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Coral Reefs and People

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CORAL REEF CONSERVATION

The term coral reef often produces images of warm climates, crystal clear turquoise waters, golden beaches and a huge array of colourful fish and other species. They are well renowned for their beauty, biological diversity and high productivity and it is this latter characteristic that makes them critical to the survival of tropical marine ecosystems and to the welfare of millions of local peoples (Berg *et al*., 1998; Hoegh-Guldberg, 1999).

Many different species make up a coral reef but the dominant feature is reef-building corals. These *hermatypic* corals are colonial and made of many millions of small anemone-like individuals interconnected to form the characteristic colonies that form the backbone of a coral reef. Hermatypic corals secrete calcium carbonate skeletons that are laid down on top of dead coral. The success of hermatypic corals is largely due to their symbiotic relationship with microscopic algae (zooxanthellae) that live as part of a mutualistic relationship within the surface layers of their coral host. The zooxanthellae are photosynthetic and provide corals with up to 98% of their energetic requirements. Therefore, corals are primarily light dependent, a trait that restricts corals to the tropical and sub-tropical belts where temperatures are constantly warm and light intensity constantly high. Corals are also, in part, heterotrophic, and are dependent on their anemone-like tentacles to feed on zooplanktons mostly during the night.

The secretion of calcium carbonate by corals, as well as a myriad of other algae and animal species, produces a highly complex three-dimensional biogenic structure that enables many different species seemingly to coexist but utilise their environment slightly differently, thereby reducing competitive stress. Corals are therefore key to the diversity of reef-based systems. Coral reefs represent one of the biggest natural structures on the planet and are home to more species than any other marine system. They are vitally important for the socio-economic welfare of hundreds of millions of people and provide a variety of services. For example, coral reefs act as natural seawall defence and buffer the coastline from the erosive forces of large storms and tidal surges. Never has this fact been more evident than when examining the impacts of the 2004 Asian Tsunami and the recent spate of Caribbean hurricane activity.

Their biological diversity alone makes coral reefs exceptionally important, but this coupled with the other functions they provide such as provision of food, bioactive compounds, other economic benefits, coastal protection and geochemical cycling results in coral reefs being one of the most important ecosystems on the planet. However, the majority of coral reefs around the world are overexploited and threatened, with up to 60% showing severe signs of decline (Wilkinson, 2002). These disturbing facts are exacerbated by the fact that pressures are likely to double over the next 50 years as populations expand, coastal zones become more developed and other anthropogenic-induced stresses and natural phenomena continue to degrade reefs around the world.

Realisation of the threats that face coral reefs is the starting point for effective strategic conservation management, which needs to be both regional as well as global in perspective. Realisation of the full economic potential of reef systems can also be a very positive outcome of trans-disciplinary conservation research and a powerful persuasive argument for the need for, and benefits of, active conservation and sustainable managed practices. Conservation actions and strategies need to be inclusive in their approach and consider both local and global issues. Direct management action is urgently required if reef resources are to be available, at least in their current state, for future generations. We face a critical time in the world of coral reef conservation with coastal populations rising, climatic conditions changing and the demand for reef-based resources increasing. Predictions suggest that up to 70% of coral reefs could be lost by 2050 and some conservation scientists have suggested that coral-dominated reefs could completely disappear over the next 50–100 years.

The loss of coral reef systems will have a major impact on over half a billion people and will be devastating for many maritime states and countries. Urgent collective action is required but first we need detailed knowledge of reef systems, their biological function and their realised economic and intrinsic value as well as their response to past, present and future conservation management strategies. Numerous studies have demonstrated that the threats to reefs are similar the world around, but the methods required to protect reefs need to consider the traditions and requirements of local user groups as well as take into consideration the global perspective.

Each dependent community has different requirements, ranging from daily subsistence needs to those related to a multi-billion dollar tourist industry. It is essential that we take into consideration these stakeholders, and it is most likely that different approaches, albeit with similar project aims, need to be considered and implemented if we are to be successful in conserving reef-based systems. A truly cross-disciplinary approach is needed to protect reefs as is an understanding of local user group requirements and attitudes towards conservation. Without such approaches it is highly unlikely that restrictive management strategies will be successful.

THE IMPORTANCE OF CORAL REEF-BASED SYSTEMS

Reefs contain more species than any other marine system. For example, approximately 25% of all marine species identified are reef associated despite coral reefs making up less than 1% of the Earth's surface. Their topographic diversity, intermediate levels of disturbance, tight resource partitioning, specialisation and high levels of competition are just some of the reasons why reefs are so diverse containing representatives from over 95% of all animal phyla as compared with only 25 to 30% characteristic of tropical rainforests. This biodiversity has both intrinsic and extrinsic values but the true biodiversity of reefs is unknown and it has been suggested that currently we only know about 10% of the true species richness with estimations ranging from 600,000 to 9 million reef-associated species worldwide. With the world facing accelerated species loss (estimated as being up to 1000 times the background rate), factors that significantly contribute to the degradation of reef systems will have global impacts with respect to the Earth's total species and gene pool.

We know that coral reefs are among the most diverse ecosystems on Earth, but the level of diversity held within reef systems is geographically dependent. The greatest coral reef diversity is found at the Indo-Pacific interface, and diversity decreases as you radiate away from this biodiversity hot spot. For example, around Raja Ampat, Indonesia, there are over 600 of the world's corals (out of a total of 800), 400 species are found in the Philippines, about 350 in the northern Great Barrier Reef, about 250 in Fiji and less than 50 in Hawaii. A second centre of biodiversity exists around the western shelves of the Atlantic, and in particular the Caribbean but here only around 60 species of coral can be found. On a more regional scale, patterns of diversity are affected by localised environmental conditions, for example, exposure levels as well as stochastic environmental features (Dornelas *et al*., 2006) and human practices.

THREATS TO CORAL REEFS

Numerous factors result in the loss of species from a coral reef and the reduction of its physical as well as biological integrity (Graham *et al*., 2006). Generally speaking we can classify the threats facing reef-based systems into *natural phenomena* and *anthropogenic stressors*. The effects of these phenomena and stressors range from negligible to catastrophic. In part, natural phenomena impacting reefs help maintain biological diversity by reducing dominance and producing temporal and spatial patchiness. Furthermore, coral reefs display a good ability to adapt to short-term natural catastrophic events through an array of physiological and behavioural plastic responses. However, coral reefs are not well adapted to survive exposure to long-term stress and thresholds exist beyond which communities can collapse, resulting in a long-term loss of diversity and productivity (and therefore natural capital). Such shifts in community structure are often termed "phase shifts" as community changes can be dramatic, for example, from a diverse and productive coral-based system to one that is overgrown and dominated by algae and therefore characterised by having reduced physical as well as biological complexity. One of the problems of course is in identifying these thresholds and introducing appropriate measures to prevent or reverse the trend.

