

Chapter fourteen

Developments in tropical reef fisheries science and management

Nicholas V.C. Polunin, Callum M. Roberts and Daniel Pauly

SUMMARY

Extinction is conceivable for some species, including aquarium-fish stocks of high value and limited geographical range, but a biological basis exists for management of such small fishes. Immediate data needs are much greater for management of stocks of larger species which most reef fishing targets. The strengths and limitations of underwater visual census are now better recognized, and there is improved capacity for fish ageing, especially through otolith analysis, as a basis for growth and mortality assessments. Information technology has grown rapidly in scope, and modelling approaches offer a better foundation now for incorporating biological data into analytical approaches to sustainability. Reef fishery dynamics may widely be simpler than implied by their 'multispecies' condition, and in spite of their assumptions, surplus production models have proved useful in stock assessment. Empirical and exploratory approaches to sustainability, however, are more desirable than ever, as uncertainty about long-term ecosystem effects of fishing increases. Work of an experimental nature urgently needs also to be directed at rehabilitation techniques, especially marine fishery reserves. Successful management is, however, unlikely without local community involvement. Understanding

the scope of such participation demands a greater social science input to reef fishery studies; here as elsewhere there are encouraging trends in recent studies.

14.1 INTRODUCTION

It is not so long since tropical reef fishery yields were thought, admittedly on the basis of limited data, to range from 0.5 to 5 t km⁻² year⁻¹ (Stevenson and Marshall, 1974; Marten and Polovina, 1982). Much higher estimates, around 20 t km⁻² year⁻¹, by Alcala (1981) and Alcala and Luchavez (1981) in the Philippines, and by Wass (1982) in Samoa, were thought at first not to be representative (Marshall, 1980). However, high yields of this order have now been independently estimated for a number of sites in the South Pacific and South East Asia (Dalzell, Chapter 7), and the higher estimates are now close to the maximum levels of fish production predicted by trophic and other models of reef ecosystems (Fig. 14.1). One research problem that remains, however, is the rigorous quantification of the effects of factors such as primary productivity, depth, sampling

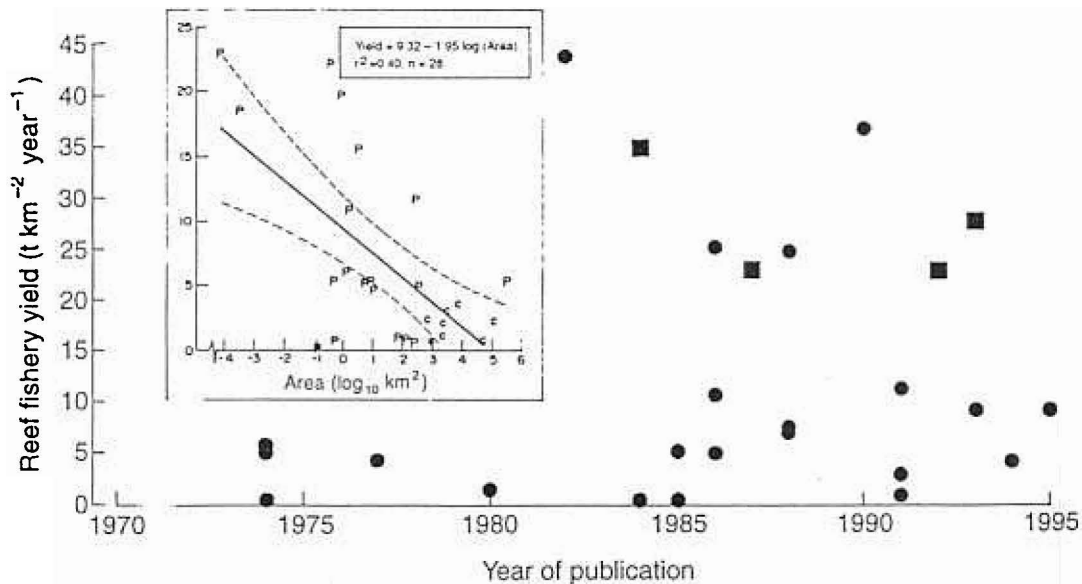


Fig. 14.1 Recent estimates of tropical reef fisheries yield have in some instances reached the high value of fish production indicated by trophic models. Filled circles (from Dalzell, Chapter 7) show observed yields, filled squares show model estimates, both plotted against year of publication. The model data are those of Galzin (1987), Polunin and Klumpp (1992b), Arias-Gonzalez (1993) and Polovina (1984). The insert, adapted from Arias-Gonzalez *et al.* (1994), shows the relationship between observed yields (t km⁻² year⁻¹) and the area from which yields were estimated, in the Caribbean (c) and the South Pacific (p).

area, or coral cover, on yield estimates (Marten and Polovina, 1982; Munro, 1984a). Figure 14.1 illustrates correlation of one such factor, sampling area, with reef fisheries yields. Beyond the capacity to predict we must also ensure that the predicted yields will be sustainable. Our ability to do so depends on the basis of our predictions, which may be either analytic or empirical. The former implies that a fishery is simulated, species by species, by incorporating all available estimates of rates (growth, mortality, recruitment, predation, etc.) into species-specific sub-models, then integrating these into a comprehensive model (Larkin and Gazeay, 1982). This approach is very data-intensive. Estimation of the parameters required for this, enhancing access to, and synthesis of the parameter values which have already been estimated, will therefore be the theme of the first section of this chapter.

