Defeo, 2001). This issue has also been addressed in a text on fisheries bioeconomics published by FAO (Seijo, Defeo and Salas, 1998).

6. THE ROLE OF HABITAT IN STOCK ENHANCEMENT AND RESTORATION

Successful stock restoration or enhancement requires harvest controls but also demands and attention to human impacts on the habitat. Reducing exploitation alone on the stock being restored will not be effective if critical habitat has disappeared. Figure 6.1 points to the role of habitat as an important sequential constraint. The use of habitat restoration to improve yields of marine coastal species does occur locally, but often has to be searched for in the literature on marine ecology, the role of marine parks, etc. Habitat restoration has on occasions been considered more feasible than stock enhancement *per se*. Hilborn (1998) for example, points to the low success rate in economic terms of most stock enhancement exercises, although most cases he refers to relate to finfish. Nonetheless, we agree with his suggestion that stock enhancement should be compared with alternatives such as habitat protection, fishery regulation and stricter enforcement before embarking on potentially costly and uneconomic large scale operations without prior experimentation. In this section we consider the first of these options. Performing a formal cost-benefit analysis on habitat restoration is feasible where artificial structures are added to the water to improve holding capacity, but basic pilot stage applications that allow us to establish feasibility, would seem required first.

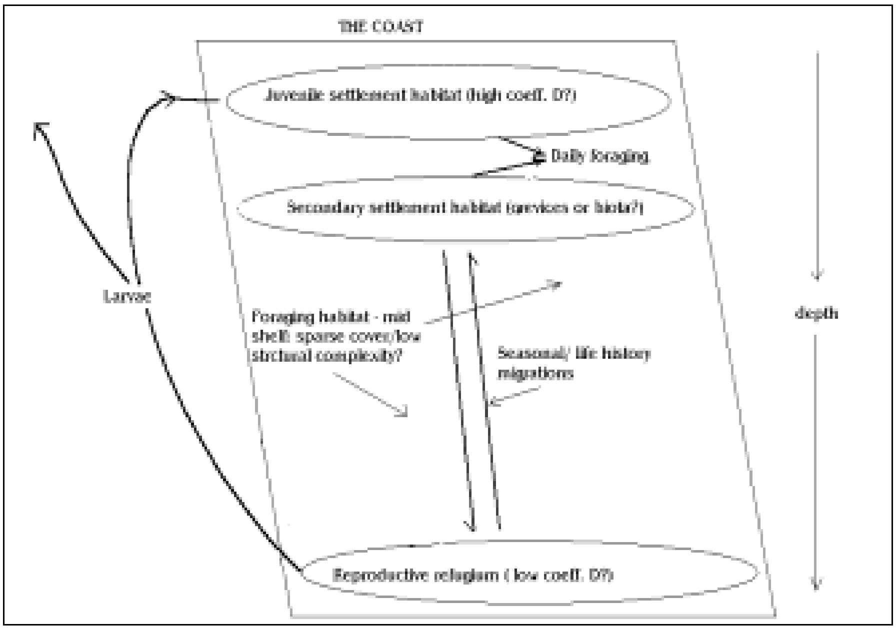

Figure 6.1 Showing how different stages of the life history (in this case of a crustacean such as a lobster) may be carried out within different habitats and depths.

Perhaps the Multispecies Virtual Population Analysis (MSVPA) experiment carried out for North Sea fish species (Sparholt, Larsen and Nielsen, 2002) provides the most eloquent argument against automatically assuming that juvenile enhancement programmes, sea ranching or restocking programmes will necessarily be effective. This extensive programme of stomach content analysis revealed that juvenile fish (as for juvenile shellfish) have a generally high vulnerability to predation, and extremely high juvenile mortalities apply in the first 1-3 years of life. While this experiment suggests that any reduction in juvenile mortality of commercially important species could pay major dividends, evidence will be presented that there may be a need to mitigate the impacts of habitat-mediated bottlenecks.

6.1 Habitat requirements

In the United States, the Magnuson-Stevens Act requires fisheries managers to describe essential fish habitat, and to minimize the impacts on this by fishing. "Essential Fish Habitat" (EFH) is defined as "those waters and substrate necessary for spawning, breeding, feeding or growth to maturity", and requires to be defined for all species under management. Mapping such areas using GIS techniques is now an essential component for deciding on closed areas, leases, enhancement areas, and in assigning priority use within an Integrated Coastal Area Management (ICAM) activity. This may involve remote sensing (see Rubec *et al.*, 1998). Figure 6.2 illustrates the complexity of decision-making in the coastal zone, and the fact that many other activities impinge on shellfish production, and will need to be reconciled in a multi-disciplinary way, involving all "players" or stakeholders in the coastal zone.

Figure 6.2 attempts to map some of the activities, causes and effects of human activities, starting with those associated with runoff from the adjacent freshwater catchment basin (upper left). This has effects listed on the right (e.g. increased red tides, shellfish toxicity etc) which impact the human activities on the right (e.g. aquaculture, fish processing etc), The fishery and fish culture activities in the box with dotted lines is thus seen as small "downstream" components vulnerable to these non-fishing anthropogenic effects.

6.2 Bottlenecks in production

Limitations of habitat may be important bottlenecks for some invertebrate populations, thus Scheding *et al.* (2001) found that the presence of sandy bottom used by ovigerous females to bury themselves in aggregations during egg incubation, may be a limiting factor for adult Dungeness crab abundance in Alaska. Experience suggests that benthic and demersal organisms of a wide range of taxonomic groups and species often pass through primary and secondary habitats in the course of their life histories, thus, American lobsters may first occupy crevices in cobble bottom areas in a completely cryptic life stage until 5-10 cm in length (Wahle and Steneck 1991), when they migrate to burrows under stones or construct burrows on more sedimentary bottom. Similarly, Palinurid lobsters settle first in finely branching substrates such as red algae or mangrove roots, before migrating to sea grass beds and later to caves under coral patches further offshore. In both cases, the extent of these critical juvenile habitats may be the bottleneck limiting overall production, given that habitat and food supplies available to older stages may not be limiting.

The giant scallop *Placopecten magellanicus* and various species of *Chlamys* have been described as settling on bryozoa before attaching with a byssus to shell gravel, even inside the umbones of dead adult scallops. Similar primary and secondary settlement episodes, and the characteristic type of bottom conditions that identify them, have also been described for *Mytilus edulis* (see e.g. Bayne, 1964, 1976). Such primary and secondary "nursery" habitats have been described for a wide range of invertebrate species, and their often restricted occurrence makes one wonder if this poses a limit to recruitment, or in other words forms a "bottleneck" in the production process. Various kinds of bottlenecks have been proposed, from the more general term "demographic bottleneck" which covers all forms of restriction on survival of a year class, to more specific "shelter-limited bottlenecks" which imply a structural limit, or "trophic bottlenecks" where food is the limiting variable. In fact Walters and Juanes (1993) show that it is the limitation of food adjacent to cover that requires organisms to be "risk-takers" and feed outside of cover.