Anthropogenic stressors, examples of which include agricultural and industrial runoff, increased sedimentation from land clearing, human sewage as well as direct physical damage, tip the balance in favour of species loss rather than maintenance and have had devastating effects around the world. It is also becoming increasingly difficult to distinguish between the natural and anthropogenic factors as the effects of natural factors could be accelerated by anthropogenic stressors. For example, coral bleaching is in part a natural response to environmental stress, in particular high seawater temperatures and high light intensity. Coral bleaching events are therefore related to prevailing climatic conditions and will be greatly affected by the predicted human-accelerated global climate change. In recent decades bleaching episodes have increased in their frequency and intensity with more than 16% of the world's coral reefs being destroyed in the last big bleaching event during the 1998 El Niño (Wilkinson, 2002).

It seems most likely that future El Niño events will have an even greater effect with predictions suggesting that up to a quarter of all coral reefs could be killed during the next one. Although we are now starting to understand the fundamental mechanisms that cause bleaching (Smith *et al*., 2005), we are far from suggesting possible remediation actions and our best strategy at present is to ensure that other factors that stress reef systems, which can be more easily managed, are done so, giving reefs the best possible chance to respond favourably to changing climate and environmental conditions.

Other "natural" factors that result in reef degradation include large storms and hurricanes. It has been shown that the growth and recruitment rate of corals are negatively impacted by storm damage (Crabbe and Smith, 2003; Crabbe *et al*., 2004). Intermediate levels of disturbance help maintain reef diversity by decreasing competitive exclusion and enhancing patchiness. However, increased frequency and intensity of storms may once again tip the balance beyond a species maintenance function to one that greatly reduces the physical and biological integrity of reef systems.

Finally, with respect to "natural phenomena", coral diseases have recently increased in their frequency of occurrence. "Disease" is a phenomenon common to all biological populations and communities and once again probably helps maintain biological diversity in the long term. However, it seems that the frequency of disease occurrence is increasing, which has been attributed to increases in organic pollution (i.e. sewage) and land pollutants washing on to reef systems. So once again we see that human impacts are adding to the weight of natural phenomena that are impacting reefs the world around. It is very difficult if not impossible to start to tackle the natural phenomena; the best alternative available to managers is to identify and alleviate the human-induced pressures thereby giving reefs a better chance of withstanding natural, degrading phenomena.

Many studies have examined anthropogenic stressors and their impacts on reef systems. These include: overexploitation of both fin and shell fisheries, the use of non-sustainable and damaging fishing practices, exploitation of coral rock and physical destruction of reef systems, coastal development leading to loss of feeder habitats including mangrove and seagrass beds, and in terms of coastal terrestrial environments, deforestation and land redevelopment. The relative importance of each of these factors varies dramatically around the world and is highly dependent on the characteristics of local people as well as the geographic location. For example, the majority of coral reefs within southeast Asia are overexploited in terms of fisheries and the use of destructive techniques such as blast fishing and poison fishing represents one of the major problems. In parts of Central America reefs such as the Honduras Bay Island reefs, part of the Meso-American Barrier Reef System, have been negatively impacted by the establishment of large plantations on the mainland, whereas off the coast of Sinai within the Red Sea, reefs are threatened by dive-based tourism.

In the majority of regions of the world coral reef habitats are overfished and/or overexploited for one reason or another. Coral colonies and brightly coloured charismatic reef fishes and invertebrates are collected for the growing aquarium and jewellery trade. Currently, over 1000 species are taken from coral reefs for the aquarium trade and, although tightly regulated in places, it is not in others, and a black market trade in species export exists. Also, quotas as well as collection techniques vary and can have major consequences for the integrity of reef systems. For example, cyanide poisoning is often used in parts of southeast Asia to stun fish either for the pet trade or the live food trade. It is extremely difficult to detect which species are caught legally as compared to species caught in an unsustainable and unregulated manner.

Probably the major threat facing reef systems is overextraction in general. As fishing techniques advance (e.g. become more efficient) and access to these techniques increases, overexploitation of an already exploited system is likely to increase.

This coupled with increases in local population sizes is likely to result in a doubling of current exploitation rates over the next 50 years or so. Fisheries pressure and the resulting impacts vary greatly with the techniques used. Some techniques such as blast fishing are completely unsustainable but although being highly illegal, they are commonly used in some parts of the world.

Overfishing in general also represents a huge problem for coral reefs. Reefs represent a limited resource in terms of the fish they recruit. Non-targeted fisheries techniques and increased fisheries effort due to technique or the actual number of fishermen could result in the loss of a single species and the collapse of a coral reef system. For example, removal of grazing fish species and hence the reduction in grazing pressure can result in fast-growing algae species overgrowing a reef. Also, non-size selective fisheries techniques, such as the many forms of fish fences common around Indonesia, can dramatically reduce recruitment of new fish biomass into the system and once again result in community as well as population collapse.

The term pollution is of course very general, and in terms of coral reefs the biggest threats really relate to those factors reducing water quality. Pollutants can be in the form of fertilisers or sewage which can elevate otherwise limiting nutrients. Normally nutrient levels are very low and this characteristic gives corals the advantage over the normally fast grown algal species. An increased nutrients status can therefore lead to algal overgrowth (Goreau, 1992). This will lead to reduced light availability to reef-building corals and overall a reduction in physical complexity. Reduced light is also common in areas where there is large run off from the land. Terrestrial soils can limit light availability to reefs and increased input of sediment is common when coastal lands become developed through deforestation or manipulation of the coastal fringe. The removal of mangrove and seagrass beds is common practice and both these habitats are connected with reef systems. They constitute important nursery habitats for juvenile coral reef species whilst also entrapping sediments (marine and land based) which may otherwise impact adjacent reefs.

Greater sediment load coupled with increased deposition rates can greatly alter the dynamics of a coral reef. The various species of coral respond differently to sediment load and deposition. At the individual level, colony corals demonstrate decreased growth rates (Crabbe and Smith, 2002, 2005) and changes in their colony structure (Crabbe and Smith, 2002, 2006). A reduction in species diversity is common and eventually, when natural rates of erosion are greater than rates of calcification, coral reef systems can completely collapse. Any factor that increases sediment input on to reefs needs to be carefully managed if there is not to be a decrease in coral diversity. Ironically some of these processes have increased due to the aesthetic value of reefs, for example tourism development, which might inadvertently be impacting their most important commodity.

Of course, the level of impact and the potential for recovery depends largely on the method and scale of the exploitation (or pollution). Some exploitative techniques are highly damaging and examples are the aforementioned blast fishing and coral mining. Trends in blast fishing have changed and now it is common practice in some parts of the world to use multiple bombs on, or "carpet" bombing of, reef systems. Many centuries of coral growth can be destroyed and although recovery rates do vary depending on local conditions, impacted reefs can take between 50 and 100 years to recover. Similarly, the use of live coral in the mining trade also reduces the carrying capacity of reefs and is similarly unsustainable.

Coral reefs represent a huge resource of natural goods and services. They are and can be exploited; however, overexploitation and non-sustainable resource utilisation could have dramatic effects on the socio-economic welfare of many millions of people. Protection and active management are required and strategies should revolve around a multidisciplinary approach. In the majority of cases, in one way or another, many of the anthropogenic factors that negatively impact a coral reef stem from economic incentives. How are we to protect reefs for future generations, if we do not provide alternatives to local communities for loss of income derived from the required restrictive conservation policies? It is imperative that we take into consideration the economic characteristics of local stakeholder communities when producing management strategies that are aimed at being (a) biologically successful and (b) socially acceptable (both in terms of agreement and compliance).