Alternatively, in the absence of detailed information upon which to base an analytic approach, experimental and exploratory approaches must be used to obtain information on productivity and sustainability. These will be the focus of the second section of this chapter. Beyond both approaches to estimating yields and their sustainability, a basis for fishery regulation must be created, and this is a more people-orientated activity. Understanding the dynamics of reef fisheries, the perceptions of reef fishers, and their historical and social contexts all have implications for management. The current status of, and needs for, scientific research in these areas will thus be the focus of the third section. While the scientific basis for good advice to managers is still weak, we will show that there are encouraging trends. These represent potential growth areas with respect to the science and management of tropical reef fisheries.

14.2 ASSESSMENT OF TROPICAL REEF FISHERIES

Knowledge of reef fish biology has been greatly influenced by ecological studies of small species (contributions in Sale, 1991). This knowledge is highly relevant to fisheries supplying the aquarium trade, but is beside the point where larger species such as groupers are concerned. Indeed, substantial gaps remain in knowledge of reproduction (Sadovy, Chapter 2), larval life (Boehlert, Chapter 3) and juvenile stages (Roberts, Chapter 4) of reef fishery species, and of the productive base of reef fisheries (Polunin, Chapter 5). There are limited catch and yield data on the fisheries themselves (Dalzell, Chapter 7), and on fishing effects (Jennings and Lock, Chapter 8). Also, much work needs to be done on the geographical, social and economic facets of reef fishing (Ruddle, Chapter 6; McManus, Chapter 10), on the dynamics of fisheries (Appeldoorn, Chapter 9), and on the roles of traditional (Ruddle, Chapter 12) and modern (Adams, Chapter 13)

institutions in development and management of reef fisheries. A traditional practice, area closure, is being revived as a modern management tool, but the underlying mechanisms still need to be explored, to derive a sound basis for functional design of reserves (Bohnsack, Chapter 11).

Sampling and reduction of stock assessment data

The visibility of many reef fishery species to scuba divers implies that underwater visual census (UVC) will continue to be a useful stock assessment technique (Harmelin-Vivien *et al.*, 1985). However, there have been only few tests of how well UVC does sample a target fish community (Samoilys, 1992; Acosta, 1993; Watson *et al.*, 1995; Jennings and Polunin, in press b). Questions of observer error and sampling design apart, the value of UVC to stock assessment or to determining effects of exploitation depends on the site and species involved. In Fiji and New Caledonia, emperors (Lethrinidae) are poorly sampled by UVC (Kulbicki *et al.*, 1987; Jennings and Polunin, in press b), and yet these species are very important in the catch (Dalzell *et al.*, 1992). It is clear that where spear fishing is common, the utility of the technique will be limited by the flight behaviour of resident fishes. Some important species, however, including groupers and snappers, appear to be more abundant in UVC than in the catches of bottom-set lines. Species exhibiting equal densities in the two techniques are likely to be few (Kulbicki *et al.*, 1987). Information on size at age is essential for stock assessment, and can be obtained in two ways: direct ageing of hard parts like otoliths (contributions in Summerfeldt and Hall, 1987), or analysis of length–frequency data (contributions in Pauly and Morgan, 1987; Appeldoorn, Chapter 9).

Annual banding is not marked in otoliths of fishes from low latitudes (Munro and Williams, 1985). This forces researchers interested in absolute ages to rely on daily rings (Pannella, 1971), especially in small species (Polunin and Brothers, 1989) and the juveniles of larger species (Lou, 1992). While the fine banding of older members of larger species is widely agreed to be difficult or impossible to discern, seasonal and/or spawning checks do occur under some circumstances (Longhurst and Pauly, 1987), along with annuli that can be validated (Manickchand-Dass, 1987; Lou, 1992). Annual banding is certainly clear enough to provide a basis for ageing at mid latitudes, and recent studies have made use of this for some important fishery species such as groupers (Bullock *et al.*, 1992; Ferreira and Russ, 1992; Sadovy *et al.*, 1992; Bullock and Murphy, 1994).

Difficulties remain, despite these and similar studies. One of the difficulties still to be addressed is the often fragmentary nature of the otolith record in large (old) fish. In those parts of an otolith sequence where daily bands cannot be counted, they have to be assumed to be of similar width

to those of areas where they can be counted and measured (Ralston, 1985). Techniques such as scanning electron microscopy can be used to increase resolution, at least to check on features in a few otolith sections, and such results can be further refined using image analysis software, which enables band counts to be based on objective densitometric criteria.

