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Latin American benthic shellfisheries: emphasis on co-management and experimental practices

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Abstract

In Latin America the small-scale fishery of marine benthic invertebrates is based on high-value species. It represents a source of food and employment and generates important incomes to fishers and, in some cases, export earnings for the countries. In the review, we define 2 key concepts: small-scale fishery and co-management. We address the temporal extractive phases which Latin American shellfish resources have experienced, and the corresponding socio-economic and managerial scenarios. We include 3 study cases in which co-management and field experimentation have been used on different temporal and spatial scales: (a) the muricid gastropod (*Concholepas concholepas*) in Chile; (b) the yellow clam (*Mesodesma mactroides*) in Uruguay; and (c) the spiny lobster (*Panulirus argus*) in Mexico. We demonstrate that co-management constitutes an effective institutional arrangement by which fishers, scientists and managers interact to improve the quality of the regulatory process and may serve to sustain Latin American shellfisheries over time. The main factors supporting co-management are: (a) a comparatively reduced scale of fishing operations and well-defined boundaries for the management unit; (b) the allocation of institutionalized co-ownership authority to fishers; (c) the voluntary participation of the fishers in enforcing regulations; (d) the improvement of scientific information (including data from fishers) to consolidate the management schemes; (e) the incorporation of community traditions and idiosyncrasies; and (f) the allocation of territorial use rights for fisheries under a collaborative/voluntary community framework. Chile is identified as an example in which basic ecological and fishery concepts have been institutionalized through management practices and incorporated into the Law. Several factors have precluded shellfishery management success in most of the Latin American countries: (a) the social and political instability, (b) the underestimation of the role of fisheries science in management advice, (c) the inadequacy of data collection and information systems, (d) the poor implementation and enforcement of management practices and (e) the uncertainty in short-term economic issues.

In the review, we also show that in Latin America, large-scale fishery experiments are starting to play an important role in the evaluation of alternative management policies on benthic shellfisheries, especially when accompanied by co-management approaches that explicitly involve the participation of fishers. Fisher exclusion experiments have demonstrated changes in unexploited versus exploited benthic shellfish populations and in the structure and functioning of communities. The information has been used by scientists to approach system elasticity. Ecological and fishery related knowledge has been translated into novel co-managerial strategies. The sedentary nature of the shellfish species analyzed in this review allowed localized experiments with different levels of stock abundance and fishing intensity (e.g., marine reserves or maritime concessions versus open access areas). This includes the establishment of closed seasons as *de facto* management experiments, which proved useful in evaluating the capacity of passive restocking of depleted areas and for the quantification of population demographic features. The precise location of fishing grounds provided reliable area-specific estimates of population density and structure, catch, and fishing effort. This allowed the allocation of catch quotas in each fishing ground. We also discuss the reliability and applicability of spatially explicit management tools. Marine Protected Areas (MPAs) and Territorial User Rights in Fisheries (TURFs) fulfilled objectives for management and conservation and served as experimentation tools. The examples provided in our review include a comparative synthesis of the relative usefulness of alternative spatially explicit management tools under a framework of management redundancy. The cross-linkage between fishery experimental management protocols and the active participation of fishers is suggested as the strategy to be followed to improve the sustainable management of small-scale shellfisheries in Latin America. Finally, we discuss the future needs, challenges and issues that need to be addressed to improve the management status of the small-scale shellfisheries in Latin America, and, in general, around the world. We conclude that for the sustainability of shellfish resources there is an urgent need to look for linkages between sociology, biology and economics under an integrated management framework. Fishers, and not the shellfish, must be in the center of such a framework.

Introduction

The artisanal, small-scale, fishery of marine wild benthic invertebrates has critical socioeconomic connotations in Latin American countries. It is based on high-value species and represents a source of food for subsistence and employment, generating important direct incomes to fisher communities and, in some cases, export earnings. However, resource sustainability has been difficult to achieve. Historical analysis reveals that, as in the rest of the world (Botsford et al., 1997), benthic shellfish populations in Latin America are becoming increasingly limited, catch has begun to drop and stocks are fully to heavily exploited, over-exploited or depleted. Among the reasons explaining this fact are the inherent characteristics of the resource, the fisher behavior and market forces (Defeo and Castilla, 1998). Above all, these fishery systems still remain poorly understood regarding the linkages between stock structure, dynamics and bioeconomic features of the fishing process (Orensanz and Jamieson, 1998).

Benthic shellfish populations are spatially explicit structured stocks with patchy distribution (Caddy, 1975; Orensanz, 1986; Defeo, 1993a; Orensanz and Jamieson, 1998). Their population dynamics are dependent on environmental conditions, which change even within small distances, and ecological interdependencies, both intra and interspecific, which regulate demographic variations in a reduced spatial scale (e.g., meters). As a result, the distribution of the fishing effort is heterogeneous, closely following the spatial distribution patterns of the resource (Hancock, 1973, 1979; Conan, 1984; Orensanz et al., 1991). Most exploited invertebrates cannot quickly redistribute themselves over a fishing ground following an extraction, for example, through the filling of gaps (patches) resulting from a sequential depletion pattern produced by heterogeneous allocation of the fishing effort. This is crucial for management, which in turn is strongly influenced by cultural, social and political factors.

Most of these fisheries have been developed under open access policies, in extended systems readily accessible to intentionally free riders, unauthorized extractors and recreational users. The above makes coastal shellfisheries difficult to manage since: a) the number of extractors cannot be readily controlled; b) operational and quota-based management measures are extremely difficult to enforce and are beyond the finances of most management agencies. Formal fisher participation in the planning, formulation and surveillance of management measures, defined here as co-management, is considered of essential importance to mitigate the above problems (Pomeroy and Williams, 1994; Berkes, 1994). Given the heterogeneous distribution of invertebrate stocks and the fishing process, intertidal and near-shore benthic shellfishes are readily accessible to evaluate the relative merits of alternative spatially explicit management strategies (e.g., reproductive refugia, rotation of grounds, natural re-stocking) through experimentation (Castilla, 2000a).

The cross-linkage between fishery experimental management protocols (McAllister and Peterman, 1992) and the active participation of fishers in them has been suggested as a useful strategy to improve the knowledge about the dynamics of the fishery system (Castilla, 1994, 1999; Defeo, 1996a). However, there are very few case studies around the world (see Prince, 1989; Sainsbury et al., 1997; Castilla and Fernández, 1998; Castilla et al., 1998) in which experimental management protocols and co-management strategies have been successfully applied. Furthermore, some of them have failed to produce useful results for policy comparison or for resolving key uncertainties in resource dynamics (Walters, 1997; Pinkerton, 1999). Experimental management has been seen as too costly, particularly in connection with the monitoring costs of experiments conducted at large spatial and temporal scales, and the risk of not obtaining substantial improvement in the knowledge of the fishery. In Latin American shellfisheries, recent successful examples are shedding light on the potential of experimental and co-management approaches working together (Castilla, 2000a).

In this review we summarize the available information on Latin American benthic shellfisheries. We define some of the main fishery concepts connected with these resources and point out the temporal phases through which they had gone. We describe three study cases in which active co-management and field experimental approaches (planned or unplanned) have been applied on different temporal and spatial scales, highlighting the diverse *modus operandi*. We detail, when information is available, how institutional management procedures have determined a gain in science and scientific fishery information and point out the advantages and disadvantages of both approaches. Finally, we discuss future needs, challenges and issues directed to improve the management status of the small-scale shellfisheries in Latin America, and, in general, around the world.

Latin American benthic shellfisheries

Small-scale fisheries constitute an important socio-economic component of the Latin American fishery. In this region there exist over 2,200 artisanal fishing communities and well over one million people directly engaged on the activity (Bermudez and Agüero, 1994). There are basically 2 types of small-scale artisanal fishery activities: (a) those related to the extraction of pelagic resources, such as finfish and squids, through spearing, netting and hooking; (b) those related to the extraction of benthic (and demersal) shellfish species, through hand-picking, diving, spearing, pot trapping and artisanal dredging (Orensanz and Jamieson, 1998). These fisheries provide an important source of employment to coastal rural communities and represent a key source of high quality food. On the other hand, the high unit value of species such as mollusks (gastropods, bivalves, octopuses), crustaceans (crabs, lobsters), echinoderms (sea urchins, sea cucumbers) compensate economically for their comparative reduced landings, as referred to those of the industrial fishing sector (Bustamante and Castilla, 1987; Defeo et al., 1993; Castilla, 1997).

Shellfisheries generate important direct incomes to artisanal communities and elevate export revenues for the countries of the region. For example, in

1994 the Chilean benthic shellfish landing amounted to approximately 160,000 metric t. This represented 2% of the total fishery landing and 11% (US$ million 140) of the total fishery export earnings (this figure does not include the domestic value of these resources, for which there is not available information: Castilla, 1997). Moreover, several commercially valuable benthic shellfish in this region show restricted geographical distributions, and are endemic or unique (Bustamante and Castilla, 1987; Defeo et al., 1993; Clarke et al., 1999).

Resources, fishers and extracting practices

A fishery system can be decomposed into three interacting subsystems (Seijo and Defeo, 1994): (a) resource; (b) resource users (fishers); and (c) management. The latter captures the whole complex dynamics of the first 2 subsystems plus external forces such as market, politics, lobby, and societal interests. The subsystems have major idiosyncrasies, which change from fishery to fishery and from place to place (even within a single fishery). Furthermore, the benthic-shellfishery system contains species with different life histories, and a wide diversity of harvesting practices and management options.

The resource

In this review, as a first approach, we have followed Orensanz and Jamieson (1998) resource categorization of shellfish stocks, based mainly on the spatial scale of their adult stages (see also Perry et al., 1999), as follows: (1) sessile regenerating invertebrates (i.e., corals), (2) sedentary benthic (e.g., mollusks, echuinoderms, barnacles), (3) mobile benthic (e.g., crabs, lobsters); and (4) highly mobile demersal (e.g., shrimps) or pelagic (e.g., squids) shellfishes. Benthic shellfish analyzed in this review belong to Orensanz and Jamieson's resource Types 2 and 3, which can be further divided according to their mobility and larval development characteristics, as follows:

Mobility

Category 1: Sessile (attached) epifaunal invertebrates (SEI), with very reduced or no mobility. This includes invertebrates such as the Chilean giant barnacle (*Austromegabalanus psittacus*), the edible tunicates (*Pyura chilensis* and *P. praeputialis*), and mussels such as *Mytilus edulis platensis* (Uruguay, Argentina) or *Mytilus chilensis, Aulacomya ater, Choromytilus chorus* (Chile and Peru).

Category 2: Sedentary infaunal invertebrates (SII) with reduced mobility, such as the scallops (*Argopecten purpuratus*) in Chile and Peru, and (*Zygochlamys tehuelcha*) in Argentina, the clams (*Mesodesma donacium*) in Chile and Peru, (*M. mactroides*) and (*Donax hanleyanus*) in Argentina, Uruguay and Brazil, the sea cucumber (*Isostichopus fuscus*) in Galapagos (Martínez et al., 1998) and the ghost crab (*Callianasa garthi*) in Chile (Marín, 1991).

Category 3: comprised of mobile invertebrates (MI), with or without territoriality or social behavior. Here we recognized two subgroups: (a) species that do not present regular migratory patterns (MIa), such as: the muricids (*Concholepas concholepas*), (*Thais chocolata*), (*Trophon gervassianus*), (*Xantochorus cassidiformis*) and the key-hole limpets (*Fissurella* spp.) in Chile and Peru; strombids (*Strombus gigas*) in Mexico and the Caribbean; volutides (*Adelomelon brasiliana*, *A. beckii*, *Zidona dufresnei*) in Argentina, Chile, Uruguay and Brazil); the sea urchin (*Loxechinus albus*) in Chile; the octopuses (*Octopus mimus*) in Chile, (*O. maya*) in Mexico and (*O. tehuelchus*) in Argentina; (b) species that exhibit well-defined seasonal migratory patterns (MIb), such as the spiny lobster (*Panulirus argus*) in Mexico.

Larval development and duration

Category 1: direct developers. Species with internal fertilization and absence of pelagic larval stages, such as the muricid (*T. gervasianus*) and the octopuses (*O. mimus*) (Defeo and Castilla, 1998), (*O. tehuelchus*) (Iribarne, 1991) and (*O. maya*) (Nepita and Defeo, 2000). These species usually exhibit parental care and low fecundity.

