishermen have lost part of their social status in the eyes of the community" (Béné and Tewfik, 2001: 182).

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Even though Béné and Tewfik claim to use an integrated, holistic approach in their analysis, they neglected to consider the domestic fishery for reef fishes. This crucial domestic fishery for groupers, snappers, grunts, hogfish and other reef fish species is used for subsistence purposes and provides an important source of income for fishers who sell their landings directly to restaurants throughout the TCI. Consideration of this component of the multi-species TCI fishery is essential for a holistic understanding of the fishers' behavior, fisheries management and policy options for conserving valuable high profile reef species like the Nassau grouper (Rudd and Tupper, 2002).

Reef Fish Occurrence and Use

Important commercial reef fish tend to occur in the same patch reef and shelf edge habitats in which lobsters are found (Tupper, 2002; Tupper and Rudd, 2002). Queen conch, on the other hand, are found primarily in sand/algal plains that provide unsuitable habitat for adult reef fishes (Tupper, 2002). Reef fishes are commonly speared opportunistically by lobster fishers (Tupper and Rudd, 2002) and most landings are 'bycatch' of the lobster fishery. Trap boats play a minor role in the TCI, accounting for only 5-10% of lobster landings (Medley and Ninnes, 1997) and minor reef fish landings (M. Tupper, University of Guam Marine Laboratory, personal communication). Like lobster, the mean size of several of the most important reef fishes tends to increase with depth (Tupper, 2002). Thus, skilled lobster divers who can free dive to 15 meters or more are most likely to encounter large, valuable reef fishes such as Nassau grouper. Because the marginal costs associated with harvesting reef fish is low for lobster fishers (essentially zero), economic theory suggests that we should observe relatively heavy landings of reef fish.

Conch and lobster are delivered to one of six export-oriented processing plants in the TCI. There has been a consistent discrepancy of about 50 to 100 tonnes between landed and exported lobster volume going back to the 1970s (Olsen, 1986), indicating that some frozen lobster is sold in domestic markets via the processing plant. Almost all reef fish landings, on the other hand, are sold by fishers directly to restaurants (Halls *et al.*, 1999; Rudd, in press a) or distributed to families or friends for subsistence (M. Tupper, University of Guam Marine Laboratory, personal communication). Therefore, most lobster and conch landings will tend to be recorded in government statistics, whereas reef fish landings are not recorded at all.

How important is the reef fish harvest for lobster divers? This question can be approached from both the supply and the demand sides. On the supply side, a dockside monitoring program (Tupper and Rudd, 2002) found CPUE of 3.2 kg/hour/fisher for reef fishes landed by lobster fishing boats operating in fished areas (n = 133 fisher interviews totaling 456 hours fishing effort). Assuming six hours per day fishing effort, this equates to 19 kg/fisher/day of reef fish landings. In lightly fished areas (the eastern shores of South and East Caicos), CPUE for reef fishes rose to 17.8 kg/hour (n = 28 fisher interviews totaling 98 hours fishing effort), although this figure was skewed upwards by some fishing trips that specifically targeted large groupers.

On the demand side, Olsen (1986) used per capita consumption figures (35 kg fish/person/year) to estimate annual domestic TCI fish consumption in excess of 310 tonnes. More recently, a restaurant survey (Rudd, in press a) estimated annual domestic restaurant consumption of local reef fishes to be 96 tonnes. Estimates of total domestic consumption of reef fish, based on a conservative per capital consumption rate (15 kg fish/person/year), are 230 and 338 tonnes for 1991 and 2001, respectively (Rudd, unpublished data).

Overall fishing effort has not changed substantially over the last decade, staying roughly in the 13,000 to 15,000 person-day range (see Béné and Tewfik, 2001 for a chart of monthly effort). If 300 tonnes of reef fish were landed annually by lobster fishers, this implies mean landings of approximately 21 kg/day, a figure that coincides closely with demand-side landing surveys from regularly fished commercial fishing grounds.

Reef Fish Value and Revenue

Reef fish has historically been intermediate in value between conch and lobster, around US $2.20 per kg over much of the last decade (see price chart for conch and lobster in Béné and Tewfik (2001)). More recently, reef fish has been fetching around $3.25 per kg although fishers may earn up to $15.00 per kg selling directly to restaurants on the tourist island of Providenciales (Rudd, in press a). Increasing numbers of South Caicos fishers are willing to make the 60-km trip when they land numerous large fish (Rudd and Tupper, 2002). Seafood used locally for subsistence is a substitute for expensive imported protein sources.

Accounting for reef fish revenue would shift the curve in Figure 8 of Béné and Tewfik, (2001) to the right. Increases of revenue of around US $44 per day (based on 20 kg/day at

US \$2.20 per kg) for lobster fishers would, in fact, shift the curve so that the intercept is close to zero (Figure 8-1). The Béné and Tewfik curve is based on their regression results, $Y = 0.0195x + 0.566$, where effort allocated to lobster fishing, y, was transformed using $Y = \ln(y/(1-y))$. The curve for the adjusted lobster fishing allocation when reef fish landings are considered is given by $Y = 0.0195x + (0.566 - 0.44)$, where 0.44 shifts the curve by the estimated average value (US \$44) of reef fish landings per person-day. In Béné and Tewfik (2001), the intercept was calculated as 63.8% (fishers allocate 63.8% of annual effort to lobster diving when there is zero difference between conch and lobster RPUE). When adjusted for reef fish landings, the intercept is 53.1%. This implies that (1) the *combined* returns of reef fish and lobsters has made lobster fishing relatively more profitable than conch fishing for all years, 1989 to 1997, and (2) fishers effort allocation to conch and lobster fishing, contrary to Béné and Tewfik's conclusion, appears to be made in an economically consistent manner.

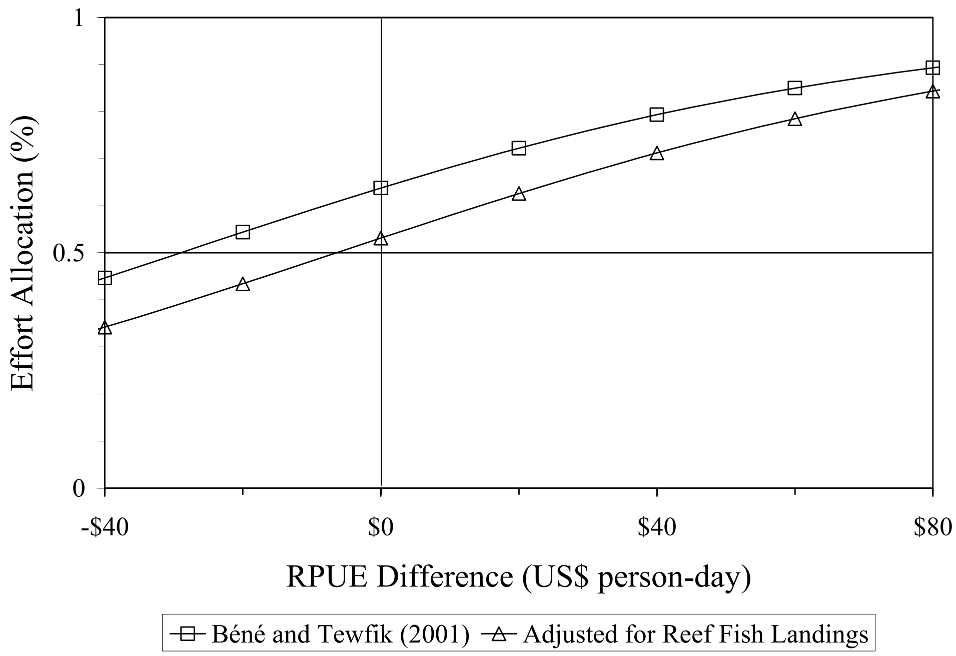

Figure 8-1 – Distribution of TCI fisher effort between conch and lobster stocks (Y-axis) as a function of Revenue per Unit Effort (RPUE) difference ($RPUE_{lobster} - RPUE_{conch}$).

Conclusions

Béné and Tewfik, (2001) provide useful background on the TCI fishery and some of the factors that influence fisher behavior. An important conclusion of their research – that lobster fishing is preferred over conch fishing because of peer pressure and community norms – does not, however, follow when all components of the multi-species artisanal fishery are considered. There is no doubt that social structure and norms play an important role in economic efficiency (Rudd, 2003), but to downplay the crucial role that economic factors play in artisanal fisher decision making can lead to faulty policy recommendations and could potentially jeopardize efforts to conserve important reef species and habitats. Like terrestrial farmers and forest users in developing countries, the decisions of artisanal fishers tend to be uncompromisingly economic in nature when all factors – information availability, risk preferences, and wealth (or lack thereof) – are considered. Fishers in the TCI tend to be marginalized ethnically or socially, and the fishery acts as the *de facto* social safety net. In circumstances such as these, fishers tend to be highly cognizant of risk and rewards even if the dockside banter centers on diving skill. Artisanal fishers' economic decision-making capacity should not be underestimated.

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CHAPTER 9

THE IMPACT OF NASSAU GROUPER SIZE AND ABUNDANCE ON SCUBA DIVER SITE SELECTION AND MPA ECONOMICS[1]

Many demersal fisheries operate at or beyond their sustainable limits (NRC, 1999) and the demand for fish continues to grow. The management of a marine fishery is a difficult task under any circumstances (Costanza *et al.*, 1998), but in the tropics, where complex dynamic coral reef – seagrass – mangrove ecosystems are increasingly stressed and management institutions are often weak, the problem is exacerbated (Roberts and Polunin, 1993; Roberts, 1997).

Tropical coral reef ecosystems provide humans with a wide variety of ecological and economic services (Moberg and Folke, 1999) and are especially important in tropical developing countries where economic opportunities are limited. Many tropical reef species are particularly vulnerable to overexploitation due to their ecology, the wide variety of fishing methods used in artisanal commercial reef fisheries, and their susceptibility to fishing pressure (Roberts, 1997). Groupers (Serranidae), for example, are highly vulnerable (Coleman *et al.*, 2000). In 1996, 21 species of groupers were proposed for the IUCN 'Red List'; of these three species are critically endangered (Hudson and Mace, 1996). Grouper stocks have been extirpated by intense fishing pressure in several parts of the Caribbean (Sadovy and Eklund, 1999).

In recent years, marine protected areas (MPAs) have received much attention as an alternative approach to traditional fisheries management (Roberts, 1997; Costanza *et al.*, 1998; Murray *et al.*, 1999). Ecologically, MPAs are thought to be able to simultaneously address problems that traditional management alone cannot. The primary goals of MPAs are to protect critical habitat and biodiversity, and to sustain or enhance fisheries by preventing spawning stock collapse and providing recruitment to fished areas (Medley *et al.*, 1993; Murray *et al.*, 1999). MPAs are thought to be important in the management and

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conservation of fisheries for vulnerable reef fish like groupers, especially when combined with additional management measures to limit effort (Chiappone *et al.*, 2000; Coleman *et al.*, 2000).

Since many fisheries are size-selective, MPAs are expected to increase both the average size and abundance of exploited species (Roberts and Polunin, 1993). In general, MPAs have proven effective in this capacity, particularly with regard to large carnivorous species such as groupers and snappers which are long-lived, slow-growing fishes with delayed reproduction (*e.g.*, Polunin and Roberts, 1993; Sluka *et al.*, 1998; Chiappone *et al.*, 2000). Since fecundity of fishes increases exponentially with length, an increase in both average size and abundance of fish within MPAs should lead to substantially greater fish production than in adjacent fished areas (Roberts and Polunin, 1993). Whether or not this production will enhance local fisheries depends on local oceanographic processes that transport larvae from protected spawning areas (Tupper and Juanes, 1999).

Coastal Management in the Turks and Caicos Islands

The use of MPAs is an important component of coastal fisheries and park management in the Turks and Caicos Islands (TCI). A National Parks Order formally established 33 terrestrial and marine protected areas in 1992 (Homer, 2000a). The primary orientation of the park system is towards recreation and human use, but some MPAs are specifically designated as fisheries reserves. The TCI commercial fishery has focused primarily on queen conch and spiny lobster over the past 40 years (Medley and Ninnes, 1998). Finfish are utilized less but have traditionally been more important in the local diet. All commercial fishing is prohibited within MPAs (Turks and Caicos Islands Government, 1994). There are, however, no size limits (other than indirect effects due to restrictions on trap mesh size), seasonal closures or species-specific fishing restrictions for reef finfish caught by commercial fishers outside MPAs. The use of SCUBA and spear guns for commercial fishing is prohibited in the TCI, so some reef fish are afforded refuge by depth.

The relatively pristine reefs in the TCI are prime attractions for tourists but increasing development, commercial fishing and tourism is putting stress on the inshore coral reef environment (Homer, 2000b). This pressure is a result of the growth of the tourist industry, which emerged as the number one industry in the country by the early-1990s. According to TCI Tourism Board figures, visitor arrivals rose 53% between 1995 and 1999, to more than 120,000 annually. Most of these tourists enjoy a variety of marine-oriented tourism activities (*e.g.*, fishing, diving, snorkeling, kayaking, sailing, and swimming) and over 6,100 tourists in 1999 indicated that diving was the main purpose of their visit.

There are potential conflicts between the tourism and fishing industries because both rely on coral reef ecosystem services for production of their respective 'commodities'. Reef fish have quasi-public good characteristics for the tourism and dive industries: the 'consumption' (*i.e*, viewing) of fish by tourists is non-subtractable until overcrowding and congestion set in (Davis and Tisdell, 1996). That is, the viewing of fish by one person does not subtract from the ability of others to 'consume' the public good at another time. The same fish, on the other hand, are subtractable when landed in the fishery – when a fish is caught, it becomes the private property of the fisher and is not available for use by any other fishers or tourists. Thus, the abundance and size of fish in an area is an important factor for both industries, although 'consumption' by the different sectors results in different physical outcomes.

Abundance and size of several commercially important reef fishes, including Nassau grouper, were measured within several habitat types in fished reefs and in a small MPA around South Caicos (Tupper, 2002). There was no difference in mean size or density of any grouper species between fished and unfished areas. It was determined that Nassau grouper prefer windward Pleistocene reef formations along the edge of the drop-off, at a depth of approximately 20-m. The MPAs around South Caicos contain relatively little of this habitat. This suggests that habitat preference may be more important than fishing pressure in determining the distribution and abundance of Nassau grouper. In order to see any effects of an MPA, the area must be subject to a lower level of fishing pressure than surrounding areas. It is possible that the level of fishing pressure around South Caicos is too low to cause a detectable difference in fish size and abundance between the MPA and surrounding areas (Tupper, 2002). Indeed, the density of all species measured at South Caicos (Tupper, 2002) rivaled or exceeded densities reported for these species within MPAs elsewhere in the Caribbean (*e.g.*, 0.35 – 0.62 Nassau groupers per 100-m^2 at South Caicos sites, compared to 0.01 per 100-m^2 in Florida and 0.16 – 0.20 per 100-m^2 in the Bahamas, Chiappone *et al.*, 2000). Thus, the local South Caicos MPA may not be effective in protecting reef fish stocks around South Caicos simply because they are not currently in need of protection.

Nassau Grouper Vulnerability in the Turks and Caicos Islands

The situation on South Caicos, however, may not be representative of the situation in other parts of the TCI. South Caicos is the center of the spiny lobster and queen conch fisheries, but is distant from the main tourism developments and restaurants on the island of Providenciales. There is little tourism and sport diving on South Caicos, while tourism in Providenciales is highly developed.

Many visitors expect to eat fresh seafood in local restaurants during their holiday. A survey of major TCI tourist hotels and restaurants (Rudd, unpublished data) estimated local demand for seafood of at least 3,000-kg per week. Grouper was the largest single product consumed, with restaurant consumption of 750-kg per week during the survey period (March and April 2000). Nassau grouper was the preferred local species for restaurant purchasers. Most seafood is supplied directly to hotels and restaurants by a small number of artisanal fishers based in Providenciales, but a substantial portion is also imported via Florida. Fresh and frozen finfish imports are, however, taxed at a rate of 40%, reducing their competitiveness in the TCI market. Local fishers enjoy higher local market prices than under free trade as a result and, according to economic theory, higher local prices for seafood should translate into higher fishing pressure on local stocks. While there are anecdotal reports of reduced abundance of groupers and other carnivorous reef fish near the island of Providenciales (M. Taylor, Coastal Resources Management Project, personal communication), no visual surveys have been completed in the region to quantify fishing impacts on reef fish abundance.

Most of the commercial fishers are based in South Caicos, about 60-km across the shallow Caicos Bank. The demand for groupers at the South Caicos fish processing plants is low; most are sold directly to restaurants or local residents for home consumption (M. Rudd, personal observation). Grouper and large snapper sold for local consumption typically fetch around US $3.25 per kg on South Caicos, compared to as high as US $15 per kg when sold fresh to hotels and restaurants on Providenciales (M. Rudd, personal observation). Given the higher value of their catch, fishers will travel the extra 120-km round trip to land their fish at Providenciales if their catch is around 100-kg (T. Morris, South Caicos Fishermen's Association, personal communication). Fishing pressure on South Caicos groupers is likely to increase in response to greater demand from Providenciales hotels and restaurants in the future.

In summary, there are three reasons for particular concern about the sustainability of the Nassau grouper fishery in the TCI: (1) the historic vulnerability of stocks in other parts of the Caribbean, (2) the lack of specific regulatory protection in the TCI, and (3) the unabated growth in the hotel and restaurant industry, where Nassau grouper is a preferred species. Large Nassau grouper may also be important attractions for visiting tourists who SCUBA dive (*e.g.*, Williams and Polunin, 2000). Depletion of Nassau grouper stocks to supply the local restaurant industry could inflict substantial external costs on the dive tourism industry if this is, indeed, the case.

Diver Preferences and Economic Value

In the tropics, the tourism industry is an important part of many economies and there has been rapid growth in nature-based tourism (Gössling, 1999). Viewing marine wildlife is recognized as one of the important services flowing from healthy ecosystems and providing economic value to nature-based tourists (*e.g.*, Shafer and Inglis, 2000; Williams and Polunin, 2000; Rudd, 2001). Large Nassau grouper are often featured prominently in dive industry advertising and promotional materials; it is therefore reasonable to hypothesize that divers derive well being from viewing Nassau grouper and that Nassau grouper, consequently, provide non-extractive economic value for dive tourists.

A number of methodologies have been used for assessing the value of ecological goods and services. Recently, a number of techniques developed in marketing research and broadly known as conjoint analysis (*e.g.*, Louviere, 1988) have been adapted for research on preferences for, and valuation of, environmental quality (*e.g.*, Farber and Griner, 2000). In these surveys, people are queried about their preferences for hypothetical environmental management policies or recreational market choices, which vary in environmental quality and price. Using these surveys, relatively subtle differences – based on heterogeneous personal preferences – in trade-off's between environmental quality and price can be assessed. Depending on the type of conjoint survey and analytical methodology used, utility-theoretic estimates of consumer welfare can be derived for use in cost-benefit analysis, or market share simulations more typical of business market research can be developed.

The strength of the conjoint approach results from of the use of nearly orthogonal survey designs that statistically isolate the effects of environmental attributes on choice. In a paired comparison conjoint analysis, for instance, survey respondents are presented with a choice of product profiles, each of which consists of a number of attributes. Each attribute can take on a number of specific levels that influence the value consumer's hold for the product. Whereas contingent valuation surveys ask respondents whether they are willing to pay a fee to improve environmental quality, a paired comparison conjoint survey asks respondents to provide a rating that specifies the strength of their preferences for one profile compared to the other.

In this research, we assessed diver preferences for viewing Nassau grouper and the marginal trade-off's that divers exhibited between fish size and abundance, dive group size and dive charter price. The research was exploratory in nature, assessing whether increased Nassau grouper size and abundance added value to the dive experience of visiting divers in the TCI. We used a simple linear utility model and ordinary least squares (OLS) to estimate regression coefficients subsequently used in market share simulations of dive charter package choice. While theoretically correct estimates of consumer welfare are important for comprehensive cost-benefit analyses, the demonstration that divers hold preferences for viewing more and/or larger Nassau grouper in market share simulations implies non-extractive use value. This alone could prove useful in policy debates in tropical countries, where the potential benefits arising from conservation of inshore marine resources are often overlooked.

Methods

The goal of the research was to assess the price sensitivity of divers to changing levels of key environmental attributes that add value to the experience of TCI dive tourists. To accomplish this, we used a self-administered full profile paired comparison survey instrument that was distributed to dive charter clients by nine local dive operators and to visiting university students at the Center for Marine Resource Studies.

The final survey instrument was comprised of five parts (a copy is available from the corresponding author on request): (1) questions about general respondent attitudes on the effects of fishing, development and marine tourism on coral reef environments; (2) a section providing background information about marine reserves and their potential effect on different types of marine animals diver may encounter in the TCI; (3) an example paired comparison question, followed by 18 paired comparison tasks for respondents; (4) a test contingent valuation methodology (CVM) question (used to assess appropriate bid ranges for future CVM surveys); and (5) questions about personal background.

Respondents (and dive operators) were assured that their answers (and input) would remain confidential.

Dive trip attributes of importance were initially identified through interviews with dive tour operators, and experienced sport and professional divers in the TCI. The appropriate levels for these attributes were chosen based on expert judgment, survey pre-testing and responses from a pilot survey. The final survey instrument used a total of five attributes and twenty levels: size of dive group (3-7, 8-14, 15-23, and 24-30 divers on the dive charter); presence of macrofauna (1 or more lobster, 1 or more sea turtles, 1 or more reef shark, or none of the above); Nassau grouper abundance (1, 3, 6, or 12 fish per dive); Nassau grouper mean size (small 2.27-kg, medium 6.80-kg, and large 13.61 kg); and dive charter price ($40, $41, $45, $50 or $60 per 20-minute single tank dive). All levels were within realistic bounds and all other attributes (*e.g.*, water depth, water clarity, coral cover, fish diversity, and unusual species) that may affect diver site choice were assumed constant across all sites.

The price of the dive was framed in terms of travel time to get to more pristine dive sites. In the pilot survey, several dive charter operators objected to queries about MPA entrance fees for dive sites with higher environmental quality. Dive operators thought it was inappropriate to equate a park entry fee, which they assumed would be collected by dive operators and add to their costs of doing business, with the maintenance of environmental quality within MPAs. In the eyes of the dive charter operators, government managers did not have the capacity to ensure that environmental quality was maintained within MPAs, even with additional funding. From an economic perspective, additional costs allocated to travel time to reach a site with higher environmental quality are theoretically equivalent to tourists paying an entry fee.

Paired comparison conjoint surveys can be cognitively challenging (Johnson and Desvouges, 1997). The design space for paired comparison questions rises exponentially with the number of attributes and levels considered. The number of questions asked must be restricted to avoid respondent exhaustion and increase the likelihood that respondents will complete and return surveys. This survey used one attribute with five levels, one with three levels, and three with four levels, yielding a potential design space of $5^2 4^6 3^2$ = 921,600 possible paired comparisons. The design challenge was to select a limited number of questions from the design space in such a way that useful information about attribute trade-off's is maximized. We used Sawtooth Software's Conjoint Value Analysis (CVA) software package (Sawtooth Software, 1996) to design a nearly orthogonal survey comprised of 18 paired comparison tasks. Every task was presented to each respondent in this research. While the number of questions for each respondent could be reduced using a blocked design (*e.g.*, asking each respondent only 8 of 32 total tasks), this would be problematic because we intended to distribute only a limited number of surveys in this exploratory research.

The CVA experimental design module identifies promising experimental designs by creating candidate pools of potential questions selected from the design space with a guided randomization process. For each candidate survey design, 108 paired comparison tasks were picked for evaluation. Tasks that contributed least to the experimental design were discarded one task at a time until the total number of tasks was reduced to 18. *D*-efficiency (Kuhfeld *et al.*, 1994) is a measure of the goodness of a specific experimental design relative to the ideal orthogonal balanced design (*i.e*, designs in which attributes vary independently and are shown an equal number of times). Each run of the CVA design module followed the iterative reduction process for five pools of comparison tasks, saving the survey design with highest *D*-efficiency. In this research, CVA runs were repeated 500 times; we thus chose the survey design with the highest *D*-efficiency of 2,500 candidate designs. A perfectly orthogonal and balanced experimental design would measure $D = 1.0$; our final survey measured $D = 0.903$.

A sample question is shown in Figure 9-1. It asks respondents to compare two dive profiles and rate the strength of their preference for one profile versus the other on a scale of 1 to 9. A rating of 1 indicates the respondent strongly prefers the first scenario, a rating of 9 indicates she strongly prefers the second scenario, and a rating of 5 indicates indifference between the choices. For many people, Option 1 was strongly preferred to Option 2 because it was a dive with smaller group size, reef sharks were present, and there were more abundant and larger Nassau grouper, for only $5 more per dive. Each of the 18 questions on the survey presented comparison tasks similar to this one, but in each question the two paired dive profiles varied due to the unique bundle of attribute values comprising each dive profile.