Estimating growth parameters (usually those of the von Bertalanffy growth function, or VBGF) is straightforward when (absolute) otolith ages are available, but should be done using one of the now widely available non-linear fitting routines, rather than through a linear transformation of the VBGF.

The other approach for obtaining (relative) ages and estimating growth parameters, the analysis of length–frequency data, can be implemented using techniques such as ELEFAN I (Pauly, 1987) or SLCA (Shepherd, 1987), both incorporated in the FiSAT software package recently released by FAO (Gayanilo *et al.*, 1994), the MIX package (MacDonald and Pitcher, 1979), or the somewhat costly MULTIFAN software of Fournier *et al.* (1990). Besides estimating the parameters L_{∞} , t_0 and k of the VBGF, several of these techniques also allow quantification of seasonal and other growth oscillations in reef fishes (Pauly and Ingles, 1981; Longhurst and Pauly, 1987; Appeldoorn, Chapter 9). Whether age- or length-based methods are used for estimation, information is lost, when, in assessments, only a single set of growth parameters is used. The growth of individual fish being variable, incorporating that variability would make the uncertainty inherent in estimates of yield, for example, more explicit. Thus the variance estimates output by the techniques listed above must be considered when performing assessments. There are many ways in which this and other elements of uncertainty and risk can be formally incorporated into stock assessment models (Hilborn and Walters, 1992).

If not accounted for, migrations and variable recruitment, both within and among years, will compromise mortality results obtained using techniques that assume a steady state (Appeldoorn, Chapter 9). On the other hand, there is no reason to believe that coral reef fish recruitment is significantly more variable than recruitment in high-latitude species, to which catch curves and similar techniques have been successfully applied (Ricker, 1975). Particularly, it would be appropriate to estimate mortality in unexploited stocks using such techniques. This is both because their assumptions are more easily met when new recruits form a relatively small fraction of overall biomass, and because mortality in such stocks consists only of natural mortality (M), a parameter otherwise difficult to estimate reliably.

Difficulty in estimating M in tropical fishes is the reason why Pauly (1980b) derived his empirical model (Appeldoorn, Chapter 9, Equations 9.13–14) for prediction of M from VBGF parameters and mean environ-

mental temperature (T). Although, at the time, only 17 estimates of M were available for reef fishes, out of the 175 used to derive his model, this group did not appear to display any significant pattern of deviation from the values predicted by the model (Pauly, 1980b). With more values of M having become available, there is growing interest in re-assessment of Pauly's (1980b) model (e.g. Trenkel, 1993). Given enough data, such analysis could also consider Ursin's (1984) point that specific groups may show distinct trends in effects of temperature on population parameters (Polunin and Brothers, 1989).

Overall, it is our impression that the complexity of reef fisheries may not always be as great as the ecological and fishery literature would have us believe. Thus, for example, although many species may ultimately contribute to the catch, a small number typically constitutes a very large part of the biomass (e.g. McManus *et al.*, 1992; Smith and Dalzell, 1993). It is likely, therefore, that the data requirements of reef fishery stock assessment may often be easier to meet than is supposed.

How can existing data be better accessed?

Information technology provides the means for storage, transfer and dissemination of the large amounts of data required for fishery stock assessment. The technology brings with it a growing potential for computer-based statistics- or graphics-orientated analyses. There is now a wealth of means for manipulating, exploring, visualizing and interpreting complex and voluminous data. There is also much information on reef fishes and fisheries in publications, both formal and 'grey', but much of this, unfortunately, is not always available when and where needed. Especially in developing countries, there is a strong tendency for valuable data sets and data-rich reports to be lost (Polunin, 1983; Janzen, 1986; Mathews, 1993). This problem is being addressed for tropical reef fisheries by comprehensive accounts of the resources of whole regions, such as the books of Munro (1983g) on the Caribbean and of Wright and Hill (1993) on the South Pacific. It is also an objective of the FishBase Project of the International Center for Living Aquatic Resources Management (ICLARM), in Manila, Philippines (Pauly and Froese, 1991). The latter project has recently released its first CD-ROM, providing fully referenced taxonomic, biological and ecological information on 12 000 fish species, of which 1980 are reef-associated (Froese and Pauly, 1994). By way of illustration, FishBase contains 189 sets of VBGF growth parameters, and the parameters of 385 length-weight relationships, for reef fishes, including those of Bohnsack and Harper (1988) and of Kulbicki *et al.* (1993). ICLARM should be contacted on how to obtain this CD-ROM or to collaborate with the FishBase Project; see the last author's address.

Has our capacity for synthesis increased?