Category 2: species with short-lived planktonic larval stages. Includes those with external fertilization, high fecundity and short-lived (hours) free-floating larvae, such as *P. praeputialis* and *P. chilensis* (Clarke et al., 1999).

Category 3: species with long-lived planktonic larval stages. Species with external fertilization, high fecundity, and free-floating larvae that can remain in the plankton for variable periods ranging from: a few days, such as *Chorus giganteus* (Gallardo, 1981); a few weeks, such as *L. albus* (Guisado and Castilla, 1987; González et al., 1986), *O. mimus* (Defeo and Castilla, 1998), the yellow clam (*M. mactroides)*

(Defeo, 1993a); or months, such as *C. concholepas* (Castilla, 1982). These species are structured as metapopulations at a large spatial scale (Orensanz, 1986; Orensanz et al., 1991). Consequently, replenishment patterns of local populations may be given by the distance of larval dispersal and the oceanic hydrodynamic features within the domain of the metapopulation (Carr and Reed, 1993; Allison et al., 1998).

Resource users and extracting practices
In Latin America, coastal inshore benthic shellfisheries are mostly operated on a small-scale by artisanal fishers. It is difficult to generalize on definitions embracing "artisanal shellfishers" and to give an exact meaning of "small-scale benthic-shellfishery" activities. Countries operate under different fishery definitions. For instance, the Chilean fishery artisanal subsector includes in the category of small-scale fishery operations, boats and vessels up to 50 gross register metric tons. The Chilean Fishery and Aquaculture Law (FAL, 1989: Ley N° 18892, Diario Oficial de la República de Chile, 1989, and subsequent modifications) characterizes shellfishers ("mariscadores") as: "fishers extracting mollusks, crustaceans, echinoderms and shellfish in general, with or without the use of a boat". Therefore, the definition includes rocky intertidal hand-pickers and sandy beach diggers (food-gathering) as well as skin and professional "hookah" (air compressor) divers.

We globally define a ***small-scale or artisanal benthic shellfishery*** as: an intertidal or subtidal benthic fishing activity operating in inshore waters, usually no deeper than 25–30 m, aimed for sale and/or subsistence by a single or reduced group of fishers (2–5), which usually incorporate fishing techniques such as hand-picking, digging, diving, artisanal dredging and trap deploying. The fishers may or may not use boats. If they do, the boats are small (wooden or glass fiber boats less than 5–7 m long) and equipped with oars, outboard or inboard engines (15–50 HP) and with onboard air "hookah" compressors. Fishing trips are normally run during the day, usually less than 15 miles from the base port.

Small-scale benthic shellfishers operate individually or in small groups (usually family relatives) and can be organized in fishing communities, such as syndicates (Payne and Castilla, 1994; Minn and Castilla, 1995), associations or cooperatives (Seijo and Fuentes, 1989). In the review we distinguish the following fisher categories:

(1) Occasional recreational intertidal food-gatheres. Women, men and children extracting intertidal benthic shellfish (handpicking or digging) for bait, food, or other purposes. They usually do not depend on the extracted resources (i.e., *P. praeputialis* at Antofagasta, Chile, see Varas, 1996; Durán et al., 1987).

(2) Regular intertidal food-gatheres. Women, men and children, who tend to operate individually during low tides (where astronomic tides are present) or under benign sea conditions at the intertidal and/or the near surf zone in microtidal coasts. The extracted products are for their own consumption or for sale (i.e., "loco", *C. concholepas,* the clams (*M. donacium*) and (*M. mactroides*) and the sea urchin (*L. albus*) (Defeo et al., 1993; Durán et al., 1987).

(3) Ordinary skin-divers. Divers operating in shallow waters, either individually or in teams. They are usually equipped with underwater suits, fins, mask and snorkel and dive to depths between 1–6 m. The handpicked or speared benthic shellfishes are for subsistence or selling into local communities (Durán et al., 1987).

(4) Professional "Hookah" divers. Divers operating from small artisanal boats, usually equipped with outboard or inboard engines and "hookah" air compressors. The divers are equipped with complete underwater gear and bags to store the handpicked or speared shellfish (Castilla, 1994; Cabrera and Defeo, 2001). They dive to maximum depths of 25–30 m and the extracted shellfish are for selling to a middleman or to "on the spot" shellfish consumers at the coves. A small percentage (e.g., in Chile approximately less than 10%, J.C. Castilla pers. obs.) of the extracted products are for self-subsistence. As an example, in Chile there are more than 12,500 "hookah" registered divers and the shellfish they extract include more than 60 species of benthic invertebrates (Bustamante and Castilla, 1987). Chilean shellfish landings (mainly "hookah" divers) in 1993–1994 amounted to 140,000–165,000 metric t (Castilla, 1997).

In Latin America the first 2 categories of benthic resource users are very low investment, but support important recreational and subsistence activities (Defeo et al., 1993; Castilla, 1994). Shellfish rocky intertidal handpicking and sandy beach digging in the rich upwelled waters of Chile or Peru can be traced back more than 8,000 years ago (Jerardino et al., 1992). Indigenous communities with knowledge

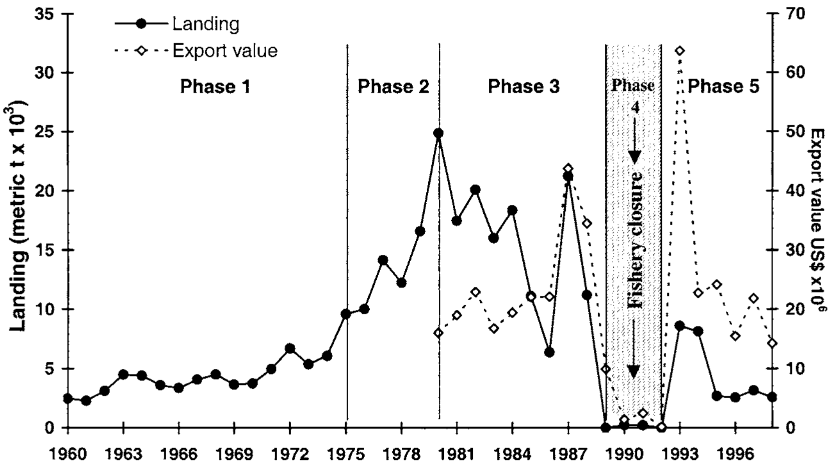

Figure 1. Landing and export values for the gastropod muricid (*Concholepas concholepas*) in Chile, showing the 5 distinctive phases between 1960 and 1998 (see text and Castilla, 1997). Data from the Anuarios Estadísticos de Pesca, SERNAPESCA, Chile, 1960–1998, and the Chilean Central Bank.

and experience gained through centuries and village-based management experiences do regularly exploit intertidal ecosystems (Balazs, 1998; see comments by Fernández and Castilla, 1997 on the organization and macroalgae extracting techniques used by Araucarian coastal communities in southern Chile). Unfortunately, statistical information on these user categories is usually lacking (Defeo et al., 1993). Something similar occurs with the skin divers (Castilla, 1989; Durán et al., 1987), though in this case there is an investment in diving gear. The professional "hookah" divers (i.e., in Chile and Uruguay operations occur under a legal permit at no cost) are better tracked statistically. The extractive activities of "hookah" divers represent important commercial enterprises (i.e., for Chile see Castilla, 1997).

Fishery temporal extractive phases

Castilla and Fernández (1998) characterized 5 contrasting exploitation phases to describe the long-term landing and export value patterns for the Chilean muricid (*C. concholepas*). In this review we provide additional information to generalize on the subject, taking into account that not all the benthic resources are subject to external market forces (export) and that there are notorious differences in country-to-country fisher idiosyncrasies and legal regulations. The recognized phases are:

(1) Initial exploitation phase. The beginning of this phase varies among stocks and is characterized by relatively low and constant landings. The fishery tends to operate under open access regimes and the products are mostly channeled to domestic markets. There are no major foreign market openings and an absence of management frameworks, other than regulations on individual sizes (in the case of *C. concholepas* this phase extended approximately between 1960 and the second half of the 70's, Figure 1). An incipient, although not reliable, statistical coverage of fishery activity exists (Defeo, 1989). Absence of information is not only restricted to this phase: it actually prevails in many Latin American shellfisheries subject to high effort levels.

(2) Expansive extraction phase. During the late 70's and early 80's, shellfish landings in Latin America showed an abrupt increase (FAO, 1995). This was the result of increasing demand from foreign markets (e.g., Asia, USA) which increased export of shellfish (in the case of *C. concholepas* the phase extended between 1976 and 1980, Figure 1). Improvements in fishery technology (diving gear, outboard engines, and air compressors) and government credits to the small-scale fishery sub-sector stimulated extraction and exportation of shellfish (Castilla, 1990). Furthermore, the increase of unemployment rates in rural zones prompted a massive migration of people to the coast and their participation in open access, low investment intertidal and subtidal food gathering activities. Domestic consumption also increased, and extractions for sale by corporate/collective organizations replaced the individual/familial scope of the activity (Defeo, 1989). These factors promoted an exponential growth

of shellfish extractions, which was not supported by management regulations founded on solid scientific knowledge, but on a variety of socio-economic factors, such as changes in market, economic and political climates (Castilla, 1990, 1997; Bustamante and Castilla, 1987; Defeo et al., 1993).

(3) Overexploitation phase. This was characterized by strong foreign market forces, which increased shellfish demand and generated higher employment, income and welfare for professional fishers and coastal communities. Exponential increase in unit prices, easy access to stocks and the open access regime stimulated investment and exponential rates of entry to the fishery in the short run, with the concurrent increase in catch volumes (Figure 1). Fishing intensity increased even under diminishing catch rates, because of low operating costs. Moreover, the easy access to resources at open coasts makes regulatory efforts expensive and ineffective (Geaghan and Castilla, 1987; Defeo, 1989; Oliva and Castilla, 1990). This is not accompanied by the concomitant increase in accumulation of scientific information and its translation to fishers and managers. The absence of studies about life history traits, demography and dynamics of stocks and the fishery leads to inadequate fishery management policies, with single operational management measures (e.g., total allowable catch levels, minimum legal sizes). This translates into shellfishery collapses (i.e., particularly during the second half of the 80's; see Castilla, 1994, 1997; Defeo et al., 1993). For instance, a sequential depletion pattern, observed for Alaskan crustaceans throughout this century (Orensanz et al., 1998), was detected in Latin America in the lapse of only two decades. The depletion of formerly high-valued species determined a shift onto formerly low-value species (i.e., in Chile: *Calyptraea trochiformis*, *Adelomelon ancilla, Odontocymbiola* spp., *Argobuccinum undatum*), thus shortening the temporal distance between fishery phases. This phenomenon is actually occurring in the coastal multispecific fishery of the gastropods *Adelomelon brasiliana*, *A. beckii* and *Zidona dufresnei* fishery of Uruguayan coasts (Riestra and Fabiano, 2000).

(4) Closure phase. Overfished or severely depleted populations determine multi-month or year fishery closures (Defeo et al., 1993; Castilla, 1997). This generates socio-economic instability for small-scale benthic shellfishers, depriving rural communities of important sources of food, employment and income. This had the utmost importance, considering the scarcity of alternative job opportunities, which is common to Latin American countries (Defeo, 1989; Castro-Suaste et al., 2000). During this phase, the demand for benthic shellfish products increases and the high unit price encourages poaching (Castilla, 1994). This occurs in fisheries that *per se* are difficult to control, due to the pseudo-individualistic characteristics of the extractive activity, the spatial scale at which artisanal fleets operate, and the lack of control at landing coves. The closure phase varies in intensity and duration according to the species fishery and the country. In some countries, isolated scientific personal efforts were benefited from the generation of scientific protocols to reduce uncertainty about the stock status and demographic properties of the targeted species. Fishery closures were thus used to evaluate the capacity for recovery of depleted populations under absence of fishing (natural or passive restocking, see Castilla, 1988; Defeo, 1993b).