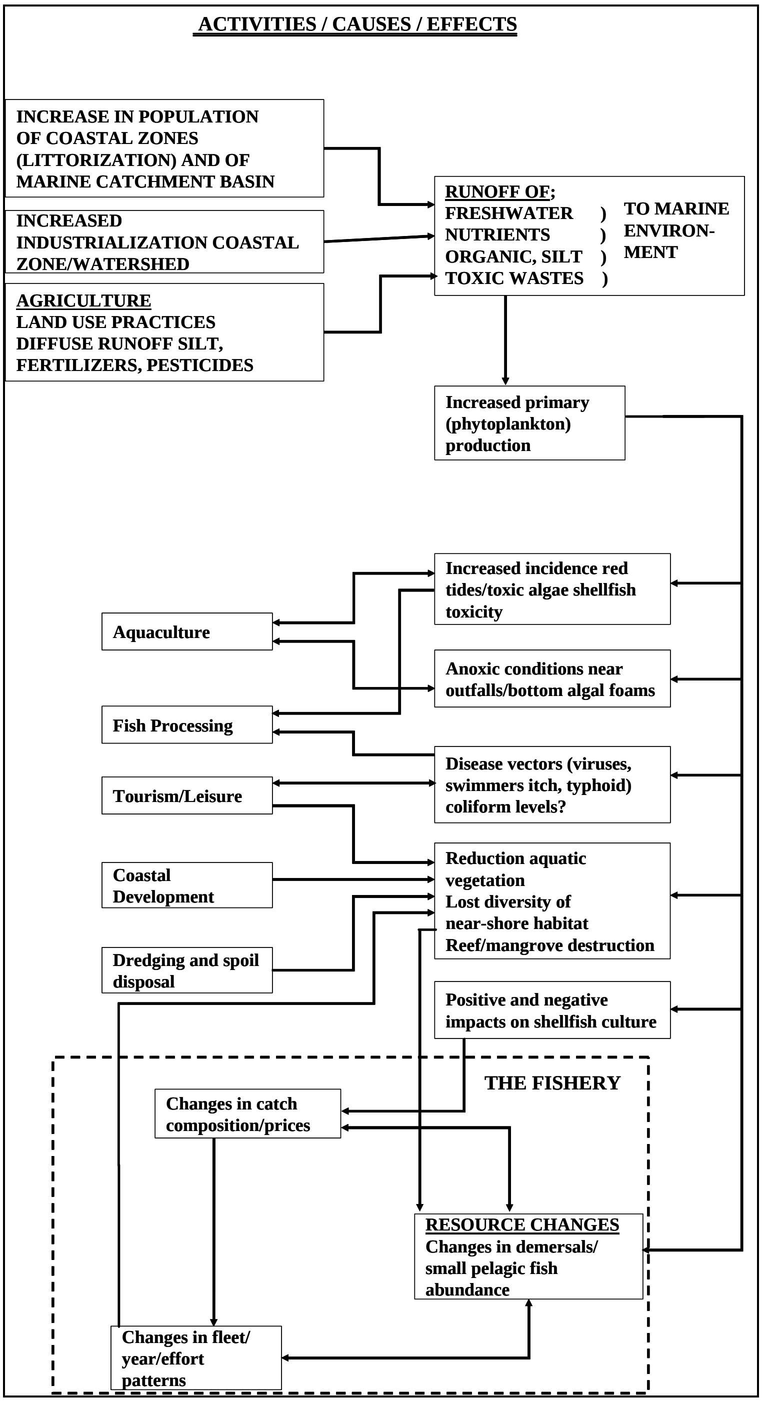

Figure 6.2 Showing a framework for policy development in the coastal zone, the socio-economic context for integrated policy formulation, and interactions of a variety of environmental factors. The interactions between different social users of the coastal environment, including shellfish production are shown. (after Caddy and Bakun, 1995).

A feature first observed by Morse *et al.* (1985) as a consequence of the fractal structure of most complex surfaces in nature, is that on a complex surface such as that of aquatic vegetation, there are "more surfaces available for small (arthropods) than large ones". Caddy and Stamatopoulos (1990) generalized this observation to show that the fractal nature of habitat structure results in small crevice sizes being much more common than large ones, hence migration or mortality by predation are the two options that await those juveniles displaced from crevices of the appropriate size while searching for much less abundant larger crevices.

Mya arenaria was introduced to Grays Harbor (Washington) during the 1880s. Palacios, Orensanz and Armstrong (1994, 2000) showed that the maximum size of members of the current population is much smaller than of clams found dead in situ from preceding decades, and which nowadays form extensive "death assemblages". Palacios, Orensanz and Armstrong (1994) concluded that extinct clams grew faster and lived longer because they occupied the best habitats available. After an extensive mass-mortality episode between 1895 and 1897 that resulted in the formation of the deposits, the population has never rebounded into its prime habitat, in spite of potential seeds being regularly available. They also showed that Dungeness crab larvae settle preferentially in these shell deposits, where the abundance of 0 + age juveniles is orders of magnitude higher than on the adjacent flats. They hypothesize that predation by juvenile crabs is the main factor that limits clam recruitment (see also Iribarne *et al.,* 1992).

The point which can only be touched on here is that where such critical habitats occur, their holding capacity will to a significant extent determine the size of the new recruiting age class to the population. Thus there will be no point attempting stock enhancement if the existing population size will subsequently be restricted by some form of bottleneck. As noted, another implication of this type of phenomenon is that even if there may be adequate food for many more adult animals than are actually present, these population levels are unlikely to be realized if there is a bottleneck at or following recruitment. This emphasizes the importance of ensuring that primary and secondary habitat types and cover characteristics are not degraded by environmental influences, and that suitable habitat is provided in extensive culture or where enhancement of depleted wild populations is underway, especially in the presence of predators for the species in question.