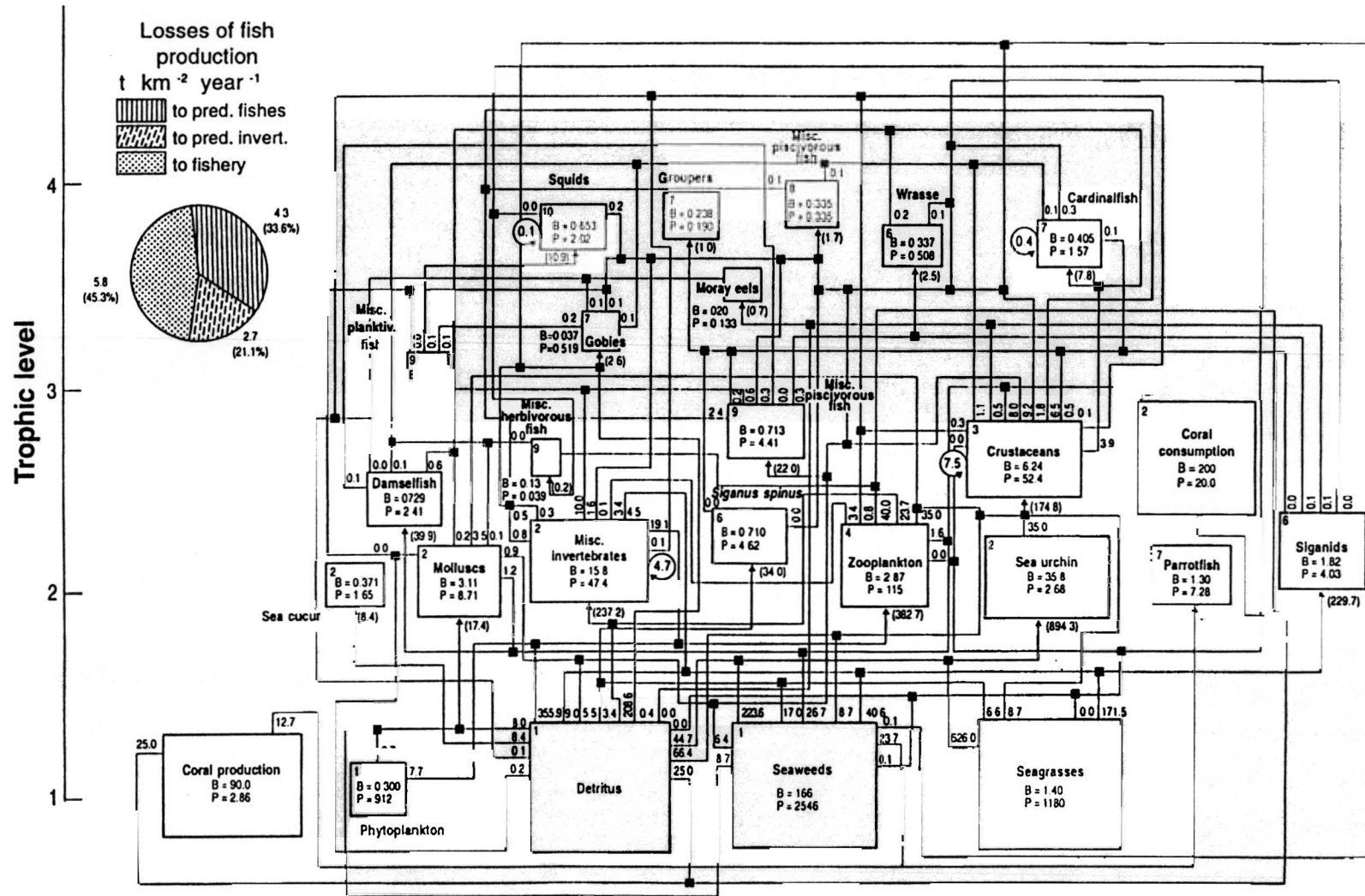
Simulation and mass-balance models are two major tools for synthesizing the multifaceted information available on tropical reef fisheries. These models may take a variety of forms, but all have in common that they simplify, sometimes strongly so, all but the process that is of interest to the model's architect (Larkin and Gazey, 1982). Still, they can be used to make sense of complex data, to derive new hypotheses and to quantify the roles of processes or factors allegedly affecting reef fisheries. We present three examples, each emphasizing one (set of) process(es), with simplifying assumptions for what may be called the 'boundary conditions'.

The first example refers to a simulation model emphasizing the recruitment to a fish stock consisting of a metapopulation, located in a number of habitat patches interconnected through larval or other means of dispersal. Man *et al.* (1995) used this to examine the role of marine fishery reserves in enhancing recruitment to fishing grounds. They showed that reserves become highly beneficial when the local extinction rate caused by fishing is large. In addition, sustainable yield was maximized when half of the patches were occupied by a population.