An ordinary least square (OLS) regression was used to estimate conjoint utilities (Sawtooth Software, 1996) as a function of the independent variables dive group size, presence of other animals, the abundance and average size of Nassau grouper, and the price of the dive. The dependent variable, preference rating, was re-scaled to a –4 to +4 scale for each of 18 survey questions. Group size, number of groupers, size of grouper, presence of macrofauna, and the price of the dive package, were coded as 0 (not present in the question), -1 (present in the left hand profile) or +1 (present in the right hand profile). The first level of each of the independent variables was dropped from the regression to avoid perfect multicollinearity. The regression coefficients from the linear model – known in marketing literature as part-worth's – were calculated and then available for use in the CVA market simulation module.

Category	**Option A**	**Option B**
Size of the Dive Group	15-23 Divers per Group	24-30 Divers per Group
Presence of Other Animals	1 or more Reef Sharks	No turtle, shark, lobster
Grouper Abundance	6 Groupers per Dive	3 Groupers per Dive
Average Grouper Size	Large Grouper: 30-lbs	Medium Grouper: 15-lbs
Price of the Dive	$45 per Single Tank Dive	$40 per Single Tank Dive

1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
A is much better		A is somewhat better		A & B are about equal		B is somewhat better		B is much better
Please circle a number from 1 to 9 that reflects your rating								

Figure 9-1 – Sample CVA question

Market simulations calculated the market share for each of four dive charter profiles that varied by price, dive group size, and Nassau grouper mean size and/or abundance. We assumed the respondent chose the single dive profile option that she most preferred, as measured by highest overall 'utility'. The utility level was calculated using regression coefficients from each individual respondent. The product with the highest overall utility for each respondent was assigned a score of '1', while all other profiles were given a score of '0'. The market simulator averaged the 'first choice' preference scores across all respondents and calculated percent market share for each hypothetical dive profile in a particular simulation.

A single baseline scenario was used in the various market simulations. In it, utility was calculated for each of four dive profiles that differed by dive charter group size and price: (1) a small group (3-7 divers) at a price of US $60 per dive per person; (2) a medium group (8-14 divers) at US $50; (3) a large group (15-23 divers) at US $45; or (4) a very large group (24-30 divers) at US $40. In the baseline profile, a single small (2.27-kg) Nassau grouper was observed per 20-minute dive. The macrofauna variables in the experiment were held constant across all simulations – no spiny lobster, sea turtles or reef sharks were observed (see Rudd, 2001 for results of macrofauna simulations).

Six simulations were conducted. In the first three, Nassau grouper abundance was increased one increment at a time (*i.e*, from 1 to 3, 6, and then 12 fish observed per dive) in the smallest, most expensive (US $60) dive profile only. Two similar simulations examined increments in mean fish size from small- to medium-size, and then small- to large-size, again for the small expensive dive profile only. The effects of the changes in size and abundance on market share for the small dive profile was calculated for the overall group and six demographic segments (male vs. female, 'younger' divers < 30 years vs. 'older' divers 30 years and over, and 'basic' divers with resort or open water certification vs. 'advanced' divers with rescue, divemaster or instructor certification). In the final simulation, Nassau grouper abundance and mean size was held constant and we examined the effect of changes in the price of the small group dive profile (falling to US $50 and $45) on market shares.

Hypothesis tests were conducted with *t*-tests, using the Bonferroni adjustment (total α-level of 0.05, so that testing multiple hypotheses did not increase the likelihood of rejecting at least one true null hypothesis). We first tested for the equality of market shares between demographic segments within each simulation (*e.g.*, could we reject the null hypothesis that the market shares for the small, expensive dive profile were equal for male and female divers when 12 groupers per dive were observed?). Secondly, we tested hypotheses regarding the equality of market shares for different dive profiles within simulations (*e.g.*, could we reject the null hypothesis that the market shares for the small US $60 dive profile and medium US $50 profile were equal when only 1 grouper per dive was observed?). Thirdly, we tested hypotheses that market shares for particular profiles were equal across simulations (*e.g.*, could we reject the null hypothesis that the market shares for the small US $60 dive profile were equal when small grouper or when large grouper were observed on the dive?). Finally, we tested the hypotheses that market shares for the small dive group profile were equal at prices of US $45, $50 and $60, when only one small Nassau grouper were observed.

Results

A total of 87 usable survey responses were employed in this analysis; this represents an overall response rate of approximately 31% based on a total distribution of 281 surveys. Because conjoint surveys are cognitively challenging, response rates are often lower than simpler surveys (*e.g.*, Farber and Griner, 2000 at 14%). Female respondents accounted for 53% (n = 46) of the 87 usable survey responses. 69% (n = 60) of the respondents were under 30 years of age. Sixty (69%) of the respondents were SCUBA certified at the basic levels. Household income was highly variable; of the 81 respondents that responded to this question, 33% (n = 21) reported household income of less than US $40,000 per year and 33% (n = 21) reported household income of more than US $125,000 per year. The self-reported most important factors influencing dive profile choice were dive group size (n = 37) and overall species diversity (n = 35); only 8% (n = 7) of respondents stated that dive price was the most important factor in their comparison tasks. The correlation between gender and certification, gender and age, and certification and age, was 0.10 (p = 0.351), 0.28 (p = 0.008), and 0.24 (p = 0.024), respectively. Younger divers tended to be female and younger divers tended to have basic certification, although gender and certification were not significantly correlated. For 81 respondents who reported income level, correlation between gender and income, age and income, and certification and income, was 0.05 (p = 0.668), 0.42 ($p < 0.001$), and 0.00 (p = 0.99), respectively. Older divers tended to have higher household income levels. Individual utility regressions were conducted for the 87 survey respondents.

The overall fit of the regressions was high, with an average R^2 = 0.97 (range 0.85 to 0.99, and one anomaly of 0.65). This was indicative of generally high levels of internal self-consistency in decision making for respondents. Market shares for all demographic segments in the baseline and the first three simulations of increasing Nassau grouper abundance (3, 6, and 12 fish per dive) are shown in Table 9-1.

Similarly, Table 9-2 shows the market shares for all demographic segments for the baseline and the two simulations where Nassau grouper mean size increased from small to medium or large.

Figure 9-2 shows the market share for the small US $60 dive profile as Nassau grouper abundance (Figure 9-2a) and mean size (Figure 9-2b) increase. Note that market share increases almost linearly as fish size increases but that there is a decline in the rate of market share increase as fish abundance increases. This indicates diminishing marginal returns for increasing grouper abundance.

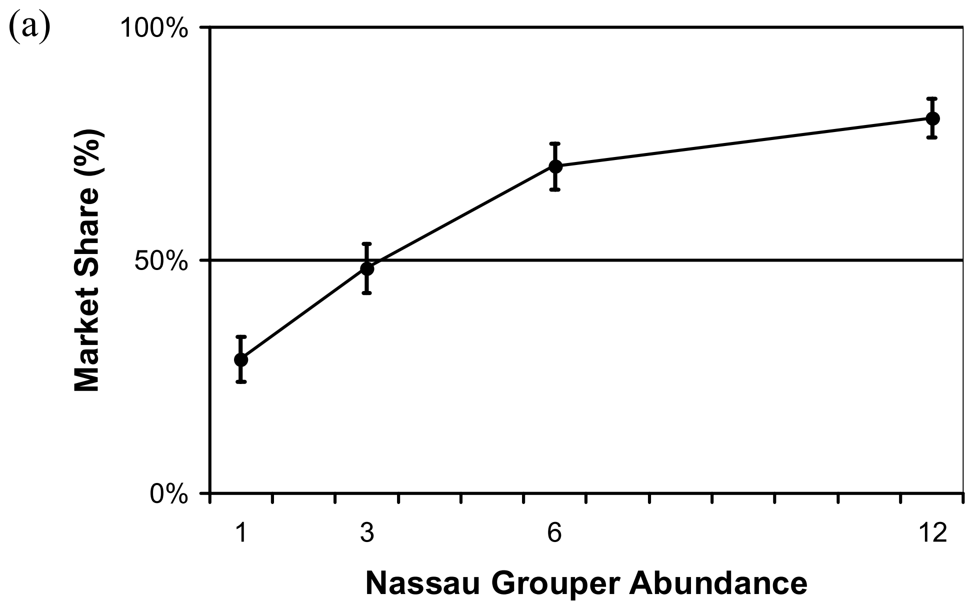
	Market Share (%) for Dive Profile			
	Small (US $60)	**Medium (US $50)**	**Large (US $45)**	**V. Large (US $40)**
(a) Baseline, 1 Nassau grouper				
Overall (n=87)	28.7	34.5	18.4	18.4
Female (n=46)	23.9	39.1	17.4	19.6
Male (n=41)	34.1	29.3	19.5	17.1
Younger (n=60)	16.7	41.7	18.3	23.3
Older (n=27)	55.6	18.5	18.5	7.4
Basic Certification (n=60)	25.0	36.7	20.0	18.3
Advanced Certification (n=27)	37.0	29.6	14.8	18.5
(b) 3 Nassau grouper				
Overall (n=87)	48.3	24.1	12.6	14.9
Female (n=46)	47.8	26.1	13.0	13.0
Male (n=41)	48.8	21.2	12.2	17.1
Younger (n=60)	43.3	26.7	11.7	18.3
Older (n=27)	59.3	18.5	14.8	7.4
Basic Certification (n=60)	48.3	26.7	11.7	13.3
Advanced Certification (n=27)	48.2	18.5	14.8	18.5
(c) 6 Nassau grouper				
Overall (n=87)	70.1	14.9	9.2	5.8
Female (n=46)	73.9	15.2	8.7	2.2
Male (n=41)	65.9	14.6	9.7	9.7
Younger (n=60)	66.7	18.3	8.3	6.7
Older (n=27)	77.8	7.4	11.1	3.7
Basic Certification (n=60)	71.7	15.0	8.3	5.0
Advanced Certification (n=27)	66.7	14.8	11.1	7.4
(d) 12 Nassau grouper				
Overall (n=87)	80.5	11.5	5.8	2.3
Female (n=46)	78.3	13.0	6.5	2.2
Male (n=41)	82.9	9.8	4.9	2.4
Younger (n=60)	80.0	13.3	3.3	3.3
Older (n=27)	81.5	7.4	11.1	0.0
Basic Certification (n=60)	83.3	10.0	5.0	1.7
Advanced Certification (n=27)	74.1	14.8	7.4	3.7

Table 9-1 – Simulation market shares (%) for four dive profiles (small, 3-7 divers per charter group; medium, 8-14 divers; large, 15-23 divers; and very large, 24-30 divers) when Nassau grouper abundance varies from 1 to 12 fish per dive

	Market Share (%) for Dive Profile			
	Small (US $60)	**Medium (US $50)**	**Large (US $45)**	**V. Large (US $40)**
(a) Small (2.27 kg) grouper				
Overall (n=87)	28.7	34.5	18.4	18.4
Female (n=46)	23.9	39.1	17.4	19.6
Male (n=41)	34.1	29.3	19.5	17.1
Younger (n=60)	16.7	41.7	18.3	23.3
Older (n=27)	55.6	18.5	18.5	7.4
Basic Certification (n=60)	25.0	36.7	20.0	18.3
Advanced Certification (n=27)	37.0	29.6	14.8	18.5
(b) Medium (6.80 kg) grouper				
Overall (n=87)	35.6	29.9	17.2	17.2
Female (n=46)	28.3	37.0	17.4	17.4
Male (n=41)	43.9	22.0	17.1	17.1
Younger (n=60)	25.0	36.7	16.7	21.7
Older (n=27)	59.3	14.8	18.5	7.4
Basic Certification (n=60)	33.3	31.7	18.3	16.7
Advanced Certification (n=27)	40.7	25.9	14.8	18.5
(c) Large (13.61 kg) grouper				
Overall (n=87)	50.6	20.7	16.1	12.6
Female (n=46)	41.3	28.3	17.4	13.0
Male (n=41)	61.0	12.2	14.6	12.2
Younger (n=60)	43.3	26.7	15.0	15.0
Older (n=27)	66.7	7.4	18.5	7.4
Basic Certification (n=60)	50.0	21.7	16.7	11.7
Advanced Certification (n=27)	51.9	18.5	14.8	14.8

Table 9-2 – Simulation market shares (%) for four dive profiles (small, 3-7 divers per charter group; medium, 8-14 divers; large, 15-23 divers; and very large, 24-30 divers) when mean Nassau grouper size varies from small (2.27 kg) to large (13.61 kg).

In the baseline scenario, the null hypotheses that (a) male and female divers, and (b) divers with basic and advanced certification, were equal could not be rejected (fail to reject H_0 at total $\alpha = 0.05$, $p > 0.05$). However, the null hypothesis that the market share for older and younger divers was equal was rejected ($p = 0.001$). Older divers exhibited stronger preferences than younger divers for the expensive small group dive, even in the baseline scenario where only one small Nassau grouper was seen. Null hypotheses that market shares were equal for male and female, and divers with basic and advanced certification could not be rejected for any of the simulations ($p > 0.05$). The null hypothesis of market share equality between the younger and older groups was rejected for the simulations when Nassau grouper size increased to medium ($p = 0.004$) and large ($p = 0.044$) mean size. The null hypothesis of equal market shares for the younger and older divers could not, however, be rejected for the simulations involving increases in Nassau grouper abundance ($p > 0.05$ in all three simulations). The balance of the results will be presented in terms of the overall group, and younger and older demographic segments only.



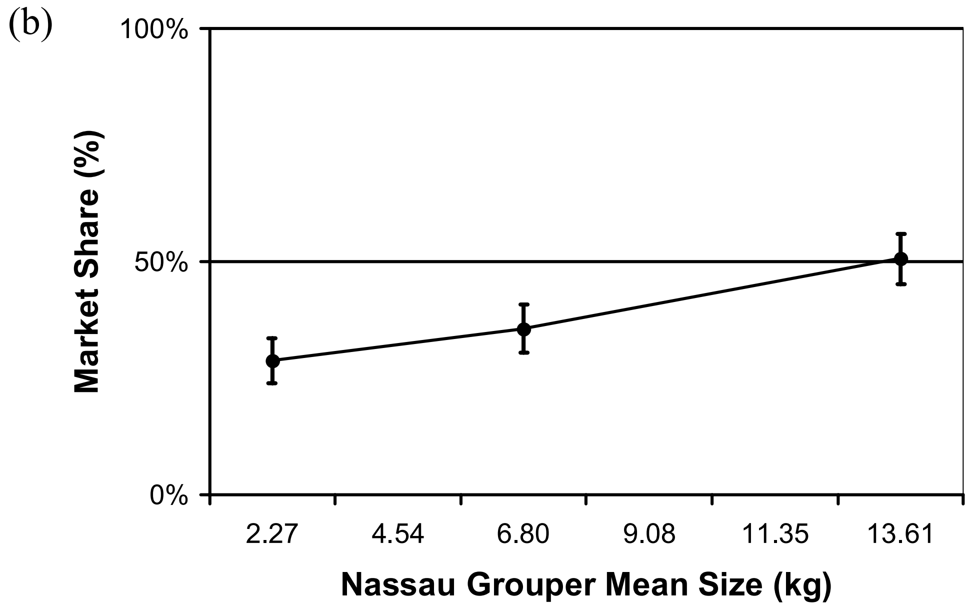


Figure 9-2 – Market share (%) and SE for small group dive profile (US $60) as Nassau grouper abundance increases from 1 to 12 fish per 20-minute dive (Figure 9-2a) and from small to large mean size (Figure 9-2b).

The second set of tests assessed the equality of different dive profile market shares within the same simulations (Table 9-3). For the baseline scenario, where one Nassau grouper was seen per dive, there were no significant differences in market shares ($p > 0.05$) between the small and medium, small and large, and small and very large dive profiles.

For younger divers, the null hypothesis that the market shares for the small and medium dive profiles were equal was rejected (p = 0.010); significantly more younger divers choose the less expensive (US \$50) medium dive group (41.7% market share) compared to the small expensive (US \$60) charter group (16.7% market share). Conversely, for older divers, the small dive group had the highest market share (55.6%) and the null hypotheses of equal market share between the small group and all the larger, less expensive dive groups could be rejected (p < 0.05 in all cases). Older divers exhibited much stronger preferences for the small group dive even though it was US \$10 more expensive than the nearest competitive dive profile and there were no difference in environmental quality.

	Market Share Test p-value (H_0: $S_i = S_j$)		
	H_0: $S_{small} = S_{medium}$	**H_0: $S_{small} = S_{large}$**	**H_0: $S_{small} = S_{verylarge}$**
(a) 1 Nassau grouper			
Overall (n=87)	0.503	0.161	0.161
Younger (n=60)	0.010	0.829	0.419
Older (n=27)	0.022	0.022	0.001
(b) 3 Nassau grouper			
Overall (n=87)	0.007	0.001	<0.001
Younger (n=60)	0.124	0.001	0.012
Older (n=27)	0.013	0.005	0.001
(c) 6 Nassau grouper			
Overall (n=87)	<0.001	<0.001	<0.001
Younger (n=60)	<0.001	<0.001	<0.001
Older (n=27)	<0.001	<0.001	<0.001
(d) 12 Nassau grouper			
Overall (n=87)	<0.001	<0.001	<0.001
Younger (n=60)	<0.001	<0.001	<0.001
Older (n=27)	<0.001	<0.001	<0.001

Table 9-3 – Test of equality of market shares (S) of small (S_{small}, 3-7 divers) and medium (S_{medium}, 8-14 divers), large (S_{large}, 15-23 divers), and very large ($S_{verylarge}$, 24-30 divers) dive profiles for the overall sample and age-based segments when Nassau grouper abundance varies. Paired t-test p-values are for total $\alpha = 0.05$.

As the abundance of Nassau grouper increased (Table 9-3), the null hypotheses of equal market share for the small dive group profile (US \$60) and other dive profile choices was rejected in all cases except for younger divers in the simulation where 3 Nassau grouper were observed (but where there had been a significantly higher market share for the medium group when only 1 fish was observed). As the abundance of Nassau grouper increased to 6 and 12 fish per dive, the more expensive small dive group profile was strongly preferred in all cases (*i.e*, null hypotheses of equal market shares were rejected at $p < 0.001$ in all cases).

Table 9-4 shows the results of hypothesis tests that the small dive group profile market shares for dives with different Nassau grouper mean size were equal within simulations. The baseline case (mean size = 2.27 kg) is the same as before, when one fish was observed. As the mean size of Nassau grouper increased to 6.80-kg, the null hypothesis that the market shares for small and medium dive group profiles were equal could not be rejected (p = 0.511). It could be, however, for small and large, and small and very large dive profiles (p = 0.017 in both cases). For the simulation with the largest size fish, the small group market share was significantly higher than other dive profile market shares except for the case of the small and medium market share for younger divers (p = 0.124).

	Market Share Test p-value (H_0: $S_i = S_j$)		
	H_0: $S_{small} = S_{medium}$	**H_0: $S_{small} = S_{large}$**	**H_0: $S_{small} = S_{verylarge}$**
(a) Small (2.27 kg) grouper			
Overall (n=87)	0.503	0.161	0.161
Younger (n=60)	0.010	0.829	0.419
Older (n=27)	0.022	0.022	0.001
(b) Medium (6.80 kg) grouper			
Overall (n=87)	0.511	0.017	0.017
Younger (n=60)	0.253	0.321	0.709
Older (n=27)	0.005	0.013	<0.001
(c) Large (13.61 kg) grouper			
Overall (n=87)	<0.001	<0.001	<0.001
Younger (n=60)	0.124	0.003	0.003
Older (n=27)	<0.001	<0.001	<0.001

Table 9-4 – Test of equality of market shares (S) of small (S_{small}, 3-7 divers) and medium (S_{medium}, 8-14 divers), large (S_{large}, 15-23 divers), and very large ($S_{verylarge}$, 24-30 divers) dive profiles for the overall sample and age-based segments when Nassau grouper mean size varies. Paired t-test p-values are for total $\alpha = 0.05$.

The results from inter-simulation tests of market share equality for the small dive group profile (US $60) are shown in Table 9-5, which shows p-values for all simulations on Nassau grouper mean size and abundance. For the overall group of respondents, the hypotheses that the small group market share in the baseline scenario was equal to the small group market shares in any of the simulations when Nassau grouper abundance or mean size increased was rejected (p = 0.013 for the medium size grouper simulation, and $p < 0.001$ for the four other simulations). Likewise, all tests of market share equality were rejected for the younger divers. For older divers, tests of market share equality could not be rejected for the first incremental increase in abundance to 3 fish per dive (p = 0.327), or for increases to medium (p = 0.327) and large (p = 0.083) mean fish size. The market shares for increases of abundance to 6 (p = 0.011) and 12 (p = 0.006) fish per dive were significantly higher than the baseline market share, however, for the older divers. Thus, in aggregate, divers exhibited strong preferences for both more abundant and larger grouper – they were willing to pay at least $10 extra for dives during which larger or more abundant Nassau grouper were observed. When broken down into demographic segments, however, the simulation market shares for older divers were not significantly different as grouper size increased.

	Market Share Test p-value (H_0: $S_{small, baseline} = S_{small, \Delta}$)				
	3 Nassau Grouper	**6 Nassau Grouper**	**12 Nassau Grouper**	**Medium Grouper**	**Large Grouper**
Overall (n=87)	<0.001	<0.001	<0.001	0.013	<0.001
Younger (n=60)	<0.001	<0.001	<0.001	0.024	<0.001
Older (n=27)	0.327	0.011	0.006	0.327	0.083

Table 9-5 – Test of equality for small dive (US $60) market shares ($S_{small, baseline}$) for the overall sample and age segments when Nassau grouper abundance or mean size increases ($S_{small, \Delta}$) from baseline (H_0: $S_{small, baseline} = S_{small, \Delta}$). Paired *t*-test *p*-values are for total α = 0.05.

Finally, we simulated the effect of dive charter price on market share for the small group profile given constant baseline environmental conditions. As the price of the small group dive profile increased from $45 to $60, there was a sharp decline in market share for the small group, from 92.0% at $45 per dive, to 71.3% at $50 per dive, to 28.7% at $60 per dive (Figure 9-3). The null hypotheses that the small group market shares were equal at prices of US $45 and $50, and US $50 and $60 were rejected ($p < 0.001$). The high degree of price sensitivity of respondents is in line with anecdotal information from dive operators and is indicative of the price competitiveness of the dive industry in the TCI.

Discussion

In order to be effective, MPAs must be planned with the ecology of target species in mind (Murray *et al.*, 1999), encompassing the habitats used by a target species and most of the species' home range (Kramer and Chapman, 1999). Nassau grouper do migrate from their home range to spawning aggregation sites (see Coleman *et al.*, 2000) and hence may spillover, for better or worse, to commercial fishing grounds along the migration path. Aggregations are highly vulnerable to fishing pressure (Sadovy and Eklund, 1999) and need protection to ensure the maintenance of long-term reproductive capacity. Larger MPAs that encompass aggregation sites and migration corridors, possibly in combination with seasonal closures during spawning and other effort controls, can increase the level of protection for reef fish that aggregate to spawn (*e.g.*, Chiappone *et al.*, 2000; Coleman *et al.*, 2000). Important spawning aggregations and migration corridors in the TCI are clearly inadequately protected at the present time by small MPAs that were arbitrarily designed.

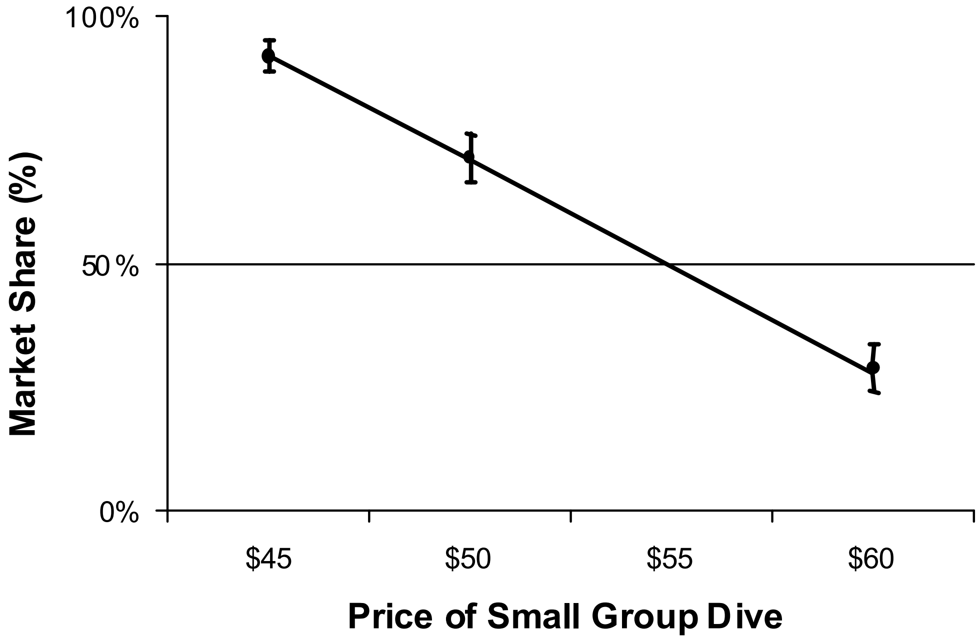


Figure 9-3 – Market share (%) and SE for small group dive profile when price increases from US $45 to $60 (given one 2.28-kg Nassau grouper per 20-minute dive for all dive profile choices).