6.3 Stock replacement, habitat rehabilitation or mitigation?

There have been many criticisms of hatchery rearing and release programmes aimed at stock enhancement, and as we have noted, the record of success is rather spotty, especially for finfish, but should not prevent careful consideration of this mechanism in certain well-studied situations. Despite the five conferences held on the issue in recent years (Grimes, 1998), there has been a relatively minor focus on related marine habitat issues, and in particular, cover and habitat complexity. A more general consideration is the widespread occurrence of density-dependent mortality in natural systems, which ensures that increases of population size above some carrying capacity will be rapidly reduced. In this sense, Heppell and Crowder (1998) suggest that before considering stock enhancement, the existence of habitat constraints should be checked for. Does the environment contain sufficient critical habitat needed for the life history stages introduced for example? They also stress the need for harvest controls to be in place. An example for a finfish in Florida nearshore waters, the gag grouper (*Mycteroperca microlepis*), showed that the existence of adequate sea grass beds is a precondition for restoration (Koenig and Coleman, 1998). For another finfish species important to anglers, the red drum, Grimes (1998) notes there is little point in attempting stock enhancement if density-dependent processes in the early life history result in high loss rates of juveniles. At the same time (and this defines one particular focus for any enhancement programme), in this case, recovery of as few as one percent of stocked fish need to be recaptured by high value sports fisheries for enhancement to be considered successful. Achieving an increase in recruitment in the range of five and ten percent by either stock or habitat enhancement would then seem to be a worthwhile and possibly achievable target. Whether this would be cost-effective as questioned by Hilborn (1998) still remains an important question, but in the case of restoration of a stock that has disappeared from its former range, such considerations must take second place to first establishing feasibility, and second, deciding what the restoration of a train of utility values into the indefinite future is worth? If the issue is to restore a train of benefits that has been lost, or otherwise would not continue into the future, and if the population would then be self-sustaining, restoration may be a feasible objective.

Habitat replacement or rehabilitation are less ambitious interventions than aiming for complete ecosystem restoration or rehabilitation, which are goals that are likely to be difficult or in some cases, impossible to achieve, and certainly more costly, and would involve whole system manipulation. Replacement or reclamation of damaged or degraded ecosystems constitutes interventions aimed at restoring economic productivity to a habitat that, due to human intervention, is currently unproductive. In this sense, a stock enhancement or replacement exercise may be a component of a more general environmental intervention aimed at improving habitat "quality", where quality is defined in the sense that it contributes to human welfare. Finally mitigation, the least ambitious type of intervention, aims to reduce the losses incurred due to ecosystem damage, and here again, shellfish enhancement schemes may play a part in a broader context, and is likely to be more feasible than, for example, restoring fish populations. Some definitions for these terms are provided in Table 6.1.

Table 6.1 Some human interventions on depleted or degraded ecosystems (after Bradshaw, 1996 and Cairns, 1994).

Societal action:	Defined as:
Rehabilitation or restoration	The action of restoring a thing to a previous condition or status
Reclamation	To bring back to a proper state (which may not be the original one)
Replacement	To procure a satisfactory substitute in place of the original
Remediation	To rectify or make good, places the emphasis on the process rather than the end point reached
Mitigation	To reduce the negative effects of change (especially habitat destruction) where these cannot be eliminated (or to soften the loss of a particular ecosystem

The importance of habitat for management of wild resources is made evident in Figure 6.3, based on Marshall (1996). This emphasizes the importance of characterizing the biological suitability of the habitat and the biological community within which the species in question is being enhanced. If the habitat has deteriorated, for example through anthropological stresses, and is biologically complex, there has to be less expectation of a successful result. Probably it will be necessary first to restore the ecosystem, to the extent possible, to a productive situation before beginning an enhancement procedure. Better still is to choose another area where these stresses or impacts are less pronounced.

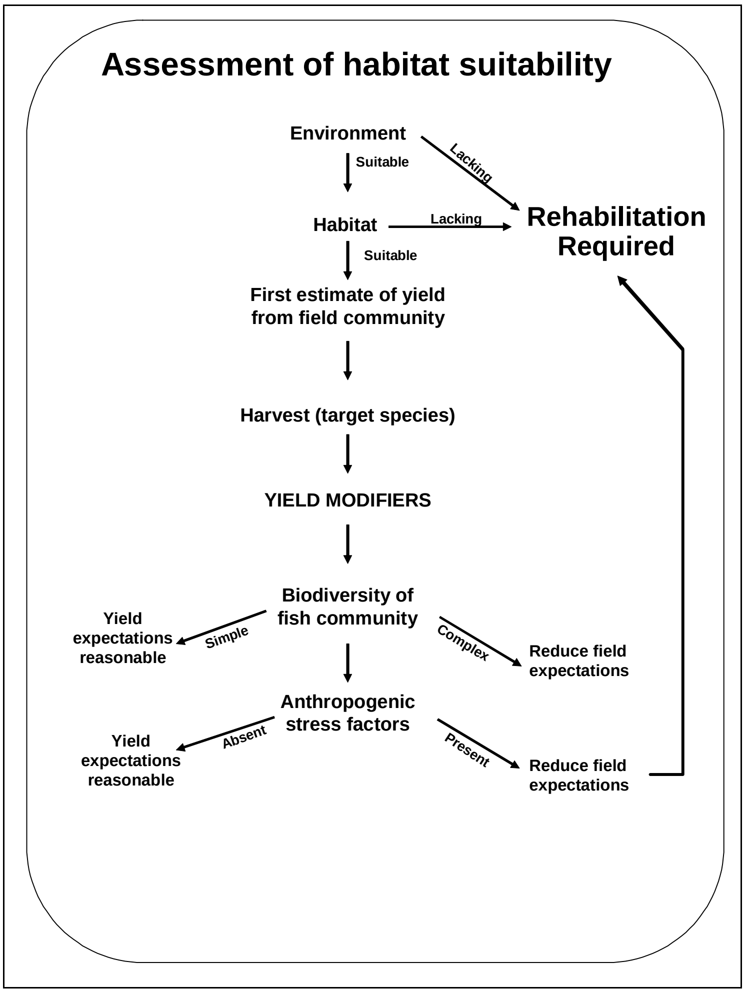

Figure 6.3 An approach to assessing habitat suitability (after Marshall, 1996).

As an example of a rehabilitation or mitigation exercise, we note that included in the goals of the US National Research Council (1992) for national aquatic ecosystems between now and 2010, is to restore ten million acres of wetlands out of the 117 million impaired or destroyed since 1800. Such wetland areas are frequently very productive, and shellfish form key components of these ecosystems and should be harvested in moderation. It is worth noting here that wholesale raking of oyster beds in the Chesapeake Bay system removed these higher relief shell or cutch banks which formerly were “islands” of hard bottom above the Bay floor, and were of ecological importance to a range of species.