Consequently, it is extremely important that we gain a full understanding of the economic value of coral reefs. It is important that we understand not only the realised value of reef systems but also their potential value. Only then can we start to study and suggest alternatives rather than additional income streams that are equivalent in economic benefit but also serve to reduce the pressures on reef systems. Such knowledge is essential if we are to ensure future sustainable coral reef exploitation and also if we are to produce workable management strategies that are agreeable to local as well as global stakeholders. Of course this requires a very much localised approach but lessons learnt from other areas can be cross-transferred to other dependent communities.

CORAL REEF ECONOMICS

Coral reefs represent an important economic resource with benefits accruing to local and global economies (Cesar, 2000). Locally, reef fisheries are a vital source of protein for millions of people, reef-related tourism is a major foreign currency earner, and reefs provide natural coastal protection from wave action and potential storm damage. On the global scale, reefs are valued for their role in the carbon and calcium cycles, their inherent existence, the consumer surplus enjoyed by Scuba divers, and for their bioprospecting potential (Spurgeon, 1992; Pendleton, 1995; Cesar, 2000).

Constanza *et al*. (1997) estimated the world's marine and terrestrial ecosystem services to be worth US$20,949 and 12,319 billion per year, respectively. Within the marine sector, it was estimated that coral reefs alone are worth US$6075 ha^{-1} per year, which equates to US$375 billion per year on a global scale. More recently it has been estimated that coral reefs provide annually around US$30 billion in net benefits in goods and services to world economies; these goods and services include fisheries, coastal protection, tourism and recreation, and biodiversity (Cesar *et al*., 2003).

Despite the numerous estimates on the value of reef-based systems the full economic potential of reefs is largely unrealised (Spurgeon, 1992). It is often the case that many of the world's ecosystem services and natural capital are given too little weight in policy decisions because they are not fully captured in commercial markets or adequately quantified (Costanza *et al*., 1997). Additionally, many of the world's richest ecosystems and poorest people are found together in the tropics so it is not surprising that human aspiration for improved living conditions often clashes with conservation and biodiversity objectives (Randall, 1991). This apparent conflict of interest between economic development and the environment has created worldwide problems. In 1983 the United Nations appointed an international commission to propose strategies for "sustainable development" defined as "development which meets the needs of human well-being in the short term without threatening the local and global environment in the long term." To achieve this aim, some form of accounting for national and international natural resources is required (Spurgeon, 1992).

Additionally, the realisation at the local scale that sustainable management of coral reefs can serve to increase their economic worth whilst also conserving biodiversity would go a long way towards the possibility of successfully implementing sustainable practices. Furthermore, the identification of viable (i.e. economically comparable) and sustainable alternative (not additional) income streams for reef-dependent communities could significantly help decrease current pressures on reefs. An understanding of the realised as well as potential economic value of reef-based systems in terms of the local user groups could help drive forward conflict resolution and help address the balance between exploitation and environmental degradation.

The role of economics in ecological protection

The disciplines of ecology and economics both have important contributions to make to the identification and solving of the global problem of overexploitation and degradation of natural capital (Perrings *et al*., 1992). Ecology can be used to establish the amount and availability of natural resources and habitats, and identify and monitor population dynamics and system changes. Economics has a major role to play in explaining resource use and degradation, measuring its impact, and designing policies to combat degradation (Pearce and Mäler, 1991; Dixon, 1997). The explanatory role of environmental or ecological economics is that we need to understand why resource degradation occurs and how economic mismanagement through wrong pricing, ill-defined property rights and incentive structures, contributes to environmental loss. Environmental economics can also be used within a policy development role to devise the best practicable solutions for sustainable development (Pearce and Mäler, 1991).

It is agreed by many environmental economists and ecologists that natural resources should be valued in economic terms when decisions involving the loss or preservation of them are made (Dixon, 1986). The main argument for this is that without true evaluation, the existence of these resources will be given zero value suggesting no net loss if they decline (Green and Tunstall, 1991). Those who oppose the economic evaluation of natural resources often misrepresent economics as being about money, which it is not. Money is simply the means of measurement to compare the relative values of different goods and services (Pearce and Turner, 1990). In economics all values given to goods are subjective and the environmental economist is seeking to derive a method of measuring the values different individuals place on different goods so that they can be compared (Green and Tunstall, 1991). Money is used as the basis for comparison because an individual's subjective values are unobservable and often incomparable;

for instance, it is difficult to measure one person's enjoyment of a natural resource as compared to another person's (Robbins, 1935).

The environment is often undervalued. Environmental benefits include many non-marketed goods and services which have no readily available monetary values. Additionally, natural habitats have off-site benefits which occur a distance away from the habitat. For correct valuation to be made, non-marketed goods and services and off-site benefits must be accounted for and recent advances in valuation techniques mean that more of these can be quantified in monetary terms providing more comparable and influential information (Spurgeon, 1992).

The role of economics in reef management

Approaches to natural resource management have been varied and continue to be adapted (Figure 35.1). In the past, natural resource and protected area management focused on understanding and managing ecosystems from a biological viewpoint with very limited user group consultation. Currently, however, management strategies have started to incorporate the wider social and economic factors, specific user groups are now actively encouraged to participate in various management actions, and cost–benefit analyses, environmental impact assessments and bio-economic models are used to support decision-making. It is thought that in the future, financial, business, legal and ethical factors will play an increasingly important role in natural resource management (Spurgeon, 2001). User groups will be more involved in the decision-making and action process, with the hope that this will allow empowerment and perceived local ownership of resources which in turn leads to a sense of pride and increased local protection of resources. With this changing and dynamic management approach, understanding the existing and potential values of coral reefs is essential to a successful outcome that maintains ecological wealth and develops sustainable utilisation.

There is widespread interest in coral reefs and their conservation, but at the same time there is also a lack of resources to provide even minimal levels of management. Part of the reason for this is the difference between economic values and monetary prices, and that people and governments have been responding to monetary price signals. Additionally, local user groups are largely unaware of the potentially increased monetary benefits of a healthy reef in the long term. Economics can be used to explain this with total economic value (TEV) calculation being one of the most useful tools. This economic value of an ecosystem is often defined as the total value of the goods and services the ecosystem provides (Cesar, 2000).