(5) Stabilization of extraction and institutionalization phase. The information collected in the previous phase improves the quality of management measures. In some cases, fisheries re-openings were accompanied by pro-active management frameworks defined by a strategic institutional structure and operational set of measures (*sensu* Charles, 1995, see detailed discussion in Orensanz and Jamieson, 1998). The former included the participation of fishers and engagement of communal organizations in the management process, which lead to innovative processes of co-management (for Chile see Castilla, 1994, 1997, 2000a; Castilla and Fernández, 1998). Operational management instruments based on area-specific management plans (minimum legal sizes, fishery gear restrictions and total catch levels per fisher and per fishing ground) are implemented (Castilla et al., 1998; this review). Management strategies such as Management and Exploitation Areas (MEAs) defined in the Chilean FAL (1989), or Coastal Concessions allocated exclusively to small-scale communities, represent important management innovations (Castilla et al., 1998; Bernal et al., 1999, and see study case on *C. concholepas*). However, there are very few examples in the region, and around the world, where the acquired ecological and fishery knowledge has been effectively institutionalized and translated into specific management tools (but for Chile see Castilla, 1994). The majority of benthic shellfish resources in Latin American countries still remain in phases 3 or 4. Even

though non-extensive detailed statistics are available for most of the small-scale benthic shellfisheries in the region, it can be stated that the above phases depict the present regional situation (Defeo, 1989; Defeo et al., 1993).

(6) Mature and consolidation phase. Sustainable exploitation over time and the consolidation of institutionalized arrangements. This phase has been seldom achieved in Latin America. The muricid (*C. concholepas*) in Chile is one of the cases closest to attaining this phase (Figure 1; Castilla, 1999, 2000a). The most significant factors precluding successful management and long term restoration of benthic shellfish resources are changing socio-political scenarios, the absence of communication among and between fishers, scientists, managers and local and world fishery agencies (Botsford et al., 1997; Castilla, 2000a) and the absence and/or imperfect enforcement of fisheries laws. Uncertainty in socio-economic and political scenarios generates uncertainty about future management actions (see below), which tends to increase fishing intensity, with the obvious effect on both stock size and profit levels. Thus, the consolidation phase has not been achieved in Latin American shellfisheries.

Socio-economic, scientific and managerial scenarios

The above problems are the result of socio-economic, scientific, managerial, and political frameworks and idiosyncrasies of Latin American countries. Some descriptions follow:

(1) Social and political instability. Continuous and sudden changes in local economic and political conditions, together with scarcity of employment, generate uncertainty about future modifications of the management process, as well as in the response of fishers to regulations (Hilborn and Peterman, 1996). Pinkerton (1999) showed that implementation of co-management in British Columbia fisheries has failed as a result of distrust of management agencies and lack of organized political support. Short-term changes in management objectives resulting from unpredictable behavior of the regulatory sector (Anderson, 1984) in Latin American's benthic shellfisheries, determined an increase in fishing intensity and in poaching (Defeo, 1989; Castilla, 1994). Without an agreement to limit investment and catches, the main result of a fisher's reduced catch rate is to lower the extraction cost of other fishers without necessarily increasing their benefits. Hence, each fisher tends to increase the catch rate to increase marginal economic benefits in the short-run, thus increasing the probabilities of stock collapse. This phenomenon, known as social trap (Seijo et al., 1998), commonly occurs in open access shellfisheries of Latin America, given the very high inter-temporal preference in resource use (Defeo and Castilla, 1998), the low welfare and the high unemployment levels in rural communities.

(2) Underestimation of the role of fishery science in management advice. Regulatory agencies and policy makers in Latin America tend to under-estimate the results of stock assessments. In general, fishery policy makers do not see science as the short-term solution for the problems of income, distribution of goods and welfare. Thus, they adopt a risk-prone attitude to satisfy the increasing requirements of societal groups involved in benthic shellfisheries (i.e., the ratchet effect, see Botsford et al., 1997). Generally speaking, this is aggravated by deficient communication between scientists, policy makers and managers, precluding the translation of ecological knowledge into fishery regulatory and management practices (Castilla, 2000a).

(3) Inadequacy of data collection and information systems. Small-scale fishery information in Latin America is, in most cases, inconsistent, weak, fragmented, inaccurate and unreliable (Bermúdez and Agüero, 1994). Lack of financial support for long-term monitoring schemes has precluded the possibility of obtaining long-term data. In the absence of new information, uncertainty about how the benthic shellfishery system behaves persists, and the ability to predict the responses of benthic stocks affected by human actions is reduced. The lack of research facilities and funds limits confidence in the inferences and recommendations derived from stock assessments. This leads to a lack of credibility and has impeded the implementation of innovative schemes such as active adaptive management, experimental approaches (*sensu* Walters, 1986) or institutional arrangements as co-management. The uncertainty about how fishery systems behave "has a cost that can be measured by how much we lose by not knowing how nature works in reality" (Parma et al., 1998). Moreover, most catch and effort data reported by the fishers are unreliable as a response of unpredictable changes in management regulations. Lastly, in the small-scale benthic fishery there is a serious problem regarding unreported and illegal catches.

(4) Poor implementation and enforcement of management practices. In some cases, the accumulated knowledge has not been efficiently translated into management regulations and benthic shellfisheries have collapsed (Defeo et al., 1993; Castilla, 1997). Sometimes, regulations are difficult to implement because of high enforcement costs. This is particularly true for coastal benthic shellfish, in which the easy access to high unit value resources in open and extended (thousands of km in Chile, Peru, Argentina and Brazil) ocean coasts make shellfisheries difficult to manage: the elevated number of fishers cannot be controlled, management measures are costly and extremely difficult to enforce, and the easy access favors poaching (Jamieson, 1993). High enforcement and policing costs attenuate efficient resource allocation over time. In this context, the legitimization of the participation of fishers in the management process is seen as the only way to promote compliance with regulations (Castilla et al., 1998, and below).

(5) Uncertainty in short term economic factors. Intra-annual fluctuations in exploitation regimes might respond to short-run bioeconomic processes, such as: (a) variations in demand in local or foreign markets and in global supply; (b) price variations with individual size; and c) changes in species price according to intra-annual fluctuations in stock abundance and accessibility (Defeo and Castilla, 1998). These short-term economic uncertainties, coupled with open access regimes, a high unit value of stocks and low operating costs, generate a short-term increase in fishing effort and make resources more susceptible to over-fishing (Bustamante and Castilla, 1987; Castilla, 1990, 1994, 1997; Defeo et al., 1993; Seijo and Defeo, 1994).

Moving ahead: experimental management and co-management into practice

The small-scale fishery of the muricid gastropod (Concholepas concholepas), "loco", in Chile

The biology, ecology and fishery of the muricid gastropod (*Concholepas concholepas*) have been intensively investigated in Chile (i.e., Castilla, 1988, 1995; Castilla et al., 1998). *C. concholepas* is a unique muricid (category MIa), restricted only to the coast of Chile and southern Peru. Economically, it is the most important benthic shellfish mollusk extracted in Chile (Castilla, 1997). The fishery saga of "loco" has served as the flagship guiding the implementation of important, novel and adaptive shellfish management schemes in the country (Castilla, 1994). *Concholepas*' fishery is exclusively a small-scale artisanal activity based on divers. At present there are over 12,500 registered "hookah" professional divers in the country and most of them extract the "loco". Landing statistics are available from as early as 1960 (although there is no reliable information on fishing effort) and 5 sequential fishery phases can be distinguished (Figure 1, also see section on fishery extractive phases). The initial extraction phase (Figure 1), which started around 1960 and ended around 1975, was characterized by landings ranging between 2,000–6,000 metric t, exclusively for domestic consumption. The second, or expansion extractive phase, occurred between 1976 and 1980 (Figure 1). "Loco" landings abruptly increased due to exportation to the Asian markets (Japan, Korea), reaching peak landings of approximately 25,000 metric t with an export value of about US$50 million per year. There was a continuing demand from these markets, and the phase was unsustainable. The "loco" overexploitation phase occurred between 1982 and 1988 (Figure 1). Landings decreased to levels seen in the first phase, as a result of low standing stocks and the implementation of a set of operational management tools. The fourth phase was represented by the total closure of the fishery for 4 years (Figure 1) and was characterized by a series of socio-economic conflicts between the fishery authority, artisanal fishers (divers) and the small–scale artisanal associations (Payne and Castilla, 1994; Minn and Castilla, 1995). During this phase, poaching and illegal exports were common (Castilla et al., 1998 calculated that during this phase, poaching of "locos" amounted to approximately 5,000 metric t per year). The fifth, stabilization phase, started in 1993 (Figure 1) and may be seen as an embryonic stabilization and institutionalization stage. It is characterized by the implementation of new co-management and fisher participatory tools for the extraction of benthic resources, such as the Individual Non-Transferable Quotas (INTQ) and the Benthic Regime for Exploitation and Processing (BREP), contained in the Chilean FAL (1989). The law included the implementation of TURFs regulations exclusively for small-scale benthic shellfish artisanal communities (Moreno et al., 1987; Castilla, 1999, 2000a; Castilla et al., 1998) linked to the MEAs (Castilla, 1994; Payne and Castilla, 1994; Payne, 1996; Pino and Castilla, 1995; Castilla

Table 1. Comparison of size and CPUE between Management and Exploitation Areas (MEA) at El Quisco, central Chile, and nearby open-access diving areas for five benthic shellfish exploited in Central Chile. Mean (± 1 standard deviation) or ranges of mean values are provided. For "locos", the values for size and CPUE are daily means and the range represents the variation over time during the ban lifting; **s** indicates summer ban lifting and **w** winter ban lifting. For key-hole limpet, size values are shown by species, but only a combined CPUE for the three species is available because divers do not distinguish among them. Sea urchin mean size is reported for the two ban liftings in the MEA and for two open-access diving areas. CPUE for "locos" was reported as catch per unit of diving time. However, information on diving time is not available for sea urchins; only sailing time was recorded. Thus, the CPUE for sea urchins in open access diving areas is a combination of lower catches and longer time (sailing/handling). "Loco" data (1993) are from Castilla et al. (1998); sea urchin (1994–1995) from Castilla and Pino (1996), and key-hole limpet data (1994–1995) from Pino and Castilla (1995). [After Castilla and Fernández (1998)]

Species	Management and Exploitation Area (MEA)		Open-access diving areas	
	SIZE (cm)	CPUE	SIZE (cm)	CPUE
Loco				
(*Concholepas concholepas*)	110 to 117	280 to 540^{s}	103 to 108	15 to 143
	107 to 118	91 to 186^{W}		
Sea urchin	102.0 (7.0)	380 (132)	78.8 (5.5)	65.33 (30.4)
(*Loxechinus albus*)	97.1 (8.1)		87.6 (8.1)	
Key-hole limpets		729 (44.4)		391.8 (129.6)
Fisurella maxima	110.3 (12.7)		83.2 (17.1)	
Fissurella latimarginata	103.1 (9.4)		69.5 (11.3)	
Fisurella cumingi	97.3 (8.9)		73.8 (11.4)	

and Pino, 1996). Furthermore, the Chilean FAL (1989) regulated diving effort through the emission of "loco's" extractive licenses only for the professional divers involved in the fishery of "locos" in 1993 (freezing of the number of divers) and the implementation of Total Allowable Catch (TAC) and individual non-transferable extractive quotas per diver, both discriminated per region of the country. Recently, special loco extractive quotas, on top of the national ones, have been assigned directly to 2 MEAs in central Chile. Between 1993 and 1998 a stabilization of landings in around 5,000 metric t per year, was observed (Figure 1). Moreover, the maximum export value of locos peaked in 1993 to over US$60 million, for a total landing of approximately 8,000 metric t. More recently, landings have stabilized at around 3,000 metric t per year and their total export value at around US$10 million per year. This trend may be considered as a consolidation phase following our historical categorization of fishery phases.