Cairns (1994) notes that: “although precise replication of predisturbance conditions will rarely be possible, achieving a naturalistic assemblage of plants and animals” (at the landscape level) “of similar structure and function should be possible in most cases”, and comments that: “It is a sine qua non that ecological repair is preferable to neglect of damaged ecosystems”. He remarks that it is not excluded that this restoration could be (in the case of wetlands) in areas which did not have them in the first place, as a way of building “ecological capital”. He goes on to say: “Necessarily, local societies must have accepted the goal of restoration and cooperate, which in turn requires greater ecological literacy”. If self-maintenance is to be achieved, the scale of restoration increases considerably, and ecological restoration is seen as “buying more time for human society to develop less threatening life styles”.

Langton *et al.* (1996) suggested a prioritization of research questions as shown in Figure 6.4, which can equally be used when contemplating an enhancement programme. If the questions in the list of “prioritized research questions” cannot be answered in advance, it is important that priority work be done on resolving them before full scale enhancement begins.

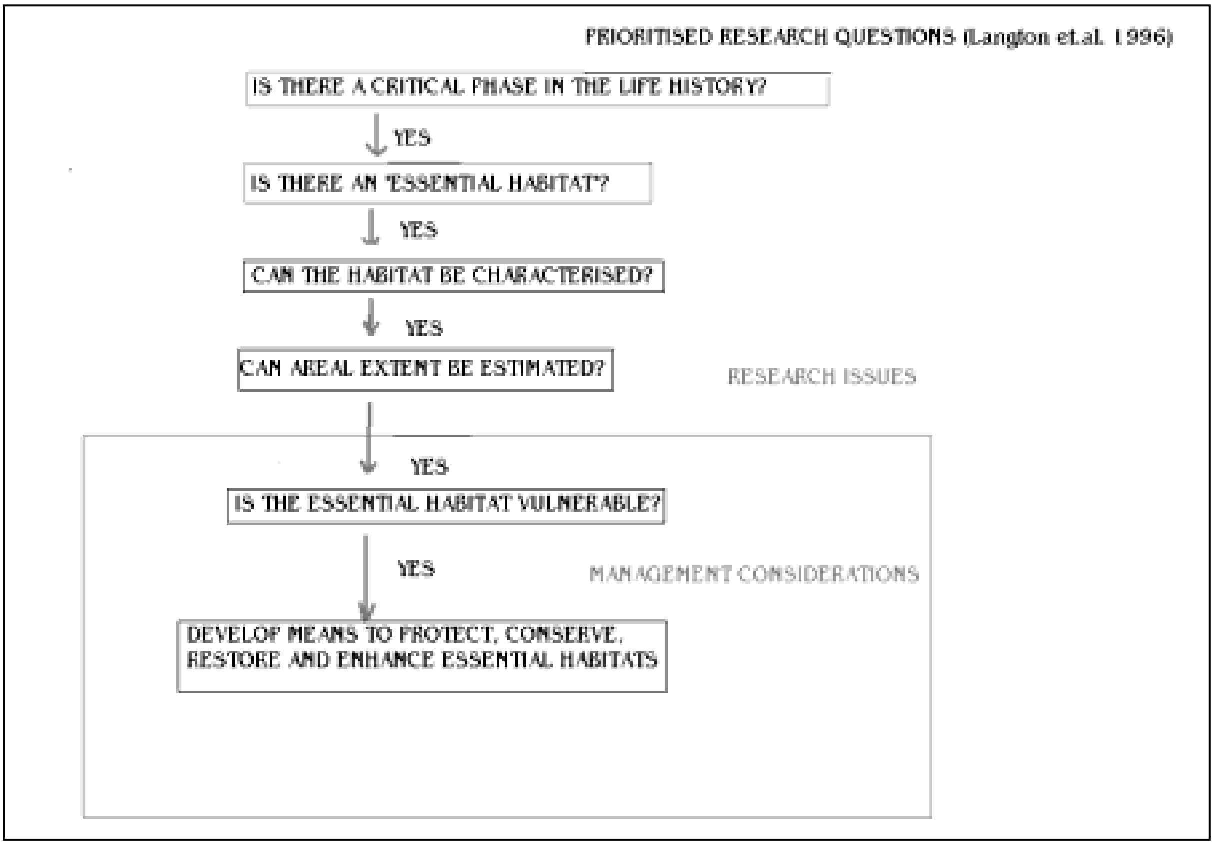

Figure 6.4 A list of research questions proposed by Langton *et al.* (1996) prior to beginning any intervention involving natural systems.

The experience with shellfish enhancement procedures as implied earlier, has not been uniformly positive, and it is instructive to consider why. Figure 6.5 places enhancement at the summit of a sequence of human ecosystem interventions, which are perhaps more realizable from the bottom of the triangle than the top, and this especially applies to marine finfish stocks.

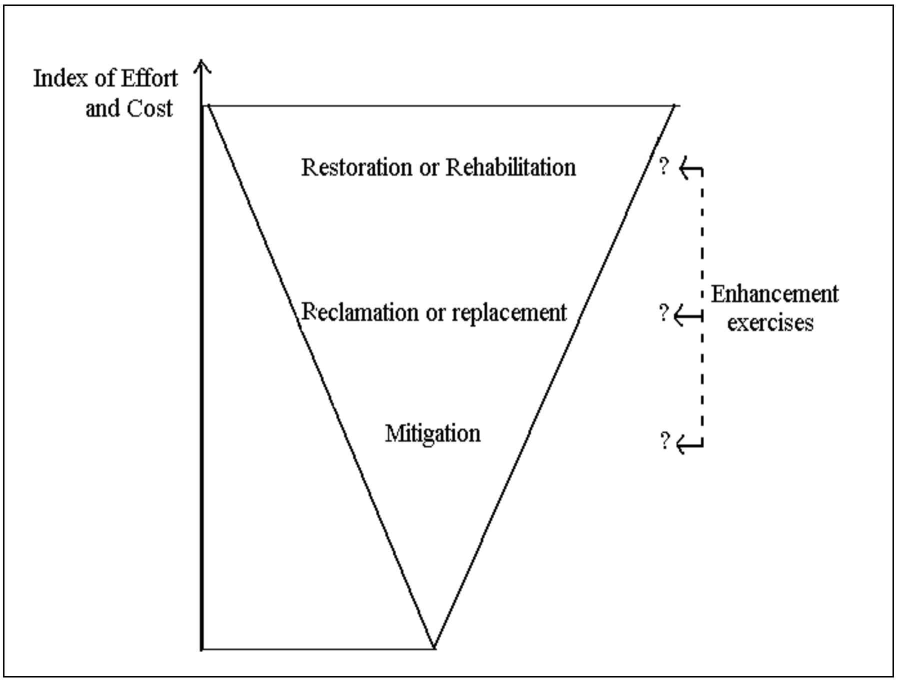

Figure 6.5 Illustrating a sequence of human ecosystem interventions, arranged from the bottom of the inverted triangle to the top according to their likely feasibility and cost. Shellfish enhancement may play a role in all of these activities, but the implication is that this is likely to be more feasible and relevant for uncontaminated ecosystems that have been otherwise impacted by human activities.