Assessing the total economic value of coral reefs

Many of the benefits associated with coral reefs are not exchanged in markets, are hard to value and therefore have often been ignored or grossly underestimated. Complete and detailed quantitative data on coral reef ecosystems are rarely available so decisions are often made without these or the involvement of local user groups (Fernandes *et al*., 1999). One problem with environmental valuation is identifying the various components of value and attaching monetary prices. For this the calculation of a TEV can be a very useful tool. It is calculated as a measure of the current

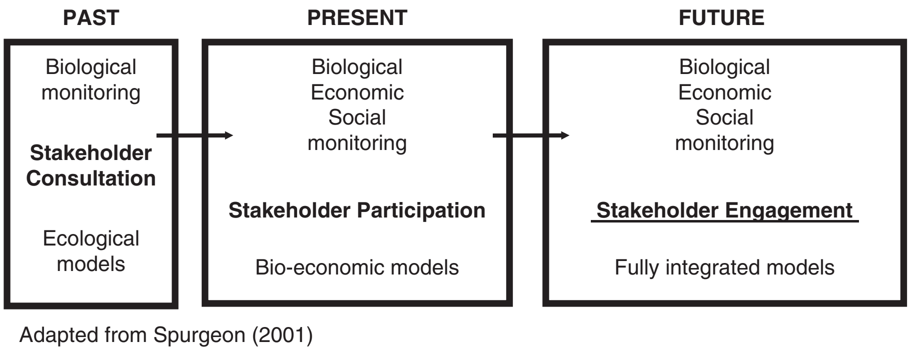

Figure 35.1 Changing approaches to natural resources management. Understanding the full current and potential values of coral reefs is critical to a successful outcome in a changing management approach

Table 35.1 Total economic value (TEV) of coral reefs

Use values			*Non-use values*		
Direct (extracted)	*Direct (in situ)*	*Indirect*	*Option*	*Existence and bequest*	*Intrinsic*
Fisheries Aquarium trade Curio trade Pharmaceuticals Construction materials	Tourism Research Education Recreation Culture Religion	Biological support Coastal protection Global life support	Maintenance of system for future use	Knowledge of system existence and continued existence for enjoyment by future generations	Biodiversity Species richness Existence with no human use

TEV = use values + non-use values.

economic worth of any resource and encompasses all direct, indirect and non-use values of that resource (Table 35.1). Direct use values relate to actual use or the goods (extractive and non-extractive) provided by the resource in question (e.g. fisheries or recreation); indirect use values are the functional uses, or services, provided by a resource (e.g. coastal protection); and non-use values represent the existence, bequest and option values, as well as the intrinsic value of natural resources and the biodiversity they support (Bateman *et al*., 2002). The economic information contained in a TEV calculation can be very powerful in making the case, to local user groups and decision-makers in particular, for the benefits of protecting and managing coral reefs sustainably (Dixon, 1997). However, it is important to note that, because some benefits of coral reefs cannot be quantified, TEV provides only a minimum estimate for their "true" value (Spurgeon, 1992).

Direct use valuation

Direct use values include both *extractive* (e.g. fisheries, aquarium and curio trade, pharmaceuticals, construction materials) and *non-extractive* (e.g. tourism, research, education, social value) uses. The direct extractive goods are, in most cases, marketable commodities, hence their economic value is relatively simple to calculate utilising existing market prices (Spurgeon, 1992).

The potential economic value of direct harvesting (*extractive uses*) can be calculated using cost–benefit analysis (CBA) where all costs and financial benefits associated with the harvesting and sale of products are taken into account. Due to natural fluctuations in "catch", variable market prices and financial outgoings, CBA is best calculated over long periods (Spurgeon, 1992) which results in the production of economic productivity values. Changes in productivity will be reflected in the monetary values calculated and may be indicative of environmental/ecological changes. It should be noted, however, that the associated costs and benefits of harvesting any product are difficult to determine and, in some cases, market prices may not reflect the true worth of a product.

Those goods collected purely for subsistence (self-consumption) have no direct financial value attached; therefore, other methods of valuation must be incorporated such as the replacement cost method, which utilises the market value of alternative potential replacement products, or contingent valuation (CV) which essentially values consumer surplus. CV uses hypothetical situations to put monetary values on non-marketed goods and services. To do this, people's hypothetical willingness to pay, or to accept compensation for, the use of a good or service is used (Spurgeon, 1992). For example, people might be asked how much they would be willing to pay for a certain reef product if they could not obtain it elsewhere themselves, or how much they would be willing to accept in compensation to discontinue use of that product. Where money is not perceived in the same way as in the Western world, the costless choice method can be used when the hypothetical bidding uses commonly exchanged goods (Dixon and Sherman, 1990).

Of the direct uses of coral reefs which are *non-extractive*, in most cases, tourism yields the greatest financial benefit, and many small island nations depend on reef-related tourism for their continued economic development. Revenues generated directly by reef-related tourism range from Scuba diving and marine park entrance fees to accommodation, food and travel costs (Spurgeon, 1992). The current value of tourism can be defined using the financial revenue (FR) approach, the contingent valuation method (CVM) or the travel cost method (TCM). The FR approach calculates the direct financial profits provided by reef-related tourism (Berg *et al*., 1998). In addition to the financial benefits of tourism, there is often a large tourist consumer surplus value, which constitutes the additional satisfaction gained by tourists in excess of what they paid for their trip. In many cases tourists visit specific reef sites for free or pay less for admission and equipment hire than they would be willing to pay (Spurgeon, 1992). To determine the extent of this additional value the CVM can be used. TCM could be employed, which assumes that the number of people travelling to a site is inversely related to the distance travelled to get there. If the number of people visiting the site and their travel costs are known, regression analysis can be used to estimate the value of that site to visitors (Spurgeon, 1992). As CVM includes social values, this approach could yield a higher value than that of FR or TCM (Berg *et al*., 1998).

Local communities gain additional benefits from reefs in a way similar to tourist consumer surplus; this social value may include cultural and heritage values representing the benefit to communities of traditions and customs that have evolved based on reef associations, or spiritual and aesthetic benefits. For instance, nomadic Bajo tribes of southeast Asia base their cultural traditions, worldviews and spiritual beliefs on the sea and its components. No quantifications currently exist for the extent of social value, but estimations could be made using an adapted CVM, surveying locals on their willingness to pay to maintain the reefs in their current condition (Spurgeon, 1992).

Indirect use valuation

The indirect values of coral reefs are the associated functional benefits, or services, provided by reefs which include coastal protection, bioprospecting potential, and biological and global life support. For example, the physical structure of coral reefs provides people with indirect economic benefits without requiring direct resource extraction. Reefs that fringe the shore provide a natural wave break and protect economically as well as environmentally valuable coastal habitats from storm damage (Berg *et al*., 1998). Reefs also provide large amounts of beach material essential for the preservation of tropical sandy shores. The coastal protection function of coral reefs can be described through the preventative expenditure approach (or replacement cost method), defined as the cost of replacing the coral reef with protective constructions, for example, groynes or underwater offshore wave breakers (Spurgeon, 1992); or by looking at the loss of property value, defined as the cost of land loss (price of lost land, buildings, roads, etc.) as a result of coastal erosion. This method also includes the loss of income resulting from lost land-use opportunities (e.g. agriculture) (Berg *et al*., 1998).

Bioprospecting can also be considered under the Indirect Use Valuation banner. There has recently been a sharp increase of interest in bioprospecting, that is, the search for naturally occurring bioactive compounds that may have commercial benefits and applications, for example, agricultural, chemical or pharmaceutical (Simpson *et al*., 1996). The hit rates, that is, the probability of finding a bioactive compound in a species, is much higher on coral reefs than in comparable (in terms of diversity) terrestrial systems, and as the true biodiversity of reefs is unknown, regulated bioprospecting has high economic potential.