The second example pertains to a simulation model of a temperate population of slow-moving molluscs (abalone), but could easily be adapted to reef species. The model, called AbaSim, incorporates, in a spatially structured context, growth, mortality and a variable recruitment to the stock, which may be exploited at different rates (Prince *et al.*, 1991). The results, presented in the form of vivid graphics, allow assessing the short-, medium- and long-term impacts of different management interventions in a way that is intuitive, yet rigorous, and thus can be used as a communication channel among fishers, managers and scientists (Sluczanowski, 1992). One can only hope that this approach will be emulated by reef researchers.

The third example pertains to the ECOPATH II modelling approach and software of Christensen and Pauly (1992a,b). This mass-balance approach, which builds on Polovina (1984) and emphasizes the trophic relationship within ecosystems, has been widely applied to reef systems (Arias-Gonzalez, 1993; Pauly and Christensen, 1993; contributions in Christensen and Pauly, 1993).

The data required for construction of ECOPATH II models are the same rates and biomass estimates that field researchers tend to publish, and hence this approach can be used to verify the mutual compatibility of such estimates (e.g. is the production rate of herbivore X at least as high as the consumption rate of its predator Y?). Further, once the trophic flows linking the various 'boxes' of an ECOPATH II model are estimated (Fig. 14.2), their topology can be analysed, for example in terms of trophic



impacts (Fig. 14.3) and compared, using the rich theory of flow network recently developed by Ulanowicz (1986) and others (contributions in Christensen and Pauly, 1993). Moreover, the newly released Windows version (3.0) of ECOPATH II includes a routine that allows entry of ranges of rates or biomasses, rather than point estimates, and of distributions (uniform, triangular, normal, log-normal). This feature allows for a Monte Carlo resampling of the distributions, leading to a multitude of model realizations. These realizations are evaluated for their conformity to constraints set by the user, and the selection of a 'best' solution in a least-squares sense. Through this new routine, called EcoRanger, ECOPATH II has acquired the ability, previously lacking, to answer 'what if?' questions.

The three examples above indicate that our capacity for synthesizing data from tropical reef fisheries has much improved in recent years. Therefore, it seems, one important task now should be to use models such as these, to answer practical questions, then use the answers to improve the models and/or develop alternatives. We suggest (along with Pauly and Christensen, 1994) that this approach beats any abstract debate on whether the complexity of tropical reefs precludes them from being modelled. Nevertheless, at the same time, efforts must continue to be made to expand the basic data available for such synthesis.

14.3 RESILIENCE AND SUSTAINABILITY

Lack of information on component populations makes prediction of overall productivity (Polunin, Chapter 5; Appeldoorn, Chapter 9) and fishing effects (Dalzell, Chapter 7; Jennings and Lock, Chapter 8; McManus, Chapter 10) difficult. Habitat is manifestly fragile. Communities may change dramatically as exploitation begins, so that virgin stocks offer a poor basis for predicting later productivity. Stocks with slow growth, low reproduction rates and high susceptibility to fishing, such as sharks, may be inherently prone to depletion (Adams, 1980).

What are maximal and sustainable yields?

Surplus-production models (Schaefer, 1954, 1957; Ricker, 1975) provide, in principle, an approach to estimate the catch that can be taken on a

Fig. 14.2 Mass-balance model of trophic interactions in the Bolinao coral reef flat of Bolinao, Philippines, constructed using the ECOPATH II software. Box sizes are proportional to (log) biomasses, expressed in t wet weight km⁻²; flows are in t ww km⁻² year⁻¹; backflows to the detritus are omitted for clarity. Adapted from Aliño *et al.* (1993).

sustained basis from a given single-species stock, given a time-series of catch and effort data. The approach has been much criticized (Hilborn and Walters, 1992; Koslow *et al.*, 1994), notably because most of its users fail to account for large and rapid changes in fishing effort (i.e. for non-equilibrium situations). It continues to be widely used, however, as it appears resistant to violation of many of its assumptions and can implicitly account for multispecies interactions (Pope, 1979a).

It is thus not any inherent feature of production models that has prevented their wide application to reef fisheries, but the absence of sufficiently long time series of mutually compatible catch and effort data. The replacement, by Marten (1979a,b), working on Lake Victoria, and by Munro (1980), working on the Jamaican reef fishery, of different times by different sites (assumed to have been similar before exploitation) was therefore a major advance in tropical fisheries research, and in fisheries research generally (Caddy and García, 1983; Pauly, 1994). Further development of this approach by Polovina (1989b) has led to a composite model, incorporating both time and space, and accounting for non-equilibrium effects. With such a powerful tool, tropical reef researchers can now estimate sustainable yield wherever enough replicates can be found in time and/or space – and if fisheries-induced changes can be assumed to be reversible, our next topic.

Reductions of predator or competitor biomass through fishing can be expected to have numerous indirect effects (Jennings and Lock, Chapter 8). One of the best-documented effects to date has been the increased incidence of low-value species in the catch, as shown by Koslow *et al.* (1988) for Jamaica, Butler *et al.* (1993) for Bermuda, and Appeldoorn *et al.* (1992) for Puerto Rico and the US Virgin Islands.