Figure 6.6 needs to be kept in mind when restoring a stock through stock enhancement, which is unlikely to be successful if the critical habitat that limits life history stages is not present, and if effort control does not allow parental stocks to build up so that natural population replenishment can occur. Trophic conditions are also important in providing overall basic requirements for food, but this alone is unlikely to be adequate in ensuring population build-up if the other two factors are not given priority. Mitigation of negative impacts is less demanding than reclamation or replacement, which in turn are less problematic than full restoration of the original ecosystem.

Enhancement, which in the strictest sense implies “improving on nature” would imply even more costs since an “enhanced” system is implied to be an improbable state differing from the equilibrium situation often characterized by the term “virgin conditions”. In fact however, enhancement becomes more feasible when it is considered in the context of restoring an ecosystem after serious stock depletion, and is also appropriate for systems which are recovering from disturbance, where for example an unfilled ecological niche may be temporarily present. It is mainly in this sense that the term is used in this paper, so the apical position of enhancement in Figure 6.6 is probably anomalous.

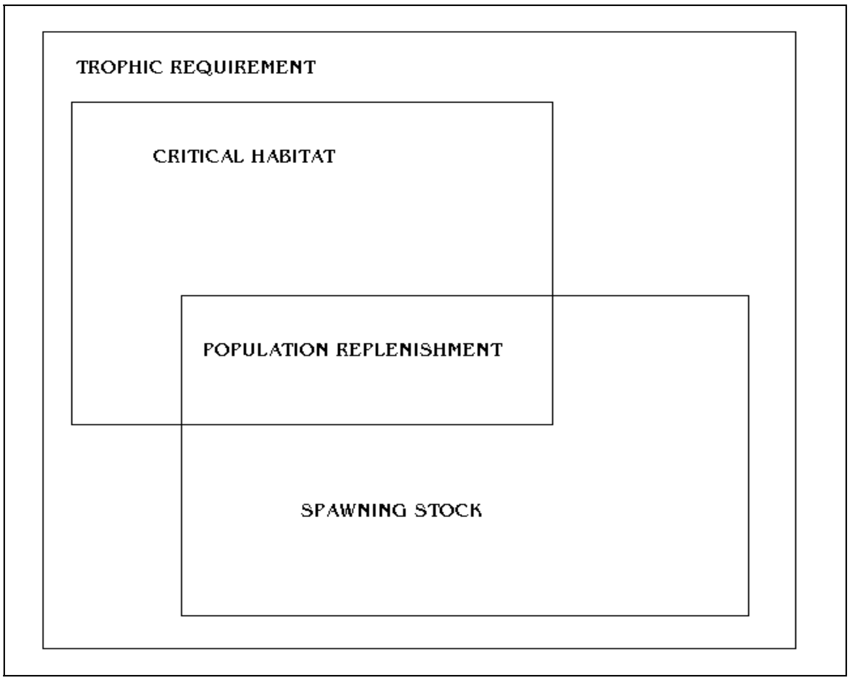

Figure 6.6 Factors that need to be considered when restoring shellfish populations through stock enhancement include basic trophic requirements and special habitat requirements, the latter more difficult to satisfy, while full restoration will require an adequate spawning stock.

In some cases (e.g. for penaeid shrimp stocks, Figure 6.7), river outflow patterns have a major role in determining shrimp production, and drainage of wet lands, cutting of mangrove forests and the use of herbicides in coastal areas can negatively affect shrimp production.

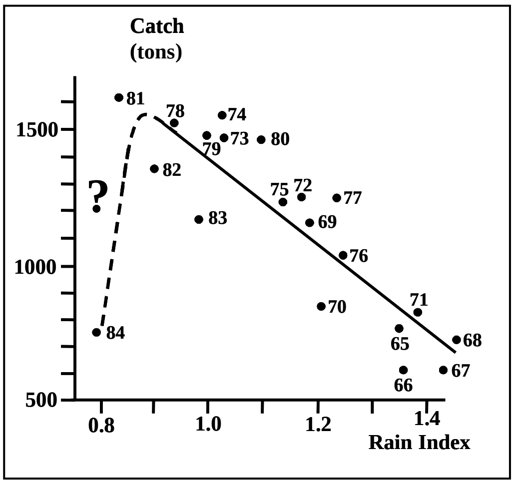

Figure 6.7 Catches of the shrimp *Penaeus notialis* in Casamance river estuary as a function of rainfall index (after Le Reste, 1992).

Determining the effects of land usage on intertidal and estuarine resources is an important precondition to producing safe shellfish products. White *et al.* (2000) mention a multi-agency project that was carried out in an Integrated Coastal Area Management (ICAM) context, to determine the effects land use in the adjacent watershed had on shellfish closures. Bacterial data were monitored and indicated increasing loadings in runoff water, with especially high levels during storm events. Dye studies confirmed that bacteria would move through the watershed over a brief time period with negligible mortality. Low levels of bacteria were found during dry weather. Most contamination came from a nearby residential area with some malfunctioning septic tanks, but also from pets and wildlife. A mitigation programme was designed using GIS, and included education, and restoration of wetlands, automated storm water monitoring, and DNA tracking of faecal sources.

A general discussion of shellfish restoration activities in the context of ecological functions is provided by Coen and Luckenbach (2000), who mention some of the "ecosystem services" provided by shellfish beds, whose value is usually underestimated. These include contributions to the filtering capacity of the water column, benthic-pelagic coupling, a role in nutrient dynamics, increased suspended sediment deposition, and stablilization of bottom sediments. Very dense shellfish beds may have negative impacts due to fouling by pseudofaeces, and Kaiser *et al.* (1998) review some of the impacts caused by dense shellfish culture. Castel *et al.* (1989) find that densely stocked oyster beds elevated organic carbon levels in adjacent sediments, sometimes producing hypoxic conditions. Site selection preferably in areas of current flow is therefore important. However, considering that shellfish beds were certainly denser before human harvesting, such negative impacts of shellfish populations are in themselves, local effects; and under natural conditions would be less evident. One reason being that through accumulation of shell material, native shellfish beds were often raised above the sedimentary level of the estuary floor and hence cleaned by stronger local currents. Sparsis, Lin and Hagood (2001) even evaluated the feasibility of using juvenile giant clams to remove dissolved inorganic phosphate and nitrate from holding tanks of ornamental or food fish. The nutrients were removed by xooxanthellae in the mantle of giant clams in lighted periods, and all species tested except *Tridacna gigas* survived, and some grew faster in effluent than elsewhere, suggesting the possibility of using giant clams in polyculture.