A high degree of connectivity exists between reefs and other tropical marine (and coastal) systems, and this characteristic has economic implications. Connected habitats and systems need to be considered during economic valuation exercises. Accurate valuation of connected systems is difficult and highly localised but estimations of a rough value may be possible using a change in productivity approach, which is essentially the difference in value of a reef-supported economic activity with and without the reef. Alternatively, the biological support value is effectively the value of the supported activity multiplied by an estimated percentage dependence of that activity on the reef's presence, referred to as the "percentage dependence technique" (Spurgeon, 1992).

Non-use valuation

The non-use values of coral reefs refer to the perceived benefits of reefs outside of the value of any goods or services they provide us with. Non-use values include the existence type values placed on reefs by humans, which is measurable, and the intrinsic value of coral reef and associated organism biodiversity.

Existence, Bequest and Option

Existence and bequest are simply the values that people place in the knowledge that a natural resource or individual organism exists and that it will continue to do so for future generations to enjoy. "The proof that these values exist is apparent in the fact that people will pay money to charities such as 'Save the Whale' even though they know they are unlikely to ever experience a whale first hand" (Spurgeon, 1992). Existence and bequest values have not been determined for coral reefs but have been measured for individual species (Pearce and Turner, 1990) and for other ecosystems. The only method of valuation is the CVM using people's stated willingness to pay for an area or species to be preserved. Measurement of the existence and bequest value of coral reefs would require an extensive CVM survey that included local, national and international population representations (Spurgeon, 1992). The greater the quality and the uniqueness of the reef on a national and global scale, the greater its existence value will be. On the local scale, population size, level of income, education and environmental perception will also greatly influence the overall value (Spurgeon, 1992).

Biodiversity

The valuation of biodiversity in its own right, that is, as removed from any association with human welfare, is complicated, controversial and incredibly difficult to achieve, but methods for its valuation do exist. It is controversial because many people feel that it is wrong for humans to place a value on biodiversity as it is of infinite value, both to human welfare and in its own right. However, making public or private decisions that effect biodiversity implicitly means attaching a value to it. Hence, monetary valuation can be used as a democratic approach to make decisions about public issues, including biodiversity ones (Nunes and van den Bergh, 2001).

Biodiversity also has an instrumental value to human society and the maintenance of biodiversity has become a popular argument for ecosystem protection for this reason (Dixon and Sherman, 1991). There are potentially enormous welfare implications related to biodiversity loss, individual organisms have direct value in terms of consumption or production, and the combination of organisms, and their role in sustaining biophysical cycles, within a framework of ecosystems, makes them of indirect value in satisfying human needs for the services of those ecosystems (Perrings *et al*., 1992).

To value biodiversity, CVM is the most commonly used technique as it is able to identify and measure passive or non-use values. Existing monetary value estimates seem to give explicit support for the belief that biodiversity has a significant positive social value, but most studies lack a uniform clear perspective on biodiversity as a distinct concept separate from biological resources, that is, the instrumental value of biodiversity. Monetisation of benefits is possible but will always lead to an underestimate of the "real" value, since the primary (intrinsic) value of biodiversity cannot be translated into monetary terms (Nunes and van den Bergh, 2001). Therefore, any calculated value may be used to justify protection measures, but will constitute only a small portion of the total value of biodiversity (Gowdy, 1997).

Empirical work at Montego Bay, Jamaica, was carried out by Ruitenbeek and Cartier (1999) in order to estimate the net present value (NPV) of Montego Bay reefs. Estimated value for direct uses included tourism and recreation (NPV of US$315 million), fisheries (NPV US$1.31 million) and coastal protection (NPV US$65 million). Therefore, the value of the reefs for direct local uses was calculated at US$381 million or US$8.93 million per hectare of reef. However, these values fail to include the non-use benefits reaped by both local residents and visitors to the area. When estimating the value of non-use benefits, Ruitenbeek and Cartier (1999) assessed the willingness to pay of local residents and tourists. They found that for typical population characteristics, and using typical visitor profiles, the non-use benefits of Montego Bay biodiversity has a NPV of US$13.6 million to tourists and US$6 million to Jamaica residents. The above values amount to a NPV of approximately US$400 million for Montego Bay reefs, or marginal benefits of US$10 million per 1% of coral abundance improvement. However, this is likely to be a lower-bound estimate as no institutional arrangement currently exists for capturing biological prospecting values although estimated values stand at around US$7775 per species (Ruitenbeek and Cartier, 1999). Compared with reefs of central and southeast Asia, Jamaican reefs have relatively low biodiversity and low resource value as alternate food and medicine sources are available to local peoples. This emphasises the vast importance of even the less diverse reefs to human populations worldwide and the immense cost to society of replacing their goods and services if lost.

Much additional research is needed into natural resource and ecosystem valuation (Costanza *et al*., 1997). Economic assessments can be used to examine the extent of the benefits directly and indirectly associated with natural resource use (Bunce *et al*., 1999), but a key problem for policymakers is the lack of quantitative models and procedures to facilitate a comprehensive economic and ecological analysis, including identification, measurement and prediction of the effects of economic activity on the environment. Specifically, the degradation of coral reefs has not been extensively analysed in a framework amenable to economic policy analysis (Ruitenbeek *et al*., 1999). Hence, there is need for adequate valuation and dissemination of this information to policymakers, and if appropriate economic methods can be developed; the values generated could be used by policymakers to implement long-term economically and ecologically sound management practices.

One of the major reasons why conservation management initiatives fail to reach their goals is lack of compliance of local communities to management rules and regulations. There are three possible reasons for reduced compliance; (a) lack of awareness of rules and regulations; (b) disagreement between stakeholders; (c) stakeholders will be significantly and negatively impacted by new rules and regulations. The actual, perceived, or even just expected, economic losses to communities utilising reefs for livelihoods and subsistence is a major concern. So a key role of management should be to improve, or at the very least maintain, the economic status of local people, which means that management strategies must consider the impact on local people. Monitoring of economic status should be included (in addition to biological monitoring) to ensure no losses occur due to economically inappropriate management schemes. To do this, a simple series of economic performance criteria, that is, testable parameters in which changes could be used to measure the success of management, would be invaluable. Alternative income streams must also be made available where incomes begin to, or are expected to, be negatively impacted. Additionally, economics could, and should, be used as a tool to encourage conservation and sustainable utilisation efforts, as the economic benefits of a healthy reef system will clearly far outweigh those of an impacted and biologically limited system.