Research on the effects of MPAs in the TCI has not provided evidence of any increases in Nassau grouper abundance and/or size within MPAs (Tupper, 2002). While evidence suggests that size and abundance of key reef species does increase in MPAs (Murray *et al.*, 1999), the effects of any particular MPA are highly dependent on site specific factors (Kramer and Chapman, 1999; Tupper and Juanes, 1999). Around South Caicos, higher densities of Nassau grouper near high-relief shelf edge habitat may reflect habitat selection or may be a function of fishing pressure (Tupper, 2002). On one hand, the MPAs near South Caicos contain little shelf edge habitat, so may not provide useful protection for Nassau grouper. Alternatively, fishing pressure may be so low near South Caicos as to render the current MPAs ineffective simply because Nassau grouper are not yet in need of protection.

From a fisheries perspective, it is not known how much fishing pressure Nassau grouper can be subjected to before stock decline and depletion occur. Coleman *et al. (*2000) recommend fishing mortality should be maintained at, or near, natural mortality for groupers. Given the history of depletion in the Caribbean (Sadovy and Eklund, 1999), it is clear that groupers are highly vulnerable. Depletion of a fishery under conditions of open access can lead to near, or total, dissipation of economic rent – fishers spend more time and money chasing fewer fish, such that they just cover their extra costs. Society as a whole is no better off economically as a result of the increased extraction. In the case of Nassau grouper fisheries, depletion of stocks can leave society worse off if the fish also have non-extractive economic value for the dive tourism industry.

According to economic theory, individual well being is based on personal preferences. This research demonstrated significant increases in market shares for dive charters with more abundant and larger Nassau grouper. This was because divers held personal preferences for viewing larger compared to smaller Nassau grouper, and more abundant compared to less abundant Nassau grouper. This implies that Nassau grouper have non-extractive use value, although the market simulation approach used in this exploratory research cannot quantify compensating surplus, the theoretically correct measure of WTP for non-market goods. We can, however, provide market simulation estimates of the value of increases in Nassau grouper size and abundance using weighted average prices (*i.e*, the sum of market share multiplied by price for each of the four dive group profiles). For the baseline scenario, the weighted average was $50.11 (28.7% share for the US $60 small group, 34.5% for the US $50 medium group, 18.4% for the US $45 large group, 18.4% for the US $40 very large group). Similarly, the weighted average price for the scenario where 12 Nassau grouper were observed on the small dive group was US $57.58, an increase of US $7.47 (14.9%) or US $0.70 per fish on average. When large (13.61-kg) Nassau grouper were observed on the small dive group, the weighted average price was US $53.00, an increase of US $2.89 (5.8%) or US $0.25 per kg on average. Taking any fish for food is very rare for recreational divers in the TCI and charter dive operators do not allow spear fishing by clients. We are therefore confident that the preferences that divers expressed imply non-extractive, rather than extractive, use values.

The rate of increase in overall market share as Nassau grouper mean weight increased was almost linear, while there was evidence of diminishing returns for increasing Nassau grouper abundance. Over 40% of divers reported that they felt that overall fish diversity was the most important single factor affecting their satisfaction with a particular diving experience. Once Nassau grouper abundance reaches a certain density, it appears that further increases in abundance added relatively little to overall diver satisfaction for our sample. Based on conversations with dive operators, it appears that divers may be implicitly correlating grouper size and overall dive quality as rarer, large Nassau grouper are indicative of unfished (Sluka *et al.*, 1998) and, therefore, superior dive sites.

Significant differences in market shares were observed in a number of simulations in which younger (< 30 years) and older (age 30+) divers were modeled separately. Older divers showed strong preferences for the smaller US $60 dive profile (56% market share), even when Nassau grouper abundance was no higher than on larger, less expensive dive profiles. Local dive charter operators and survey respondents suggested that this difference was likely because older divers tended to implicitly correlate the ability to observe any type of marine wildlife with the size of the dive group. If one was diving with a large group, the likelihood of seeing any groupers decreased because other divers were likely to frighten fish away after first contact. The correlation between certification and age was positive (0.24) and significant ($p < 0.05$), so more experienced older divers might hold stronger preferences for small group dives based on past experiences with large group dives. Older divers also had significantly higher household incomes than younger divers (correlation 0.42, $p < 0.001$) and may thus be willing to pay more for small group dives. Market shares were not significantly different from baseline for older divers when Nassau grouper varied in size, but they were when there were increases in abundance to 6 or 12 fish per dive. In simulations of macrofauna abundance, market shares for older divers were not significantly different in the presence or absence of spiny lobster but did exhibit significant differences from baseline in the presence of both sea turtles and reef sharks (Rudd, 2001). The differences in market shares for simulations with different animals suggest that the results are likely due to diver preferences rather than congestion effects.

There are a number of limitations with this exploratory survey. First, we do not know how many of the charter dive operators actually distributed our surveys. The dive operators in Providenciales tend to specialize and several have very distinct market niches (*e.g.*, some cater to visitors who come to the TCI exclusively for dive vacations, some operate small 'boutique' operations, and others operate large boats that cater to 'casual' divers at all-inclusive resorts where diving is just one of many possible outdoor activities in which to participate). Over-representation of one type of diver could have an important impact on paired comparison conjoint results. Similarly, the inclusion of visiting university students (who tended to be environmentally aware and had lower average household income than tourists) could bias the results. In general, paired comparison simulations tend to over-emphasize the value of individual attributes, resulting in an exaggeration of the importance of attributes that might be less salient in reality. Finally, the market simulations did not account for cross-attribute interactions (*e.g.*, what would the impact of fish that were both larger and more abundant be on market share?).

Still, for the purposes of this exploratory research, we believe this sample is adequate for illustrating our main point. That is, an increase in Nassau grouper abundance and/or mean size adds value to the dive experience because most divers hold preferences for viewing more fish and many divers hold preferences for viewing larger fish. Dive operators could charge higher prices and increase revenue by taking clients to sites with increased grouper abundance and/or larger mean size. Alternatively, MPA entrance fees could be used to capture consumer surplus resulting from increased grouper abundance. An increase of $5 per dive, for instance, should have little impact on the number of dives charter packages sold if there were abundant Nassau grouper at a relatively pristine dive site. The policy challenge in the TCI will be to tap into the typically unrealized and remarkable WTP for nature-based tourism (Gössling, 1999).

Our survey showed that divers are relatively price insensitive when animal abundance was high but that there was strong resistance to higher prices for dives that differed only by dive group size. This is consistent with the experiences of dive operators in the TCI; they were very cognizant of the importance of dive price as a factor in client decision making. Dive operators fully realize that maintaining reef quality – coral diversity and fish abundance – is crucial to the success of their businesses. At the same time, however, they tended to be very wary of any increases in dive price that might be caused by MPA user fees. Their caution stems from a wariness of the government's ability to actually transform MPA revenue into concrete actions to protect the reefs.

Government, on the other hand, has a mandate to manage public goods for the citizens of the TCI. The emerging National Park Service (NPS) is now funded by a 1% value-added tax on hotel accommodation and meals. Revenue for 2000 was projected at approximately $550,000, but that amount was unlikely to finance the management of all TCI MPAs (F. Homer, Coastal Resources Management Project, personal communication). Homer (2000a, 2000c) estimated that recurring direct costs to manage three existing MPAs near Providenciales (Princess Alexandra Land and Sea National Park, Northwest Point Marine National Park and West Caicos Marine National Park – all of which are used extensively by dive charter operators) would range from US $113,000 to $185,000 annually after first year costs of US $319,000 for establishing management capacity. Thirty additional parks, reserves and heritage sites, several in remote areas, also need to be managed by the NPS. Ensuring adequate protection of environmental quality in the face of budget constraints is difficult, especially when growing tourism will likely lead to higher fishing pressure that is difficult to monitor and enforce, and increasing reef damage in high use areas.

What do we yet need to fill in the balance of the economic puzzle? This survey does not provide a quantitative estimate of consumer surplus, which will be needed in the future. The logical follow-up to this exploratory study is further research that quantifies theoretically based measures of consumer surplus using an expanded conjoint survey or choice experiment (*e.g.*, Johnson and Desvouges, 1997). The goal of that survey would be to quantify the WTP of tourists for specific marine attributes important to their TCI experience. A broader pre-trip survey would also be very useful in assessing how environmental quality enters into the decision of divers to travel to the TCI, and if business and government support for MPAs have value as a signal to environmentally conscious consumers interested in high-quality diving. In addition, research is needed on both fisheries and dive industry production economics. This type of research is rare for recreational fisheries and tourism (see Rudd *et al*, in press) but essential for the quantification of total social welfare. Producer surplus (the appropriate measure of producer welfare) is simply given by the area above the supply curve and below price for a certain production level – it, thus, depends on the production costs within an industry. It is an important component of overall economic welfare that is often ignored in recreational and environmental economics, where the focus is most often on consumer welfare.

Under open access conditions typical of Caribbean fisheries (Christy, 1997), the economic rent generated by using renewable natural resources can be totally dissipated. In the TCI, the depletion of Nassau grouper could cause welfare losses for all major actor groups. In the face of declining stocks, fishers could dissipate rent by chasing fewer fish and dive charter operators would face rising costs as a result of price competition or from incurring higher costs to travel farther afield to pristine sites. Consumers would lose because the cost of fresh fish in restaurants would rise, as fresh fish become scarcer. Divers would lose because they derive less satisfaction and well being from their dive experience. The government of the TCI, which raises a substantial portion of it's revenue through a 9% sales tax on accommodations and restaurant meals, also stands to lose if the TCI loses comparative advantage in the Caribbean dive tourism market.

A lesson arising from the TCI research is to start economic valuation with the obvious. We know that quantifying the ecological effects, and hence the extractive use value, of MPAs is difficult in the tropical inshore environment. It is relatively easier, however, to assess any increase in size and abundance of animals within an MPA. In a country like the TCI, where tourism is the most important part of the economy, it makes logistical and economic sense to start valuation efforts by focusing on the non-extractive use value that tourists hold for the natural environment. The non-extractive economic value of viewing marine wildlife within MPAs is, however, only one of the services that MPAs may provide in the TCI. It is also clear that increased efforts should be directed towards understanding the ecological systems that MPAs are meant to protect if the full economic costs and benefits of MPAs are to be calculated and used for policy purposes.

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Appendix 9A – Paired Comparison Survey Methodology

Paired comparison conjoint analysis has a long history in market research and psychometrics (Green and Srinivasan, 1978; Green *et al*., 2001) and has roots in Lancaster's (1966) theory of value. Lancaster postulated that utility is derived from the characteristics that goods possess, rather than the inherent nature of a good itself. That is, a good can be characterized by certain attributes that are the source of a consumer's utility. In conjoint analysis, it is common to refer to a product 'profile' as being composed of distinct 'attributes', each of which may take on a number of distinct 'levels'.

In paired comparison conjoint analysis, survey respondents are asked to express their preferences for one profile relative to another using a rating scale. In this dive survey, each dive profile was composed of a bundle of attributes that were important in determining overall dive quality but that varied in level between the two profiles. This permitted an assessment of the marginal trade-offs that divers made regarding key characteristics of the dive experience.

In environmental economics, there has been a move towards the use of choice experiments to value environmental amenities (Louviere *et al*., 2000; Hanley *et al*., 2001). Choice experiments have a similar format to paired comparison surveys in that two (or more) profiles are compared, but instead they ask survey respondents to make a single choice about their most preferred option. This design uses random utility theory to model choices and can be used to derive utility theoretic measures of consumer surplus. Paired comparison conjoint analysis can be used to derive consumer surplus only with certain designs (*e.g.*, Johnson *et al*., 2000); in general, paired comparison conjoint studies cannot be used to calculate consumer surplus.

Why, then, use a paired comparison conjoint analysis to assess diver preferences if consumer surplus measures can't be estimated? Paired comparison surveys do have some advantages in exploratory studies such as this one where there is little guidance from previous research on even what attributes are important to consumers. Paired comparisons elicit more information about the strengths of people's preferences. In this research, funding limitations made it important to gain the most information possible about preferences from what was expected to be a limited pool of respondents. The general intent was to use this study to further develop a choice experiment of MPA value based on insights developed about diver preferences.[2]

Further, even when theoretically correct calculations of consumer welfare are available, they are most often used to inform policy debates within a broader social context. A 'blue-ribbon' panel (Arrow *et al.* 1993) recommended that willingness to pay figures derived from contingent valuation surveys are halved as a starting point for cost-benefit analysis or litigation purposes. Economic efficiency is an important consideration in the policy process, but it is only one of many that influence policy decisions in political arenas (*e.g.*, Weimer and Vining 1998). From a policy analysis perspective, we are interested in understanding people's incentives and behavior – a paired comparison conjoint analysis can help achieve this objective, even without welfare estimates.

Choice of Survey Attributes and Levels

The survey was developed with input from TCI commercial dive operators, a focus group, and two pilot surveys of visiting university students and tourists. We found from preliminary feedback that several major attributes influenced the actual dive experience (rather than the entire 'tourist experience' in the TCI). Because paired comparison surveys are cognitively challenging and can lead to respondent exhaustion (Huber, 1997), it is important to limit the number of questions in a survey. There is an implicit trade-off between the amount of information one can gather in a paired comparison survey and the likelihood that respondents will complete the survey. After considering a variety of different attributes that divers consider important, five key attributes were chosen for inclusion in this survey: the size of the dive group; the price of the dive; the presence of macrofauna (reef shark, sea turtle or spiny lobster); Nassau grouper abundance; and mean Nassau grouper size.

Experimental Design

The survey was designed using the Sawtooth Software Conjoint Value Analysis (CVA) software (Sawtooth Software, 1996). An optimally efficient paired comparison survey would be both orthogonal (*i.e*, attributes vary completely independently) and balanced

[2] Unfortunately funding and support for a further study in the TCI was not forthcoming, but there is currently (2003) a follow-up to this study underway in Micronesia. The current research uses a fractional factorial choice experiment to calculate non-extractive use value and non-use value of Napoleon wrasse, an icon reef fish in the Pacific of similar stature to the Nassau grouper in the Caribbean.

(*i.e*, each attribute is shown an equal number of times). This survey used five attributes with a total of 20 levels, yielding a design space of over 917,000 comparisons.

A full factorial experimental design is not feasible for this number of attributes and levels. Further, even fractional factorial designs can require a substantial number of paired comparisons, especially if the design is of sufficiently high resolution to allow estimation of all main effects and two-way interactions.[3] As this research was exploratory in nature and a limited number of respondents were being surveyed (*i.e*, limiting the feasibility of distributing surveys with blocks of manageable comparison tasks), we instead chose to use a nearly orthogonal experimental design.

Efficiencies for nearly orthogonal designs can be measured in a number of ways (Kuhfeld *et al.,* 1994), including the measure known as *D*-efficiency that is used by Sawtooth's CVA. For an (N_D x p) design matrix, $\boldsymbol{X}$, *D*-efficiency calculation is based on the information matrix $\boldsymbol{X'X}$. Specifically, it is a function of the geometric mean of the eigenvalues, $|(\boldsymbol{X'X})^{-1}|^{1/p}$. The eigenvalues provide a measure of the 'size' of the variance-covariance matrix. If a design is balanced and orthogonal, it has optimum efficiency and, conversely, the more efficient a design, the more it tends towards balance and orthogonality. Using the CVA experimental design module, a nearly orthogonal and balanced experimental design consisting of 18 survey questions was constructed by choosing the design with the highest *D*-efficiency:

$$D = \frac{100}{N_D \left|(X'X)^{-1}\right|^{1/p}}$$

where N_D = number of choice tasks (=18); p = number of attributes (=5); and X = design matrix using orthogonal coding.

Candidate surveys were generated using five pools of 108 randomly chosen conjoint comparisons each. The CVA algorithm excluded one question at a time for each of the five pools, deleting the task that contributed least to the efficiency of the overall design, until 18 questions remained. This survey generation procedure was repeated 500 times,

[3] Although we did not examine two-way interactions, it is possible to use commercially available hierarchical Bayesian analysis software to estimate all two-way (and higher level) interactions in paired comparison analyses.

resulting in the selection of a final survey instrument with $D = 0.903$ (where a score of 1.0 is fully orthogonal and balanced).

Table 9A-1 shows the final design and the ratings of respondent 1. For each question, the survey design shows 20 possible levels in total. A coding of '-1' indicates that level was present in the left-hand profile, '1' indicates that level was present in the right-hand profile, and '0' indicates that level was not used in the survey question. For example, choice task 1 asks respondents to compare a left-hand profile composed of levels 1, 6, 9, 13 and 19 (3-7 divers; 1 or more spiny lobster; 1 grouper; mean grouper size – small; $50 per dive) with a right-hand option composed of levels 4, 8, 10, 14 and 16 (24-30 divers; 1 or more reef sharks; 3 grouper; mean grouper size – medium; $40 per dive). The respondent's rating for each of 18 paired comparisons is shown in the final column; '1' indicates that the respondent strongly prefers the left-hand option, '9' indicates the respondent strongly prefers the right-hand option, and '5' indicates that the respondent is indifferent between the two options. This same design was used for all survey respondents – there was no blocking – so each respondent answered the same 18 paired comparison survey questions. Table 9A-2 shows summary statistics for the ratings of all respondents.

An ordinary least square (OLS) regression was conducted for each survey respondent. The preference rating for each of 18 survey questions, the dependent variable, was re-scaled to a –4 to +4 scale. The first level of each attribute was dropped from the regression to avoid linear dependency (Table 9A-3).

In the marketing literature, the individual regression coefficients are referred to as 'part-worth utilities' or just 'part-worth's' (the marginal valuations of choice variables) and are used to calculate linear, additive utility measures used in market simulations. Note that the CVA software calculated an intercept, divided it by the number of attributes and added the quotient to every regression coefficient. Summary statistics for the individual regressions are shown in Table 9A-4.

Survey	Divers				Macrofauna				Grouper Abundance				Grouper Size				Price				
Question	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	Rating
1	-1	0	0	1	0	-1	0	1	-1	1	0	0	-1	1	0	1	0	0	-1	0	5
2	0	0	1	-1	0	0	1	-1	1	0	0	-1	1	0	-1	0	1	0	-1	0	3
3	1	-1	0	0	1	-1	0	0	0	1	-1	0	1	-1	0	0	-1	0	0	1	2
4	0	-1	1	0	0	1	0	-1	0	0	-1	1	-1	0	1	0	-1	1	0	0	6
5	-1	1	0	0	1	0	0	-1	0	-1	1	0	-1	0	1	0	0	-1	0	1	2
6	0	-1	0	1	-1	0	1	0	0	0	1	-1	0	-1	1	0	-1	0	0	1	2
7	0	1	0	-1	-1	1	0	0	-1	1	0	0	1	-1	0	0	0	-1	1	0	5
8	1	0	-1	0	0	0	1	-1	0	0	1	-1	0	1	-1	0	0	0	1	-1	7
9	1	0	-1	0	-1	1	0	0	0	-1	0	1	1	0	-1	0	0	0	-1	1	6
10	-1	0	0	1	0	-1	0	1	0	-1	0	1	1	0	-1	0	0	-1	1	0	3
11	0	1	-1	0	0	1	-1	0	1	0	-1	0	-1	1	0	-1	0	1	0	0	4
12	0	0	-1	1	1	0	0	-1	0	1	-1	0	0	1	-1	1	0	-1	0	0	3
13	0	-1	1	0	-1	0	0	1	1	-1	0	0	-1	1	0	0	0	-1	1	0	4
14	1	0	-1	0	0	0	-1	1	1	0	0	-1	0	-1	1	1	0	0	0	-1	9
15	0	0	1	-1	1	0	-1	0	0	-1	1	0	0	1	-1	0	-1	1	0	0	4
16	1	0	0	-1	0	-1	1	0	-1	0	0	1	-1	1	0	-1	0	0	0	1	5
17	-1	1	0	0	0	-1	1	0	0	0	-1	1	0	-1	1	0	0	1	0	-1	7
18	0	-1	0	1	1	0	-1	0	-1	0	1	0	-1	0	1	-1	1	0	0	0	5

Table 9A-1 – Coded survey design and response for a single survey respondend

	Task 1	Task 2	Task 3	Task 4	Task 5	Task 6	Task 7	Task 8	Task 9	Task 10	Task 11	Task 12	Task 13	Task 14	Task 15	Task 16	Task 17	Task 18
Mean	5.01	4.71	2.43	3.41	2.31	3.95	6.93	6.34	6.54	3.56	4.34	1.74	5.18	6.94	4.43	7.59	6.71	3.34
Standard Deviation	2.38	2.24	1.85	2.22	1.28	2.26	1.32	2.18	1.62	1.95	2.07	1.10	1.76	2.11	1.92	1.45	2.34	1.73
Median	5	4	2	3	2	3	7	7	7	3	5	1	5	7	4	8	7	3
Mode	7	3	1	3	2	2	7	7	6	3	5	1	5	9	3	9	9	3
Minimum	1	1	1	1	1	1	4	1	2	1	1	1	2	2	1	1	1	1
Maximum	9	9	8	9	6	9	9	9	9	9	9	5	9	9	9	9	9	8
Range	8	8	7	8	5	8	5	8	7	8	8	4	7	7	8	8	8	7

Table 9A-2 – Survey responses to paired comparison questions (n = 87 respondents).

Survey	Divers		Macrofauna			Abundance			Size			Price			Scaled
Question	2	3	4	6	7	8	10	11	12	14	15	17	18	19	20	Rating
1	0	0	1	-1	0	1	1	0	0	1	0	0	0	-1	0	0
2	0	1	-1	0	1	-1	0	0	-1	0	-1	1	0	-1	0	-2
3	-1	0	0	-1	0	0	1	-1	0	-1	0	-1	0	0	1	-3
4	-1	1	0	1	0	-1	0	-1	1	0	1	-1	1	0	0	1
5	1	0	0	0	0	-1	-1	1	0	0	1	0	-1	0	1	-3
6	-1	0	1	0	1	0	0	1	-1	-1	1	-1	0	0	1	-3
7	1	0	-1	1	0	0	1	0	0	-1	0	0	-1	1	0	0
8	0	-1	0	0	1	-1	0	1	-1	1	-1	0	0	1	-1	2
9	0	-1	0	1	0	0	-1	0	1	0	-1	0	0	-1	1	1
10	0	0	1	-1	0	1	-1	0	1	0	-1	0	-1	1	0	-2
11	1	-1	0	1	-1	0	0	-1	0	1	0	0	1	0	0	-1
12	0	-1	1	0	0	-1	1	-1	0	1	-1	0	-1	0	0	-2
13	-1	1	0	0	0	1	-1	0	0	1	0	0	-1	1	0	-1
14	0	-1	0	0	-1	1	0	0	-1	-1	1	0	0	0	-1	4
15	0	1	-1	0	-1	0	-1	1	0	1	-1	-1	1	0	0	-1
16	0	0	-1	-1	1	0	0	0	1	1	0	0	0	0	1	0
17	1	0	0	-1	1	0	0	-1	1	-1	1	0	1	0	-1	2
18	-1	0	1	0	-1	0	0	1	0	0	1	1	0	0	0	0

Table 9A-3 – Dummy variable OLS regression coding and rating results for a single survey respondent. The first level of each variable was dropped and respondent ratings were recoded to a '-4' to '+4' scale (scaled rating = ordinal rating – 5)

Attribute and Level	Variable	Mean	Std Dev	Median	Min	Max
3-7 Other Divers	x_1	0.081	0.182	0.030	-0.236	0.855
8-14 Other Divers	x_2	-0.462	0.636	-0.512	-1.839	0.835
15-23 Other Divers	x_3	-1.297	1.063	-1.245	-3.275	0.829
24-30 Other Divers	x_4	-1.977	1.363	-1.868	-5.137	0.662
No other macrofauna	x_5	-0.008	0.078	-0.015	-0.236	0.185
1 or more spiny lobster	x_6	0.850	0.704	0.918	-0.829	2.277
1 or more sea turtles	x_7	1.568	1.235	1.510	-1.644	4.463
1 or more reef sharks	x_8	1.827	1.004	1.778	-0.888	4.371
1 Nassau grouper	x_9	-0.081	0.190	-0.027	-0.703	0.185
3 Nassau groupers	x_{10}	0.375	0.575	0.345	-0.694	1.781
6 Nassau groupers	x_{11}	0.840	0.773	0.750	-0.674	3.272
12 Nassau groupers	x_{12}	1.091	0.911	0.965	-0.665	3.292
Small grouper - 5 lbs	x_{13}	-0.107	0.166	-0.067	-0.600	0.185
Small grouper - 15 lbs	x_{14}	0.156	0.430	0.061	-0.580	1.713
Small grouper - 30 lbs	x_{15}	0.483	0.611	0.430	-0.555	2.511
$40 per single tank dive	x_{16}	0.246	0.328	0.130	-0.198	1.334
$41 per single tank dive	x_{17}	0.083	0.551	0.108	-1.788	1.331
$45 per single tank dive	x_{18}	-0.064	0.620	0.019	-1.808	1.311
$50 per single tank dive	x_{19}	-0.522	0.772	-0.470	-2.492	1.105
$60 per single tank dive	x_{20}	-1.259	1.055	-1.168	-4.345	1.101
Individual Regression R^2		0.969	0.048	0.980	0.650	1.000

Table 9A-4 – Summary statistics of respondent part-worth values (n=87)

Market Simulations

The CVA market simulation module was used to model the market share for various hypothetical dive profiles. The strength of various types of conjoint analyses is that they allow the modeling of market share for products not currently ‘on the market’; this explains their potential for valuing and modeling consumer choices about new products not yet in the market (Anderson and Bettencourt, 1993). In these simulations, total utility for each alternative hypothetical dive profile was calculated using the regression coefficients for each survey respondent. Each respondent was assumed to choose the dive profile with the highest overall utility in the simulation. The individual choices were aggregated to determine market share (% of respondents choosing the option) for each dive profile.