Although Jennings *et al.* (in press) and Jennings and Polunin (in press), in combined fishery and UVC studies, found no evidence for increased abundance of prey fishes in the Seychelles or Fiji, sea urchins do appear widely to have increased in abundance following removal of predators by fishing (McClanahan, 1992, 1994). Increased sea-urchin density with increase of fishing effort may not appear to be of immediate consequence to fisheries (unless sea urchins are consumed locally or can be exported), but there are potential side-effects for the ecosystem, for example through bioerosion of the reef (McClanahan and Muthiga, 1988). In such a case,

Fig. 14.3 Trophic impact matrix of the reef ecosystem model in Fig. 14.2, estimated by the ECOPATH II software and showing how an increased consumer biomass would affect the biomass of all other elements in the ecosystem, assuming that diet compositions are conserved. The changes (increases above, or decreases below the baseline) are relative, but comparable within rows.

exploitation, through its impact on a keystone species (Paine, 1966) or group, may lead to ecological changes, and the yields predicted by surplus production models, however sophisticated, would not be sustainable.

Changes may also occur because of nutrient removal by fishing, although the evidence is equivocal. After years of contention, it is still unclear whether tropical reefs are widely 'nutrient-limited' or not (Polunin, Chapter 5). Internal stores of nutrients have been little studied (Entsch *et al.*, 1983) and external sources have rarely been quantified (Meyer *et al.*, 1983; Lewis, 1987; Hamner *et al.*, 1988), but it seems possible in any case that space is a major determinant of algal biomass and productivity (Grigg *et al.*, 1984). Intense fishing evidently would remove a large part of the store of nitrogen and phosphorus in reef biomass. If external supply is slow, and internal cycling fast, the effects could be large and long term, and conversely otherwise.

There is indication that nutrient export through fishing may be important for oceanic atolls. Thus, Salvat *et al.* (1985) calculated that the lagoon of Takapoto could export only 3.5% of the biomass of bivalve molluscs per year without nitrogen deficit. Here again, strong exploitation would not be sustainable.

We can perhaps generalize these observations by suggesting that extensive harvesting should be limited to areas with high nutrient inputs, such as continental margins, compared with nutrient-poor regions such as oceanic gyres. This would match the situation occurring in non-reef fisheries, which have high catches in areas with high value of new production and low catches in areas where regenerated production predominates (Longhurst and Pauly, 1987).

How readily do stocks and systems recover?

Two key aspects of sustainability are the recovery rate of depleted species, once fishing pressure is relaxed, and extinctions, which preclude recovery.

Intensive episodic exploitation of small reef areas by spear fishing and leaf-sweeps in Yap (Smith and Dalzell, 1993) rapidly depletes fish biomass, but there are indications that this grows back to pre-exploitation levels in 1 or 2 years. In the Philippines, abundances of several reef fishes have increased in small reserves within a few years of their establishment (White, 1988a; Russ and Alcala, 1994), although recovery in numbers of fish is much faster than recovery of biomass, especially in larger species such as groupers. About 50% of the target species on two Caribbean reefs apparently recovered in protected areas which were less than 2 km across, within 4 years of the fishery being closed (Polunin and Roberts, 1993). Roberts (in press) recorded a 60% increase in overall fish biomass within the Saba Marine Park in the Caribbean, within 2 years. These findings

indicate that local populations have the potential rapidly to recover from depletion. Tagging and tracking work (Holland *et al.*, 1993) indicates that most reef fishes are strongly site attached. However, many larger species, such as jacks (Carangidae) and some snappers (Lutjanidae) are mobile, and movement of large fishes is another mechanism of local recovery following fishing (DeMartini, 1993).

Estimating the rate of recovery from, and reversibility of, fishing effects over large reef areas appears more difficult. Where growth overfishing predominates, recovery following effort reduction may be rapid if the fish in question are fast-growing, as in the case of mullids (Garcia and Deme-tropoulos, 1986). Recovery may be slower if the biomass reduction was due to recruitment overfishing, as it takes time to rebuild adult spawning biomass and high fecundities (Polunin and Morton, 1992). Regarding extinctions, McAllister *et al.* (1994) found that 17% of the 800 reef fish species which they reviewed had been recorded from areas of less than 50 000 km². Many species occur in only a single island or cluster of islands. Widespread heavy fishing could cause global extinctions of some such species, particularly where there is also habitat damage. A complicating factor is that the majority of species with restricted ranges are of high value to the aquarium trade. Species-by-species restrictions on capture and trade (through the CITES lists) may be appropriate to prevent over-exploitation, but at present there are few controls on the industry (Wood, 1992). Issues relating to the aquarium trade can be expected to increase in importance for reef fishery managers. Fortunately, much of the basic information to devise management strategies for these fisheries already exists. The same cannot be said for species such as sharks, which grow and reproduce slowly.