CORAL REEF MANAGEMENT

Highly diverse landscapes, the primary focus of conservation efforts, have remained occupied by human populations throughout history, and will continue to be into the future. Conservationists in the field have to make concise decisions on the realistic options available for the management of resources within these systems, aside from the speculation of management options widely discussed in theory (Alcorn, 1993). Despite long-term dependence on reef resources by coastal communities in particular, it has become evident that efforts to govern and sustain reef fisheries have frequently failed, especially in tropical fisheries where exploitation is intense and equipment diverse, and often even destructive. Transferable quotas are difficult to implement in artisanal fisheries whereby data are incomplete, reef ownership ill-defined and landings go unrecorded (Rudd *et al*., 2003). Thus, in a deviation from previous trains of thought, policymakers, conservationists and academics alike have recently come to consider the importance of community in resource management, as a consequence of the poor outcome of government efforts.

A community is perceived as a spatially defined unit with a well-defined social structure (Agrawal and Gibson, 1999). In the past, intrusive strategies and externally planned regulations excluding community stakeholders have overruled traditional systems of management often at great costs to local communities with limited conservation benefits yielded. Shortcomings have included government instability, monitoring and enforcement limitations, and inaccessible, unavailable or outdated science. Hence, there has been a recent surge of resources into community-management schemes from international agencies, including the World Bank, Worldwide Fund for Nature and The Nature Conservancy (Agrawal and Gibson, 1999).

Coral reef goods and services fall into the pool of common resources, with high subtractability by its users and difficulty in excludability of external non-authorised exploiters (Rudd *et al*., 2003). Management of coral reef systems, like many other global resource pools, has predominantly been in the hands of those harbouring its goods and services historically. For ninth-tenths of human existence on Earth, hunting and gathering pressures exerted on any ecosystem have had to be constrained for the survival of the community and wildlife population through to the present day. Local resource users, particularly on isolated oceanic islands, are often keenly aware of their own impact on local ecosystems, observing the differences between natural and anthropogenically derived fluctuations (Drew, 2005). Thus, self-management practices, originated and legitimated locally (Feit, 1988), have evolved. These community–ecosystem reciprocal partnerships evolve from the equitable sharing of environmental benefits and the reinforced exclusion of locals.

Traditional management practices

Self-management practices derive from a sense of ownership and responsibility for the resource combined with an in-depth knowledge of the system, its components and their interactions, reinforced by a set of socially accepted informal rules. These rules comprise the backbone of the community with respect to worldview and are termed social norms. Some have even evolved more recently in a local response of resistance to government-enforced management efforts (Feit, 1988). Losing a portion of their individuality to fight for common interests, through reinforcing internalised behaviour norms, can act to influence the direction of self-management (Agrawal and Gibson, 1999). For instance, where established norms act to limit fishing to certain areas of the reef at certain times and protect areas of regeneration such as aggregation and spawning sites, community norms ensure management is self-sustaining, but where accepted norms promote excessive exploitation, for instance, in Indonesia where certain marginal tribes believe mined coral used in house foundations makes the building stronger, norms may be to the detriment of the environment. Rural Fijians, for instance, believe that reef is a supernatural occurrence, thus the introduction of conservation-oriented practices would prove futile in such a culture (Rudd *et al*., 2003). Strategies for the intrinsic alteration of long-existing community norms are unknown.

Social norms held by the islanders of Ahus Island, Papau New Guinea, include prohibiting spear and net fishing, and strictly limiting invertebrate harvests, within six demarcated areas, *tambu*, of the reef lagoon upon which they rely. Tambu fishing is only permitted for significant ceremonial occasions that occur up to three times annually. At these times, tambu areas are opened very briefly (two to three hours) for intensive exploitation to provide food for consumption at the ceremony. As a result, target species extracted from the restricted areas are over 20% larger and fish biomass caught, 60% higher. Such management practices are imbedded in tradition rather than being conservation oriented, but succeed in yielding the same results. Despite intense resource dependence for both sustenance and economic incentives, compliance is successful through the perceived legitimacy of behavioural norms in benefiting the whole of society, combined with moral influence from peers. Also, exclusive ownership rights over reef resources in the direct vicinity induce a sense of community responsibility reinforcing community compliance and conservation incentives (Cinner *et al*., 2005). A similar situation was unveiled in New Island province of Papau New Guinea by Wright (1985). This region practises a tradition of restricting fishing access to a reef area during the mourning period for an influential community member (lasting up to several years). This acts to replenish local fish stocks for a ceremonial feast concluding mourning and sustains community harvests into the future.

The treatment of the reef, according to these norms, devolves into a local pattern of customary, and thus expected, behaviour by which everyone conforms as thought to be in the best interests of the community. No government enforcement is required, but instead local taboos exist, reducing cost expenditure by government bodies on enforcement sanctions and the monitoring and punishment of rule breakers that arises with other forms of management (Rudd *et al*., 2003). This evolution of traditional practices that act to sustain the reef system in its complexity forms a self-management system that is adaptive to phase shifts in the environment as they begin to emerge. Costs of making collective decisions are reduced between individuals sharing a spatial unit and communal norms in frequent contact with one another, and although social stratification is inevitable in any human coercion, even in terms of gender relations, power differences are likely to be far less pronounced and far better understood within an established community (Agrawal and Gibson, 1999).

Many self-management systems have proved successful at maintaining a resource over a long historical period (Feit, 1988). Over a lengthy period, any system of management would have to adapt to the dynamic nature of an environmental system. For a coral reef this may include altered species composition, sedentarisation, nutrient influx and shifts in water temperature. Traditional self-management systems contain a delicate feedback system that allows for subtle changes in the environment to be detected and accounted for in the relevant adjustment of practices. This is the key component by which a management system is deemed successful over time, and requires an intricate understanding of the ecology of the system.

Hence, life by the sea requires a dynamic fluidity of practice attuned to adapting to seasonal and longer-term environmental transitions; although not every community member may hold this intricate knowledge, within the community exists a way of being within the world that allows for respectful coexistence and information transfer (Tyrrell, 2005). In successful self-management systems, ecological knowledge of the coral reef has co-evolved with established practices and changing environmental conditions. This local knowledge is the intellectual antecedent of such practices (Drew, 2005). By observing the delicate balance between natural and anthropogenic pressures impacting upon the reef, resource users can enhance their knowledge *of* the sea whilst *at* sea (Tyrrell, 2005).

The *I-Kiribati* tribe, indigenous to Kiribati Island situated in the tropical Pacific, have a long history of self-management over their reef resources through the sustainment of customary practices. Traditional systems of management diminished during British colonisation, and development projects evolved that interfered with bonefish spawning grounds, a key local food resource, long known to the I-Kiribati. As a result, six of the seven spawning runs were depleted. The elders, knowledge acted to advise researchers on phase shifts that had occurred and subsequent areas of the water that needed restoration. Consequently, the ecosystem was replenished as were its resources. This example exemplifies the importance of local knowledge – ignoring this resource is only to the detriment of the ecosystem (Drew, 2005).

Through traditional biomonitoring techniques that involve observing shifts in catch species size and abundance, artisanal fishers can improvise upon pre-existing practices to accommodate such changes, often by self-imposed restrictions or altered efforts, and thus successfully manage the resources of the reef system for future generations. In addition to biotic changes, traditional management systems have had to adapt to anthropogenic changes as other peoples have come into the area, often fishing local waters with altered incentives. External threats to resources from outsiders have become more frequent in recent decades, but throughout history resource users have been forced to extend management practices into new areas to adapt to these impositions, and will continue to regulate external access into the future (Feit, 1988; Agrawal and Gibson, 1999). Hence, self-management of reef systems does not have a history of continual success in the light of environmental change, but rather a history of disruption, adaptation and self-renewal (Feit, 1988).