For example, Table 9A-5 shows part-worth summations for two survey respondents. When other attributes are held constant, Diver 1 would chose a dive in which reef sharks were seen, while Diver 2 would chose a dive where sea turtles were seen.

To develop simulations of increasing Nassau grouper abundance, utilities are first calculated for all respondents when Nassau grouper abundance is set equal to one and dive price at \$40. This process is then repeated when the dive price is increased to \$45, \$50, and \$60, resulting in a series of market shares for dives where only one grouper is observed. This process is then repeated for scenarios where grouper abundance increases to 3, 6 and 12 fish observed per dive. The mean market share and standard error for the \$60 dive package can then be shown as a function of grouper abundance (Figure 9-2) and differences between product profile can be assessed using paired *t*-tests.

		Part-Worth Utilities		Scenario				Diver 1 Utility				Diver 2 Utility			
Attribute and Level	Variable	Diver 1	Diver 2	No other macrofauna	1 or more spiny lobster	1 or more sea turtles	1 or more reef sharks	No other macrofauna	1 or more spiny lobster	1 or more sea turtles	1 or more reef sharks	No other macrofauna	1 or more spiny lobster	1 or more sea turtles	1 or more reef sharks
3-7 Other Divers	x_1	-0.055	-0.086												
8-14 Other Divers	x_2	-1.328	-0.229	1	1	1	1	-5.312	-5.312	-5.312	-5.312	-0.916	-0.916	-0.916	-0.916
15-23 Other Divers	x_3	-1.874	-0.249												
24-30 Other Divers	x_4	-1.894	-0.857												
No other macrofauna	x_5	-0.055	-0.086	1				-0.055				-0.086			
1 or more spiny lobster	x_6	0.486	0.499		1				0.486				0.499		
1 or more sea turtles	x_7	0.433	1.896			1				0.433				1.896	
1 or more reef sharks	x_8	0.652	-0.888				1				0.652				-0.888
1 Nassau grouper	x_9	-0.055	-0.086												
3 Nassau groupers	x_{10}	0.115	0.185	1	1	1	1	0.460	0.460	0.460	0.460	0.740	0.740	0.740	0.740
6 Nassau groupers	x_{11}	0.917	0.473												
12 Nassau groupers	x_{12}	1.507	1.192												
Small grouper - 5 lbs	x_{13}	-0.055	-0.086												
Small grouper - 15 lbs	x_{14}	0.113	0.163	1	1	1	1	0.452	0.452	0.452	0.452	0.652	0.652	0.652	0.652
Small grouper - 30 lbs	x_{15}	0.775	0.717												
$40 per single tank dive	x_{16}	-0.055	-0.086												
$41 per single tank dive	x_{17}	-0.541	-0.510												
$45 per single tank dive	x_{18}	-0.704	-0.605	1	1	1	1	-2.816	-2.816	-2.816	-2.816	-2.420	-2.420	-2.420	-2.420
$50 per single tank dive	x_{19}	-1.469	-0.625												
$60 per single tank dive	x_{20}	-3.172	-1.725												
Total Utility								-7.271	-6.730	-6.783	-6.564	-2.030	-1.445	-0.048	-2.832
Maximum Utility											-6.564			-0.048	

Table 9A-5 – Calculation of first choices for two divers for a dive simulation where the presence of macrofauna varies. Total ‘utility’ is calculated by a linear summation of part-worth’s for two survey respondents when hypothetical dive profiles vary only according to the presence of macrofauna and all other variables are help constant. Diver 1 derives maximum utility from a dive where a reef shark is observed, while Diver 2 derives maximum utility from a dive where a sea turtle is observed.

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CHAPTER 10

SPECIES-SPECIFIC IMPACTS OF A SMALL MARINE RESERVE ON REEF FISH PRODUCTION AND FISHING PRODUCTIVITY IN THE TURKS AND CAICOS ISLANDS[1]

In recent years, the establishment of marine reserves closed to fishing has been promoted as a cost-effective means to protect exploited species from overfishing (Bohnsack, 1993; Russ and Alcala, 1996; Murray *et al., 1*999; Roberts *et al., 2*001). The potential ecological advantages of marine reserves are thought to be the maintenance of a critical spawning-stock biomass to ensure recruitment supply to fished areas, and the possible enhancement of yields in areas adjacent to the reserve via emigration of adult fish (Johnson *et al., 1*999; Roberts *et al., 2*001). Marine reserve proponents have argued that they are simple and inexpensive to monitor and enforce, thereby having cost advantages over more traditional effort- or catch-oriented fisheries management alternatives (Bohnsack, 1993; Roberts and Polunin, 1993).

Whether or not marine reserves achieve their ecological and economic potential depends partly on the behaviour of local fishers. If compliance is poor, reserve benefits may prove difficult to achieve because of unsustainable fishing pressure and/or escalating enforcement costs (Mascia, 2000). The probability of compliance will increase in common pool resource systems when local users, who bear most of the costs of an area closure, derive direct benefits from that closure (Ostrom, 1990). For many areas with limited opportunities for economic diversification, it will be critical that fishers benefit from improved fishing opportunities arising from the emigration of commercially important fishes from marine reserves if they are to be viable. Numerous studies have shown that marine reserves contain a higher abundance and/or mean size of fish than adjacent fished reefs (*e.g.*, Koslow *et al., 1*988; Russ and Alcala, 1989, 1996; Polunin and Roberts, 1993; Rakitin and Kramer, 1996; Wantiez *et al., 1*997; Johnson *et al., 1*999; Tupper and Juanes, 1999; Roberts *et al., 2*001), and several studies have shown an increase in catch-per-unit-effort (CPUE) in fishing grounds adjacent to marine reserves (Alcala and Russ, 1990; Bennett and Attwood, 1991; McClanahan and Kaunda-Arara, 1996; McClanahan and Mangi, 2000; Roberts *et al., 2*001; Kelly *et al., 2*002). However,

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to date there is little evidence that marine reserves can increase total catches, such that the loss of fishing grounds is mitigated by increased catches outside the reserve (McClanahan and Mangi, 2000).

The degree of emigration or 'spillover' from marine reserves, which should increase fishery landings and/or reduce CPUE in adjacent fishing grounds, depends on the rate of fish migration across reserve boundaries (DeMartini, 1993). Reef fishes are generally considered sedentary, although the scale of movement varies among species (Chapman and Kramer, 1999, 2000; Meyer *et al., 2*000). Some studies have shown that many species of fish migrate considerable distances to forage (Hobson, 1973; Bryant *et al.*, 1989; Helfman, 1993; Burke, 1995) or reproduce (Shapiro, 1987; Bolden, 2000). In contrast, however, other research has found no emigration reserves (Buxton and Allen, 1989) or that the difference in density between fished and protected reefs was not related to species mobility (Chapman and Kramer, 1999). Whether a marine reserve is a preferred policy tool will, thus, depend on species- and site-specific factors. Implementing reserves for species, or in areas, where size and/or abundance increased within the reserve but where spillover was insignificant would amplify incentives for fishers to disregard reserve regulations. At best this would be inefficient, because of the need for costly monitoring and enforcement and, at worst, ineffective for either fisheries or conservation purposes. In these situations, it is likely that other policy tools provide higher fisheries benefits than marine reserves and, thus, gain the support from fishers necessary for successful implementation.

In this research, we examine evidence for emigration of three commonly targeted fish, Nassau grouper (*Epinephelus striatus*), hogfish (*Lachnolaimus maximus*), and white margate (*Haemulon album*), from a small marine reserve near South Caicos, Turks and Caicos Islands (TCI). Nassau grouper are the preferred target for local consumption (Rudd and Tupper, 2002) and fetch the highest price at dockside (up to US$3.50 per kg). White margate are the most commonly landed fish due to their higher relative abundance. Landing prices are typically around US$2.20 per kg for the smaller margate and hogfish. The rapid development of tourism on the nearby island of Providenciales has recently increased demand for reef fish and fishers may sell Nassau grouper directly to restaurants at up to US$15.00 per kg (Rudd and Tupper, 2002). Some South Caicos fishers have begun to target grouper, as the value of the catch is often worth the expense of traveling 60 km to land the catch in Providenciales when their catch exceeds about 100 kg.

The objective of this study was to investigate the role of fishing pressure on density, size and biomass of reef fishes in and adjacent to a small marine reserve (the Admiral Cockburn Land and Sea National Park) in the TCI. In addition, we monitored CPUE of reef fish from waters adjacent to the reserve and from occasional confiscated catches within the reserve, as a function of distance from the reserve center. The study addressed the following specific questions: (1) Are spatial variations in fish density, size and biomass attributable to habitat structure or to level of protection? (2) Assuming that the efficiency of spear fishing decreases with depth, do the effects of protection differ among reefs at different depths? (3) Does CPUE of fish differ among zones of different fishing pressure? (4) Does CPUE of reef fish decrease with increasing distance from a protected area? We discuss our results, which suggest that there are species-specific differences in the conservation and fisheries benefits that small marine reserves provide, with reference to the viability of marine reserves as a conservation policy option for the TCI.

Methods

Study Sites and Species

The study area was located at South Caicos, on the eastern end of the Caicos Bank, Turks and Caicos Islands (Figure 10.1). The Caicos Bank is a shallow, oolitic limestone platform that rises abruptly from depths of 2000-4000 m. The platform is bordered by extensive coral reefs that are distinguished by their steep, abrupt drop-off. These shelf edge reefs typically occur at 15-20 m depth and drop almost vertically to a depth of several hundred m. The Caicos Bank also supports extensive shallow sand flats, mangroves, seagrass beds, and shallow patch reefs.

The study area was broadly divided into three zones: (1) The Admiral Cockburn Land and Sea National Park (ACLSNP) from the southeastern tip of South Caicos (High Point) to the southwestern tip of Long Cay (SWLC). This 4 km^2 zone was closed in 1992 to all fishing except recreational hook and line fishing from shore, although some poaching occurs (M. Tupper, personal observations, 1999). An adjacent marine reserve, the East Harbor Lobster and Conch Reserve, has an area of approximately 12 km^2, consisting mainly of shallow sand and algal plains. (2) The area north of the ACLSNP from High Point to Plandon Cay. This zone is only lightly fished due to rough sea conditions and the prevalence of sharks. (3) The area south of the ACLSNP from SWLC past the Fish Cays (Figure 10-1), to the Ambergris Cays. Moderate fishing pressure is concentrated in this area, which is shallower and less turbulent than the windward zone north of the ACLSNP.

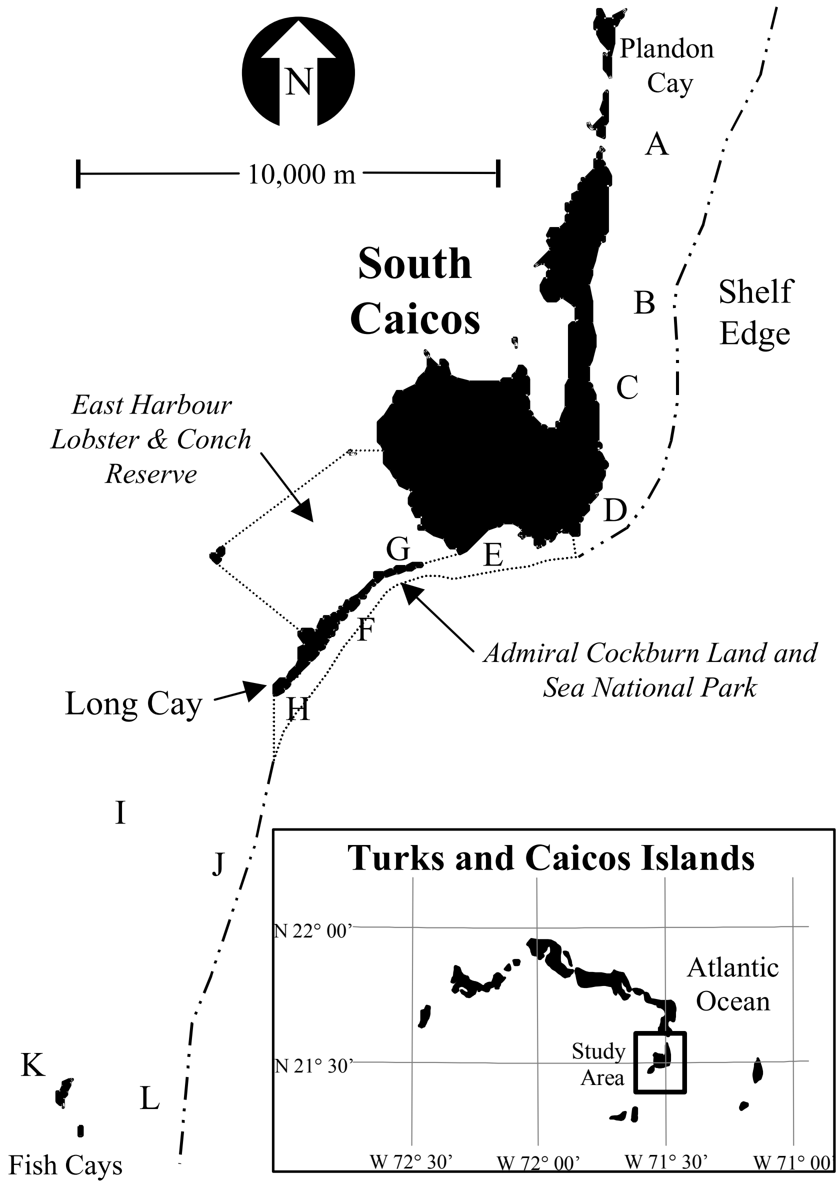

Figure 10-1 – Admiral Cockburn Land and Sea National Park, South Caicos, Turks and Caicos Islands. Study sites include: A – Coast Guard (low fishing pressure, shallow depth); B – Catacombs (low pressure, deep); C – Horseshoe Reef (low pressure, shallow); D – Highland House (low pressure, deep); E – East Bay Spur (marine reserve, deep); F – Troy's Dream (marine reserve, deep); G – Admiral's Aquarium (marine reserve, shallow); H – South End Long Cay (marine reserve, shallow); I – Trophy Hall (high fishing pressure, shallow); J – South Slope (high pressure, deep); K – Fish Cay (high pressure, shallow); L – Fish Cay Wall (high pressure, deep).

Within these three zones, 12 sites representing two depth strata were surveyed. The depth strata included shallow reefs 3-4 m in depth and deep reefs at approximately 15 m depth. Within each depth stratum, similar habitats were surveyed. Shallow reefs consisted of fringing reefs dominated by *Montastrea annularis* and *Acropora palmata*. Deep reefs were situated along the shelf edge and were dominated by high relief spur and groove formations, primarily consisting of *M. annularis*. At each site, percent cover of benthic substrata was quantified along four line-intercept transects, each 10 m in length. Substrata were characterized as macroalgae (fleshy, filamentous turf, calcareous green, branching coralline or encrusting coralline), live coral, octocoral, sponge, hard substrate (dead coral, rubble, limestone pavement), and sand. In addition, maximum vertical relief and substrate rugosity were measured on each transect. Substrate complexity was estimated by fitting a fine-link brass chain to the bottom contours along the transect line. The total distance covered by the chain was then divided by 10 m (the horizontal distance covered by the transect line), producing an index of substrate rugosity (Tupper and Boutilier, 1997).

Census Technique

In order to compare our results to those of previous studies in other regions, we replicated the survey methods used by Sluka *et al.* (1996a) and Chiappone *et al.* (2000). At each site, abundance and total length (to the nearest cm) of target fish species were visually estimated along ten haphazardly placed transects measuring 20 m x 5 m. Observers were trained in transect width estimation prior to conducting surveys; transect width estimation was conducted according to Sluka *et al.* (1996a). Observers were also trained to estimate total length of fishes by visually estimating the size of fish models to the nearest centimeter (see Sluka *et al.,* 1996a). Observers were not permitted on research dives until their size estimations of models were within 5% of the actual value. For each transect, biomass of individual species was estimated from numbers and lengths of fish using published length-weight relationships (Froese and Pauly, 2001). The biomass for each species was then summed to arrive at an estimate of total fish biomass for the transect.

Dockside Monitoring

All fishers interviewed in this study used similar gear (a Hawaiian sling). Fishers were interviewed upon their return to processing plants to unload their catch. Each fisher was asked where they had fished that day and how many hours they had spent fishing as opposed to traveling between sites. Where multiple fishers occupied one boat, hours spent fishing was recorded for each individual. Sites visited were categorized as being either north or south of the reserve as described above. Since fishers must make the decision to travel north or south early in the morning, depending on weather and sea conditions, each boat fished only one zone per day.

After interviewing all persons aboard the boat, the interviewer examined the catch. All fish were identified to species level, measured to the nearest cm fork length (FL) and weighed to the nearest g using an electronic scale. Fish larger than 10 kg were weighed to the nearest 0.5 kg using a hand-held spring scale. Since fishing pressure was expected to influence both size and abundance of exploited fishes, catch per unit effort (CPUE) was calculated as both the number of fish caught per fisher per hour and the weight of fish per fisher per hour. Both measures of CPUE were calculated for hogfish, Nassau grouper and white margate individually.

Statistical Analysis

Data were tested for normality of distribution using the Wilke-Shapiro test (Sokal and Rohlf, 1995) and homogeneity of variance using Levene's test (Zar, 1999). For each species and for combined species data, density, fork length, and biomass were non-normally distributed and variances were heterogeneous. Following log(x+1) transformation, data were retested and found to meet the assumptions of parametric analysis of variance (ANOVA). Density, fork length, and biomass were then analyzed using a two-way ANOVA. The fixed factors were level of protection (lightly fished northern zone, ACLSNP, and fished southern zone) and depth (shallow fringing reefs vs. deep shelf-edge reefs). Tukey's Honestly Significant Difference (HSD) was used as a post-hoc multiple comparison test. For each of the three study species, least squares regressions of biomass against the measured habitat variables were conducted to determine the effects of habitat characteristics on fish distribution.

For all species, CPUE was square root transformed after adding a constant of 0.5 to account for zero catch data (Johnson *et al., 1*999). Transformed CPUE data met the assumptions of parametric analysis and were compared among zones using one-way analysis of variance (ANOVA). Data on the distance from reserve center was non-normal despite monotonic transformations. Spearman rank correlations were therefore used to determine the relationship between CPUE and distance from the reserve center. Correlations were conducted for pooled data and separately for distance north and south of the reserve center.

Results

Effects of habitat, depth and fishing pressure

The two-way ANOVAs comparing habitat characteristics between different depths and zones of fishing pressure indicate differences among zones of fishing pressure and/or depths for all substrates except sponge, crustose coralline algae and sand (Table 10-1). Post-hoc comparisons revealed that fleshy algal cover (primarily brown algae of the genera *Sargassum*, *Dictyota*, *Turbinaria* and *Lobophora*) was significantly higher in the fished southern zone than elsewhere (Tukey's HSD, $p < 0.05$) and was also higher on shallow reefs than on deep reefs (Tukey's HSD, $p < 0.05$). Cover of filamentous algal turf was higher in the ACLSNP than outside (Tukey's HSD, $p < 0.05$). Cover of calcareous green algae was lower in the fished zone than in the ACLSNP or the lightly fished zone (Tukey's HSD, $p < 0.05$). Branching coralline algae was more common in the lightly fished zone than elsewhere (Tukey's HSD, $p < 0.05$). Both stony coral and octocoral cover were higher within the ACLSNP than outside (Tukey's HSD, $p < 0.05$). The coverage of hard substrate was lower within the ACLSNP than in either fished area (Tukey's HSD, $p < 0.05$). Topographic complexity and vertical relief were both higher on the northern windward reefs than in the ACLSNP or the southern zone (Tukey's HSD, $p < 0.05$), and vertical relief of deep reefs was higher than that of shallow reefs in all zones.

Comparisons of fish length, density and biomass among levels of fishing pressure and depths indicate that both parameters had a marked influence on some species, while Nassau grouper were unaffected by fishing pressure (Table 10-2). Length, density, and biomass of hogfish were all significantly higher on deeper reefs. Density of Nassau grouper was higher on deeper reefs, although their length did not differ between depths (Table 10-2). Density of white margate was higher on shallow reefs, but their mean length was greater on deeper reefs. Length, density and biomass of hogfish were all significantly lower in the fished zone, but did not differ between the reserve and the lightly fished zone (Tukey's HSD, $p > 0.05$ for all comparisons). The significant interaction effects shown in Table 10-2 result from lower mean length and biomass on deep reefs in the fished zone than on shallow reefs in the lightly fished and protected zones (Tukey's HSD, $p < 0.05$). Length of white margate did not vary with fishing pressure. Density of white margate was greater in the reserve than in the fished zone (Tukey's HSD, $p < 0.05$). Biomass of white margate was significantly lower in the fished zone than the lightly fished or protected zones (Tukey's HSD, $p < 0.05$) but did not differ with depth (Table 10-2).

	Mean Coverage (% ± 1 standard deviation)						Two-way ANOVA					
	ACLSNP		Lightly Fished		Fished		Protection		Depth		Interaction	
Benthic Substratum	Deep	Shallow	Deep	Shallow	Deep	Shallow	F	*p*	F	*p*	F	*p*
Fleshy algae	10.9 ± 6.2	12.2 ± 5.9	12.4 ± 7.3	14.1 ± 9.2	27.4 ± 16.3	39.6 ± 12.9	33.1	<0.001	18.2	<0.01	2.4	NS
Filamentous (turf) algae	7.2 ± 2.2	7.4 ± 4.4	6.3 ± 4.3	6.5 ± 5.4	6.7 ± 4.6	6.0 ± 2.1	12.4	<0.01	3.3	NS	5.1	NS
Calcareous green algae	3.6 ± 2.1	2.8 ± 1.6	2.4 ± 1.8	3.2 ± 1.9	1.6 ± 1.2	1.4 ± 1.1	9.1	<0.01	2.1	NS	2.0	NS
Branching coralline algae	1.3 ± 0.9	1.1 ± 1.7	1.1 ± 1.3	1.5 ± 1.6	0.4 ± 0.3	0.6 ± 0.5	12.7	<0.01	3.1	NS	2.3	NS
Crustose coralline algae	5.2 ± 3.3	5.1 ± 3.7	4.6 ± 2.8	4.3 ± 4.9	4.8 ± 3.6	4.1 ± 2.5	3.4	NS	2.8	NS	1.8	NS
Stony coral (live)	35.1 ± 12.2	38.7 ± 12.0	24.3 ± 10.0	25.4 ± 9.8	22.4 ± 8.3	24.4 ± 12.9	39.3	<0.001	2.7	NS	1.3	NS
Octocoral	5.8 ± 1.4	6.4 ± 2.1	6.7± 2.7	2.3 ± 1.3	3.7 ± 1.8	2.0 ± 1.7	16.0	<0.001	15.3	<0.001	9.5	<0.01
Sponge	2.4 ± 1.1	1.2 ± 0.8	2.0 ± 1.5	0.9 ± 1.0	1.5 ± 0.7	0.7 ± 0.7	2.5	NS	16.2	<0.001	1.2	NS
Hard bottom	58.7 ± 8.6	51.7 ± 7.7	68.9 ± 9.0	64.0 ± 8.9	56.8 ± 9.9	63.8 ± 8.7	8.8	<0.01	1.3	NS	1.1	NS
Sand	8.2 ± 5.7	7.6 ± 7.7	8.7 ± 7.6	6.4 ± 5.5	7.2 ± 6.6	7.7 ± 6.2	2.0	NS	1.8	NS	1.9	NS
Topographic complexity	2.5 ± 0.9	2.6 ± 0.8	3.2 ± 0.7	3.1 ± 0.8	3.1 ± 1.0	2.6 ± 0.8	7.1	<0.01	2.6	NS	2.7	NS
Vertical relief (cm)	106.4 ± 23.3	66.8 ± 6.7	154.8 ± 47.9	124.9 ± 36.7	114.5 ± 33.6	81.9 ± 31.0	14.3	<0.001	15.9	<0.001	1.1	NS

Table 10-1 – Two-way ANOVAs comparing benthic habitat variables between deep and shallow reefs at varying levels of protection from fishing. NS: not significant at Bonferroni adjusted α (0.01).