Specific fishery, ecological, economical and community organization (societal) results obtained from several MEAs located in central Chile have been reported (Fernández and Castilla, 1997; Castilla and Fernández, 1998). We highlight the following: (1) joint fisher-scientist control over the evaluation of benthic invertebrate stocks within the MEAs, and an increase in credibility and linkages between fishers, scientists and managers. Fishers had a much better understanding of managerial regulations and the opportunities to challenge such regulations (Castilla et al., 1993, 1998); (2) Demonstration to fishery authorities and fishers of "loco", key-hole limpets and sea urchin, of the enhancements attained within MEAs, through a comparative analysis of CPUE and individual sizes between MEAs and open access areas (Table 1 and Castilla and Fernández, 1998). MEAs are closed for different time periods, based on management plans presented by the small-scale artisanal communities to the government; (3) Reinforcement of communal bounding association (syndicates within the Caleta units) and strengthening of the leadership in communities. This has led to greater protection of the resources (inside and outside the MEAs), including the implementation, by fishers themselves, of surveillance procedures to stop poaching and extractions by outsiders, and to establish participatory and regulatory rules within the communities. A sense of ownership, responsibility, pride and hope for the sustainability of the exploited resources has arisen. Furthermore, MEAs have helped the communal association of fishers economically (Payne and Castilla, 1994; Castilla et al., 1998). In general, MEAs, due to their reduced surface (up to approximately 200 hc), may represent approximately 25–30% of small-scale fisher earnings. Nevertheless, they have attached other important non-economic values greatly appreciated by fishers; and (4) Joint collaborations between scientists and fishers within the MEAs have facilitated the joint planning of biological, ecological and fishery observations, experiments and fishery ecosystem approaches (Fernández and Castilla, 1997; Castilla, 2000a). In MEAs, fisher-auto-control on the enhancement, exploitation and co-management of benthic shellfish exists, and as such they may be used as experimental fishery units (Castilla and Fernández, 1999).

The institutionalization of benthic shellfish co-management and community-based regulations in the Chilean FAL (1989), was initially aimed almost exclusively at managing the "loco" fishery. It has been extended to other benthic resources such as key-hole limpets (*Fissurella* spp.) (Pino and Castilla, 1995), the sea urchin (*L. albus*) (Castilla and Pino, 1996) and crabs (Fernández and Castilla, 1997). Castilla and Fernández (1999) reported that the government of Chile selected 168 coastal areas along the Chilean territory to be designated as MEAs. Approximately 20 benthic invertebrate MEA management plans have already been presented to the authority by small-scale fisher associations, including co-management of resources such as *C. concholepas*, *Fissurella* spp., *Thais chocolata, Aulacomya ater, Loxechinus albus*, the crab *Homalaspis plana* and the tunicate *Pyura chilensis*. Nevertheless, it must be pointed out that in spite of a few successful examples regarding the use of MEAs/TURFs/BREP/and co-management tools, and the advanced legislation contained in the Chilean 1989 FAL, the country is still far from implementing a significant enough number of MEAs to rationalize benthic fisheries. According to Montecinos (2000), from a total of 4,533,442 hc of MEAs made available by the Chilean government, the small-scale fishers' organizations have requested 66,920 hc so far, from which only 32,768 are effectively functioning. Chile has an important task for the future regarding the implementation of MEA's; so far, the large majority of the benthic shellfish catches originate from open access areas.

Castilla (2000a) has suggested that the combination of MEAs and the designation of Marine Protected Areas (MPAs, Reserves, Parks) along the Chilean coast may serve as a managing model integrating the rational use of benthic coastal resources and conservation (see also Castilla, 1996; Castilla and Fernández, 1999). Finally, the cooperation and transfer of knowledge between scientists, managers and small-scale fishers needs to be enhanced and linked through appropriate channels of communication. In the case of the Chilean MEAs and the implemented co-management strategies this has been done using a tailored Bulletin, aimed at the small-scale fishers: REMA Bulletin (**RE**population and **MA**nagement; see Oliva and Castilla, 1992; Castilla, 2000b).

The implementation of fisher participation in a framework of co-management, linkages with scientific and technical improvements, enhancement of communication channels and activation of social mechanisms are the essential factors for the rational exploitation of shellfish, and for building ecosystem approaches and understanding of their elasticity (resilience).

The yellow clam Mesodesma mactroides *of Uruguay*

The yellow clam (*M. mactroides*) (Mesodesmatidae) is a sedentary infaunal bivalve (SII) distributed along the warm temperate intertidal of the Atlantic coast of South America, from Brazil (24°S) to Argentina (41°S). This fast-growing, short-lived species (< 4 years: Defeo et al., 1992) is artisanally exploited (shovels and hand-picking) along hundreds of kilometers of sandy beaches in Brazil and Argentina, and along 22 km of Uruguay. *M. mactroides* is sympatric with the wedge clam (*Donax hanleyanus*, Donacidae) in a benthic community biomass, dominated by the former. Although *D. hanleyanus* is not commercially exploited, it is incidentally affected by harvesting (Defeo and de Alava, 1995).

The yellow clam fishery has been analyzed by the National Institute of Uruguay (INAPE) since 1983. Historical extractive phases of the fishery closely resemble those described earlier in our review. Catches were low before the decade of the 80's, in which official fishing statistical coverage did not exist. The expansion extractive phase began in the 80's. Landings increased up to 3.5 times in five years (from 62 metric t in 1981 to 219 metric t in 1985). Catches decreased more than 100% from 1985 to 1986, and in the first quarter of 1987 only 11 metric t were caught (overexploitation phase: see Defeo, 1987). The fishery was closed for 32 consecutive months, from April 1987 to November 1989 (Defeo, 1993a). The main idea underlying fishery closure between 1987 and 1989 was to consider this closure as a management experiment to investigate the effects of fishing on yellow clam demography. Coastal marine authorities, scientists and a well-defined group of local fishers, participated in the experiment, with the latter group also involved in enforcing regulations (Defeo, 1996a). The yellow clam in Uruguay is only found along the 22 km of sandy beach mentioned, in which the nearshore, benthic environment is affected by the freshwater discharge (Andreoni Canal) from a wide plain basin used for agriculture and cattle rearing (Lercari and Defeo, 1999). This gradient in habitat quality determines a continuous spatial gradient in abundance (Defeo, 1993a; Defeo et al., 1986) generating heterogeneous spatial distribution of fishing effort (Defeo, 1989). This precluded the possibility of performing a large-scale management experiment with spatial controls, thus testing hypotheses through time, using temporal controls as predicting units. Nevertheless, taking into account the longshore variability, from 1983 the area was divided into 4 fishing grounds to evaluate spatial differences in population structure and abundance, as well as in daily catch/effort data (Defeo et al., 1986, 1991). Fishery-independent surveys were done at least seasonally between March 1983 and March 1991 (see details in Defeo, 1996a, 1998; Defeo and de Alava, 1995).

Contrasting effort levels in the long term generated important changes in the abundance and population dynamics of the wedge and yellow clams. Fishing influenced the demography and abundance of *M. mactroides* and *D. hanleyanus* beyond the effects of exploitation; thus highlighting the ecological implications of humans in the system as extractors and also as a source of physical disturbance. Results are summarized as follows (see details in Defeo, 1996a, 1998): (1) the great recovery of *M. mactroides* was observed during fishery closure (Figure 2b). Mean adult density was lowest in 1987, just before the fishery was closed (16 ± 1 ind·m^{-2}), whereas it was more than 16 times higher two years later, at the end of the closed season (257 ± 30 ind·m^{-2}); (2) *M. mactroides* exhibited an overcompensatory stock recruitment-relationship (SRR). Highest adult densities, which occurred at the beginning of the third year of the closed season (1989) and immediately after the reopening of the fishery (1990), produced extremely low recruitment, whereas maximum recruitment occurred from moderately low and medium densities of adult stock (see also Lima et al., 2000). The wide range of fishing effort levels resulting from a variable spawning stock size, together with the closed season, allowed the determination of a dome-shaped recruitment curve with a relatively short time data series. Longshore trends in the abundance of adults and recruits showed a common response to gradients in habitat quality (Defeo, 1993a), and the spatial dynamics of fishing followed these spatial variations. This longshore pattern was also observed for the unexploited sympatric wedge clam (*D. hanleyanus*) (Defeo and de Alava, 1995) and the mole crab (*Emerita brasiliensis*) (Lercari and Defeo, 1999); (3) A maximum width of the clam bed, and also of the total area covered by the stock, suggested a limitation of available space at an adult density close to 120 ind·m^{-2}. This value was consistent with those suggested by the SRR as indicators of compensation. At a “quadrat scale” (0.0625 m^2), highest densities of recruits were never coincident with highest adult densities. This negative relation suggested a maximum carrying capacity of the system, with upper limits of the relationship representing maximum adult densities for varying levels of maximum recruitment (Defeo, 1996b; Brazeiro and Defeo, 1999); (4) Long-term individual growth patterns were density-dependent, with young-of-the-year *Mesodesma mactroides* growing slowest at highest adult densities achieved during fishery closure. Spatial analysis showed that growth rates and maximum sizes were lower towards the freshwater discharge. This spatial trend was also observed for the wedge clam (Defeo and de Alava, 1995) and the mole crab (Lercari and Defeo, 1999); (5) Natural mortality (***M***) rates of the young-of-the-year were low under null or low extracting levels, and were significant and positively correlated with the amount of fishing effort and catches per fishing ground. This suggests that extracting activities (viz digging with shovels) and related sediment modifications might constitute a source of incidental mortality. Fast growth in conjunction with low mortality during fishery closure (1988) led to a strong stock recovery; (6) Abundance of the sympatric unharvested wedge clam *D. hanleyanus* rose steadily throughout the fishery closure. Adult density (Figure 2b) was consistently below 40 ind·m^{-2} from 1983 to 1987, during the active fishery period. Immediately after fishery closure, adults increased by more than 435% between summer (26 ± 4 ind·m^{-2}) and spring (113 ± 35 ind·m^{-2}). After that, density increased to more than 100 ind·m^{-2}, peaking in the

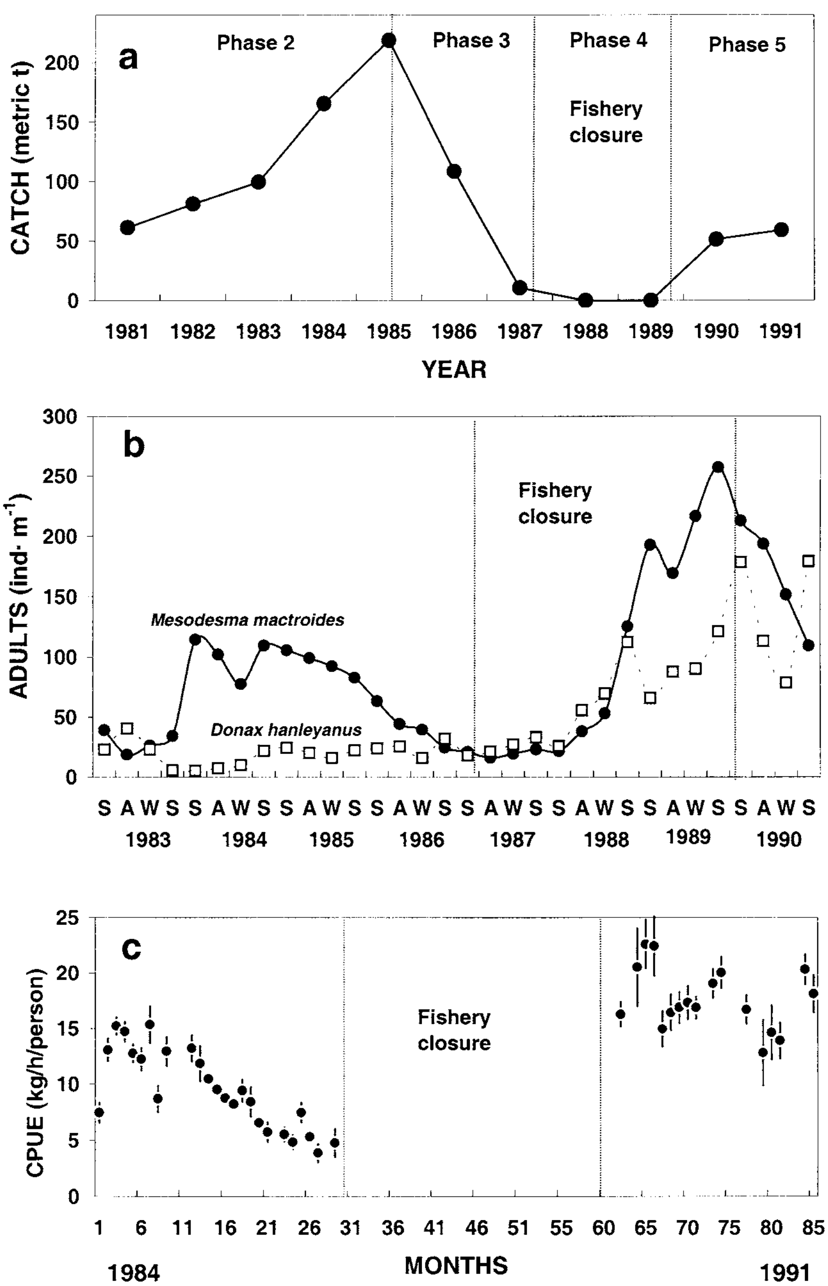

Figure 2. Yellow clam (*Mesodesma mactroides*) fishery of Uruguay between 1984 and 1991: (a) annual catches; (b) seasonal fluctuations in density of adults of *M. mactroides* (solid line) and *D. hanleyanus* (dashed line); (c) mean ($\pm$ standard error) monthly CPUE values (kg/hour/fisher). The closed fishery for *M. mactroides* (March 1987–November 1989) is indicated. [After Defeo (1996a, 1998)].

spring and summer of 1990 (179 ind·m^{-2}), immediately after the reopening of the fishery (Defeo and de Alava, 1995); (7) Recruitment of *D. hanleyanus* showed no evident relationship with the size of the parental stock. However, it was explained by a monotonically decreasing exponential function of yellow clam density. This "interspecific stock-recruitment Relationship" was evident only from 1983 to 1988. In 1989 and 1990, during or immediately after fishery closure, highest densities of both species occurred, suggesting that wedge clam recruitment could be influenced by the amount of fishing on yellow clam (Defeo, 1996b; Defeo and de Alava, 1995); (8) Spatio-temporal variation in wedge clam (*D. hanleyanus*) abundance was inversely correlated with fishing intensity over *M. mactroides*. Adult density during the exploited period (1983–1987) was negatively correlated with *M. mactroides* catches and fishing effort. Spatial analysis also showed that density of wedge clam recruits was inversely related to fishing effort exerted on *M. mactroides* per ground. A reflector of the same phenomenon was the inverse relation between recruitment density and yellow clam catch.