Despite an assemblage of successful self-management systems, with survival of the community and reef acting as testimony of not only local knowledge but also its effective application in a particular time and place despite relative isolation in the past (Feit, 1988), sustainability is not an automated response to self-management. Thus, it may be possible to re-establish equitable partnerships that meet similar conservation goals today. As already outlined, for a management system to be successful and self-enforcing it requires four components: intricate knowledge and understanding of the ecosystem, ownership responsibility, adaptive capacity and the social institutions capable of enforcement. The latter was demonstrated by Hoffmann (2002) when reef management in the Rarotonga region of the Cook Islands greatly improved, in terms of both reef system health and species diversity, upon the re-implementation of the traditional marine social institution, responsible for governing social rules and relationships, known as *Ra'ui*. Where one of these components is lacking, self-management is likely to collapse. Thus, self-management of a system is not automatically indicative of sustainability, but rather the capacity of a community to learn from mistakes and feel a continued sense of responsibility in the light of sometimes rapid change, and where communities lose this capacity, management breakdown follows.

The inshore fisheries of Fiji continue to provide a key source of protein to the locals under traditional self-management practices. Here, the government lacks the financial and institutional resources to impose resource restrictions, leading to local fisheries, networks being left to their own management devices. Well-established social channels act to facilitate knowledge devolution, community decision-making and self-enforcement, with non-conformists threatened with harsh treatment. Thus, social capital enforces successful self-management where government capacity is weak and governance is successfully decentralised to local decision-making bodies at the grassroots level (Rudd *et al*., 2003).

Self-management in itself refers to the competency of the resource users in sustainably exploiting the resources in their local area. It requires no external intervention and is often tied up within the culture of the society. Local devolution of power is thus devoid of regional costs of decision-making and resolution of disputes by government bodies (Agrawal and Gibson, 1999). The long-awaited intrusion from state and market pressures has the capacity to generate despoiling communities from the image of ecological primitives in harmonious balance with their ecosystems (Agrawal and Gibson, 1999). Although this image may be somewhat thwarted by idealism, the shift as a result of external forces is certainly not.

Increased consumption pressures from population expansion combined with technological innovations and institutional alterations have the capacity to deplete traditional practices. Additionally, marketisation acts to place cash values on common property systems. This has the potential to increase volumes of resource extraction that may in turn generate environmental degradation. These pressures combined with a weak non-specific property rights act to renegotiate resource extraction incentives (Agrawal and Yadama, 1997; Agrawal and Gibson, 1999). For instance, the sustainability of giant clam (Tridachnidae) stocks is highly dependent on local knowledge with sedentary populations developing site-specific requirements, and yet community management of these stocks more often than not fails. Rudd *et al*. (2003) assert this exception to the high market prices of giant clam, overwhelming traditional social norms, controlling opportunism, with economic incentives.

Centralised conservation management

Sense of ownership is key to self-management capacity as it evokes a sense of responsibility over resources by which access and user rights remain in the hands of locals (Feit, 1988). Ownership is key to exercising authority over the resource via the construction of regulations, subsequent implementation, and resolutions of disputes that arise, without which, the benefits of such efforts are nullified (Agrawal and Yadama, 1997). When ownership changes hands, through privatisation for instance, especially against the will of the resource users, any responsibility felt towards the resource is lost or abandoned, often in spite of the new management body. When privatisation occurs, subtractability of the resource subsequently increases as does the excludability of local users (Figure 35.2). A shift in control implies an incapability of traditional practices and an inferiority of their knowledge compared with

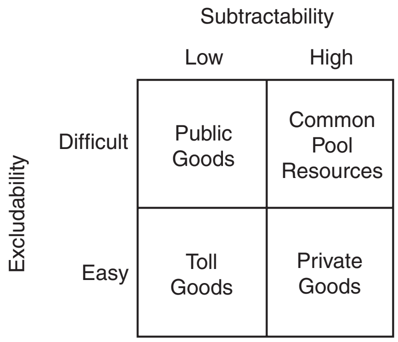

Figure 35.2 A classification of resources based on public ownership in relation to the ease of exclusion of non-authorized users and the degree of subtractability (adapted from Ostrom *et al.*, 1994)

that of outsiders. Although new institutions have in mind a sustainable management concept, by disregarding local knowledge, environmentally and culturally, enforcement is jeopardised.

This situation arose in the Turks and Caicos Islands where marine resources had been sustainably managed for generations under the responsibility, knowledge and cooperation of coastal fisher families. In complete disregard for this system, the central government imposed its authority by allocating marine reserves excluding community involvement without any local consultation. Consequently, locals hold government-imposed restrictions in low regard to the extent that the start of the lobster season is now termed The Big Grab for its free-for-all nature with everyone maximising their catch illegally. Many lobsters taken now fall 95% under size as a result (Rudd *et al*., 2003). Thus, despite the mutual desire of all stakeholders for resource sustainment, government-imposed top-down management can evoke a form of rebellion to the detriment of government management bodies, local communities and local ecosystems, although the largest costs will be felt by local resource dependants. This introduces state management as an alternative to self-management.

State management is a form of management deriving from the legal authority of the nation state. When national interests lie in conserving a local resource, management of that resource is often taken out of the hands of local users and transferred to the hands of professional policy-makers and scientists (Feit, 1988). State management exerts authority from a distance and overrides the benefits of territoriality and personal interests held by settlements local to the resource. Instead of being locally derived, the knowledge system state management relies upon usually evolves further a field, for instance, in Western scientific institutions. Thus, it relies on a different, some might say disconnected, set of components to local systems of community management.

Management practices imposed in state systems are often based on a conceptual theory of sustainability, rather than generations of observations learned *of* the sea combined with personal experience gained at sea. In Indonesia, as in many other developing regions, development experts given control of inshore reef fisheries are those with a background of formal education, often with minimal understanding of reef systems. Nevertheless, they view locals, for their lack of literacy skills, as their students rather than their teachers of the local waters. Formal education is assumed justification for the accuracy of their decisions, whilst the absence of the written word from local folk knowledge acts as justification for its own inadequacy (Dove, 1988). This logic assumes that a teenager who has gone through formal schooling is more adept to the changes and needs of the local reef fishery than a community elder who has spent his life fishing those waters. In truth, no outsider can hope to acquire a fraction of the botanical, climatological and ecological knowledge and understanding of the community elder, for in the government's attempt to identify an expert for management advice, it overlooks the greatest one. For as long as governments continue to view traditional folklore as an obstacle to resource management in the light of conservation, state management systems will be doomed to failure (Dove, 1988).