	Mean Density (± 1 standard deviation)						Two-way ANOVA					
	Reserve		Lightly Fished		Fished		Protection		Depth		Interaction	
	Deep	Shallow	Deep	Shallow	Deep	Shallow	F	*p*	F	*p*	F	*p*
Nassau grouper												
Length	62 ± 9.5	61 ± 12.8	57 ± 13.8	59 ± 5.7	61 ± 6.5	58 ± 14.5	1.2	NS	1.4	NS	1.4	NS
Density	0.61 ± 0.3	0.55 ± 0.5	0.90 ± 0.4	0.65 ± 0.5	0.65 ± 0.4	0.45 ± 0.4	1.9	NS	9.8	<0.01	1.5	NS
Biomass	3660 ± 1619	3300 ± 1217	4895 ± 764	3535 ± 570	3900 ± 1261	2448 ± 1243	1.3	NS	11.7	<0.01	1.0	NS
Hogfish												
Length	31 ± 7.2	24 ± 8.3	32 ± 7.8	25 ± 8.8	24 ± 8.6	21 ± 7.6	148.1	<0.001	35.2	<0.001	5.7	<0.01
Density	1.9 ± 1.2	1.1 ± 0.2	1.1 ± 0.9	0.7 ± 0.4	0.3 ± 0.2	0.2 ± 0.1	44.0	<0.001	31.5	<0.001	1. 5	NS
Biomass	728 ± 478	177 ± 61	421 ± 279	121 ± 79	48.4 ± 19	32 ± 15	30.0	<0.001	74.9	<0.001	8.3	<0.01
White margate												
Length	28 ± 5.5	26 ± 6.2	30 ± 5.2	24 ± 4.4	30 ± 6.1	26 ± 6.2	0.6	NS	12.8	<0.001	1.2	NS
Density	9.3 ± 2.0	20.3 ± 4.7	10.0 ± 1.4	10.7 ± 2.5	2.6 ± 0.6	4.6 ± 2.2	10.3	<0.01	5.8	<0.01	1.7	NS
Biomass	2922 ± 1369	6378 ± 1992	3142 ± 1676	1726 ± 450	817 ± 337	742 ± 189	22.2	<0.001	1.2	NS	1.5	NS

Table 10-2 – Two-way ANOVA comparing the influence of depth and level of protection on total length (cm), density (individuals * 100 m^{-2}), and biomass (g wet wt. * 100 m^{-2}) of commonly exploited reef fishes on deep and shallow coral reefs in the Turks and Caicos Islands. NS: not significant at Bonferroni adjusted α (0.01).

Least squares regressions of fish biomass on habitat variables indicated that fleshy algal cover had a weak but significant negative effect on the biomass of hogfish on both shallow (r^2 = 0.24, p <0.05) and deep (r^2 = 0.22, p <0.05) reefs. Similarly, biomass of white margate was inversely related to the cover of fleshy algae on both shallow (r^2 = 0.29, p <0.01) and deep (r^2 = 0.32, p <0.001) reefs. Nassau grouper was unaffected by fleshy algal cover, but showed a weak positive association with vertical relief on both shallow (r^2 = 0.30, p <0.001) and deep (r^2 = 0.34, p < 0.001) reefs.

Dockside monitoring

Over the course of this study, local government fishery officers caught three poachers and confiscated their catch. This represented approximately five hours of effort (Table 10-3). Although the sample size of catches from the reserve is low, the opportunity to sample illegal catches from protected areas is rare and so the data have been included in this analysis. For hogfish, Nassau grouper and the total catch, CPUE was much lower (half or less) in the fished zone than in the lightly fished or protected zones but did not differ between the latter two zones (Tukey's HSD, p < 0.05 for all comparisons). Only white margate had a lower CPUE within the ACLSNP than outside, but it also supported a higher CPUE in the lightly fished than the fished zone. It should be noted that, when questioned, two of three poachers indicated that they were specifically targeting hogfish and Nassau grouper. Thus the CPUE of margate within the ACLSNP may not reflect its actual distribution.

	Level of Protection			**One-way ANOVA**	
	Marine Reserve	**Lightly Fished**	**Heavily Fished**	**F**	***p*-value**
Fishers Interviewed	3	28	113		
Hours Fished	5	98	456		
CPUE (kg/hr/person)					
Nassau grouper	0.5 ± 0.6	0.4 ± 0.6	0.7 ± 0.9	1.3	NS
Hogfish	9.1 ± 2.4	8.2 ± 3.1	0.8 ± 0.8	50.0	< 0.001
White margate	0.2 ± 0.4	3.2 ± 2.0	0.7 ± 0.4	16.3	< 0.001
Total catch	17.9 ± 6.2	17.8 ± 6.8	3.2 ± 2.1	29.5	< 0.001

Table 10-3 – One-way ANOVAs of catch per unit effort (CPUE) for commonly exploited reef fishes of the Turks and Caicos Islands. NS: not significant

The effects of distance from the ACLSNP center on CPUE of exploited reef fish are shown in Table 10-4. CPUE of hogfish decreased with increasing distance from the reserve and with increasing distance across both the southern and northern boundaries. No relationships were found between CPUE of Nassau grouper or CPUE of the total catch and distance from the reserve center. CPUE of white margate was not related to distance in general or distance across the southern boundary, but increased with distance across the northern boundary.

Direction	Species	Spearman R	*p*-value
North and South	Nassau grouper	0.13	NS
	Hogfish	-0.41	<0.001
	White margate	0.07	NS
	Total catch	0.06	NS
North only	Nassau grouper	0.09	NS
	Hogfish	-0.39	<0.001
	White margate	0.62	<0.001
	Total catch	0.06	NS
South only	Nassau grouper	0.19	NS
	Hogfish	-0.47	<0.001
	White margate	-0.20	NS
	Total catch	-0.22	NS

Table 10-4 – Spearman rank correlations of the catch per unit effort (CPUE, $kg*fisher^{-1}*hr^{-1}$) vs. distance from the center of the Admiral Cockburn Land and Sea National Park (ACLSNP). Data are presented for overall distance, distance north from the ACLSNP center (towards the lightly fished zone) and distance south from the ACLSNP center (towards the fished zone). NS: not significant at Bonferroni adjusted α (0.005).

Discussion

Influence of habitat

The most noticeable difference in habitat among the three zones of fishing intensity was the much greater coverage of fleshy macroalgae in the fished zone. The higher coverage of fleshy macroalgae on the reefs in the southern zone was partially a result of past storm damage (Tupper, unpublished data), but to some extent it may also have stemmed from coral death due to destructive fishing methods, particularly the widespread, intensive use of chlorine and detergents to drive spiny lobster from their shelter sites (W Clerveaux, Turks and Caicos Department of Environment and Coastal Resources, personal communication). In addition to having the highest coverage of algae, the shallow southern reefs suffered the highest fishing pressure and supported the lowest density and biomass of hogfish and white margate.

It is difficult to separate the effects of fleshy algal cover and fishing pressure on the biomass of hogfish and white margate. McClanahan *et al.* (2000, 2001) determined that benthic habitat structure had a greater effect than management (*i.e* control of fishing pressure) on abundance of herbivorous reef fish in Belize. They reported increases in abundance of six fish species, including four herbivorous species and and two species that fed primarily on invertebrates, following experimental reductions of fleshy macroalgal cover on patch reefs. In our study, however, the large differences in density and biomass of hogfish and white margate between the fished zone and reserve, coupled with strong, bi-directional spillover of hogfish from the marine reserve, suggest that fishing pressure was an important factor in determining the distribution and abundance of these two species. Furthermore, neither species exhibited an affiliation for substrates such as live coral or turf algae, the coverage of which might be reduced by fleshy algal growth (Hughes, 1994). Since both species feed primarily on infaunal invertebrates in sandy bottoms, rather than foraging on the reef (Humann, 1994), it seems likely that fleshy algal cover would have a lesser effect on their distribution in comparison to herbivorous species or species that feed on reef-associated invertebrates.

The highest percent cover of live stony coral and octocoral occurred within the ACLSNP, but this is probably not a function of protective management. Rather, the site of the ACLSNP was chosen based on the health of its reefs (which are visited regularly by live-aboard dive boat operations), queen conch spawning habitat and adjacent inshore conch and lobster nursery grounds. Regardless, the higher coral cover within the reserve had no effect on the density or biomass of the three study species, none of which were associated with live coral cover.

A previous study at South Caicos (Tupper, 2002) found no differences in Nassau grouper abundance in a variety of habitat types (channel reefs, fringing reefs, patch reefs and shelf edge reefs). Similarly, Sluka *et al.* (1996b, 1997) found no habitat associations for Nassau grouper in the Exuma Cays. The results of this study support previous studies on Nassau grouper that suggest it may be more important to protect reefs in general than to attempt to protect "optimal" grouper habitat from fishing (Sluka *et al.,* 1996a, 1996b; Tupper, 2002).

Influence of depth and fishing pressure

In general, fish were larger and more abundant on deeper (15-20 m) reefs than shallow (3-4 m) reefs. However, the lack of significant interactions between depth and fishing pressure suggest that response to fishing pressure does not vary with depth over the range of depths. While depth may afford some protection from spear fishing in the form of reduced harvesting efficiency, it is perhaps more likely that these species move to deeper water as they grow (Appeldoorn *et al.,* 1997). In the case of Nassau grouper, no differences in size, abundance or biomass were found between zones of different fishing intensity (see below), so fishing intensity would not explain the greater abundance and biomass on deeper reefs.

For two of the three species in this study, fishing pressure appeared to have a significant influence on fish populations. Hogfish were smaller and less abundant where fishing was most intense, and biomass of white margate was higher inside the ACLSNP than outside. Moreover, CPUE of hogfish declined with increasing distance from the MPA center. This suggests that spillover from the ACLSNP can enhance local fishery yields outside its boundaries.

The presence of the South Caicos marine reserve, however, appeared to have no impact on the distribution and abundance of Nassau grouper in this study. In contrast, Sluka *et al.* (1996a) found higher density of Nassau grouper in the protected Exuma Cays Land and Sea National Park than on fished reefs outside the park. Why would the MPA have an impact on white margate and hogfish, but not Nassau grouper? It is possible that the size of the reserve relative to home range of the fishes plays an important role in the results.

Despite a recent increase in research effort (Holland *et al.,* 1993, 1996; Samoilys, 1997; Zeller, 1997; Chapman and Kramer, 1999, 2000; Meyer *et al.,* 2000), little is known of the specific movements or home range size of exploited coral reef fishes (Kramer and Chapman, 1999). In general, it is understood that the longer the time spent outside the reserve, the more vulnerable fish become to fishing mortality (Kramer and Chapman, 1999) and that the extent of home range is most strongly influenced by body size. Large and schooling species have larger home range sizes (Samoilys, 1997; Zeller, 1997) and tend to move farther than small or solitary species. Larger fishes such as grouper are therefore more likely to cross reserve boundaries, while smaller species may spend all their time within MPA boundaries (Holland *et al.,* 1993, 1996; Meyer *et al.,* 2000).

The area covered by the ACLSNP is only 4 km^2 (slightly larger when considering suitable habitat within the adjoining East Harbour Lobster and Conch Reserve). Home range sizes of Nassau grouper, hogfish and white margate are currently unknown, but Kramer and Chapman (1999) analyzed the relationship between body size and home range size for 29 species of reef fish, including members of the families Labridae and Haemulidae. By pooling data for these species they determined that home range area in m increased with the 3.53 power of body length in cm. Assuming an average fork length of about 25 cm for both hogfish and white margate, these regressions result in home range areas of 0.9 km^2 for the two smaller species. Assuming an average fork length of about 60 cm for Nassau grouper, home range would be in the 19 km^2 range. These are obviously rough estimates, taken from a conglomerate picture of other species. However, it is apparent that the home range of adult Nassau grouper is markedly larger than the ACLSNP, while hogfish and white margate probably have home ranges smaller than the protected area. Nassau grouper density was higher in the Exuma Cays Land and Sea Park (ECLSP) than in surrounding fished areas (Sluka *et al., 1*996a), but the area covered by the ECLSP is 442 km^2. Thus, the differences in response to protection may relate to the home range size of reef fish relative to the size of the marine reserve.

The lack of a measurable protective effect on Nassau grouper may also stem from its long-distance spawning migration. Nassau grouper may travel tens to hundreds of km to participate in spawning aggregations (Bolden, 2000). Nassau grouper from South Caicos travel approximately 40 km to a large spawning aggregation at Phillips Reef, off the island of East Caicos around the full moon in January (T. Morris, personal communication; author's personal observations). However, the aggregation is rarely fished due its remote location and rough seas, making it unlikely that fishing mortality outside the reserve during the spawning migration is a factor.

Finally, the failure of the ACLSNP to enhance grouper biomass within its boundaries may have been exacerbated by poaching. Poaching appeared to be a relatively rare occurrence over the course of this study (1999-2000). However, poachers may have operated at night and on Sundays and holidays, when enforcement officers were not active.

Management Implications

The ACLSNP may not be large enough to effectively protect large reef fishes such as the Nassau grouper, which have large home ranges and/or undergo seasonal spawning migrations. However, even at fairly low levels of fishing pressure, smaller, more sedentary species, such as hogfish and white margate can apparently benefit from small marine reserves (see Kramer and Chapman, 1999; Meyer *et al., 2*000). The effectiveness of the small reserve in increasing fish size, biomass and emigration suggests that local fishers derive some economic benefits from ACLSNP. It is not clear, however, if the economic benefits from spillover exceed the opportunity cost of closing the area.

The lack of protection provided to Nassau grouper by small marine reserves is disturbing. Small marine reserves are functionally the only control on fishing pressure for inherently vulnerable Nassau groupers throughout much of their range (*e.g.,* Chiappone *et al., 2*000; Tupper, 2002). If the reserves are ineffective, the fishery operates under *de facto* open access: there are no annual fisheries landing limits, individual trip limits, size regulations, or gear limitations outside of these reserves. In addition, Nassau grouper has non-extractive economic value for the dive tourism industry (Rudd and Tupper, 2002). Depletion of stocks could impose economic costs on the dive industry because divers' willingness to pay for dive charters (or MPA entry fees) decreases as grouper size and abundance decrease.

If ACLSNP is ineffective for Nassau grouper conservation purposes because of its small size, what policy options exist that might protect Nassau grouper stocks around South Caicos? An obvious option would be to increase the size of ACLSNP, to encompass an area that provides adequate protection for small and large reef fish. The likelihood of fishers and government adopting this option is low, however, because a larger no-take reserve close to South Caicos would impose high opportunity costs on lobster fishers. A larger reserve would close important lobster fishing grounds adjacent to South Caicos and south to Fish Cays, and could become very difficult to monitor and enforce.

Another option would be implementation of a seasonal closure on spawning grounds where Nassau grouper are known to aggregate. Closures of grouper (predominantly red hind, *Epinephelus guttatus*) spawning aggregation sites have been successfully implemented in the U.S. Virgin Islands (Beets and Friedlander, 1999). However, recent research indicates that Nassau grouper spawning aggregations may exhibit plasticity in timing and location, such that an aggregation which occurs at a certain time or place in one year may be shifted by several hundred meters to a few kilometers another year, or may occur one or two months earlier or later (M. Tupper, unpublished data). If a closure is implemented at a given spawning site or for a particular period, there is a chance that in subsequent years, the spawning aggregation may occur outside the spatial or temporal boundaries of the closure. Thus, seasonal spawning closures might have to be several months in length (*e.g.* November through March) in order to be effective.

The use of catch controls, especially setting an appropriate total allowable catch (TAC), possibly in conjunction with seasonal closures during the spawning season, seems to hold more promise. A conservative TAC could have pragmatic advantages over other options: local fishers would maintain access to fishing grounds for lobster and reef fishes other than Nassau grouper and enforcement efforts might focus on shore-based restaurant buyers, reducing more expensive field enforcement costs. A further analysis would be required to fully understand the incentives of various stakeholders and the likelihood of various policy measures successfully protecting Nassau grouper stocks in the TCI.

In conclusion, the small marine reserve near South Caicos appears to provide effective protection for the small reef fishes, white margate and hogfish. Our results are confounded somewhat by the overgrowth of fleshy macroalgae in the fished zone, which negatively affected the abundance of these species. An inverse relationship between distance from the center of the reserve and CPUE for hogfish also strongly suggests spillover of commercial fish from the reserve. The lack of difference in size and abundance of Nassau grouper inside and outside the reserve, however, illustrates the need to consider site- and species-specific factors in policy design. Given the ecological, cultural and institutional context in which South Caicos fishers operate, marine reserves are unlikely to provide sufficient protection for Nassau grouper as fishing pressure continues to increase. Conservation of this species may require stronger regulation and alternative fisheries management tools such as seasonal spawning closures, conservative TACs and/or commercial trade bans.

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CHAPTER 11

AN INSTITUTIONAL ANALYSIS OF POLICY OPTIONS FOR NASSAU GROUPER CONSERVATION IN THE TURKS AND CAICOS ISLANDS[1]

In the tropical North Atlantic, Nassau grouper (*Ephinephelus striatus*) have been subject to heavy fishing pressure, to the point of commercial extinction in many areas (Huntsman *et al*., 1999; Sadovy and Eklund, 1999). They have traditionally been consumed for subsistence purposes and provided income to artisanal fishers selling fish in local markets (*e.g*., Olsen, 1986). Nassau grouper have become an icon species for the dive tourism industry, providing non-consumptive economic value to scuba divers who derive increased utility from viewing more abundant and/or larger Nassau grouper during dives (Rudd and Tupper, 2002). Top piscivores such as groupers may also play a key role in maintaining ecosystem balance by controlling numbers of herbivore fishes (*e.g*., Watson and Ormond, 1994). In the absence of controls, excessive grazing can lead to reef damage, degradation of overall reef productivity and a loss in ecosystem resilience (McManus *et al*., 2000). Failure to account for the full range of market and nonmarket services that Nassau grouper provide humans could, as a result, lead to an under-emphasis on policies that protect remaining stocks and inadequate investment of societal resources in conservation (Rudd *et al*., 2003). Further, failure to account for the incentives of influential actors in renewable resource systems may also severely compromise the likelihood of successfully implementing effective conservation policies (Ostrom, 1990; Mascia, 2000).

The Turks and Caicos Islands (TCI) is one of the few areas in the tropical North Atlantic that still has relatively healthy Nassau grouper stocks (Tupper, 2002; Tupper and Rudd, 2002). However, given the inherent vulnerability of Nassau grouper to fishing (Sadovy and Eklund, 1999; Coleman *et al.*, 2000), the history of stock depletion in other areas (Sadovy and Eklund, 1999; Chiappone *et al*., 2000), and evidence of rapidly increasing local market demand for reef fishes in the TCI (Rudd, in press a), strong measures will be needed to ensure the conservation of Nassau grouper stocks in the TCI.

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[1] This chapter is currently in review as: Rudd, M.A. Policy options for Nassau grouper conservation in the Turks and Caicos Islands. *Biological Conservation.*

The TCI Department of Environment and Coastal Resources (DECR) is responsible for managing reef fisheries but has a very limited program for reef fishes due to historically low export demand for finfish. There is a prohibition on the use of scuba gear and spear fishing for all commercial fisheries (over 90% of commercial fishing for spiny lobster (*Panulirus argus*), queen conch (*Strombus gigas*) and reef fish is conducted by free divers) and this affords some protection for large, deep-dwelling Nassau grouper for most of the year. There is also a series of small marine reserves in the country, although Tupper and Rudd (2002) found no difference in abundance of Nassau grouper inside and outside one important reserve, likely due to the small size of the reserve relative to home range of Nassau grouper. Additional fisheries management tools that might help control Nassau grouper exploitation have not been implemented. Thus, the reef fish fishery essentially operates under *de facto* open access conditions.

In this paper, I use the Institutional Analysis and Development (IAD) framework (Ostrom, 1990, 1999) to organize an assessment of the TCI Nassau grouper fishery and identify possible policy packages that could help conserve local Nassau grouper stocks. Two fisheries management tools (conservative landing limits and larger, more effective marine reserves) and one policy directed at the domestic tourist-oriented restaurant sector (a Nassau grouper trade ban) are considered relative the status quo situation of de facto open access and import tariff protection. The goal of the analysis is to identify policies that are most likely to protect Nassau grouper stocks given ecological, economic, social and political realities within the TCI.

While the focus of this case study is a single, small island nation, one finding – that marine reserves are unlikely to receive the necessary support needed for successful implementation due to their relatively high transaction costs – may have broader relevance in the debate over the use of marine reserves for tropical fisheries management and conservation. Marine reserves are widely viewed as an all-purpose, low-cost tool for fisheries enhancement and conservation (Roberts, 1997; Murray *et al.*, 1999), yet the TCI case suggests that there are market-oriented policy alternatives – in this case, a commercial trade ban on Nassau grouper in tourist-oriented restaurants – that are (1) more likely to be effectively implemented compared to marine reserves because the policies are better aligned with the interests of resource users and managers and (2) more likely to endure because they have lower transaction costs of management relative to marine reserves. It would be prudent for the TCI government to initiate a policy experiment in which a Nassau grouper commercial trade ban for tourist-oriented restaurants was implemented before lasting damage is inflicted on Nassau grouper stocks in the TCI.

Methodology

Study Site

The TCI is located at the southern end of the Bahamian archipelago, about 160 km north of the Dominican Republic (see Figure 6-1) and has a population of under 25,000 residents. Virtually all commercial artisanal fishing takes place on the Caicos Bank, a shallow, oolitic limestone platform covering an area of about 6,140 km^2 and comprised of sand (64%), mixed coral and algae (18%), coral reefs (7%), and other habitats (11%) at depths typically 1-5 m (Olsen, 1986). Extensive coral reefs fringe the shelf edge and are characterized by steep drop-offs with high-relief coral formations favored by adult Nassau grouper (Tupper, 2002).

The Caicos Bank supports export-oriented fisheries for queen conch and spiny lobster, and a domestic fishery for 'scale-fish' (primarily reef fish, including Nassau grouper). Reef fishes are most often landed opportunistically by lobster fishers. South Caicos is the traditional home of the artisanal fleet, but landings of conch and reef fishes near the island of Providenciales ('Provo') have increased over the last two decades as the island has been developed for tourism. Nassau grouper are a historically favored species for local consumption (Olsen, 1986).

Institutional Analysis

This research uses the Institutional Analysis and Development (IAD) framework, a multi-tiered conceptual map that can be used to organize the universal elements that should be considered in policy analyses (Ostrom, 1990, 1999). It is a robust framework that has been used extensively to test theories and models linking institutions and the sustainability of ecological-economic systems. Rudd (in press, b) incorporates the concept of capital assets (natural, manufactured, human, social and economic capital) within the basic IAD framework to provide a platform specifically for ecosystem-based fisheries management policy design, monitoring and communication.

Institutions – the 'rules-in-use' regarding resource access and utilization – are crafted by people to increase predictability and provide order in uncertain environments (Ostrom, 1999). Institutions are comprised of formal rules (*e.g.*, regulations, laws) and/or informal prescriptions (*e.g.*, social norms) that permit, prohibit or require certain actions or outcomes (Crawford and Ostrom, 1995). It is possible that a variety of institutions (means) might be crafted to achieve any particular societal objectives (ends) and that the various institutional arrangements may vary substantially in cost-effectiveness (Rudd *et al.*, 2003).

Ostrom (1990, 1999) identifies seven generic types of rules that are commonly encountered when studying common pool resource systems: boundary (entry and exit); position; authority; information; aggregation; scope; and payoff rules. The configuration of these rules, in combination with resource availability and the exogenous socio-institutional environment help shape the incentives of people within the resource system or 'action arena'. Institutions influence individual and collective behavior that, in turn, has impacts that may threaten the sustainability of the system itself (Figure 11-1).