The information obtained was included in a spatially explicit bioeconomic simulation model (Seijo and Defeo, 1994) coupled with a multiple criterion optimization algorithm (Seijo et al., 1994). Simulation of alternative management strategies suggested that, if the population is closed to fishing activities for more than two years, high adult density could inhibit recruitment success and hence the magnitude of the stock available for fishing. Because of very low levels of the unit cost of effort, bioeconomic equilibrium occurs at high levels of fishing intensity, thus increasing the risks of fishery collapse.

The yellow clam fishery was reopened from December 1989 onwards, with a global quota and three additional operational management strategies (Defeo, 1993a): (a) a minimum catch volume per fisher; (b) a maximum number of fishers, estimated as the ratio between the net value of the overall quota and two times the minimum wages (Defeo, 1989). Priority was given to fishers with longer activity in the fishery; (c) A spatial management scheme, considering heterogeneity in resource abundance and fishing effort. Thus, fishing grounds with lower productivity were used as exploitation units when demand for clams was low (e.g., austral autumn and winter), whereas the grounds with highest stock densities were used in spring and summer (Defeo et al., 1986, 1991). The stabilization phase, after the closed season, lasted 2 years (1991–1992). During this phase, catches varied between 50 and 60 metric t per year, but CPUE was two times higher than in preclosure years (Figure 2c). Due to a decision by INAPE, the fishery was left as an open access system in 1993, which caused another collapse. Thus, the 6th fishery phase (consolidation) was not fulfilled.

Before this work, spatio-temporal patterns of distribution, abundance and dynamics of sandy beach macroinfauna seemed to be fully explained by wave climate, sand particle size, and tide range (McLachlan et al., 1995, 1996; Defeo et al., 2001). However, there was no quantitative evaluation of the effect of human harvest on the fauna of these systems, and information on biological interactions was also lacking. Hypotheses tested on population dynamics and demography of exploited and unexploited bivalves by manipulating fishing effort, showed that human activities and endogenous density-dependent factors play important roles in structuring sandy beach populations. Thus, the "physical stressed ecosystem" paradigm of sandy beach ecology and the way that ecologists and managers perceive this system was challenged after this study.

*The spiny lobster (*Panulirus argus*) fishery of Punta Allen, Mexico*

The spiny lobster (*P. argus*) is one of the most important benthic shellfishes along the Yucatan Peninsula (Mexico). Catch volumes registered in the States of Quintana Roo and Yucatan (770 metric t in 1997) situates this fishery as one of the most important artisanal lobster fisheries around the world. The unit price (tails) is about 10 US$/kg, even though during the second half of the 80's and the beginning of the 90's it reached values twice as high (Seijo and Fuentes, 1989; Castro-Suaste et al., 2000). "Hookah" divers equipped with outboard or inboard engines and air compressors dominate the fishery, even though skin divers are also present (Cabrera and Defeo, 2001). In the Mexican Federal Law of 1947, the spiny lobster fishery was mainly assigned to artisanal fishing cooperatives. In this context, a successful example of community-based management is given by the "Vigía Chico" Cooperative lobster fishery at Punta Allen (Ascension Bay, Quintana Roo State, Yucatan Peninsula, Mexico). The cooperative was formed in 1969 by 49 fishers to fish for the spiny lobster. This is the only significant economic activity at Punta Allen and the high degree of isolation promoted a high motivation on a self-help approach to community development (Seijo and Fuentes, 1989). Fishing operations within Ascension Bay are limited only to cooperative members, reflecting the traditional range of fishing operations. This area is also inserted within the range of Sian Ka'an Biosphere Reserve (528,147 ha), which assures low human intervention levels (Briones and Lozano, 1992). One of the main characteristics distinguishing Punta Allen lobster fishery from others in the area is the informal division of the inner Ascension Bay and back-coral reef areas in "campos" or fishing lots, within a depth interval of one to four fathoms (Miller, 1982). The Vigía Chico cooperative has divided the fishing area granted (324 km^2) into approximately 150 individual fishing lots (see Miller, 1989, 1994; Seijo and Fuentes, 1989). Each lobster fisher pertaining to the cooperative has exclusive rights to extract within the assigned boundaries. The temporary (renting) or permanent (selling) transfer of individual rights to fishing lots involves simple artisanal transactions: a specific payment is done according to ground size and its perceived relative productivity in previous years (Seijo, 1993). Permanent transfer of fishing lots among cooperative members may include monetary payments and/or barter transactions. As the spiny lobster has spatial variations in abundance, ownership or access rights to the most productive areas are given to those fishers with longer and continuous activity. Fishing lots are inheritable and might be transferable according to community-established rules (Seijo, 1993). A variety of penalties imposed by community rules, recognition of leadership and self-policing strategies have assured a relatively stable development of Punta Allen lobster fishery during the last decades.

The spiny lobster fishery for the State of Quintana Roo showed an initial phase that occurred from 1955 to 1961, with low (less than 20 metric t of tails) and constant landings (Figure 3a). After 1961, the fishery shows an expansive phase characterized by a strong increase in landings of lobster tails: catches increased about eight times between 1961 and 1988, mainly as a result of a strong demand from foreign markets (e.g., USA) and from newly developed tourist centers in the Caribbean coast of Mexico (Castro-Suaste et al., 2000) and also as a response to the introduction of habitat shelters or "casitas" for stock enhancement in 1968 (Miller, 1982). Lobster landings drastically decreased between 1988 and 1993. The underlying factors may be attributed to a combined effect of fishing intensity, low market prices, recruitment failure (Briones, 1994) and notably the environmental detrimental effects caused by Hurricane Gilbert (September 1988; Figure 3a) on the resource, on the "casitas" shelters and on the seagrass (*Thalassia testudinum*) used by the lobster as refugia (Briones and Lozano, 1992; Arceo et al., 1997 and references therein). Between 1994 and 1998 a fishery stabilization phase was seen. This was also observed for the large-scale spiny lobster fishery of the Yucatan State (Castro-Suaste et al., 2000). In contrast, the lobster fishery of Punta Allen's managed fishing ground showed fairly constant catches between 1975 and 1987, but the effect of Hurricane Gilbert was also noticeable in 1989 (Figure 3a). Low catches extended between 1988 and 1995, after which landings increased back to the 70's and early 80's level.

The Punta Allen's lobster fishery has been by far the most productive lobster cooperative along the Caribbean coast of Mexico, accounting for one seventh of the production of the Quintana Roo State (Miller, 1994). This high production could be attributed to the informal allocation of property rights among cooperative members. A comparative analysis carried out by Cabrera and Salas (1999) showed that Punta Allen had significantly higher catch and catch rates than other lobster cooperatives (e.g., Celestun and Ria Lagartos) along the Yucatan Peninsula (Figure 3b), where there is no strategy for regulating access to lobster fishing grounds. This had strong economic connotations: during the 1987–1988 Punta Allen's fishery seasons the total economic revenues were over US$1 million, with a peak price of US$20.50 per kg of lobster tail (Seijo and Fuentes, 1989; Lozano et al., 1989). Net revenues accounted for more than US$368,000, with an annual mean of approximately US$3,500 per cooperative member. At Punta Allen, poaching is minimal and operational management regulations (e.g., a closed season between March and June and a minimum legal size of ca. 8 cm of cephalotorax length $\approx$ 14.5 cm of tail length) are respected (Figure 3c). CPUE values (kg/boat/day) recorded through the 1981–1990 fishing seasons showed a maximum at the start (July) and a minimum at the end (February) of the fishing season (Figure 3c; Lozano et al., 1989; Lozano, 1991; Cabrera and Salas, 1999). Fishers outside the cooperative who violate the "campos" boundary are punished by the cooperative and the local marine authorities, in a sort of responsibility sharing or informal co-management procedure. At Punta Allen, stock

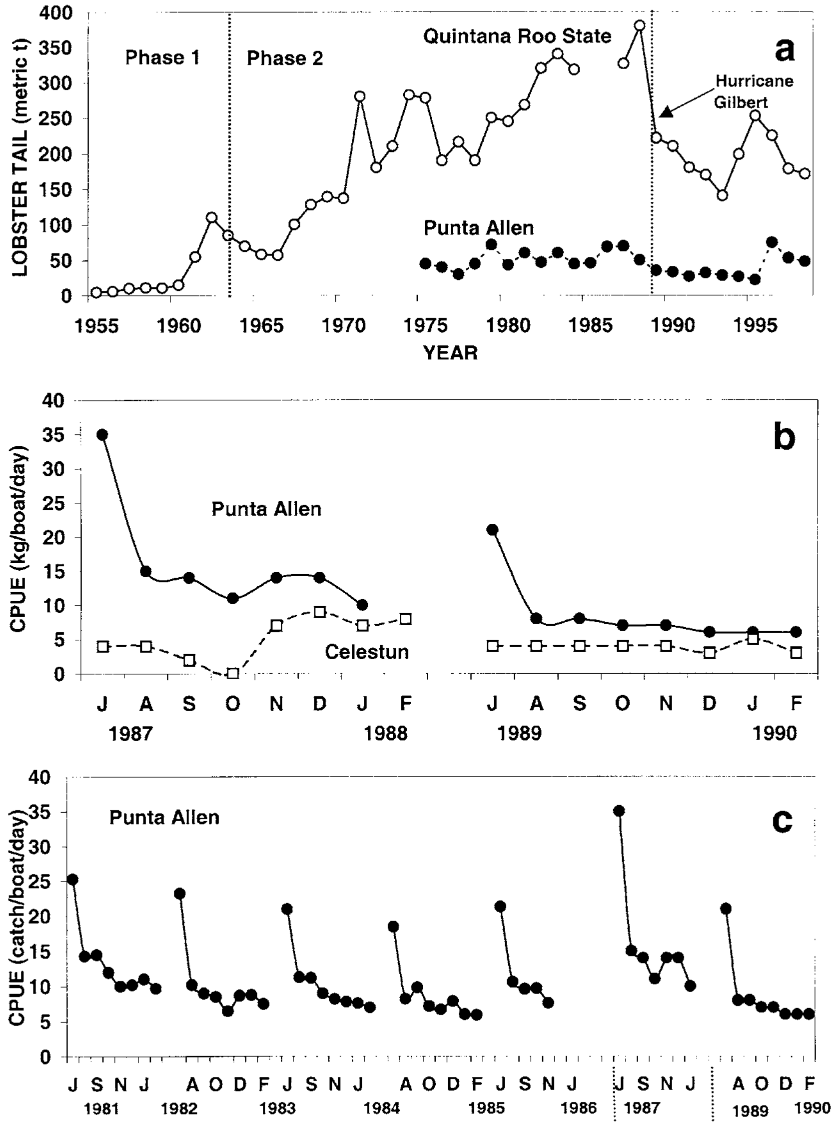

Figure 3. Spiny lobster (*Panulirus argus*) of Yucatan Peninsula, Mexico: (a) landings of tails for Quintana Roo State (1955–1998) and Cooperativa "Pescadores de Vigía Chico", Punta Allen (1974–1998); (b) CPUE (kg of tails/boat/day) for fishers of Punta Allen and Celestún for the fishing seasons 1987–1988 and 1989–1990; (c) monthly mean CPUE values for 7 fishing seasons (1981–1985, 1988 and 1989) between July and February. [After Lozano et al. (1989) and Cabrera and Salas (1999)].

enhancement is promoted through artificial enhancement habitats, "sombras" or "casitas" (Miller, 1982, 1994 and references therein). By 1988 there were 14,210 "casitas" pertaining to Punta Allen cooperative. Although the spiny lobster is a mobile shellfish (MIb), its territorial behavior together with the use of "casitas" enhance the probability of remaining longer within a fishing lot. The differential productivity favors the allocation of territorial use rights to community members in specific fishing lots.