Real closure to fishing of areas at the scale of hundreds of km will be difficult in most cases. A mix of protective and rehabilitative measures may be more desirable, particularly where valuable species are depleted, while the fishery as a whole is still healthy (Munro, Chapter 1; Ruddle, Chapter 6). The rearing and wild release of giant clams may be considered in this context, along with the careful use of artificial reefs (Polovina, 1991b; Bohnsack, Chapter 11). Another possibility is the provision of shelter habitat to reduce mortality in species such as spiny lobsters (Eggleston *et al.*, 1992). Important management questions remain also as to the rates and mechanisms of recovery of communities and ecosystems. The extent and nature of these effects is only now being appreciated; few case studies or time series data exist which could be used to draw inferences on these processes. Data sets relating the state of reef fishery resources to episodes of pulse fishing (Smith and Dalzell, 1993), changes in fishing effort over time (Koslow *et al.*, 1988; Alcalá and Russ, 1990; Russ and Alcalá, 1994) or spatial differences in effort (Jennings and Polunin, 1995) are very few. Assembling such data sets is an urgent task.

More on the multispecies question

In tropical multispecies fisheries, catch composition usually changes as fishing pressure increases. Vulnerable species decline, but overall catches may be maintained through species replacements, or by shifts in gear use (Pauly, 1979; Pope, 1979a; Munro, 1980). Such maintenance of overall yield may prevent important actions from being taken until it is too late, for example when grouper stocks are heavily depleted (Butler *et al.*, 1993).

Low gear selectivity is often cited as a problem in that it renders application of single-species approaches to stock assessment and management of questionable value. However, even if more selective gears could be devised, they would be of limited value as yields of single species would be small. Besides, it would be difficult, given highly variable growth and mortality schedules (Adams, 1980; Buesa, 1987; Kirkwood *et al.*, 1994), to devise implementable management strategies that would simultaneously optimize overall yield, consisting mainly of small species, while protecting stocks of larger, vulnerable target species. Marine fishery reserves, in combination with restrictions on gear and take, offer a way out of this dilemma, and this is the reason why they are given so much emphasis in this contribution and in this book.

14.4 MANAGEMENT IN SOCIETAL, ECONOMIC AND GLOBAL CONTEXTS

Beyond the natural science base of tropical reef fisheries lie their little-explored, but crucial, social science aspects. In addition, the global context of tropical reef fisheries needs consideration.

What management measures can be implemented?

To be successful, management measures must appear reasonable and advantageous to the fishing communities concerned, i.e. to the major users of fisheries resources (Ruddle, Chapter 12). This perception will obviously depend on many factors, including the condition of a fishery (Appeldoorn, Chapter 9), which may dictate interventions ranging from mild to harsh, and objectives which, as we have seen, may be determined at different levels in the community (Adams, Chapter 13). Feasibility will be governed by many factors, including political considerations and costs and benefits of action, which may be assessed in a variety of ways. In countries lacking strong administrative and scientific infrastructures, simplicity and cost-effectiveness will be major determinants of success.

Such criteria mitigate against techniques such as individual transferable quotas, fashionable as they may be among fisheries scientists (Pauly, in press), and favour blanket measures such as area closure (Roberts and Polunin, 1991; Medley *et al.*, 1993; Bohnsack, Chapter 11).

Another important aspect of proposed reef fisheries management schemes is whether they be made to relate to traditional management systems. The strengths of these systems lie in their focus on spatial and temporal structures of gear deployment (Beeching, 1993) and resource allocation, and on the fact that decisions are enforced by local moral and political authority (Ruddle, Chapter 12). While the origin and nature of traditional, community-based systems for managing reef fisheries is debated, the key issue is whether or not these systems can meet modern pressures.

Cooke (1994) quantified fishing pressure and management aptitude in eight Fijian customary fishing rights areas, or *qoliqoli*. Of the *qoliqoli* identified as being subject to high subsistence and commercial fishing pressure, half showed low aptitude and half high aptitude to management (Fig. 14.4). Thus, even within one relatively small country, large differences occur in the way those responsible for tenured areas meet the challenges of management.