Restoring whole ecosystems where the environment has changed due to human intervention or climate change is a much less certain prospect, and in this case, actions to mitigate the damage may be in order. An example from other than shellfish enhancement comes from the North American Great Lakes (Regier *et al.,* 1988), where eutrophication and overfishing had destroyed the valuable deep water lake trout fishery, but subsequently established an abundant but low value resource of small forage fish. The introduction of a west coast salmon species which could prey on these forage fish completed the transformation of the system into an "exotic" pelagic ecosystem (where effectively no economic resources of surface waters existed before) and this now has high economic value for sports anglers. This example is not an apologia for habitat degradation, just that restoring the Great Lakes to their pristine condition was probably infeasible, and would anyway have meant radical changes to the life of several million inhabitants within the catchment basin of the Great Lakes, hence restoring a productive, if alternative, system appears the most feasible option available. For this reason, fully restoring the native lake trout population is a costly option compared with the current sports fishery for introduced coho salmon. This example of a successful mitigation of human impacts shows that the more general objectives of restoring habitat quality for a range of purposes (recreation, quality of life issues etc), will usually have to precede the restoration of a population of organisms of value to humans, and these may be different from those originally present.

Another example can be cited in the case of the Black Sea. At the start of the twentieth century this was a mesotrophic body of water with oxygenated waters on shelf areas, and a rich fauna of indigenous species (see Caddy and Griffiths, 1995). The eutrophication of the basin led to a cascade of ecological effects described in Zaitsev (1993). Seasonal anoxia of shelf areas and the littoral in summer led rapidly to the elimination of many indigenous species. Eutrophication coincided with the likely accidental introduction of two exotic species, the clam *Mya arenaria*, and a gastropod *Rapana* sp. both adapted to eutrophic conditions. This latter species, a predatory snail introduced from Japan with oyster imports, has acclimatized, and now supports a fishery of some 40 000 t/year. So far *Mya* is unexploited but is believed to support a very large biomass. The fate of a large subtidal population of the mussel, *Mytilus galloprovincialis* in the Northwest of the Black Sea is also instructive. This was the target of a Russian fishery, but subsequent events suggest it may have played a major role in controlling turbidity of shelf waters (Sorokin, 1993). With seasonal hypoxia, this deeper water mussel population has largely collapsed, and coincidentally, a subtidal population of a red alga, *Phyllophora* sp. in the same area disappeared, presumably due to poor light penetration caused by algal blooms. Again Sorokin (1993) deduced that both of these declines played a significant role in oxygenation of shallow shelf waters and reduction of suspended material, thus improving light penetration. This example illustrates that the services provided by bottom fauna and flora, including populations of filter feeders, have a significant role, in this case in controlling suspended sediments and phytoplankton and coastal eutrophication, apart from their importance for human food.

Hypoxic conditions can of course be detrimental to shellfish enhancement operations, depending on species tolerances. Diaz and Rosenberg (1995) found that commercial species varied considerably in their resistance to hypoxia; thus, bivalves *Arctica islandica* and *Mytilus edulis* were most resistant, the clams *Mercenaria mercenaria* and *Spisula solidissima* were intermediate, while benthic crustaceans, *Nephrops norvegicus, Crangon crangon, Carcinus maenas,* and *Spisula solida,* a clam typical of clean sand, were the most sensitive.

The effects of bottom-water hypoxia on the population density of the clam *Macoma balthica* was estimated using a survival-based approach by Borsuk, Powers and Peterson (2002). They used a Bayesian parameter estimation to fit a survival model to times-to-death corresponding to multiple dissolved oxygen (DO) concentrations assessed by scientific experts, and combined the survival model with a model describing the time dependence of DO. Under current conditions, the mean summer survival rate was predicted to be only 11 percent. However, if sediment oxygen demand (SOD) is reduced as a result of nutrient management, survival rates increased, reaching 23 percent with a 25 percent reduction in SOD and 46 percent with a 50 percent SOD reduction (Borsuk, Powers and Peterson, 2002).

Lenihan *et al.* (2001) tested the hypothesis that mobile consumers have the potential to cause a cascading of habitat degradation beyond the region that is directly stressed, by concentrating in refuges where they intensify biological interactions and can deplete prey resources. They worked on structurally complex, species-rich biogenic reefs created by the eastern oyster, *Crassostrea virginica*, in the Neuse River estuary, North Carolina. Bottom-water hypoxia and fishery-caused degradation of reef habitat induced mass emigration of fish, thus modifying community composition in refuges across an estuarine seascape. Moreover, oyster dredging reduced reef height and exposed the reefs located in deep water to hypoxia/anoxia for more than two weeks, killing reef-associated invertebrate prey and forcing mobile fishes into refuge habitats. High-density accumulations of refugee fishes on reefs in oxygenated shallow water depleted epibenthic crustacean prey populations. Thus, the interaction of reef habitat degradation through fishery disturbance and extended bottom-water hypoxia/anoxia caused oyster mortality and influenced the abundance and distribution of fish and invertebrates that use this reef habitat (see also Lenihan and Peterson, 1998). The authors concluded that physical disturbances can impact remote, undisturbed refuge habitats through the movement and abnormal concentration of refugee organisms that have subsequent strong trophic impacts. In this context, they highlight the implications of MPAs as critical refuges.

The upper Adriatic Sea, an area of predominantly fine bottom sediments acts as an “outer estuary” by receiving some thousand tonnes a year of nutrients from the very eutrophic Po river. Italian workers (e.g. Bombaci, Fabi and Fiorentini, 2000) have focussed on use of artificial reefs colonized densely by *Mytilus* as a way of making use of these high productivities and precipitating suspended material from the water in the pseudofaeces of mussels. The potential role of mytiliculture here is not only to provide considerable economic add-on food value, but also to act as a depurator of estuarine discharges and the precipitation of sediments and algae from the water column: a function of importance to bathing resorts in the Adriatic. These kinds of ecosystem functions that may be achieved through a shellfish enhancement programme deserve further economic analysis. The dramatic increase in shellfish production in the Mediterranean shown from FAO statistics (Figure 6.8), especially in the upper Adriatic and Gulf of Lions under the influence of the Po and Rhone river outflows, needs attention. As noted by Caddy (1993a) and de Leiva Moreno *et al.* (2000), European inland seas such as the Baltic, Adriatic, Mediterranean and southern North Sea, depending on their degree of enclosure and the extent of the catchment areas feeding them, have, become eutrophic to different degrees, under anthropogenic influences from adjacent catchment basins. Molluscan shellfish production to a significant extent has benefited from this situation, although issues of depuration and the control of transmission of disease vectors through untreated shellfish have also become important. Thus, macroenvironmental trends in an area provide an important context for the shellfish enhancement activities we have been discussing here. It may be noted that in terms of the production of animal protein, shellfish cultivation does not depend on resources of fish meal or agricultural products as for most (carnivorous) species used in marine finfish culture, and as we have noted, if used strategically, mollusc shellfish stocks can play an important role in locally enhancing water transparency and hence in restoring aquatic vegetation.