Enforcement becomes more difficult as the number of fishers, landing sites and regulated species increases. For instance, the successful example of Punta Allen cannot be extrapolated to the whole lobster fishery of the Caribbean coast of Yucatan Peninsula, where the fishery is over-capitalized and the conflicts with recreational fishers are frequent (Miller, 1994). The isolation of Punta Allen, the well-organized cooperative and the exclusive access to the fishing lots are the best explanations for its success and long-term persistence as a community-based management system. Fishers have also participated in, and directly benefited from, research efforts related to geographic specification of fishing lots and estimates of fishery yield (Seijo, 1993; Cabrera and Salas, 1999).

Innovations

Co-management, self-government and property rights

We showed that co-management constitutes an effective institutional arrangement by which fishers and managers could interact to improve the quality of the regulatory process and to sustain Latin American shellfish over time. The most important factors supporting this statement can be summarized as follows:

(1) A comparatively reduced scale of fishing operations and well-defined boundaries for each management sub-unit. Invertebrates comprised within categories SEI, SII and SMIa, constitute spatially explicit structured stocks with "strong and persistent" spatial structure (*sensu* Orensanz and Jamieson, 1998; see also Caddy, 1975; Conan, 1984; Orensanz, 1986). Spatial distribution patterns of stock abundance are heterogeneous, and the spatial dynamics of the fishing process follows closely these spatial variations in abundance when considering small sub-areas (e.g., the area of a cove or a fishing ground). Thus, the fishery could be composed of several discrete fishing grounds in which the dynamics of the stock and the fishing process are studied independently, treating each ground as a small-scale management unit defined by discrete physical boundaries (Defeo et al., 1991; Defeo, 1998; Castilla et al., 1998). The scale of the management unit should ideally be that of each fishing community, thus facilitating the application of co-management, as demonstrated by the successful Chilean examples. However, features related to the life history of the shellfish should be considered when delineating the spatial scale of the management unit (Orensanz and Jamieson, 1998; Fernández and Castilla, 1997; Prince and Hilborn, 1998).

(2) Allocation of institutionalized co-ownership authority to fishers. The study cases illustrated the importance of fishers' co-ownership and responsibility in shellfish management decisions and actions (Pinkerton, 1994; Gimbel, 1994; Pomeroy and Williams, 1994). Participation of fishers improved shellfish management: the perception of ownership by the fishers is the most important focal point determining co-management success (Castilla, 1994, 1997; Castilla et al., 1998). The related experiences also showed that co-management must be preferred over open access. Under common property, the community has the right to exclude other fishers (Berkes, 1994), thus closely resembling "a private property regime for a group of co-owners" (Bromley, 1991; Waters, 1991; Mantjoro, 1996). Co-management encompasses different degrees of power sharing between fishers and managers (Hanna, 1994). This ranges from an institutionalized framework (Chile) and grant of "campos" to groups of fishers (Mexico) to informal, extra-legal establishment of communal power assigned to local fishers to defend the resource against outsiders (Uruguay). However, informal government recognition is not enough for allocation of territorial or fishing rights and *ad hoc* implementation of co-management or community-based systems. The yellow clam example (Uruguay), which included the voluntary participation of the fishers in enforcing regulations, became unsuccessful years later, due to changing management policies. Fishers felt themselves unprotected under an uncertain management environment, and changed a long-term, "sustainable" perspective of the fishery to a short-term, profit-maximizing behavior. Thus, shellfish co-management needs to be institutionalized in a legal framework including well-defined fishers' rights, responsibilities and a clear identification of the communal role in the management process. Castilla (1994, 1997) highlighted the importance of clarifying local responsibility and authority of the artisanal community, which in turn increases the willingness of the fisher to participate. The author showed that the small-scale fishery management rules institutionalized in the 1989 Chilean FAL were critical to define the role played by coastal MEAs and the future role foreseen for MPAs (Castilla, 1996, 2000a). The legitimacy of co-management and the perception of ownership by Chilean small-scale fishers (divers) override the expectations of the benefits to be derived from shellfish extraction. The assignment of fishing grounds to well-defined groups of fishers ("Sindicatos") represents recognition of the role of small-scale communities in conservation and management. Legal mechanisms included in the 1989 Chilean FAL and related regulations have promoted a decentralization of the responsibility to take care of the allocated MEAs.

(3) Communal ownership encourages cooperation among fishers. Results of the Chilean experience have shown that well organized fisher communities take good care of their assigned fishing grounds by preventing illegal extractions. In the case of the resource "loco" this had major repercussions in yield levels, product quality (individual sizes far above the minimum legal size permitted) and economic returns (Castilla, 1997; Castilla et al., 1998). On the other hand, those Chilean "Caletas" (coves) with unorganized communities were affected by illegal fishing and did not yield the catch quota after fishery reopening (Payne and Castilla, 1994). The relatively stable development of the Punta Allen common lobster fishing grounds and some Chilean communities (e.g., El Quisco) rely on the solid motivation for a collective organization, and penalties imposed by strong operational rules specified, enforced and controlled by local fishers. Furthermore, in the case of Punta Allen the relative isolation of the community and the restricted scale of the territorial permit favor the implementation of self-policing strategies and a voluntary cooperative action to avoid infringement of rules (Seijo, 1993).

(4) Improvement of the quality and quantity of fishery information. Cooperation among scientists, fishers and managers exponentially increased the quality and quantity of fishery information, with clear management connotations (McCay, 1989). Information flow under co-management reduced misreporting and uncertainties inherent to stock assessment (particularly when the fishers form part of stock assessment teams, as in the Chilean MEAs), mainly related to fishing effort estimates, landings and landing-based estimators (commercial sampling) of population structure per fishing ground. Information on the spatial dynamics of fishing effort and economic indicators (fixed and variable operating costs, ex-vessel species prices) has also been improved (Defeo and Castilla, 1998). Cross-fertilization between large-scale and long-term field experiments and co-management constituted a synergistic complement to acquire knowledge about the dynamics of the stock and the fishery (Pinkerton, 1994, 1999; Jentoft et al., 1998).

Fishers play an outstanding role in the implementation and surveillance of regulations, diminishing information and enforcement costs. High transaction costs in fisheries include information, enforcement and contractual costs, which attenuate an efficient allocation of resources over time (Seijo et al., 1998). Information costs are often high and vary according to stock characteristics and the nature and extent of the data collected to monitor the fishery. In this setting, transaction costs incurred in a fishery are lower in a bottom-up or co-management approach, when compared with a top-down approach based on scientific information gathered in a centralized management control (Hanna, 1994). For example, joint discussion between scientists and fishers on fishing effort components (e.g., handling, travel and searching times) provides valuable information about how to interpret CPUE estimates, improving the quality, quantity and reliability of stock assessments (Prince, 1989; Defeo, 1993a). This information flow implies lower monitoring and enforcement costs, and provides fine-grained signals about resource status, which allows setting spatially explicit management measures (e.g., ground closures). Implementation of regulatory measures in a co-management context generates a great incentive to fishers to adhere to and get involved with enforcing regulations, thus reducing the probability of occurrence of free-riders and illegal fishing (Defeo, 1989; Castilla, 1994). The fact that each fisher group permanently resides near the management unit improves information flow and interaction with scientists and enables the identification of each extractor to study short-term differences in fisher's behavior. The individualistic character of the small-scale shellfisheries, and generally speaking, the lack of by-catch problems, also simplify estimates of the effective fishing effort applied and allow estimating between-fisher variability in fishing power.

(5) Existence of community fishery traditions. Fisher communities that have taken the responsibility of managing coastal shellfish resources, exhibit old traditional roots (Castilla, 1994; Balazs, 1998; Johannes, 1998). As shown in the Chilean and Mexican examples, ancient collective voluntary organizations in coastal shellfisheries include strong community rules and self-policing tools. This constitutes a key condition to mitigate the fisheries social trap (*sensu* Schelling, 1978) of inconsistent short-run micromotives (e.g., short-run profit maximizers) with long-run desired results (e.g., resource sustainability) (Seijo et al., 1998). The marked development and expansion (see fishery phases) of Latin American shellfisheries intensified rates of extraction and encouraged traditional communities to change their way of commercialization from subsistence to semi-industrial

or industrial production. Even though this requires a different management perspective, institutionalization of co-management or community-based management must consider traditional community ways of managing the resource, as well as ancient social and cultural factors influencing the development of fisher organizations. There is a high risk of displacement of traditional rules by some legislation that does not consider the customary sea tenure system traditionally enforced by the fishers. The size of the group of fishers is a relevant factor affecting the avoidance of this social trap. Ideally, there must be a limited number of fishers and few external threats (Seijo et al., 1998). Small groups with clearly defined members and leadership, as mentioned for the spiny lobster fishery (Seijo, 1993), the "Caletas" of Chile (Castilla et al., 1998), and the yellow clam in Uruguay (Defeo, 1987, 1989), encourage cooperation and promote the identification and exclusion of non-contributing users. Thus, trust among fishers and group cohesion are necessary conditions to improve co-management (Pomeroy and Williams, 1994).

(6) Increasing the possibility of allocation of territorial use rights in fishery (TURFs). The examples reported here on Chilean shellfisheries and the spiny lobster fishery of Punta Allen illustrate the strong potential that the apportionment of TURFs has, when accompanied by a co-management approach. In Chile, the allocation of TURFs among communities that extract benthic shellfish is an efficient tool to cope with overexploitation concerns generated during the second half of the 80's. Allocation of TURFs to fisher organizations ameliorated the weaknesses of enforcement regulations and the high transaction costs in a country with more than 4,200 km of coastline (Castilla, 1994, 1996; González, 1996), improved the status of shellfisheries (e.g., increasing abundance and individual sizes; see Castilla et al., 1998). The formal allocation of TURFs to fisher organizations such as the collectively managed spiny lobster fishery of Punta Allen (Mexico) constitutes another sound example (Seijo, 1993). Another particularly well-documented example in developing countries is the traditional management of fishing grounds by the Para community, in Indonesia (Mantjoro, 1996).

Given the current state of Latin American benthic shellfish stocks, the recent collapse of some of them, and the continuing declining trend of many resources, there is a need to refocus efforts on management structures. Successful examples described here suggest that shellfish management might be a hybrid of traditional and modern arrangements. The community may allocate extraction quotas, access rules and self-policing strategies among fishers (e.g., Punta Allen, El Quisco), whereas the government should retain the authority to modify the management plan by setting or modifying operational management measures (e.g., legal sizes, closures, gear regulations).

The absence of co-management practices is a critical factor that generated the collapse of coastal small-scale benthic fisheries in Latin America. Scientists and policy-makers must learn from the various forms of community-based management traced for centuries by the traditional fisher communities. This is precisely the opposite way international fishery management bodies have operated over the past 30–40 years. The fruitful interaction among fishers, policy makers and scientists provide a comprehensive course of action in scope.

The short/medium term solution to the dangerous overexploitation levels in Latin American small-scale benthic shellfisheries should not exclusively come from computer-intensive and sophisticated modeling techniques developed by "in-house" fishery biologists. The best management experiments will fail when exclusively based on a hundred sentences of a computer program aiming to describe the short and long-term behavior of fishers, without integrating the fishers as direct participants in the flow and interpretation of information concerning their own behavior in the field. It is not only by setting "a precautionary approach to management" (Caddy and Mahon, 1995) that the effectiveness of coastal benthic shellfish management will be improved. Easy access to resources, high transaction costs and the typically high unemployment levels in rural Latin America promote illegal fishing and poaching, thus dissipating the economic rent generated by the fishery, even under risk-averse management schemes. In this context of uncertainty, it is imperative to develop and establish a legal framework formalizing community responsibility in the management process. This should preserve traditional rights of use and access to the resources. Once this strategic institutional arrangement is implemented, additional, risk-averse, precautionary management schemes could be set.