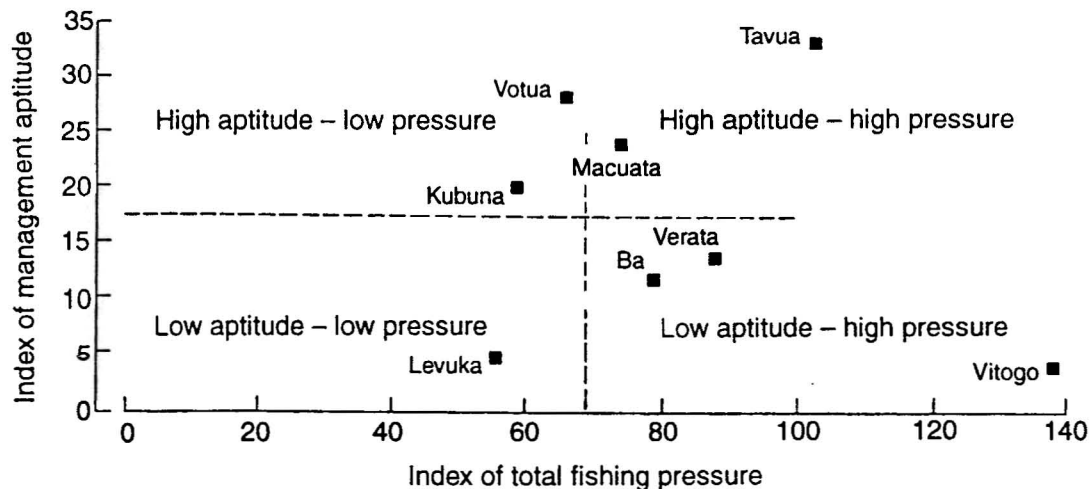


Fig. 14.4 The management aptitude of Fijian customary fishing rights areas (*qoliqoli*) varies with overall pressure on fishery resources within the area (Cooke, 1994). The index of management aptitude was derived from the sum of indices of: ability to marshal relevant information + approach to goodwill payments + management measures taken + consultation/cooperation with the Fisheries Department + patrolling and enforcement. Total resource pressure was measured as the sum of commercial fishing pressure + poaching supplement + subsistence fishing pressure. Figure redrawn by permission of Andrew Cooke Esq.

The better-managed grounds included be those which collaborated actively with the Fisheries Division, although at least one rights-owning group showed high management aptitude without state support (Cooke, 1994). In some cases, therefore, such an approach may help identify those rights owners in need of state assistance with management.

Modern governance of reef fisheries

When central governments can offer expertise and facilities, while the fishing community can provide knowledge and a monitoring function, co-management (Pinkerton, 1989) may be considered. However, such conditions do not necessarily occur in all developing countries; it is not by accident that the co-management concept emerged in Canada. In many of the developing countries where reefs occur, government cannot offer scientific expertise for local fisheries management, and neither are all fishing communities committed to sustainable resource utilization, or able effectively to protect 'their' resources. Thus, fisheries reserves in the Philippines stand or fall at the whim of local political forces (Alcala and Russ, 1990; Russ and Alcala, 1994). One reason for this may be the basic, if implied, tenet of co-management, emanating from its very definition (contributions in Pinkerton, 1989). This is that central governments and fishing communities are, with regards to fisheries resources, the only legitimate users.

For reef resources, which can be extracted, but may also form the basis of ecotourism (e.g. support scuba diving resorts and their staff) or be conserved as sources of germplasm, there are clearly more users than the fishing communities and governments. Modern concepts of governance (contributions in Kooiman, 1993) offer a framework for arrangements enabling different groups of users to interact, and governments to limit their role to providing a level field for these interactions.

Whether such governance arrangements would be conducive to adaptive management schemes (Russ, 1991; Hilborn and Walters, 1992; Medley *et al.*, 1993) is unclear. This, and related studies, dealing with the appropriateness of adaptive management schemes for subsistence fisheries, seem worthwhile areas for future socio-economic and policy research on tropical reef fisheries.

Tropical reef fisheries in a global context

The data now available (e.g. Fig. 14.1) suggest that (potential) tropical reef fisheries catches are, on a global basis, much higher than estimated by Smith (1978). Within specific regions, such as the South Pacific, present catches already are significant, although to a modest population scattered over a vast area (Dalzell and Adams, 1994).

Escalating human populations are increasing pressure on reef resources throughout most of the intertropical belt. At the same time, human activities are causing widespread degradation, especially on coastlines and to coral reefs. To this must be added the possible consequences of global climate change for reef fisheries, which are likely because the reefs themselves stand to be affected (Glantz and Feingold, 1990).

In many places, simply maintaining present fisheries production in the face of these pressures, as opposed to increasing yields, may be the primary challenge to managers. However, as mentioned above, fisheries management extends far beyond fisheries biology. Where the pressures on fishery resources are great, regulation will demand broadly based and imaginative approaches, encompassing both human and natural resources. Successful management schemes will increasingly have to look beyond a narrow fishery focus. In high-pressure areas, they will have to deal with the causes of massive entry into fishing by displaced farmers (Pauly, 1994) and of inshore habitat degradation, through comprehensive coastal zone management strategies (see McManus *et al.*, 1992, for a model study). In addition, these schemes will have to incorporate means of providing alternative livelihoods to displaced fishers, such as in cases where fishery reserves are designated. Only through use of such integrative approaches are reef fishery yields likely to be sustained over the long term.

ACKNOWLEDGEMENTS

We are grateful to the other authors in this book and to many colleagues for the ideas which they have shared with us. We thank Drew Wright, John Caddy, John Tarbit and Chris Mees for their comments on drafts. The senior author thanks the Overseas Development Administration for funding through its Fish Management Science Programme. This is ICLARM Contribution No.1129.