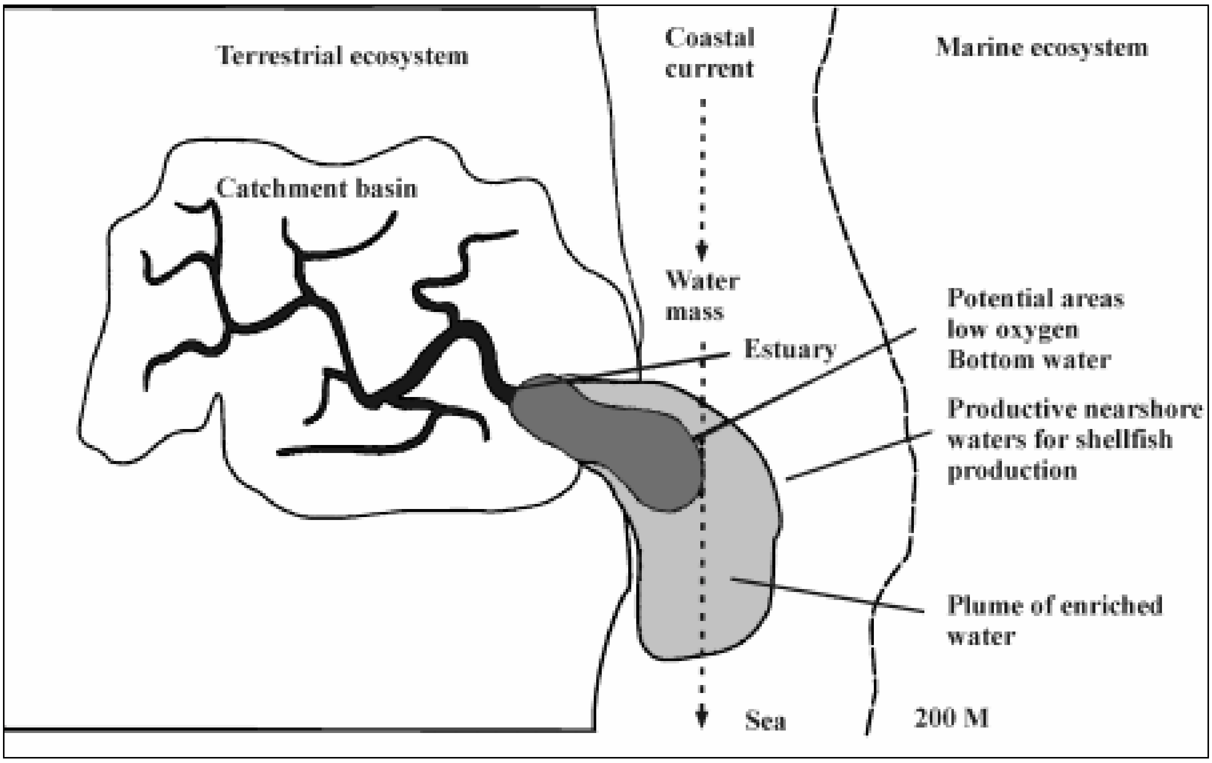

Figure 6.8 Illustrating the role of nutrients from river catchment areas and river plumes in enriching nearshore shellfish fisheries (after Caddy, 2000b).

6.4 Choosing a site for enhancement activities

The selection of adequate habitats is crucial for the development of enhancement programmes. Recognition of gradients in habitat quality is important in defining the extent of the area available for seeding. A range of habitat sites must be analysed to evaluate likely differences in seeding success according to habitat suitability. Consider for example that a seeding programme for an intertidal soft-bottom bivalve is started with the intention to colonize new areas and develop a new fishery. In this case, some critical variables detailed below will give useful insights as to which sites would be optimal.

6.4.1 Sediment properties

Exposure (exposed-sheltered). The choice of an adequate habitat for seeding should be a trade-off between different factors acting simultaneously. For example, when considering intertidal oysters, the level of the intertidal chosen to seed is critical, because lower tidal levels are more susceptible to predation. Alternatively, higher tidal levels commonly have a higher degree of silting which increases mortality and lower growth rates. Quantity and quality of food might depend on exposure and the degree of turbidity. Some exposed sandy coasts could constitute semi-closed ecosystems in which high concentrations of surf phytoplankton occur in waters rich in nutrients and oxygen. These sites could be useful for enhancement operations of intertidal suspension feeders (including passive restocking, see Defeo, 1993b, 1996a). However, wave action and speed of currents could act as negative forces that could preclude spat settlement. Once again, a trade-off between these different factors must be evaluated. If a sandy beach mollusc is selected, the definition of beach morphodynamics will be critical, i.e. if it is reflective or dissipative (McLachlan *et al.,* 1996).

Grain size preferences (fine - coarse). Settlement rates could be accelerated in the presence of a suitable substrate. For example, Tong, Moss and Illingworth (1987) reported that juveniles of the abalone *Haliotis iris* tend to settle in almost all cases associated with the encrusting algae *Lithothamnium*. A careful selection of sites for seeding can help reduce mortality rates. In some cases (e.g. oysters), ground selection according to consistency of the bottom can also reduce silting mortality. Shifts in habitats, especially burial by sand, in the abalone *Haliotis iris* (Schiel, 1993), led to high mortality rates and negative rates of return in the enhancement operation.

Other sediment features, such as face slope, substratum penetrability, sediment water content, texture and roughness, are also important agents defining settlement variation among sites. Space availability in proportion to the amount of spats to be seeded should also be assessed.

Knowledge of sediment preferences at different life stages helps the shellfish farmer use adaptive behaviour of shellfish in protecting them from abiotic (e.g. hydrodynamic factors, exposure) and biotic (e.g. predation) factors. By carefully choosing sites, within-site sources and levels of natural mortality due to abiotic and biotic factors can be minimized. Historically productive fishing grounds, which have high probability of recolonization and generally low mortality levels, could serve as potential sources for seed replenishment (Caddy, 1989b). The quality and quantity of food present, or added as supplements, can be critical if economic losses are to be avoided due to density-dependent processes. Food availability often depends on habitat quality, and mortalities will occur due to starvation if seeding is conducted in the wrong place (see e.g. Tegner, 1989; Kristensen and Hoffmann, 1991).