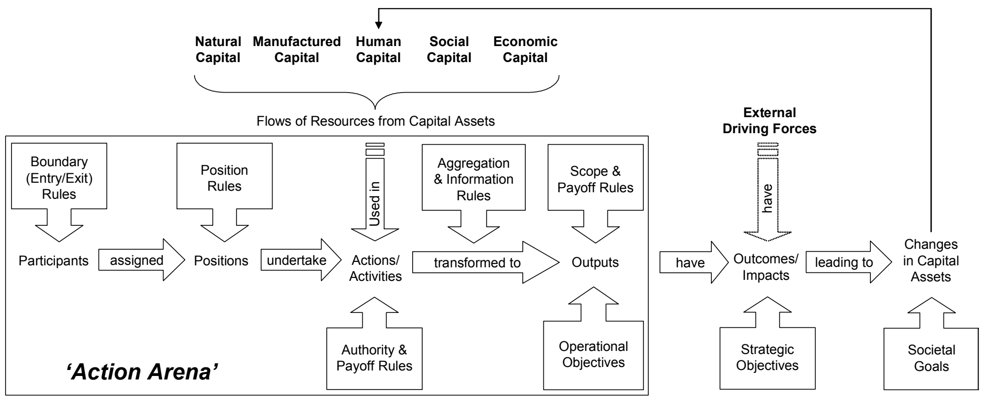

Figure 11-1 – Categories of rules that influence the incentives, behavior and outputs of 'action arena' participants

Potential participants may face boundary rules that define their eligibility to participate in resource extraction (*e.g.*, limited entry licensing based on geographic residence). Participants can be assigned to different positions that define their role and their responsibilities. Authority rules define actions or activities that are required, permitted or prohibited by participants (*e.g.*, rules about where, when and how fishers may fish). Participants consider the costs of various actions, the expected benefits of outputs and their impacts, and then engage in activities, that may or may not comply with the formal rules, to transform resources into outputs that help them improve their well being. Information rules (*e.g.*, mandatory landing slips) or aggregation rules (*i.e*, relating to how site access decisions are made) may come into play. Scope rules define what outputs or states-of-the-world are permissible or not (*e.g.*, size, bag or landing limits). Payoff rules specify sanctions or rewards associated with certain actions or outputs.

The aggregate effects of participants going about their daily activities as well as exogenous driving forces have impacts on the state-of-the-world, contributing to the depletion or accumulation of different types of capital. Whether or not changes in capital assets pose a "threat" depends on individual goals (*e.g.*, profit maximization) and broader societal goals (*e.g.*, conservation of public goods). When a threat exists, different sectors of society can respond by investing scarce resources directly in the capital assets themselves (*e.g.*, stock enhancement or habitat rehabilitation) or in institutions that might mitigate the threat (*e.g.*, increased monitoring or a change in the management plan) (Rudd, in press b).

Results

Driving Forces

The TCI is developing into an upscale tourist destination; tourism arrivals have increased from 40,000 in 1990 to 165,000 by 2001 (CDB, 2002). Consumption of local grouper (almost exclusively Nassau grouper) in tourist-oriented restaurants has risen to approximately 25,000 kg annually and another 25,000 kg are estimated to be used for subsistence purposes and in 'native' restaurants (Rudd, in press a). By 2000, tourist-oriented restaurants in Provo were willing to pay up to US $15 per kg directly to fishers for fresh local finfish, compared to the US $2.20 to $3.00 that fishers would typically receive in local markets. The growing tourist sector is the primary driving force that threatens Nassau grouper stock status and the policy challenge in the TCI is to insulate Nassau grouper stocks from further increases in fishing pressure due to the growing market demand from tourist-oriented restaurants.

Management Context: Capital Assets and Resource Flows

Nassau grouper densities, an indicator of natural capital, in the South Caicos area are in the range of 0.45 to 0.90 individuals per 100 m^2 (Tupper, 2002; Tupper and Rudd, 2002). In dockside samples, Tupper and Rudd (2002) found CPUE for all reef fish was 3.2-kg per hour over 456 hours fishing effort (when lobster was the primary fishing target) in regularly fished grounds; Nassau grouper CPUE comprised 0.7-kg per hour of the total. TCI densities are higher than those reported in the Exuma Cays Land and Sea Park (0.35 individuals per 100 m^2) and non-protected areas in the Bahamas (0.16 to 0.20 individuals per 100 m^2) (Chiappone et al., 2000). Thus, Nassau grouper stocks in the TCI are still in relatively healthy condition although the lack of baseline data on Nassau grouper abundance from, say, three or four decades past make it difficult to interpret what a truly "healthy" stock actually is (Pauly, 1995).

Manufactured capital – fishing technology – in the TCI is simple. Small 14-ft fiberglass skiffs equipped with 70- to 110-hp outboards are popular for fishing as they handle waves well, are maneuverable in patch reefs and can be used for fishing either conch or lobster. Crews of one driver and two divers work together, with skilled free divers descending to as deep as 25 m searching for spiny lobster. Nassau grouper are most commonly taken opportunistically by lobster fishers (Tupper and Rudd, 2002). Spear guns and scuba gear are banned in the TCI, although Nassau grouper can be easily speared using Hawaiian slings. Trap boats play a minor role in the TCI, accounting for only 5-10% of lobster landings (Medley and Ninnes, 1997) and minor reef fish landings (author's personal observation). There have been no formal studies of the total capacity or capacity utilization of the artisanal fleet in the TCI but it is likely that substantial latent capacity exists and that it could be used to target reef fishes should market conditions dictate.

Like lobster, the mean size of several of the most important reef fishes tends to increase with depth (Tupper, 2002). Thus, skilled lobster divers, those who possess high levels of human capital and can free dive to 15 m or more, are most likely to encounter large Nassau grouper. The more skilled divers tend to be willing to take risks, especially with regards to frequent shark encounters along the fringing reefs where Nassau grouper and large spiny lobster are most abundant.

From a social capital perspective, Bennett *et al.* (2001) observed dense social networks that facilitate information flow in the TCI and give people easy access to decision-makers in government and business. Despite the dense networks, however, there is a high degree of mistrust amongst many fishers, between fishers and fisheries managers, and between Belongers and non-Belongers (expatriate North Americans and Europeans, and immigrants from Haiti and the Dominican Republic). Rampant drug smuggling in the 1980s may have encouraged a culture of distrust and disregard for authority in the TCI. The problem was so serious that the TCI constitution was suspended for two years in 1986 after then Chief Minister, Norman Saunders, was convicted in Miami in 1985 on conspiracy charges. Saunders, since re-elected to the Legislative Council by South Caicos voters after serving prison time in the United States, accepted cash payments in return for providing drug smugglers 'safe haven' en route from Columbia to the United States

(President's Commission on Organized Crime, 1986). Community capacity to successfully participate in fisheries governance is thus limited (Rudd *et al*., 2003).

The TCI government is now quite capable by Caribbean standards even if dependent on outside help for key bureaucratic functions. As an Overseas Dependency of the United Kingdom, an appointed Governor wields real power, including the authority to appoint the majority of cabinet members from the elected Legislative Council. The TCI receives some external aid and technical support for fisheries management (*e.g.*, Ninnes and Medley, 1995; Medley and Ninnes, 1997, 1999; Halls *et al.*, 1999). Legal infrastructure in the TCI is quite strong and inter-departmental policies quite consistent, a reflection of its British colonial heritage, the political strength of the governor, and the ease with which government officials can communicate in a small country.

Fishing is often viewed as the employer of last resort in the TCI, the de facto social safety net. Economically, fishers usually have limited financial capital to re-invest in improvements in fishing technology or other endeavors. Dive operators and tourist restaurant operators, on the other hand, need to be extremely well financed to operate in the TCI due to the high cost of doing business. In the public sector, the TCI government faces severe financial constraints and was facing a deteriorating budget situation as early as the 2000-01 fiscal year (CDB, 2002), before a sharp drop in tourism resulting from the terrorist attacks of September, 2001. There are no income, business or property transfer taxes in the country, so the government has limited revenue generation capacity, relying on licenses, fees, and import tariffs. Normally, one would consider the financial capital that civil society organizations bring to bear on marine conservation and fisheries issues in an institutional analysis, but foreign and domestic non-governmental organization activity in the TCI is negligible.

Relevant Actors

It is important to identify individuals or organizations that have impacts on the day-to-day patterns of resource use and/or have influence over the choice of which rules govern a fishery. Fortunately, the ecological, socio-economic and governance aspects of TCI fisheries are relatively self-contained, so identification of key participants is straightforward.

In the public sector, there are three main participants: the Ministry of Finance ('Finance'); the fisheries division ('Fisheries') of the Department of Environment and

Coastal Resources (DECR); and the parks division ('Parks') of DECR. From a whole-of-government perspective, all departments are committed to enhancing economic benefits for TCI 'Belongers' (a local term referring to native islanders).

Finance is the 'senior' government department and is headed by the Chief Minister, the top elected official in the TCI. It has responsibility for government fiscal policy and budget administration. The situation with Parks and Fisheries is transitional due to the ongoing evolution of DECR (Campbell, 2003). The goal of Fisheries includes the optimization of financial and social benefits to the TCI, as indicated by fishery sector contributions to economic output (Ninnes and Medley, 1995). The orientation is on 'traditional' productivity-maximizing management of the export-oriented queen conch and spiny lobster fisheries (*e.g.*, Medley and Ninnes, 1997, 1999). Parks, on the other hand, has more of an ecosystem-based management orientation and mandate to manage for non-extractive recreational benefits within the existing marine reserve system (Homer, 2000; Campbell, 2003).

In the market sector, TCI fishers respond to market price signals and allocate their effort between conch and lobster/reef fish in an economically consistent manner (Rudd, in press c). Fish processors, who would be important actors in an assessment of export-oriented TCI conch and lobster fisheries, have a very limited role in the reef fish fishery as virtually all reef fishes are sold dockside or directly to restaurants (Halls *et al*., 1999). Both dive charter operators and tourist-oriented restaurateurs operate in a competitive business environment.

There are, then, six main participants to consider in the TCI institutional analysis: Finance, Fisheries, Parks, fishers, dive operators, and restaurant operators. Note that the power to influence decisions in the TCI declines from Finance, at the top of the list, to restaurant operators, at the bottom.

Institutional Options for Nassau Grouper Conservation

In this section, I explore the feasibility of different institutions and address how they affect, and are affected by, the incentives of the TCI actors. That is, which set of rules is most likely to have the capacity to technically achieve conservation objectives *and* to be successfully implemented and sustained given the ecological, social and political context outlined in the previous sections?

Status Quo Situation

Formal Nassau grouper fishing rules in the TCI are currently minimal. The use of scuba gear and spear guns are illegal for all commercial fishing and this rule is virtually universally respected by TCI residents. The only other controls on Nassau grouper fishing are a series of small marine reserves in which fishing is prohibited. The design of the reserves in the early 1990s was *ad hoc* (Homer, 2000) and enforcement vigilance is inconsistent at best (Olsen, 1986; Halls *et al.*, 1999; Rudd *et al.*, 2001).

The other important status quo rule affecting Nassau grouper conservation is a 40% tariff on imported seafood. The tariff makes local reef fishes relatively more attractive to restaurants and has led to higher prices for Nassau grouper than would otherwise be the case in the absence of tariff protection (Rudd, in press a). This has prompted increased fishing effort for reef fishes near Provo and has induced some commercial fishers from South Caicos to make the 60-km trip across the Caicos Bank when they are able to land the 100 kg or so of fish needed to make the trip financially worthwhile. While an import tariff is economically inefficient for the economy as a whole, the government gains tariff revenue and local fishers capture resource rents at the expense of consumers, whom are largely foreign tourists and expatriates.

Policy Alternatives

The combination of a *de facto* open access fishery for Nassau grouper and artificially inflated local market prices due to import tariff protection is potentially very dangerous. The import tariff exacerbates an already risky situation by increasing financial incentives for local fishers to sell Nassau grouper directly to tourist-oriented restaurants. Other resource users – local residents and ‘native’ restaurants using Nassau grouper, and dive tourism operators that use the fish non-extractively – are not a major threat to resource conservation at the current time, although this may have to be re-examined in the future should the situation change. This suggests that conservation policies should be directed at one of two intervention points: (1) the tourist restaurants that are the ultimate consumer of local reef fish or (2) the artisanal fishery that supplies the tourist restaurants with their product (Figure 11-2).

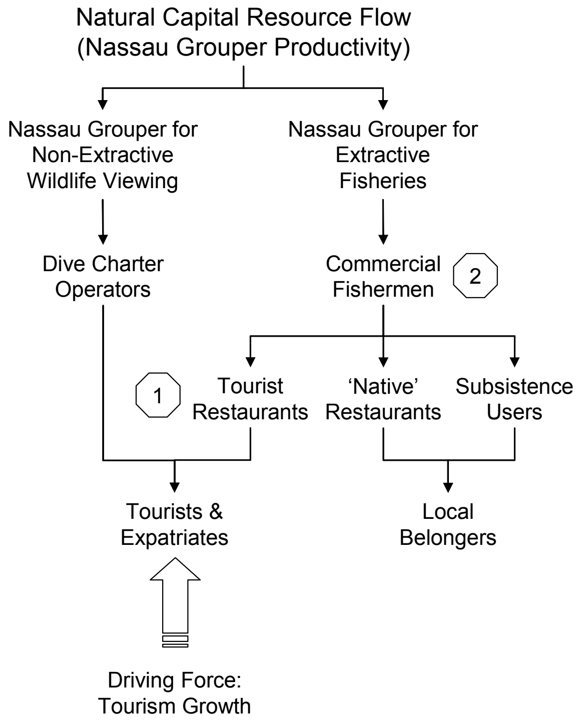

Figure 11-2 – Potential policy intervention points for Nassau grouper conservation in the Turks and Caicos Islands

Fisheries management regulations usually aim to control the supply of fish being harvested by either controlling fishing effort (*e.g.*, input-oriented technological, seasonal and area restrictions, or output-oriented catch limits) or the age at which fish recruit to the fishery (*e.g.*, input-oriented gear restrictions or output-oriented size limits), thereby ensuring adequate reproductive capacity is maintained (Medley *et al.*, 1993). It is useful to view these different types of rules in terms of the IAD framework (recall Figure 11-1); this analysis focuses specifically on five types of those rules (boundary, authority, information, scope and payoff) that could be used to ensure Nassau grouper conservation.

Boundary Rules

Boundary rules are those that stipulate conditions for entry or exit of action arena participants. Limited entry policies meant to control fishing effort are typically the main type of boundary rule used for fisheries management. Commercial fishers must meet two conditions in the TCI: they must be Belongers and they must pay a nominal fishing license fee. As Nassau grouper are mainly caught opportunistically by lobster fishers, limited entry rules would need to apply to lobster fishing, a policy that would draw strong political opposition from all sectors of local society. More fundamentally, limiting entry provides no guaranty Nassau grouper landings will be contained because of the flexibility generalist fishers have in targeting lobster, conch or reef fish in the TCI (Medley and Ninnes, 1999; Rudd, in press c). Even a limited number of fishers specifically targeting Nassau grouper could boost landings to unsustainable levels. As a result, limited entry licensing of participants in the Nassau grouper fishery is unlikely to be a feasible, or desirable, policy option.

At the restaurant level, industry entry and exit is largely determined by access to financial capital and economic performance. In the free-enterprise environment of the TCI, it is extremely unlikely that limits could be placed on how many restaurants were allowed to operate. Similarly to the fisheries situation, limits on industry participation would not necessarily control the utilization of Nassau grouper as restaurants have flexibility to substitute different seafood products.

Authority Rules

Three popular types of authority rules for fisheries management impose limits on fishing behavior: (1) restrictions on fishing vessels and gear; (2) seasonal closures; and (3) area closures. Given the existing ban on scuba gear and the simplicity of gear needed to fish for Nassau grouper (Hawaiian slings), efforts to impose technological rules for conservation are unlikely to have any incremental impact.

Seasonal closures play an important role in the management of grouper fisheries in other regions, especially where grouper spawning aggregations have been targeted (Sadovy and Eklund, 1999; Johannes *et al*., 1999). The timing of grouper spawning is quite predictable (Sadovy and Eklund, 1999) and large Nassau grouper can undergo long migrations (Bolden, 2000) along known corridors to reach spawning grounds. In the TCI, the main aggregation site (Phillip's Reef, off the eastern end of East Caicos – Tupper, 2002) is remote and rough weather during spawning season has restricted fishing activity to date. Seasonal closures may play a role in TCI management plans in the future but, in the short-term, are not significant factors for Nassau grouper conservation.

Marine reserves are widely viewed as a robust tool for the conservation of vulnerable reef fish such as groupers (Roberts, 1997; Murray *et al*., 1999; Rudd *et al.*, 2003). Tupper and Rudd (2002), however, found no difference in Nassau grouper density inside and outside of the Admiral Cockburn Land and Sea National Park (ACLSNP), South Caicos and hypothesize that this is due to the small size of the reserve (4 km^2) relative to the home range of typically-sized Nassau grouper. Given that only a small fraction (roughly 5% or 200,000 m^2) of the ACLSNP contains habitat suitable for adult Nassau grouper (M. Tupper, unpublished data), and that home range size scales exponentially with body size (Kramer and Chapman, 1999), large adults may require substantially more habitat area than exists within the ACLSNP. The three other TCI marine reserves in areas that may be important for Nassau grouper conservation (Northwest Point Marine National Park, 10 km^2; West Caicos Marine National Park, 4 km^2; and Princess Alexandra Land and Sea National Park, 26 km^2) have similar types and area of habitat suitable for adult Nassau grouper. Assuming that properly designed marine reserves included enough area to at least contain several times the normal home range of the species, this would imply ACLSNP (the reserve adjacent to the fishing port of South Caicos) would need to be expanded several-fold in size and other reserves would require similar expansions in area. Larger marine reserves are technically possible and one of the policy options considered further in this analysis.

A second policy option that is technically feasible is a trade ban. In tourist-oriented restaurants, a commercial trade ban is an authority rule that would prohibit restaurants from buying, possessing or selling local Nassau grouper; it could act as a demand-side proxy for a conservative supply-side landing limits. Local Belongers could still be allowed to fish Nassau grouper for subsistence and sell fish to 'native' restaurants that cater to Belongers. Trade restrictions have been used effectively for the conservation of some export-oriented marine products in the Pacific Islands (*e.g.,* Johannes *et al*., 1999) and are used more widely in international trade (*e.g.*, Best and Bornbusch, 2001) but do yet not appear to have been applied within a country to domestic fisheries at risk.

Information Rules

Currently no rules specify information that must be reported for either the fishery or restaurant sector. Landing slips are commonly used in developed countries to record information about fishing location, effort and price (and are often a condition of licensing agreements) but are rare and difficult to enforce in developing countries. For artisanal fisheries, minimum information management options are usually preferred (Johannes, 1998). Restaurant data would be much easier to collect as tourist-oriented restaurants usually keep detailed records on food supply purchases. Information rules will need to be considered in conjunction with authority or scope rules for both sectors.

Scope Rules

Output-oriented scope rules define what states-of-the-world are required, prohibited or permitted. Size limits and landing limits are two popular types of scope rules for fisheries. Nassau grouper are slow growing, large, fecund fish (Sadovy and Eklund, 1999; Coleman *et al*., 2000) so minimum size limits are unlikely to prove effective for conservation because of the continued vulnerability of the older, larger fish most important for gamete production. For the restaurant sector, scope rules are likely inappropriate as it would be difficult to specify meaningful rules about restaurant outputs.

Fishery landing limits may be applied to single trips or for entire fishing seasons using Total Allowable Catch limits (TACs). While trip limits may have limited applicability in a tropical lobster/grouper fishery, setting a TAC is a potentially feasible management option for Nassau grouper. In the face of the shortage of information about Nassau grouper population abundance and dynamics in the TCI, a conservative proxy for sustainable yield may be historical landing levels. It is known that historical fishing pressure has had a relatively limited impact in the TCI relative to other parts of the Caribbean (Tupper, 2002; Tupper and Rudd, 2002). Rudd (in press, a) estimated current Nassau grouper consumption at approximately 50,000 kg per year, so a conservative TAC may be in the 25,000 kg per annum range, more in line with pre-tourism domestic consumption.

Payoff Rules

The 40% tariff on imported seafood products is the main payoff rule that indirectly affects Nassau grouper conservation in the TCI as it alters incentives for both restaurant operators and fishers. Sanctions are not a direct factor in the current Nassau grouper fishery simply because it operates under *de facto* open access.

Table 11-1 summarizes the main policy alternatives considered in this analysis.

A Comparative Analysis of Conservation Options

What is the likelihood of implementing various technically feasible policies? Understanding the feasibility of choosing, implementing and sustaining each of the policy options outlined above requires that we consider the incentives of participants involved in choosing the rules as well as the incentives of resource users.

	Policy Options			
	Status Quo (de facto Open Access)	**Total Allowable Catch (TAC)**	**Enlarged Marine Reserves**	**Tourist Restaurant Trade Ban**
Fishery-Oriented Rules				
Limited entry (boundary)	Nominal	Nominal	Nominal	Nominal
Gear restriction (authority)	Scuba/Spear	Scuba/Speargun	Scuba/Speargun	Scuba/Speargun
Closed areas (authority)	Ineffective	Ineffective	Effective	Ineffective
Landing slips (information)	None	Required	None	None
Landing limits (scope)	None	Imposed	None	None
Restaurant-Oriented Rules				
Trade restrictions (authority)	None	None	None	Imposed
Paper trail (information)	None	None	None	Required
Import tariff (payoff)	40%	0% to 40%	0% to 40%	0% to 40%

Table 11-1 – Policy option configurations for Nassau grouper conservation in the Turks and Caicos Islands

Should/Can the Import Tariff be Eliminated?

The first policy question to consider is whether the current seafood import tariff should be eliminated, allowing restaurants to import fish products at a 40% price reduction? This would exert downward pressure on the price of local reef fish and, consequently, reduce fishing pressure on Nassau grouper. Clearly this would be a desirable step contributing to Nassau grouper conservation in the TCI.

At first examination, one might expect restaurants to be strongly opposed to the import tariff because it increases their cost of doing business. While restaurant purchasers indicated that a reduction in the import tariff would change their purchasing habits (Rudd, in press a), restaurants can pass on price increases to their customers, who seem to be relatively price insensitive (*e.g.*, is a $5 increase in the cost of a meal an important consideration for a tourist paying $250 or more per night for a hotel?). Thus, restaurants are likely not as supportive of eliminating the import tariffs as one might initially think. The political and economic reality in the TCI is such that the import tariff is likely to remain in effect for the foreseeable future.

Proposals to eliminate the import tariff would likely meet with strong opposition from Finance because of the loss of government revenue. Similarly, strong opposition could be expected from fishers because the market price for local reef fish could fall substantially (Rudd, in press a).

Impacts of Maintaining the Status Quo

It is important to consider that the marginal cost of fishing Nassau grouper is low for lobster divers as they spear Nassau grouper opportunistically: the cost is comprised mainly of the opportunity cost of time lost for lobster fishing (returning to the boat for a sling and landing the fish). In the case where there is a dockside market outlet for the fish, we should expect to see landings of most marketable fish that are encountered as marginal revenue is greater than marginal cost. Indeed, bycatch of all reef fishes was estimated to be almost 20-kg per day based on dockside sampling of 456 hours fishing effort in regularly fished grounds (Tupper and Rudd, 2002).

If prices rise in an open access regime, the effect should be to reduce the threshold volume at which it makes economic sense for South Caicos fishers to land fish in Provo. We would expect to see Provo fishers, on the other hand, expanding the geographic range of fishing activity as prices rise. Anecdotal evidence from restaurant buyers in Provo does, in fact, suggest that fishers now have to go farther afield than they used to in search of Nassau grouper (Rudd, in press a). Under the *status quo* regime, we would therefore expect to see ever-increasing effort targeted at Nassau grouper and an increase in the geographic scope of fishing activity (precisely the same pattern that has occurred in the less vulnerable spiny lobster and queen conch fisheries in the TCI – Olsen, 1986; Medley and Ninnes, 1999; Rudd *et al*., 2001).

Fishers may capture some resource rents generated by the Nassau grouper fishery, depending on their cost structure and the amount of fish they land and transport to Provo. With increasing depletion of Nassau grouper stocks, rent capture by fishers would fall to zero as the costs of fishing just offset revenues. Restaurants would need to compensate for increasing Nassau grouper scarcity by importing seafood (Rudd, in press a). Increased imports would result in increases in tariff revenues for Finance, thereby creating a situation where resource depletion could actually provide perverse financial incentives for government to maintain high import tariffs and oppose conservation efforts.

Impacts of a Total Allowable Catch Limit

If successfully enforced, a conservative TAC of around 25,000 kg would limit the amount of fish that are landed annually. As many fishers deliver reef fish directly to restaurants (Halls *et al*., 1999) and the market price would remain artificially high due to import tariffs on competitive products, there would be substantial incentives for fishers to cheat on the TAC. Effective monitoring and enforcement of a TAC would be problematic for Fisheries as it would require a substantial increase in resources. Requiring all vessels to land fish at official landing sites could help to make a TAC more workable.