The dynamics of a reef system can never be fully understood by those external to it (Tyrrell, 2005), and since effective sampling of all of the world's ecosystems is impossible, state management theory is based on results generated from periodic surveys. These tend to be isolated in a specific time and place, incorporating few of the environmental parameters at work in an ecosystem. Interpretation of these data outside of their environmental context generally acts as the foundation of imposed state regulations (Feit, 1988). Hence, government agencies lack the ecosystem experience of local resource users and focus on quantitative predictions of potential outcomes (Lundquist and Granek, 2005). Thus, the knowledge available to these two different forms of management is inherently different and culturally bound if government bodies stay on this isolated route (Drew, 2005).

The synthesis of local knowledge on site permits anticipation and response to environmental alteration, unlike state theory whereby changes are only detected when they reach large-scale phase shifts. State management responds to such changes by returning to desktop modelling based on data from similar ecosystems and relying on assumptions to fill in gaps in the data, cutting itself off from local ecological understandings and cultural norms in a situation where it can ill-afford to do so (Feit, 1988). Thus, self-management has the capacity to provide more rapid responses to sudden and often complex environmental changes.

State management combines Western science with government institutions of enforcement. The values derived from a local sense of ownership are replaced by a government's need to assert authority, and little adaptive capacity exists within this imposed externally enforced set of regulations, especially when those in a primary position to monitor resource changes are those with the least power. Where the fish production of a reef is predictable over time and space, users have the ability to predict the levels of resource withdrawal that can be permitted without depleting the system, thus making them effective managers. But where local knowledge and understanding of a reef system has been depleted, the scientific theoretical basis of state management may be the only alternative. As predictability of a system decreases with loss of local knowledge systems, uncertainty as to the levels of withdrawal increases, and thus management tends more towards theoretical insurance functions to prevent environmental degradation. Hence, management capacity shifts from the realms of local control to state control and its basis of scientific theory (Rudd *et al*., 2003).

State management originally devolved from an illogical misconception that conservation is dependent on exclusive protection, and this is not possible where resources are being utilised within livelihoods, as human communities lack the capacity to exploit with due constraint. And even if this capacity was once a reality, the past is long gone. Protected area designation of Marine National Parks for instance is the strongest example of this Westernised way of thinking (Agrawal and Gibson, 1999). Recent decades have seen a trend in widespread protected-area designation under state management, many employing exclusion theory. This entails excluding all local resource users from future access to reef systems, undermining their ability to exploit at sustainable levels as they have been doing for generations. Thus, local costs of such designations have been considerable, whilst conservation benefits have yet to be seen.

Hence, paper parks abound in which constraints are neither rewarded nor violation punished (Rudd *et al*., 2003). Paper park designation are routes by which governments are being seen to designate conservation areas that undermine local management capacity and fulfil policy targets, but fail to manage them effectively to the benefit of global biodiversity (Alcorn, 1993). What these paper designations have shown us though is that the success of forced coercion into culturally external, disciplined resource use strategies is limited. The imposition of state management erases the incentive of community benefits upon good stewardship of the sea. This deficiency is precisely the reason for reconsideration of community involvement and its benefits (Agrawal and Gibson, 1999).

Self- and state-imposed management depict the two ends to the scale that is resource control. A grey area exists between these two extremes under the title of co-management and has the potential to combine the assets of the two systems to differing degrees. For instance, decentralised state control may play a larger role in reef management where local knowledge is required but community capacity is low. Thus, a mutually respectful working relationship between government agencies and resource users is forged to form multicultural action plans (Drew, 2005). This route appreciates the role of stakeholder involvement at every stage of planning, especially if stakeholders are to contribute support and commit to monitoring and enforcement. This is particularly important where governments lack stability or sufficient resources leading to management breakdown. Post-implementation, a common management failure is regulation enforcement due to inaccessibility, funding limitations, poor assignment of responsibilities and lack of public support. Community engagement buffers these effects by continuing management practices and enforcement in the light of government declining support (Lundquist and Granek, 2005).

Successful co-management effectively combines the traditional partnerships between locals and their resources with supporting policy to legally enforce this relationship, although steps have to be taken to establish the institutions, power relations and communications necessary for this balance. Action must be taken to avoid government agencies creating a false sense of power in local communities but in fact withholding decision-making and resolution rights for themselves (Alcorn, 1993). Where this happens successfully, local decision-making combined with government implementation techniques may prove a winning combination (Agrawal and Gibson, 1999).

The Samoan Fisheries Act (1988) succeeds in achieving this balance. By passing an Act of Government legally enforcing local reef-fishing practices dictated by societal norms, and thus securing local ownership rights and resource security, the government has enhanced local incentives to conserve and enforce regulations (Zann, 1999). Trust between stakeholders is essential in creating this relationship essential to co-management. Social norms reinforced through relations of trust ensured successful implementation of a marine reserve by coastal communities of Apo Island, in the Philippines. Where trust fails and each fisher maximises his catch disregarding local authority regulations, management breaks down, as happened on the nearby Sumilon Island (Russ and Alcala, 1999). Thus, both microlevel "community capacity" exhibited through social networks and norms, and macrolevel "institutional capacity" exhibited through designated property and resource rights are necessary for co-management. Lacking one of these, management implementation will fail on community or government level, or both (Rudd *et al*., 2003).

Changes in technology and demography are not likely to make systems of self-control devoid, but instead are likely to illicit management adaptations as have been made repeatedly in the past in the face of both anthropogenic and natural pressures. More subtle changes instead are likely to draw on co-management options. These include a shift from traditional institutions and social systems responsible for the gathering and transfer of knowledge of local reef dynamics to systems of formal education, locally detached with a view towards commercialism (Feit, 1988). Thus, co-management of coral reef systems is likely to become a more prevalent option in the future as local knowledge systems shift in the light of development. Local input remains essential to fill in state gaps in knowledge, to determine realistic culturally-viable objectives and to build stakeholder ties. Thus, upon recognising their autonomy, authority and in-depth environmental understanding, legal force should be applied to pre-existing community practices for their self-enforcing, adaptive and sustainable success throughout history (Feit, 1988).

SUMMARY

For a long time coral reefs have been recognised for their beauty, biological diversity and high productivity; they are only now being recognised as an important economic resource bringing benefits to both local and global economies. Unfortunately, despite their importance, reefs around the world are being degraded or destroyed by human activities and more damage is expected as anthropogenic impacts, particularly as a result of population growth and economic activities, continue to increase. Realisation of the full economic potential of reefs could provide a strong argument for conservation and sustainable utilisation of reef resources. Hence, management strategies are required that take into account both ecological and economic impacts. Local understanding of the full economic potential of reefs when sustainably utilised, and provision of socially and economically acceptable alternatives where resource use must be limited are both important to gaining and strengthening local support. Additionally, successful management requires compliance from local people. Therefore, active engagement of local communities by promoting self-management and continuation of traditional practices is essential along with government support where necessary, in the form of legal backing through policy provision and logistical support with implementation strategies. Self-management strategies require local knowledge, local resource ownership, capacity to adapt to a changing environment and provision of social institutions capable of rule enforcement. These self-management objectives must be instated now whilst generations of ecological knowledge remain prevalent, as future management prospects without local knowledge threaten to rely on theory and state alone, endangering reefs and the diversity of life they support.

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