Experimental management

Controlled field experimentation has unfortunately played a minor role in developing fisheries management theory (Caddy, 1999). Experimental design in the management of fisheries was reviewed by McAllister and Peterman (1992). The authors concluded that most of the approaches to fishery management have been non-experimental and results (unexpected or expected) derived from fishery management actions often lead to confusion, since management manipulations are not originally designed to distinguish rigorously between alternative hypotheses (see also Larkin, 1978, 1984). A step forward on the matter is recognized in the adaptive management approach, originally proposed by Holling (1978) and later on implemented by Walters (1986) and coworkers (i.e., see Walters and Holling, 1990; Walters, 1997 and references therein). Adaptive management can be defined as a structured process of "learning by doing" involving a modeling step and the design of management experiments (Walters, 1997). The term "adaptive" refers to a dynamic management strategy, by which a sequential reassessment is done as more accurate information about the fishery is obtained. Walters and Holling (1990) defined 3 adaptive management approaches: (1) *deferred action*, by which the fishery system is managed by "trial and error" decisions until sufficient information about the stock and the fishery is achieved; (2) *passive adaptive*, where previous information about the fishery is used to build a deterministic fishery model and perform management actions to be improved through time; (3) *active adaptive*, by which the short and long-term fishery performance is continuously improved by experiments consisting of different management policies. Thus, management decisions are adjusted on the basis of updated information (also see Halbert, 1993). An adaptive process regularly evolves to the recommendation of additional management experiments. This has led to the interchangeable use of the concepts of "adaptive management" and "experimental management" (Walters, 1997). Walters and Hilborn (1978), Walters (1986) and Walters and Holling (1990) have discussed the main challenges to implement and design fishery experimental management programs through an adaptive process. However, after 20 years from the initial proposition, Walters (1997) concluded that, in the case of riparian and coastal ecosystems, 'adaptive management programs have failed to produce useful models for policy comparison or good experimental management plans for resolving key uncertainties" (but see successful example in Sainsbury et al., 1997).

In this review we showed that, fortunately, large-scale fishery experiments (Castilla et al., 1998; Castilla, 2000a) do play an important role in the evaluation of alternative management policies in Latin American benthic shellfisheries, especially when they explicitly involve the participation of fishers in field experimentation. The exclusion of humans from rocky shores (Reserves) in Chile allowed testing effects of handpicking and fishing (diving) on shellfish abundance and community elasticity (Moreno et al., 1984, 1987; Castilla and Durán, 1985; Castilla and Bustamante, 1989). Unreplicated experiments in Central and Southern Chile demonstrated that humans, as specialized top predators, constitute the key factor (Moreno et al., 1984; "capstone" *sensu* Castilla, 1993), altering exploited and unexploited benthic coastal populations, generating ecological cascading effects that affect the structure and functioning of communities (Castilla, 2001). Varying extraction intensity on species of different trophic levels may translate into different community structures, thus enhancing the identification of linkages and strengths of ecological interactions. This information was used by scientists to understand system elasticity and to translate ecological knowledge into management strategies.

Benthic shellfish showing low mobility and/or territoriality may facilitate the experimental requirements of replication, control and contrast between treatments in a relatively small and manageable spatial scale. This also enables each management area to be treated as a separate experimental unit. Hence, benthic shellfish best suited for practical-experimentation are those with persistent spatial structure (Orensanz and Jamieson, 1998), which fall in the categories SEI, SII (yellow clam), MIa ("loco") and MIb (spiny lobster). Highly mobile benthic-demersal species are not as tractable, and could obscure the results of management experiments because the appropriate spatial scale for experimentation is not considered. For example, Fernández and Castilla (1997) failed to detect differences in CPUE and individual sizes in the mobile stone crab (*Homalaspis plana*), between MEA's and open access fishery grounds in Chile.

Concerning larval attributes, species showing direct development or short-lived planktonic larvae should be most amenable for conducting experimental work, because the degree of connectance between experimental units through larval dispersal could be considered negligible or low. This has consequences in evaluating the effect of different fishing intensity and stock sizes on productivity between experimental units. For example, the spatial relationship between resident adult abundance and recruitment in each unit could be ideally estimated as in a closed self-sustained population (e.g., see Clarke et al., 1999), thus attenuating the uncertainty given by the highly variable exportation/importation rates of larvae derived from "source-sink" metapopulation dynamics (Rowley, 1994). The sedentary nature of the species analyzed in this review allowed the performance of localized experiments with different levels of stock abundance and fishing intensity (e.g., MEA's versus open access areas). The low exchange of adult organisms among experimental replicates is a useful feature that facilitates the evaluation of population responses that might accompany dissimilar fishing levels (Prince, 1989, 1992; Castilla et al., 1998, Lima et al., 2000). This holds true even for mobile shellfish with high territoriality patterns such as the spiny lobster. Habitat enhancement and substrate preference experiments performed in Punta Allen (Eggleston et al., 1990, 1992), strongly contributed to the knowledge on stock enhancement activities actually carried out by local fisher communities along the Caribbean Sea and Yucatan Peninsula (Castro-Suaste et al., 2000). Such results highlight the potential role of applied ecological research and experimental marine ecology with regards to the establishment of marine protected areas (MPAs) for conservation and management purposes (Bohnsack, 1993; Castilla, 2000a).

The location and discrimination of experimental grounds and their boundaries were easy to achieve both by scientists and fishers, particularly when the latter were integrated into collaborative management (Defeo et al., 1991; Castilla et al., 1998). The precise location of grounds provided reliable area-specific estimates of population density and structure, catch, and fishing effort. Fishery-independent surveys conducted in the loco and yellow clam fisheries, provided reliable estimates of stock status that facilitated allocation of catch quotas for each MEA (loco) or fishing ground (yellow clam). Measurement and estimation errors were minimized, as both quantities were estimated *in situ* for each fishing ground (Castilla et al., 1998). This reduces uncertainties in the outcome of a management policy (Prince, 1989; Prince and Hilborn, 1998; Iribarne et al., 1991; Seijo and Defeo, 1994). Management experiments focused on fast growth, short-lived benthic shellfish, such as the yellow clam (Defeo et al., 1992), were especially useful to evaluate the short-term rate of stock rebuilding following a perturbation, as well as area-specific variations in growth and mortality rates. This allowed a fine-tuning of management measures according to the spatial structure of the stock and its population dynamics (Defeo, 1993a, 1998).

The wide variation of fishing effort on benthic shellfish populations enabled us to acquire critical scientific information to improve management. This includes the establishment of closed seasons as a *de facto* management experiment, which was particularly useful in the yellow clam case for evaluating the capacity of passive restocking of depleted areas and for quantifying demography features of targeted (yellow clam) and sympatric unexploited (wedge clam) species. Identified mechanisms include overcompensation (yellow clam) and an interspecific stock-recruitment relationship (wedge clam) resulting from competitive asymmetries (see references in the *Mesodesma* study case). These ecological results, obtained through varying levels of exploitation intensity, were easily incorporated in simulation modeling, which facilitated decision-making (Seijo and Defeo, 1994).

The establishment of "no take areas" such as ECIM at Las Cruces (Chile) and Mehuín, in southern Chile, and the comparison of stock abundance and shellfish demographic features both within and outside human-excluded areas was a useful tool for management and conservation purposes (Moreno et al., 1984; Castilla, 1990, 1994; Castilla et al., 1998). This experimental approach could be especially attractive in newly developed fisheries, in which little or no information on stock dynamics is available (Jamieson and Caddy, 1986; Cobb and Caddy, 1989; Perry et al., 1999).

Finally, experimental management, when used jointly with co-management, had the potential to promote conservation and enhancement of benthic shellfish stocks. The reliability of the acquired information and the management alternatives resulting from this information are legitimated by the participation of scientists, decision makers and fishers. For instance, experiments conducted at the Chilean MEA at El Quisco generated sound and reliable scientific information about the dynamics of the stock and the fishery, including information (notably traditions) coming from the fishers. We suggest that the key factor through which benthic invertebrate experimental management (e.g., Chile) is showing success, when compared to active adaptive management (*sensu* Walters, 1997), is the fact that the former is being developed in a co-management scenario where self-governing and property rights are the corner stones.

Spatially explicit management framework into practice

The intensity of spatial structure in a shellfish stock defines the nature and extent of allocation of fishing effort, which in turn affects management options. Thus, shellfisheries should be analyzed by identifying homogeneous areas able to be treated as independent management units (Orensanz, 1986; Defeo et al., 1993; Seijo et al., 1993). This is the reason why conventional aggregated management measures such as global quotas (e.g., TAC) tend not to be useful: shellfish management criteria should be spatially explicit. The definition of the adequate spatial and temporal scales of implementation of management tools should incorporate features of the life history and the fishing process (Caddy, 1989; Fernández and Castilla, 1997; Orensanz and Jamieson, 1998).

The reviewed Chilean and Uruguayan cases showed that the spatially discrete analysis of stocks, the surrounding environment, and the fishing process were useful tools to: (i) assess the spatial dynamics of catch and fishing effort; (ii) detect changes in length or age composition of the catch that could explain variations in economic benefits; (iii) monitor changes in stock abundance and structure, as well as in the biological community composition and environmental variables; (iv) quantify area-specific variations in growth, recruitment and fishing and natural mortality; and (v) evaluate the relative importance of regulation mechanisms. These discrete bioeconomic results were integrated afterwards to develop a comprehensive management scheme (Seijo and Defeo, 1994).

Orensanz and Jamieson (1998) followed the main concepts developed by Charles (1995) to suggest operational and institutional strategies to set shellfish management regulations. They include institutional strategies mentioned in this review: government-based management, co-management, community-based management and different forms of allocation of property rights (e.g., individual quotas). Spatially explicit benthic shellfish management measures include: TURFs, refugia for spawners or other life-history stage or habitat, MPAs, rotation of fishing grounds and area closures, adaptive and experimental management with spatial controls, and area-specific stock enhancement activities. Minimum size limits and quota settings should also be set on the basis of spatial differences in productivity, growth and mortality rates, mean size/age at maturity and recruitment patterns.

TURFs and MPAs are key operational instruments (see the Chilean example). MPAs might fulfill objectives for conservation and management, and, at the same time, serve as robust experimentation tools (Castilla, 1999, 2000a). Establishment of TURFs in Chile was also successfully used to regulate access to benthic resources by divers through the development of the MEA concept. The same holds true for the allocation of grounds among fishers of the Punta Allen lobster fishery in Mexico. Prince and Hilborn (1998) and Prince et al. (1998) also suggested that TURFs offer the greatest potential benefit as a regulatory scheme for the Tasmanian abalone (*Haliotis* sp.), pertaining to our shellfish category (MIa).

Our results show that some degree of management redundancy (Caddy, 1999) is needed to achieve a sustainable exploitation of benthic shellfish resources. Management redundancy is defined as the implementation of operational and institutional management instruments directed to achieve the common long-term goal of sustainable exploitation. The examples provided in our review include a comparative synthesis of the relative usefulness of alternative spatially explicit management tools under a framework of management redundancy. Hence, in Chile, the implementation of TURFs, accessed only by duly organized artisanal fisher communities, was accompanied by resource size limits, area-specific quota settings, and the institutionalization of co-management through the 1989 FAL (Castilla, 1994 and this review). In Uruguay, individual quotas per fisher and global quota settings were accompanied by size limits and a spatial management scheme including between-ground differences in stock productivity and seasonal differences in market demand and resource accessibility. Rotation of fishing areas (Pfister and Bradbury, 1996) and granting of TURFs together with stock enhancement activities through natural restocking, seeding and transplanting, constitute another useful way of redundancy in management regulations (Caddy, 1999). Spatially explicit management tools are not mutually exclusive but should be simultaneously used to diminish the risk of overexploitation. Operational management measures alone, when not accompanied by co-management and spatially explicit allocation of property rights, offer too little management redundancy. When global catch/effort quotas were implemented by a centralized, government-based "top-down" approach, the exclusion of fishers from policy formulation and implementation during the management process precluded success and thus resources remain overexploited (Defeo, 1987, 1989).