Impacts of Large Marine Reserves

Expanding the size of marine reserves could effectively protect Nassau grouper. This strategy would also create incentives for fishers to target Nassau grouper stocks outside of reserves because market demand still exists unchanged while the availability of potentially catchable fish declines. Properly designed marine reserves that effectively increase Nassau grouper abundance and/or mean size within reserve boundaries may also attract poachers, making it easier to land a sufficient volume of fish to make it economical to deliver fish directly to restaurants in Provo. Most importantly, however, the expansion of marine reserves in prime lobster fishing grounds would impose a large opportunity cost on legitimate local fishers. Large no-take reserves would force fishers out of prime lobster fishing grounds (Nassau grouper and large lobsters tend to be found in similar habitat along reef drop-off's) and farther from port, potentially increasing fishing costs, decreasing landings, increasing conflict over site access in remaining grounds, and decreasing overall wealth generation within the fishery. Conflicts between fishers and fisheries officers would likely rise as a result of more active monitoring and enforcement relative to the status quo.

Successful marine reserves can have a significant impact on the abundance of fish within their boundaries (*e.g.*, Polunin and Roberts, 1993). Marine reserves would, therefore, be popular with dive charter operators as the larger and more abundant reef fish stocks within the reserve could allow them to charge higher prices for trips and/or increase the number of clients they host (Rudd and Tupper, 2002). Alternatively, the government of the TCI might capture those resource rents generated by marine reserves by imposing park entry fees for divers. Current conflicts between divers and fishers (Bennett *et al*., 2001) could be reduced if fishers and dive boats were spatially segregated.

Impacts of a Trade Ban in Tourist-Oriented Restaurants

Imposing a trade ban on tourist-oriented restaurants while retaining the import tariff should have similar biological effects as the imposition of a conservative TAC, with fishing effort being limited to a level known to be historically sustainable. Fishing effort directed at Nassau grouper would decline as the local market demand would be halved without the tourist-oriented restaurant trade. The absence of a supply of local Nassau grouper for restaurants would induce an increase in imports of substitute products (Rudd, in press a), hence slightly increasing tariff revenue for Finance. As enforcement moves ashore to the restaurants, there should be less conflict on the water between fishers and fisheries officers relative to marine reserves or TACs. Most importantly, the costs of monitoring and enforcement should fall dramatically for Fisheries. Given the serious consequences that might befall restaurants for a rule infraction (*e.g.*, fines and/or high opportunity costs resulting from the suspension or loss of business license), there should also be relatively high levels of compliance from the restaurant industry.

Table 11-2 summarizes the anticipated transaction costs and the impacts of sanctions on resource users for the four policy alternatives.

Summary of Support for / Opposition to Policy Packages

Assuming that the 40% seafood import tariff remains in place, four main policy options were considered in the policy analysis: the status quo open access situation; a conservative TAC based on historical consumption levels; large, ecologically effective marine reserves; and a commercial trade ban directed at tourist-oriented restaurants. The strategic objectives of various participants will influence their willingness to support the policy alternatives.

Finance

As the TCI is a tax-free haven, Finance generates national revenue largely via license fees, various service charges, tariffs on imported goods, and value-added taxes on hotel accommodations and restaurant meals. Thus, Finance is best viewed as being a revenue maximizer that has no other explicit objectives regarding resource conservation or other capital assets. Finance is responsible for the TCI budget, has substantial power over resource allocation, and participates in the administration of other government departments; it is clearly the most powerful of the three relevant government agencies. Finance would therefore likely veto any proposal to eliminate the seafood import tariff and support any option that retains the tariff. A trade ban should induce a slight increase in imports and tariff revenue, slightly increasing Finance's support for a combination of trade ban and import tariff as a preferred policy package even though it has no explicit conservation objectives.

	Policy Options			
	Status Quo	TAC	Reserves	Trade Ban
Management Transaction Cost				
Information				
Finance	Low	Low	Low	Low
Fisheries	Low	High	High	Moderate
Parks	Low	Low	Moderate	Low
Fishers	Low	Moderate	Low	Low
Dive Operators	Low	Low	Low	Low
Restaurant Operators	Low	Low	Low	Moderate
Monitoring				
Finance	Low	Low	Low	Low
Fisheries	Moderate	High	High	Moderate
Parks	Moderate	Moderate	High	Moderate
Enforcement				
Finance	Low	Low	Low	Low
Fisheries	Moderate	High	High	Moderate
Parks	Moderate	Moderate	Moderate	Moderate
Sanction Probability/Costs				
Fishers				
Pr (Detection)	Low	Low	Moderate	n/a
Pr (Charges|Detection)	Low	Moderate	Moderate	n/a
Pr (Sanctions|Charges)	Moderate	Moderate	Moderate	n/a
Sanction Direct Cost	Low	Low	Low	n/a
Sanction Opportunity Cost	Low	Low	Low	n/a
Restaurant Operators				
Pr (Detection)	n/a	n/a	n/a	High
Pr (Charges|Detection)	n/a	n/a	n/a	Moderate
Pr (Sanctions|Charges)	n/a	n/a	n/a	Moderate
Sanction Direct Cost	n/a	n/a	n/a	Moderate
Sanction Opportunity Cost	n/a	n/a	n/a	High

Table 11-2 – Anticipated transaction costs and expected sanction costs of Nassau grouper conservation policy options (Pr x|y indicates the probability of x given y having occurred).

Fisheries

Fisheries is motivated by concerns for the resource (particularly the maintenance of extractive fishery benefits) and by the need to minimize resource management expenditures. As such, we would expect to see Fisheries support any of the conservation measures philosophically. A TAC and trade ban may be preferred to larger marine reserves from a fisheries productivity perspective because benefits are distributed over the entire islands rather than concentrated in specific locales. A central concern of Fisheries, however, relates to expenditure minimization as the departmenmt has very limited resources. Any policies that significantly increase the cost or complexity of fisheries management will meet with resistance or, if officially adopted, may be practically neglected.

Amongst the technically feasible policy options, TACs would probably have the highest costs for Fisheries as monitoring and control measures would have to be implemented to track landings at multiple landing sites (involving extensive dockside monitoring). The costs of enforcing expanded marine reserves may be nearly as high, given the costs of maintaining patrol boats and undertaking surveillance activities. The status quo situation requires little monitoring and, from an expenditure minimization perspective, is attractive for Fisheries. The commercial trade ban should also be quite attractive to Fisheries as enforcement is moved ashore, reducing surveillance costs. In addition, the paper trail requirements of a commercial trade ban would be the responsibility of restaurants, allowing Fisheries to 'download' that cost to the private sector, and monitoring would become a simpler matter of ensuring appropriate documentation and periodic site inspections of restaurant facilities. Thus, in balance, we would expect to see Fisheries showing support for the commercial trade ban option and reticence to adopt other management measures despite the potential ecological benefits they may have for Nassau grouper stock productivity.

Parks

Parks motivations center on conservation for tourism purposes (*i.e*, non-extractive use value) and expenditure minimization. As such, we expect to see philosophical support for policies that lead to increases in abundance of icon reef species like Nassau grouper. This support would be strongest for expanded marine reserves because they are most likely to lead to appreciable increases in Nassau grouper (and other fish species) abundance in specific locales important for dive tourism. A trade ban and TAC would tend to have more limited impacts on fish abundance in specific locales although, philosophically, Parks would also support those measures. Like Fisheries, Parks has very constrained resources and faces strong incentives to minimize expenditures. Because Parks plays a role marine reserve monitoring and enforcement, they would face increased costs with the implementation of large marine reserves. TACs and a commercial trade ban would be more attractive for Parks, on the other hand, as the monitoring and enforcement costs for those measures would fall squarely on Fisheries.

Commercial Fishers

Commercial fishers have strong short-run incentives to maintain the status quo open access situation, as they benefit from higher market prices, freedom of access and flexible sales opportunities. Any reduction in the 40% import tariff would be opposed. If effective conservation regulations are to be developed, the issue is one of determining what rule set is least offensive to fishers, most of whom who regard open access as a basic citizen's right in the TCI. The imposition of any conservation rules in the TCI would need to be accompanied by longer-term education and awareness-building activities about the benefits of conservation. This is especially the case given the history of biological and economic over-exploitation (*e.g.*, during the "Big Grab", the first three weeks of lobster season, up to 95% of lobster harvested in some fishing grounds are below minimum size limits – Rudd *et al*., 2001) and poaching (CDB, 2002) of other species in the TCI. A conservative TAC is likely to raise some opposition from fishers but expanded marine reserves would face the stiffest opposition because they would close fishing in traditional fishing grounds and impose high opportunity costs on fishers. Fisher opposition to a commercial trade ban should be less strident as the direct and opportunity costs of the policy are lower relative to other options. Some fishers may even slightly support a tourist-oriented restaurant trade ban from an equity perspective (*i.e*, the costs are primarily borne by non-Belongers while fishers and their families can continue to use Nassau grouper for subsistence purposes and sell them to local native restaurants).

Dive Charter Operators

The situation for dive charter operators is fairly simple. They would oppose the status quo open access and support restrictions on fishing because any restrictions will help maintain fish stocks that are vital for providing their clients with a satisfying dive experience. Expanded MPAs that had more abundant and/or larger fish in a concentrated geographic area would be the preferred option. A trade ban or conservative TAC would be welcome but would have less visible impact on Nassau grouper abundance and mean size in the main dive sites relative to marine reserves.

Tourist-Oriented Restaurants

A trade ban would increase red tape and impose some costs on restaurants although profitability – the restaurant's primary motivation – should not be impacted strongly by a Nassau grouper trade ban. Other types of fish are readily substituted for local Nassau grouper (*e.g.*, local snapper and pelagics, imported snapper and grouper – Rudd, in press a). Restaurant operators would likely be neutral with regards to other fishery management options.

Table 11-3 summarizes the support of key participants for the different policy options. Policy options are ranked from most preferred (1) to least preferred (4) for each actor. Even though the rankings do not reflect the power of actors nor the strength of their preferences, note that the trade ban is ranked as most preferred by the three public sector actors and least preferred by only restaurant operators, the least-powerful private sector actor.

Actor	Objectives	Policy Ranking Status Quo	TAC	Marine Reserves	Trade Ban
Finance	Revenue Generation	2	2	2	1
Fisheries	Fishery Production	4	1	3	1
	Expenditure Minimization	2	4	3	1
Parks	Recreational Benefits	4	2	1	2
	Expenditure Minimization	3	1	4	1
Fishers	Revenue Targets	1	2	4	2
Dive Operators	Profit Maximization	4	2	1	2
Restaurateurs	Profit Maximization	3	1	1	4

Table 11-3 – Ranking of actor support for Nassau grouper conservation policy options (1 = most preferred; 4 = least preferred).

Discussion

From a conservation perspective, it would be preferable to eliminate the seafood import tariff as a starting point for a Nassau grouper conservation program in the TCI. The revenue generation goals of Finance and the price support that the tariff provides for fishers means, however, that there are strong incentives for both government and fishers to oppose this measure. Unfortunately, living with import tariff market distortions (at least in the short- to intermediate-term) needs to be accepted as a political reality in the TCI and conservation strategies for Nassau grouper must take this into account.

Given the presence of the import tariff, a ban on the commercial trade of Nassau grouper in tourist-oriented restaurants is the preferable policy alternative. A conservative TAC might be biologically effective, but the high costs of enforcement for Fisheries and high likelihood of cheating means that this is a risky option from a conservation perspective. Expanded marine reserves may also effectively protect Nassau grouper within reserve boundaries if adequately enforced, but migration outside of reserve boundaries (*e.g.*, Bolden, 2000) could put stocks at risk without additional effort restrictions (Rudd *et al.*, 2003). Further, the incentives for fishers to fish illegally within marine reserves would increase because those areas are productive, providing high quality habitat that both Nassau grouper and large, adult spiny lobster utilize. Monitoring and enforcement of expanded marine reserves can take substantial resources and constant vigilance. Mascia (2000), for example, noted that fishers in the neighboring Bahamian fishery flocked into marine reserves to fish illegally each time Bahamian patrol vessels broke down. Again, these factors translate into a high degree of risk that conservation objectives will not be met in the TCI because of the difficulty in successfully implementing reserves and sustaining the investment necessary to manage them on an ongoing basis.

A commercial trade ban on tourist restaurants would make it illegal for those businesses to purchase, possess or sell local Nassau grouper. This policy would have several significant advantages over the use of TACs, marine reserves or other traditional effort-oriented fisheries management tools. First, a trade ban aimed at tourist restaurants would allow for on-going subsistence fishing and sales to 'native' restaurants, thereby respecting traditional use and making the policy politically palatable to TCI Belongers and politicians. Second, a trade ban would circumvent the problem of imposing high opportunity costs on fishers caused by closing prime lobster grounds and would, as a result, address a major source of opposition to the policy. Third, a trade ban aimed at tourist restaurants should also, at a minimum, allow Finance to maintain the revenue flow they currently capture from import duties on seafood products. It may even increase government revenue should more imports enter the country to compensate for the drop in supply of local Nassau grouper to tourist restaurants. The trade ban policy option, thus, possesses the very attractive property that it is congruent with the strategic objectives of Finance, the most powerful government department, even though Finance has no explicit strategic objectives regarding environmental conservation. Finally, and perhaps most importantly, a trade ban aimed at tourist restaurants would move enforcement ashore, sharply reducing the costs of gathering information, monitoring, surveillance and enforcement for Fisheries, and reducing confrontations and conflicts between fishers and fisheries officers.

Other policy options that are possible have not been explored in this analysis. Some – fisheries enhancement, building management capacity, awareness and education programs about the benefits of conservation – are long-term in focus, so would not mitigate the short-term threat posed by increasing tourism in the TCI. It may well be the case that clients of restaurants – who ultimately pay the price for a trade ban policy – would support conservation measures directly if an awareness-building campaign accompanied the ban. Many of the consumers are, after all, also divers and snorkellers who derive well being from seeing abundant reef fish in the water and may be willing to pay higher meal prices to support conservation efforts. Strengthening quasi-property rights for fishers (*e.g.*, individual quotas) may induce long-run stewardship in some fisheries but in the TCI, as in many developing countries, options for establishing property rights are limited due to substantial information requirements and the lack of established markets to trade quota. Community quotas would also be problematic due to very limited community management capacity and low levels of trust in the TCI (Rudd *et al.*, 2001).

Why should a policy focused on the restaurant trade be more effective than traditional fisheries-focused management tools for conservation of a species like Nassau grouper? The role of credible threats plays a central role it seems. In the absence of social norms that constrain opportunism, fishers' incentives are a function of (1) the probability of being observed fishing, (2) the probability of being charged, (3) the probability of being convicted (a function of investigation effort and the strength/integrity of the legal system) and (4) the severity of the sanction applied at the end of the day (Crawford and Ostrom, 1995; Nielsen, 2003). Effective deterrence of poaching requires an effective suite of monitoring, control, surveillance and enforcement tools that are not available in the TCI and only rarely available in tropical developing countries. When credible threats are absent, we see, for example, a proliferation of 'paper parks' where users violate marine reserve access rules with impunity (Alder, 1996; McClanahan, 1999). For restaurants, however, the probability of being observed, charged and convicted of trade infractions is likely far higher than for fishers because surveillance is simpler. Further, possible sanctions for restaurants include non-trivial fines and/or a suspension of the firm's business license. Unlike the situation for fishers, the opportunity costs of violating rules could be very high for restaurateurs.

This analysis has not touched on the role of rewards versus sanctions. Recall from Figure 11-1 that payoff rules can involve either and can be directed at either prescribed behaviors (authority rules) or outputs (scope rules). Direct rewards for resource users ('carrots') can be more effective than sanctions ('sticks') in achieving strategic conservation objectives (Ferraro and Kiss, 2003). In the TCI, public sector financial capital is so scarce that payments for meeting conservation standards would likely need to be financed with outside assistance. Should it be demonstrated that healthy TCI Nassau grouper stocks 'export' larvae 'downstream' to the Bahamas or Florida, there may be opportunities for international compensation schemes to be developed. These, however, are not short-term options but would require substantial time and effort to implement. As such, conservation payments should not be ruled out for the longer-term but the focus in the short-run should remain on regulatory options until the potential role for compensation mechanisms is better defined.

One final comment is in order regarding the popularity of marine reserves as a tool for ecosystem-based fisheries management. A common view is that marine reserves are a transaction cost-minimizing conservation policy because it is a simple matter of observing, yes or no, whether illegal fishing activity is taking place in a marine reserve (*e.g.*, Roberts, 1997). Considering the *expected* benefits of a marine reserve requires, however, that an analyst take account of both the potential benefit of a policy and the likelihood that that the beneficial outcome will be achieved. This likelihood depends on implementation and operational phases.

In the TCI, it appears that the likelihood of successful implementation and the likelihood of securing the necessary financial capital needed for ongoing management of marine reserves are both low relative to the commercial trade ban option. Further, larger, ecologically effective marine reserves do not appear to have inherent biological advantages relative to an effective, conservative TAC or a commercial trade ban. Therefore, the expected conservation benefit of a commercial trade ban for Nassau grouper appears to be greater than the expected benefit of a policy relying on marine reserves. This is simply because the likelihood of trade ban implementation is higher and the management costs associated with onshore enforcement of a trade ban lower than the fisheries patrols needed for marine reserve monitoring in the TCI. More generally, if market-oriented policies can deliver fisheries conservation results, as it seems they can, then those policies should be considered much more widely than they are now, especially in developing countries where financial resources for information-intensive, fisheries management approaches are scarce and/or when a relatively small number of market outlets are available to local fishers.

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CHAPTER 12

CONCLUSIONS

In the conclusion, I want to return to the goals and objectives of the thesis. These were to (1) integrate insights from the disciplines of ecology, economics and political science in a way that facilitates the design and assessment of ecosystem-based fisheries management experiments involving MPAs and/or the devolution of fisheries governance; and (2) to use a case study of Nassau grouper conservation in the TCI to illustrate the utility of institutional analysis applied to complex fishery problems requiring an ecosystem-based fisheries management approach. As most of the important points have been made previously in the individual chapters, the summary here can be quite brief. I start by reviewing the lessons from the TCI case study, then turn to the broader issue of using marine reserves as multi-purpose tools for ecosystem-based fisheries management. I particularly want to focus on the role of social capital as a key variable that impacts the potential effectiveness of MPAs and to point out the danger of trying to 'promote' MPAs as multi-purpose tools that simultaneously provide long-run conservation and short-run fisheries benefits. Finally, I return briefly to the issue of designing and assessing ecosystem-based fisheries management experiments using the modified IAD framework.

The Turks and Caicos Island Case Study

Studying artisanal fisheries can be difficult due to the lack of existing data (*e.g.*, basic data on landings and fishing location) and challenges in gathering primary data (*e.g.*, lack of cooperation and time/budget constraints). The goal of this part of the research was to gather enough information on contextual variables, the effectiveness of existing fishing rules and regulations, and the incentives of key actors with interests in Nassau grouper fishing or conservation, so that a policy analysis examining options for Nassau grouper conservation could be conducted. Clearly, more data could be gathered and more rigorous field studies conducted (*e.g.*, assessing consumer surplus of dive tourists using choice experiments, assessing fisher decisions about site choice and target species, and further evaluating the ecological effectiveness of other MPAs in the TCI).

While this additional research would be valuable, for current purposes the various components of this study have demonstrated that there is an increasing threat to Nassau grouper because of growing tourism and existing trade barriers in the TCI (Chapters 6 and 7). Restaurant buyers already indicate that fishers based in Provo have to go farther afield to find Nassau grouper and that they are less abundant than they were just a few years ago and that more fishers from South Caicos are making the trip across Caicos Bank to deliver Nassau grouper directly to restaurants. Reef fish appear to play a role in the annual allocation of fishing effort between commercial conch and lobster fishing, as most reef fishes are landed opportunistically by lobster fishers (Chapter 8). This implies, albeit indirectly, that we can expect to see increasing fishing pressure on Nassau grouper in the future as monetary incentives for fishing increase. A complicating factor arises in the TCI because Nassau grouper are also popular fish for wildlife viewing by dive tourists. Divers hold preferences for seeing more abundant and larger Nassau grouper (Chapter 9). That is, more abundant and larger Nassau grouper increase the well being of dive tourists and, as a result, provide them with non-extractive use value that might be extracted by local dive charter operators or the government.

Given the inherent vulnerability of Nassau grouper, there will have to be management responses soon if Nassau grouper stocks are to avoid the extirpation other parts of the Caribbean have experienced. Management of reef fishes is, however, a low priority for the DECR. The Fisheries division focuses limited resources more on science-based management of the export-oriented conch and lobster fisheries. The only protection afforded Nassau grouper stocks results from restrictions on fishers using scuba and from a limited number of small MPAs. TCI scuba restrictions are of limited value because many Nassau grouper occur at shallow depths and are accessible to free divers. Unlike some other countries in the tropical western Atlantic (Belize, Cayman Islands and Bahamas) that have implemented or are planning seasonal closures, spawning site MPAs or bag limits to protect spawning aggregations of Nassau grouper (Society for the Conservation of Reef Fish Aggregations Newletter, December 2002), the TCI has no such regulations in place or planned.

Looking specifically at the issue of MPA design, it is evident that the Admiral Cockburn Land and Sea National Park is too small to provide effective protection for Nassau grouper, although there was evidence that it is effective for smaller hogfish and white margate (Chapter 10). While other MPAs in the TCI were not examined, those small reserves are also likely to have a limited impact on Nassau grouper. Knowing that well-designed, larger MPAs have proven very effective for protecting predatory reef fish like Nassau grouper in other parts of the world, the logical question to ask is whether larger MPAs could be successfully implemented in the TCI.

The high opportunity costs for local fishers (*i.e*, the loss of productive lobster fishing grounds) and expenditure-minimizing incentives for some government departments are likely to hinder adoption and implementation of larger MPAs. The Fisheries division, in particular, would face a significantly increased enforcement burden if expanded MPAs were to be implemented. This finding is in contrast to the common wisdom that MPA enforcement is simple and, as a result, economical (Roberts, 1997). Incentives for fishing illegally in MPAs are a function of (1) the probability of being observed fishing, (2) the probability of being charged, (3) the probability of being convicted (a function of investigation effort and the strength/integrity of the legal system) and (4) the severity of the sanction applied at the end of the day (*e.g.*, do judges – or other community members – take poaching seriously?). To effectively deter poaching (when social norms do not) requires an effective suite of monitoring, control, surveillance and enforcement tools that are rarely available in tropical developing countries. Knowing whether a boat is fishing in an MPA may seem a simple matter of observation but to change fisher behavior – and hence achieve conservation objectives – requires much more effort and resources.

The relatively healthy state of Nassau grouper stocks in the TCI suggest that the historically low population in the islands has not had a strong enough impact on the Nassau grouper stock to lead to serious stock depletion. This, in turn, suggests that the imposition of very conservative landing limits, roughly equal to historical landings, could insulate Nassau grouper stocks from tourism-driven market pressure. Ecosystem-based fisheries management proponents correctly argue that setting and enforcing TACs is more risky and costly than simpler measures like MPAs (*e.g.*, Roberts, 1997; Johannes, 1998). This led me to consider whether there may be demand-side proxies for a conservative TAC that could limit fishing effort to sustainable historical levels while reducing the transaction costs of fisheries management for key government actors and opportunity costs for fishers.

The management experiment I propose in Chapter 11 is a commercial trade ban that prohibits the purchase or possession of Nassau grouper by any tourist-oriented restaurant. This could be implemented while leaving seafood import tariffs in place. From a conservation perspective, the best policy option would be to eliminate the import tariff as well but there are strong incentives for government and fishers to oppose this measure; living with import tariff market distortions will need to be accepted as a politically motivated reality in the TCI. Market-oriented export bans have been used effectively in the Pacific (Johannes, 2002). The Convention on the International Trade of Endangered Species (CITES) has also used threats of trade restrictions very effectively to reduce market incentives that encourage the trade of species at risk. In fact, queen conch is a CITES Appendix 2 listed species (Mulliken, 1996) and trade is allowed only when conch fisheries are scientifically managed in a sound manner. The TCI is one of the largest exporters of conch and, though not a signatory to the CITES agreement, abides closely to management guidelines for fear of losing access to the important United States market.

A commercial trade ban on tourist restaurants would make it illegal for those businesses to purchase, possess or sell local Nassau grouper. An adequate paper trail would be needed for enforcement purposes and sufficient sanctions authorized by government to significantly alter the incentives for restaurants buying fish directly from fishers (*e.g.*, substantial fines and/or a loss of business license). This policy would have several significant advantages over the use of MPAs or other traditional supply-side fisheries management tools:

- A trade ban aimed at tourist restaurants should reduce fishing pressure on Nassau grouper in all areas, not just within designated MPAs and could, as a result, provide more widespread benefits to the dive industry and local subsistence fishers than MPAs alone;

- A trade ban aimed at tourist restaurants would allow for on-going subsistence fishing and sales to 'native' restaurants, thereby respecting traditional use;

- A trade ban aimed at tourist restaurants should not impose high opportunity costs of lobster fishers by closing prime fishing grounds inhabited by both spiny lobsters and Nassau grouper. This would reduce a major source of opposition to the policy and potential conflict on the water;

- A trade ban aimed at tourist restaurants should, at a minimum, allow the TCI Ministry of Finance to maintain the revenue flow they currently capture from import duties on seafood products and may even increase their revenue should more imports enter the country to compensate for the drop in supply of local Nassau grouper to tourist restaurants;

- A trade ban aimed at tourist restaurants should prompt restaurants to pass on any increase in their costs to tourists and expatriates who, in the TCI, tend to be very affluent and would likely not find a small increase in the cost of meals a serious imposition (with an accompanying awareness campaign, a ban might even be supported because of its conservation benefits). Policies that pass on costs to non-Belongers are likely to be politically popular with the local Belonger population; and

- A trade ban aimed at tourist restaurants would move enforcement ashore, sharply reducing the costs of monitoring, control, surveillance and enforcement for the Fisheries and potentially reducing confrontations and conflicts between fishers and fisheries officers.

This combination of potential advantages should be something that key actors could support. Implementation of a commercial trade ban would move enforcement ashore, so this policy experiment should be feasible without lengthly debate over resource access. Key performance measures could be adequately monitored at reasonable cost and options should, in fact, exist for shifting the burden of compliance costs to the restaurant industry itself.

Marine Reserves and Fisheries Governance[1]

Marine reserves can protect essential fish habitat and vulnerable fish species, increase animal size and/or abundance within reserve boundaries, and, in principle, induce larval 'export' and density-dependent emigration ('spillover') across reserve boundaries (Murray *et al.*, 1999; Roberts *et al.*, 2001; Halpern, 2003). They are also thought to reduce management costs because of their simplicity and ease of enforcement, making them attractive multi-purpose tools for ecosystem-based fisheries management (Roberts, 1997). Marine reserves may be particularly important tools for implementing ecosystem-based fisheries management in tropical inshore fisheries for these reasons and because of the difficulty in applying traditional fishery management tools (*e.g.*, gear restrictions, total allowable catch limits, limited entry licensing) in data-poor artisanal fisheries (Johannes, 1998).

Whether or not marine reserves are, in fact, efficient policy tools likely depends on (1) whether the primary objective of the reserve is for the maintenance of ecosystem resilience in the face of uncertainty or the enhancement of fisheries productivity and (2) whether the 'State' and 'Community' have the capacity to engage in effective fisheries governance. Insights from transaction cost economic theory (Williamson, 1999) suggest that the validity of the commonly held view – that marine reserves can simultaneously provide effective insurance against uncertainty and efficient enhancement of fishery production – is suspect. The discriminating alignment hypothesis states that transactions, which differ in their attributes, are assigned to governance structures that vary according to their competencies and organizational costs, so as to minimize the transaction costs of governance.

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1 The short essay has been accepted, subject to appropriate revisions, as: Rudd, M.A. and Tupper, M.H. Marine reserves and fisheries governance. *Conservation Ecology*.

The key attribute of the marine fisheries transaction – sustaining natural capital that provides flows of valuable ecosystem goods and services – is the degree of uncertainty it entails. Governance structures for this transaction need to be based on hybrid co-management regimes that balance Community and State responsibilities because private ownership of common pool resources is usually not ecologically or socially feasible in artisanal fisheries. The optimal balance depends on transaction costs (planning, information gathering, monitoring, enforcement and *ex post* opportunism).

When ecological or socio-economic predictability is low, capable hierarchical organizations (transparent, honest, responsive State agencies) tend to have a comparative advantage in the provision of insurance-oriented services. When predictability is low, the goal of using marine reserves should be oriented towards ensuring the conservation of biodiversity and maintenance of ecosystem structure, function and resilience.

Community should have a comparative advantage enhancing fisheries production, on the other hand, when predictability is relatively high and there is rapid, tangible feedback between management investments and fishery productivity. Community governance capacity is a function of structural and cognitive social capital (social networks, protocols, norms and values) that facilitate mutually advantageous collective action (Rudd *et al*., 2003).

When the three factors, predictability (P), State capacity (S), and Community capacity (C), are considered together, a governance matrix emerges with eight case types: [PSC]; [PS~C]; [P~SC]; [P~S~C]; [~PSC]; [~PS~C]; [~P~SC]; and [~P~S~C], (~ indicates 'not'). General policy prescriptions vary by case type.

When predictability is relatively high, the discriminating alignment hypothesis suggests that efficient governance systems will generally tend to be locally managed and production-oriented. When State and Community capacity are both high [PSC], the generic policy prescription is for devolution of management authority from State to Community and implementation of formal individual or common property rights. Marine reserves may well be part of a Community's management package, but there is no reason to expect that marine reserves will be necessary or sufficient for fisheries production in relatively predictable environments. In fact, economic modeling suggests that when fishers' spatial behavior is accounted for, reserves likely only provide economic benefits in areas that have been dramatically overexploited (Sanchirico and Wilen, 2002). For example, the coral reefs surrounding San Salvador Island, Philippines had been devastated by destructive fishing practices and *de facto* open access through the 1980s (Katon *et al*., 1999) and the relatively quick recovery in reef fish abundance was predictable given effective management. A Filipino NGO invested in building community capacity, municipal and national governments passed enabling legislation, and the community implemented a marine reserve and restrictions on destructive fishing methods (cyanide, dynamite, fine-mesh nets). As a result, live coral cover, fish abundance and fish species diversity rose dramatically. San Salvador adopted a reserve as part of their package, but the strong recovery in live coral cover (23% in 1988 to 57% in 1998) suggests that halting destructive fishing also played a key role in the recovery.

When only Community capacity is low [PS~C] (*e.g.*, the deteriorating South African abalone fishery – Hauck and Sweijd, 1999), the State should take account of fishers' incentives to engage in short-run opportunism and implement regulations or market mechanisms to align social and individual incentives, and, over the long run, invest in structural social capital. If State capacity is low [P~SC], strong social networks may restrict access to fishery resources, but there is a need to develop legal infrastructure to enforce property rights and efficiently resolve disputes, thus minimizing resource rent dissipation. In Indonesia, some communities have used customary law ('sasi') to manage stocks of economically valuable mother-of-pearl shell (*Trochus niloticus*) for local economic gain (Ruttan, 1998). Local institutions have often come under pressure, however, as the central government has proven incapable of providing coordinated fisheries management support, adequate enforcement of national rules, or legal backing for traditional management systems (Novaczek *et al*., 2001). In predictable environments where State and Community lack capacity [P~S~C], there is an important long-term role for donor agencies and NGOs in helping build capacity for devolved governance regimes.

In unpredictable environments under fully functional governance [~PSC], transaction cost economic theory suggests that precautionary policies, including marine reserves, be used to fulfill insurance functions. The State should retain a prominent role due to the need for policy experiment coordination, engaging in scale-matching at regional levels (often requiring international negotiation), and compensating local people who bear the opportunity costs of conservation. When State capacity is low [~P~SC], government capacity-building and decision-maker education will be needed while Communities draw on their social capital to cope in an environment with little opportunity for formal recourse against cheaters from outside the local community. Local communities manage customary fishing rights areas (CFRAs) in Fiji, for instance, using a variety of tools for integrated upland and coastal zone management, including reserves (Cooke *et al.*, 2000).

Despite official government recognition of CFRAs, enforcement capacity and inter-department coordination is low, so some CFRA managers feel helpless to prevent poaching and others use more aggressive, and technically illegal, enforcement methods. When predictability and Community capacity is low [~PS~C], local norms discouraging opportunism can be absent, leading to an erosion in compliance and prompting a prescription of strong regulation and norm seeding by the State if conservation objectives are to be met in an uncertain environment. In the Turks and Caicos Islands artisanal fishery for spiny lobster, for example, up to 95% of lobsters landed from some fishing grounds are undersize during the first weeks of lobster season (a period referred to as 'The Big Grab' by locals – Rudd *et al.*, 2001). When both Community and State are weak [~P~S~C], international organizations may need to resort to coercion to protect ecosystems that provide regional insurance services while building national governance capacity.

Marine reserves are undoubtedly valuable policy tools that can provide insurance services in uncertain environments and will play a central role as a robust policy tool for ecosystem-based fisheries management. The common perception that marine reserves can efficiently fulfill both fisheries enhancement and conservation roles in highly devolved governance regimes may, however, be in error. The discriminating alignment hypothesis strongly suggests that substantially different governance systems are needed to achieve different objectives.

This further implies that efforts to 'sell' marine reserves that are ultimately insurance-oriented on the basis of their fisheries production benefits are risky and may hinder the evolution of appropriate governance regimes for fisheries conservation. Capable public sectors will likely retain substantial governance rights and duties because of their comparative transaction cost advantage when uncertainty is high. Where marine tourism potential exists, the non-extractive economic value provided by focal species within marine reserves may provide economic benefits for local communities (Rudd and Tupper, 2002) and help reconcile the tension between society's insurance-oriented conservation objectives and economic objectives held by local resource users. In regions with limited tourism potential, a "no promises" approach may be more useful. For example, the government of American Samoa (Daschbach, 2002) encourages village councils to establish conservation-oriented marine reserves for the benefit of future generations, with no fisheries enhancement benefits promised over the short term.

Under precautionary ecosystem-based fisheries management, experimental management is needed to build further understanding about complex fishery system processes (Walters, 1997). I suggest that Community and State capacity should be explicitly considered in the process of developing and testing hypotheses about the effectiveness and efficiency of marine reserves in comparative policy experiments. This points to an important institutionally oriented marine conservation research agenda for the future.

Experimental Design and Monitoring for Ecosystem-Based Fisheries Management

Successful implementation of ecosystem-based fisheries management policies requires that managers consider conservation, economic and social goals and objectives in transdisciplinary policy experiments. There is a need for an analytical framework that can be used to both identify and design policy experiments that will guide adaptive ecosystem-based fisheries management and to monitor the status of the fishery system through quantifiable indicators. In this dissertation, I have proposed that a modified IAD framework is used for this purpose.

The modified IAD framework, which incorporates capital assets into the framework structure and uses PSR ('pressure-state-response') terminology, has several important benefits for ecosystem-based fisheries management policy design and monitoring. Most importantly, societal responses to threats against the five types of capital assets are clearly differentiated as investment choices (financial and in kind) that various sectors of society – public, private and civil society – make in response to those threats. Threats are defined based on societal values and objectives. Defining what constitutes a threat requires that society engage in dialogue about long-term goals and vision through participatory democratic processes, a process that has a positive, self-reinforcing spin-off effect of further building the social capital necessary to engage in the objective-setting process (Rudd, 2000). Because resources are scarce, however, economic efficiency will always be an important factor in ecosystem-based fisheries management no matter what ultimate ecological, social or governance objectives are chosen as top priorities.

Using the IAD approach, it is possible to systematically consider and compare a variety of possible investment responses – from each of the public, private and civil society sectors – that would typically be overlooked using traditional fisheries management approaches. For example, besides standard investments in monitoring, control, surveillance and enforcement that top-down fisheries management departments typically focus on, private, public and civil society organizations can now consider investments in social networks and trust- and norm-building activities or recording and expanding local ecological knowledge as important investment activities that contribute to achieving broad ecosystem-based fisheries management objectives. While these investments are not new, the IAD framework provides a conceptual and theoretical link between ecosystem-based fisheries management 'best practices', on the one hand, and ecosystem, social, economic and governance outcomes, on the other hand, that has been lacking. This enhances our ability to conduct the directed policy experiments crucial for experimental ecosystem-based management.

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SAMENVATTING

Overal ter wereld ondervindt de tropische visserij tegenwoordig de grootste problemen vanwege het feit dat zowel de meeste waardevolle, bij riffen voorkomende vissoorten als de habitat waar ze afhankelijk van zijn aan het verdwijnen zijn. De complexiteit van de visserij in de rifgebieden en het gebrek aan de voor het visserijbeheer benodigde hulpmiddelen in ontwikkelingslanden hebben samen geleid tot een afname in de potentiële effectiviteit van regels die maar voor één soort van toepassing zijn. Daarom wordt het over het algemeen aangenomen dat de *mariene protected areas* (MPA's), of zeereservaten, het voornaamste hulpmiddel vormen voor een op ecosystemen gebaseerd visserijbeheer. Men gaat hierbij van uit dat men met de MPA's in staat zou zijn verschillende ecosysteembeheersdiensten tegelijk te kunnen beschermen of verbeteren en er worden vaak argumenten naar voren gebracht dat ze kost-effectiever zouden zijn dan andere beheersmogelijkheden omdat men er gemakkelijk toezicht op zou kunnen houden en ze op naleving zou kunnen controleren.

Het is mijn doel om met dit proefschrift de inzichten uit de ecologie, de economie en de politieke wetenschappen zodanig met elkaar te integreren dat het ontwerp en de evaluatie van experimen-ten met het, met behulp van MPO's en de devolutie van visserijregelementen naar de lokale gemeenschap, op ecosystemen gebaseerd visserijbeheer vergemakkelijkt wordt. In het eerste, theoretische gedeelte van dit proefschrift wordt de rol van het sociale kapitaal – de normen, netwerken en de bepalende infrastructuur waarmee de wederkerig voordelige acties en de reductie van de transactiekosten van het beheer vergemakkelijkt worden – benadrukt in experimenten met een op ecosystemen gebaseerd visserijbeheer.

Na de introductie in hoofdstuk 2 van de rol van economische analyse ontwikkel ik in hoofdstuk 3 de connecties van het sociale kapitaal met een op ecosystemen gebaseerde benadering voor het visserijbeheer. De argumenten voor het gebruik van MPA's worden meestal gebaseerd op pragmatische kostenbesparingsvoordelen wat betreft het op peil houden van de reserves en de toezicht op de naleving. MPA's vormen echter maar één mogelijkheid in een hele serie beleids-opties voor conservatie en andere doelstellingen van het visserijbeheer en die nog vrijwel nooit de focus zijn geweest van een rigoureuze beleidsanalyse waarbij gelet wordt op het totale bereik van de economische kosten en voordelen, zoals bijvoorbeeld de transactiekosten van het beheer. Als men geen geloofwaardige analyses uitvoert bestaat er een gevaar dat het huidige enthousiasme voor mariene reservaten af zou kunnen nemen als ze economisch gezien niet opleven naar hun veronderstelde potentieel.

Om de sociale drijfveren te ondergronden die tot veranderingen in het milieu leiden moeten we specifiek de rol bijhouden die gespeeld wordt door sociale interacties, de ontwikkeling van gedragsnormen en de institutionalisering van regels en normen en de ontwikkeling van het 'sociale kapitaal'. In hoofdstuk 4 wordt het nut aangetoond van de sociale kapitaal-theorie door de verbanden te benadrukken die bestaan tussen de menselijke besluitvorming op zowel individueel als collectief niveau en de sociale visie die een belangrijk onderdeel vormt van het op resultaten georiënteerde en op ecosystemen gebaseerde visserijbeheer.

Het IAD-stelsel is een robuust stelsel waar uitvoerig gebruik van wordt gemaakt voor het ontwerpen van beleidsexperimenten en het empirisch testen van theorieën en modellen waarmee een verband wordt gelegd tussen ecologische en economische systemen, instellingen en de duurzaamheid van de gemeenschappelijke voorraad hulpmiddelensystemen. In hoofdstuk 5 ontwikkel ik een aangepast IAD-stelsel dat als platform dient voor het ontwerp van en de controle op een experiment met een op ecosystemen gebaseerd visserijbeheer-beleid. Met een institutionele aanpak van het visserijbeheer wordt het gemakkelijker gemaakt de belangrijkste afsteekpunten onder de loep te nemen, zoals bijvoorbeeld de aannamen waar duurzaamheid uit bestaat en hoe de markt, de regering en de maatschappelijke organisaties gebruik maken van strategische investeringen in kapitaalbezit en -instellingen met het doel een toekomstige duurzaamheid te bereiken. Het feit dat er nadruk wordt gelegd op kapitaalbezit, zoals bijvoorbeeld het sociale kapitaal, houdt de aandacht gericht op het relatieve voordeel van alternatieve investerings-mogelijkheden in beleidsexperimenten.

Ik laat in het tweede deel van dit proefschrift het nut zien van een institutioneel georiënteerde en op ecosystemen gebaseerde benadering voor het visserijbeheer door het geval onder de loep te nemen van de bescherming en het visserijbeheer van de Nassau tandbaars (*Epinephelus striatus*) in de Turks & Caicoseilanden. De Turks & Caicoseilanden is een dun bevolkt eilandenstaatje aan de zuidkant van het Bahamaanse archipelagio. In hoofdstuk 6 wordt een overzicht gegeven van de visproductie en -handel in de Turks & Caicoseilanden in de afgelopen 100 jaar.

In hoofdstuk 7 wordt de lokale vraag naar vis uit de rifgebieden behandeld die van de ambachtelijke kustvisserij in de TCE afkomstig is. De lokale visserijsector wordt beschermd door invoerrechten van bijna 40% op ingevoerde visproducten. Aangezien ingevoerde visproducten daardoor duurder zijn zou dit in theorie moeten leiden tot een toename in de vraag naar lokale visproducten. In dit hoofdstuk wordt een gezamelijke-gecombineerde opiniepeiling gebruikt in de restaurants in de TCE om uit te vinden welk effect veranderingen in de invoerrechten hebbben op de vraag naar verse en lokale vis in vergelijking met de vraag naar zowel ingevoerde diepvriestandbaars als andere producten die potentieel de plaats in kunnen nemen van tandbaars. Ik heb ontdekt dat de vraag naar Nassau tandbaars significant verhoogd wordt door de invoerrechten en waardoor de visserijdruk op de Nassau tandbaars ook sterk verhoogd wordt.

De lokale vissers kunnen voor hun multi-soort ambachtelijke visserij allerlei verschillende soorten vistuig gebruiken en zich binnen een etmaal in hoog tempo van plek naar plek en van soort naar soort verplaatsen. Men zegt dat de TCE-vissers een hogere sociale status en meer prestige bereiken als ze hun vispogingen op een economisch onvoorspelbare wijze uitvoeren. Dat argument wordt in hoofdstuk 8 aangestipt en er wordt uiteengezet dat een logische conclusie voor die denkwijze – namelijk dat men de voorkeur zou geven aan de kreeftvisserij boven de conchvisserij vanwege de sociale prestige – hier niet geldig is omdat het besluit ergens te gaan vissen in de rifgebieden afhangt van de omvang van de vangst. Hiermee is, weliswaar indirect maar toch duidelijk bewezen dat de visserij in de TCE gevoelig is voor lokale marktprijzen en dat de mede door het toenemende toerisme en de door de regering ingestelde invoerrechten toenemende vraag naar de Nassau tandbaars tot een vergrootte Nassau tandbaars-visserijactiviteit leidt.

Aangezien men in de visserij vaak op grootte selecteert verwacht men dat er met de instelling van MPA's een toename zal plaatsvinden van zowel de gemiddelde grootte als de overvloed van de geëxploiteerde vissoorten. Door de toename van de gemiddelde grootte en/of de overvloed van de beschermde vissoorten in de MPA's hoopt men ook op een niet-onttrekkend economisch voordeel voor toeristen. In hoofdstuk 9 ga ik na welke voorkeur scubaduikers hebben als ze de Nassau tandbaars komen observeren en wat de bijkomstige voordelen voor duikers zijn wat betreft de grootte en de overvloed van de Nassau tandbaars, de grootte van de duikgroep en de duikprijs in de TCE. Uit simulaties van marktaandelen met behulp van de resultaten van een gezamelijke-gecombineerde opiniepeiling is significant gebleken dat de duikers voorkeur hebben aan plaatsen met meer en gemiddeld grotere Nassau tandbaars. Daaruit volgt dat de Nassau tandbaars een niet-onttrekkend economisch voordeel betekent voor duikers, wat van belang is voor de economische levensvatbaarheid van MPA's in de toeristenconjunctuur van de TCE.

In hoofdstuk 10 geef ik de resultaten weer van een ecologisch onderzoek in het Nationale Zeepark *Admiral Cockburn.* Het onderzoek bestond uit een studie in Zuid Caicos naar de invloed van de visserijdruk en de habitat op de biomassa en vangst per eenheid inspanning (CPUE) van drie soorten geëxploiteerde en in de rifgebieden voorkomende vis. Het bleek dat zowel de gemiddelde grootte als de dichtheid en de biomassa van de zwijnslipvis groter waren in het kleine (4 km^2) mariene reservaat dan op de riffen eromheen aar men normaal vist. Dit was ook het geval voor de *White Margate* (Haemulon album). De CPUE van de zwijnslipvis was omgekeerd evenredig met hoe ver ze van het centrum van het reservaat verwijderd waren, waaruit blijkt dat de lokale visvangst gunstig beïnvloed wordt door de nabijheid van het reservaat. Aan de andere kant had de visserijdruk kennelijk geen invloed op de Nassau tandbaars. Het zou kunnen dat grotere vissoorten of vissen die normaal migreren om kuit te schieten, zoals de Nassau tandbaars, zich over te grote afstanden verspreiden om effectief beschermd te worden door een klein marien reservaat. Hoewel kleine reservaten dus niet alle soorten vis beschermen kunnen we wel de conclusie trekken dat ze leiden tot een toename in de biomassa van de kleinere standvissoorten en dat ze dus erg goed zijn voor de bescherming en het beheer van juist die vissoorten.

In hoofdstuk 11 gebruik ik een *Institutional Analysis and Development*-stelsel voor de ordening van een vergelijkende vaststelling van de beleidsopties voor de bescherming en de TCE van de Nassau tandbaars. Uit de resultaten blijkt dat de Nassau tandbaars waarschijnlijk het beste beschermd kan worden met behulp van een verbod op de commerciële vishandel speciaal voor toeristenrestaurants, aangezien het op die manier waarschijnlijker is dat zo'n beleid (1) met succes nagevolgd, en (2) gemakkelijk een tijd lang volgehouden kan worden. Visserijbeheerders besluiten vaak maar al te gauw dat op de consument gerichte handelsbeperkingen geen mogelijke hulpmiddelen zijn voor een op ecosystemen gebaseerd visserijbeheer, maar hier wordt aange-toond dat handelsrestricties in sommige gevallen de meest veelbelovende optie vormen voor de bescherming van bedreigde vissoorten van de door de vraag opgedreven visserijdruk.

Hoewel ik me in deze studie concentreer op slechts één klein eilandenstaatje geloof ik dat de resultaten – dat er voor zowel de privé-sector als de regeringsorganisaties substantiële redenen bestaan om zich te verzetten tegen de mariene reservaten – van groter nut zijn voor de algemene discussie over het gebruik van mariene reservaten voor het beheer van de tropische visserij en de bescherming van tropische vissoorten. Mariene reservaten worden over het algemeen gezien als een kost-effectief allround hulpmiddel voor de verbetering van de visserij en de bescherming van bedreigde vissoorten. Uit mijn resultaten blijkt echter dat er beleidsalternatieven bestaan, zoals in dit geval bijvoorbeeld een verbod op de commerciële handel in Nassau tandbaars voor toeristenrestaurants, die naar grote waarschijnlijkheid veel gemakkelijker in te voeren en substantieel kost-effectiever zijn dan mariene reservaten.

Een op vraag gebaseerd beleid mag in het op ecosystemen gebaseerde visserijbeheer niet onderschat worden. In gevallen waarin het lokale sociale kapitaal te kort komt kan het zelfs leiden tot groter succes in het bereiken van beschermingsdoeleinden en meer kost-effectief zijn dan slecht-beheerde mariene reservaten of 'parken van papier'.

PERSONAL BACKGROUND – MURRAY ALAN RUDD

Born 22 October 1958, Mindemoya, Canada

Education

Mansholt Graduate School, Wageningen University, The Netherlands. Ph.D. Program in Social Science (1999-2003)

Indiana University, Bloomington, Indiana. Joint Doctoral Program in Political Science and Policy Analysis (1998-1999)

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Professional Background

Senior Economic Analyst, Fisheries and Oceans Canada – Maritimes Region (2002-present)

Center Director, School for Field Studies, Center for Marine Resource Studies, Turks and Caicos Islands (1999-2000), and Adjunct Lecturer, Boston University.

Resident Faculty (Environmental Policy), School for Field Studies, Center for Coastal Studies, Canada (1998), and Adjunct Lecturer, Boston University.

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General Manager, Saga Seafarms, Canada (1986-1989)

Manager, Whitehorse Rapids Salmon Hatchery, Canada (1984-1986)

Publications

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