The short and long run spatial behavior and resource perception of artisanal fishers is a key issue in the definition and implementation of shellfish management actions. For instance, fisher decisions that determine the spatial pattern of effort allocation respond to the spatial pattern of shellfish abundance, distance from port, traditional or legal territorial boundaries, accessibility and market demand (Caddy, 1975, 1999; Defeo et al., 1991; Defeo and Castilla, 1998). Intra-annual variations in area-specific harvesting levels are found according to short-term variations in resource demand and accessibility. Thus, spatial and temporal coupling of management measures through specific "area-season windows" (Caddy, 1999) are needed to consolidate a sustainable management framework for shellfish.

Constraints and future needs

Logistics, scales, and the design of management experiments

Controlled field experimentation could play a major role in developing fisheries management solutions for shellfish stocks. The application of scientific protocols in a rigorous falsificationist procedure or in a Bayesian perspective (Hilborn and Mangel, 1997; Castilla, 2000a and references therein) should aim to legitimate possible explanations for the impact of fishing in benthic shellfish marine populations and communities. Ideally, there should be several experiments and controls over temporal and spatial variability, randomization, contrasting treatments (e.g., fishing effort levels) and replication. However, the appropriate spatial and temporal scales for replication should be sufficiently large to make the experimental falsificationist procedure difficult to follow or prevented by prohibitive logistical constraints (Underwood, 1990, 1997; McAllister and Peterman, 1992). This particularly occurs in long-lived and/or highly mobile shellfish, in which ecological responses of populations or communities to contrasting fishing effort levels arise at decadal time scales or at spatial scales larger than the selected management units.

The above strongly suggests that the scale at which a management experiment makes sense for improving shellfish stock status might vary according to characteristics of the life history of the species targeted, notably larval duration and habitat requirements, and the fishing process. For example, the scale for setting a management unit or implementing TURFs will increase from SII species and direct developers to MIb species with external fertilization, high fecundity, and long-lived, free floating larvae. In these species, structured as metapopulations at large spatial scales (Orensanz et al., 1991), the design of MPAs or TURFs should rely on replenishment patterns of local populations (mesoscale), given by the distance of larval dispersal and connectance between populations, and the hydrodynamics features within the domain of the metapopulation (Carr and Reed, 1993; Allison et al., 1998). In this context, very little is known about dispersal duration and mechanisms influencing meroplanktonic larval phases of exploited benthic shellfish in Latin America, and the point at which physical forces become sufficient to override any active component, has not been adequately examined (Defeo, 1996b). Alternative hypotheses should test effects of mechanisms of retention or dispersion of larvae and the long-term variability in area-specific repopulation mechanisms of the parental area, despite the dispersal of planktonic phases. Lack of information on these issues could seriously affect the outcome of ecological and fishery experiments and the efforts directed to the implementation of MPAs and TURFs.

The boundaries selected for each management subunit are generally based on the physical structure of the coastline (e.g., Chilean "caletas"). Even though fishers, managers and scientists could easily observe and understand the limits of "caletas", these are not the boundaries defined for the life history habits of benthic shellfish, particularly those falling in categories MIa and MIb. For instance, Fernández and Castilla (1997, 2000) demonstrated a lack of differences in CPUE and individual sizes of the stone crab (*H. plana*) between open access zones and MEAs, because habitat complexity and wave action, the main factors explaining the species distribution, were not taken into account for designing MPAs or assigning MEAs, initially selected for "locos" (*C. concholepas*). This becomes most difficult in transboundary shellfish with long-lived pelagic larvae such as the spiny lobster (i.e., 6 to 11 months, see Briones, 1994). The above suggests that mobile species may offer a new challenge to design and implement area-specific management tools. In such cases, however, measuring the effects over relatively large space-time scales is costly and/or slow, and risky, and becomes a trade-off given by the potential benefits and costs of the experiment.

In summary, scales for designing management experiments or for the implementation of operational and institutional management arrangements should be in agreement with the scales of dynamics of the benthic shellfish stock to be managed. Experimental management units (e.g., TURFs) should be adapted to a fine-tuning procedure that must take into account area-specific differences in the spatial structure and population dynamics of the stock. In multispecies shellfisheries however, a hierarchical approach should be used to select the criteria for experimentation or TURFs implementation, on the basis of the life history of those highest valued for the society. This implies a trade-off between conflicting criteria given by the scale of implementation of a given area-specific management measure and the probability of response of the benthic shellfish species (all categories combined) to fishing pressure. An optimization process might be used to assist in ensuring that the best possible trade-offs are made, thus involving seeking "Pareto optimal solutions" to this multiple criterion optimization problem (Seijo et al., 1998).

One implication of a multispecific approach is that the components of fishing effort become more heterogeneous and therefore more difficult to link to a single population. Thus, experiments directed to promote stock enhancement could consider competitive release or a diminution of predation effects on species highest-valued in the market. An experimental manipulative approach would provide more reliable means of guiding community structure to a desired state than when based on predictions of a single species model (Sainsbury, 1988; Sainsbury et al., 1997). However, experiments must be designed with caution, because the complex dynamics in multispecies assemblages precludes a synthetic forecast of the ecological outcome of manipulation of species abundance.

Conservation and management

The conservation and management of wild shellfish, coastal communities and ecosystems are part of human goals. Conservation can be defined as the administration (management) of the biosphere by humans so that the protection, use and sharing of benefits is done in a sustainable way (Castilla, 1996). Therefore, conservation and management should not be seen as contradictory but complementary concepts (Castilla, 2000a). In the past century, this has not been the mood and "protection/conservation of marine resources and habitats" has been seen as the antonym for "exploitation/management of marine resources and habitats". In our view, this has been one of the main constraints to develop theoretical and practical comprehensive approaches to solve the paradigm of sustainable use of marine resources and habitats.

On the other hand, culture of invertebrates and fish has shown that truly freshwater aquaculture, based on herbivorous or filter-feeder species directed to "*create protein*" and increase production, has not occurred at the predicted rate (the best examples are found in Asia and Africa). Instead, mariculture of high trophic level species, such as salmon and trout, which "*transform protein*", has been more successfully developed. The intensive farm-raising of high-value species, such as shrimp and salmon, is far from trouble free (Castilla, 2001). The commodity value of these products is very high; nevertheless, aquaculture and mariculture had not significantly helped (with seldom localized exceptions) to solve the problem of the lack of protein in the population, partly derived from the overexploitation fishery crisis (Botsford et al., 1997).

In marine systems, where property rights do not exist (but see review on TURFs), the joint administration of wild shellfish resources and/or habitats for conservation and management represents an important challenge. Furthermore, the integration of fishers into both processes appears to be a key factor. The fishers (shellfishers) and not the fish should be the main target. Countries where this situation has been partially achieved, due to the existence of long standing fishery traditions or legislation (Castilla, 1994; Johannes, 1998; Castilla et al., 1998) are experiencing outstanding results derived from managerial tools such as co-management, community-based management, the implementation of TURFs or Individual Transferable or Non-Transferable Quotas. So far, these tools have been tested mainly with small-scale shellfishery activities, particularly on sessile or slow moving benthic invertebrates. There is a need to combine the use of management with conservation, as exists in MPAs (Castilla, 2000a).

In connection with the future sustainable conservation and use of shellfish resources we identify 5 main tasks ahead: (1) the geographical expansion, within Latin America and other regions, of SEI, SII and MIa invertebrate co-management and community-based management schemes; (2) The further expansion of the above to mobile MIb invertebrate species, such as spiny lobsters and crabs (also see

Russ and Alcala, 1998 for coral reef fishes); (3) The implementation of adaptive and experimental fishery approaches, including replication and the expansion of the spatial scale (McAllister and Peterman, 1992); (4) The improvement linkages (communication) between ecological and fishery findings, risks assessments, management and conservation decisions and associated social and cultural values (Norton, 1998; Castilla, 2000a); and (5) The joint approach to the management and conservation of coastal resources (and habitats) through spatially combined schemes, such as the one proposed for the Chilean coast (Castilla, 2000a).

Legal aspects

Co-management success depends on various factors, among which the definition and establishment of legal instruments formalizing community rights and responsibilities through the institutionalization of management practices are essential (e.g., Chilean FAL, 1989; Castilla, 1994; Bernal et al., 1999). This will help preserve traditional rights of resource use and access and enhance local welfare (White et al., 1994; Castilla et al., 1998). Legal frameworks and the allocation of property rights (ancient rights or privileges) and a set of clearly identified and respected management rules are the key factors in successful co-management, which have broken away with resource open access policies and the so called "tragedy of the commons" (Hardin, 1968). For instance, the legal setting of global quotas (e.g., TAC), regulated by law or decrees, although far away from active co-management schemes, can be wisely used as an important step forward towards the sustainable use of shellfish. A trichotomy appears: (1) If quotas are administrated through ITQs and used to solve small-scale management within traditional fishery societies, this may lead to "virtual communities", since fishers do not necessarily trade "quotas" at their long-term benefit (Annala, 1996; Pauly, 1996; Caddy, 1999); (2) If TACs are administrated through INTQs, then fishers would be forced to use the "quotas" more wisely in a long-term perspective (Castilla and Fernández, 1998); and (3) If a TAC is regulated through INTQs, and shellfish extractions are authorized exclusively inside protected areas (e.g., MEAs in Chile for *C. concholepas*), then the global quota may reinforce co-management procedures. All the above require legislation and enforcement of legal frameworks, which need to be adapted to countries and idiosyncrasies.

Economics, sociology and links with experimental and co-management

Large-scale experimental options often involve substantial economic and political costs (Walters, 1997). Moreover, methodology for objective, economic comparison of experimental management options is poorly developed, and there is no general consensus about how to value or weight alternative experimental outcomes. Economic analysis is needed to evaluate the performance of a given TURF (MEA in Chile) and this should include a multispecific approach together with the expected economic outcome of alternative management strategies, given the inherent characteristics of the life history of the strategies mentioned earlier in this section. Bioeconomic analysis should also incorporate non-monetary criteria such as the conservation of marine biodiversity, protection of threatened species and communal and/or societal non-economic gains (assets for community bond-linkages and organization). This leads to cross-connection and mutual fertilization between sociological, management, economic and conservation gains.

Tangible economic and social benefits have been considered as key motivating factors to initiate community-based management schemes (White et al., 1994). Fishers have an expectation that the benefits derived from participation and compliance with co-management schemes will exceed the costs of investments in such activities (Pomeroy and Williams, 1994). However, the Chilean experience also showed that small-scale fisher communities have an unpriced value given by the willingness to pay for the existence of a maritime concession to be owned by the fisher community, since there is extra-societal value involved (Castilla, pers. inf.). This unpriced value needs to be internalized in a benefit-cost analysis. For example, the allocation of TURF has a "societal dimension", as shown by the Chilean example (Castilla, 1997). The legislator implemented a set of management procedures balancing "the commons, communal and individual rights"; accounting for the existence of "real fisher communities" incorporating fisher traditions (i.e., respecting the Caletas traditional fishery grounds). Co-management and communal-rights (viewed as alternative approaches) in Latin America are in their infancy, but preliminary results are promising. In any case, the key to success must always be to understand not only how the shellfish, but also how the fishers behave: rational fisheries do not exist if both components are not well tuned.

As a consequence of small differences in fisher behavior, management regulations that are successfully applied to one location may fail in another. In fact, strong differences in the management of the El Quisco and Las Cruces MEA were observed in Central Chile (Payne and Castilla, 1994; Minn and Castilla, 1995). Both Unions, separated only by about 15 km, obtained legal rights to their private MEA ground in 1993. However El Quisco Union voluntarily banned diving activities two years prior (1991) and enforced the 1993 regulation, while the Las Cruces Union never enforced the ban and as a consequence fishers never complied with their own rules. This could be attributed to different strategies: (1) The Las Cruces fishers are short-term $PUE (profit per unit of effort) maximizers, optimizing their only ability (diving) during the short time window available for invertebrate diving extractions, having no choice with regard to sale opportunities; (2) El Quisco fishers appear as long-term $PUE optimizers, maximizing their income by switching among fishing gears, resources, and private versus open access fishing grounds, depending on sale opportunities. This highlights the future need to look for linkages between sociology, biology and economy, under an integrated management framework.

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