Substrate preferences for many crustacean and molluscan larvae are often rather specific: thus Stevens (2003) found that late larval king crab demonstrated preferences for structurally complex substrates and a low preference for sand where mortalities were higher. This illustrates the importance of leaving “biological oases” where bottom contact fishing gear such as dredges and trawls are prohibited. Tegner *et al.* (2001) tied declines in abalone production in Southern California not only to overfishing but to the cessation of growth of the alga *Macrocystis pyrifera* which provides food through drift of debris under the canopy, as well as providing habitat for abalones; such cessation coinciding with the warm, nutrient-poor waters associated with El Niño events.

6.4.2 Hydrodynamic factors

Hydrodynamic factors act to generate spatio-temporal settlement patterns in shellfish populations. As population patterns and processes in shellfishes are scale-dependent, depending on the stage of the life history involved (Orensanz, 1986; Thrush, 1991; Defeo, 1996a, b), analysis of the physical-biological coupling at different scales is useful in the context of enhancement programmes. Peterson, Summerson and Luettich Jr. (1996) showed for the scallop *Argopecten irradians* that larvae larval settlement drops off sharply as a result of physical transport of their short-lived pelagic larvae. This has important implications in regulating population size in the system, as well as in developing adequate strategies for enhancement.

Oceanographic factors lead to site-specific variations in the local abundance of larvae and subsequent successful settlement, and thus determine the optimum times and sites for seed collection. For example, lack of suitable hydrographic conditions for supplying and retaining large numbers of larvae in the vicinity of collectors could lead to the failure of seed collection. Often, large settlements occur in semi-enclosed bodies of water or enclosed bays, which can also be good sites for early survival. These places are recurrent sites for successful settlement since they avoid mortalities due to flushing of water masses in the area. This is an important consideration when considering metapopulations, in which the capacity of larval dispersal over the fishing grounds is often dependent on the intensity and direction of wind-driven currents. In this context, self-recruiting, “source” areas (Carr and Reed, 1993) should be differentiated from “sink habitats” (see Chapter 3). As each ground has its distinct regime of primary production, nutrients, food availability, predation and disturbance, in theory, between-ground differences in these features could be quantified.

Enhancement operations will be increasingly affected by pollution on the coastal zone. Nearshore environments are more and more vulnerable to harmful algal blooms, sewage discharges, oil spills and so on. The quality of the site for seeding must be assessed from these points of view also, because their occurrence implies increasing variable costs and investment.

6.4.3 Carrying capacity and habitat suitability

For shellfish resources, it is common to find some portions of the habitat more densely populated than others as a response to gradients in habitat quality. Such spatial variations might be assessed to determine distributional patterns common to adults and recruits, in order to select a site for restocking. The following mechanisms could explain patchy distribution patterns and should ideally be evaluated: (1) active larval choice, and ability to colonize areas of habitat with greater environmental stability where population growth is maximized; (2) occurrence of higher mortalities operating before and after settlement due to adverse environmental effects (e.g. low salinity); and (3) a major incidence of hydrodynamic forces, which determines passive transport of larvae to a given receiver site.

Population regulation may be habitat-dependent, as demonstrated for shellfish populations by the positive covariance between density, resources and environmental harshness (e.g. salinity, seston, food availability). Density-dependent habitat limitation within the seeding area could greatly reduce the potential benefit of any restocking programme. Shelter/space availability may control the size of many shellfish populations (see e.g. Caddy and Stamatopoulos, 1990; Beck, 1995). An attempt to investigate experimentally whether hatchery-reared animals displace natural stocks should aim at testing the hypothesis of habitat limitation.

The above facts clearly suggest that the optimum individual size for restocking and the carrying capacity of the system will depend on site quality and extent: suitable hydrographic conditions, absence or rarity of predators, food availability and available shelter could be some of the factors affecting habitat quality and thus carrying capacity. Carrying capacity will also differ at different stages of the life cycle (Orensanz, 1986; Fréchette, 1991) and restocking operations must take this into account. For example, if a natural stock is already present, the total biomass of wild plus enhanced stock should not lead to compensatory mortality and depressed growth rates as a result of stocking. Dijkema, Bol and Vrooland (1987) found that high population densities of the cockle *Cerastoderma edule* in Netherlands determined density-dependent growth rates and that at very high densities, some individuals are pushed out of the sediment and subsequently die. The re-seeding of cockles on an experimental scale demonstrated the major advantages to thinning very dense natural cockle beds in order to improve growth rates. All of these factors affect production and thus the economic viability of the operation (see Schiel, 1993 and Brand *et al.*, 1991 for examples).

Maller (1990) and Blackburn, Lawton and Perry (1992) developed a simple and effective method to determine the slopes of the upper boundary (maximum densities) of the relationship between density and body size. Even though the original procedure was conceived for scaling body size to density, an issue also relevant for stock enhancement initiatives as an indicator of available energy in the ecosystem, the methodology equally applies to any combination of variables in which a Constraint Envelope Pattern (CEP: *sensu* Marquet, Navarrete and Castilla, 1995) has a real biological meaning. The procedure involves dividing the X-axis into intervals of equal length and recording the maximum value of the response variable on the Y-axis for each X interval (see also Marquet, Navarrete and Castilla, 1995 and Blackburn and Gaston, 2001 for additional theory). Because the value of the slope depends on interval size used in the X-axis, it is suggested to consider a range of interval sizes from, say, 0.1 units of the X variable up to a value that renders at least three values of Y (Marquet, Navarrete and Castilla, 1995). Then, the nature of the relationship defined by the points of the upper boundary is visualized through a simple scatterplot and then the appropriate (linear or non-linear) model is fitted. The upper limit corresponds to optimal combinations of the X and Y variables, whereas values within this "envelope", well below the upper ceiling, represent a wide range of suboptimal conditions. Typically, the CEP has a well-defined upper boundary with a negative slope indicating an inverse relationship between X and Y.

The above methodological approach has been suggested as a simple way of estimating carrying capacity through the use of a scatter diagram of adult and/or recruits density in each sampling unit or quadrat. At this small-scale of spatial resolution, a boundary of carrying capacity for both population components (adults and recruits combined) could be estimated (Orensanz, 1986). This must be done for different degrees of fishing pressure, and by site. Defeo (1996a) estimated carrying capacity with and without fishing activities. However, in the context of stock rebuilding initiatives, this approach is useful for evaluating optimal levels of abundance of the different population components at a given site, and thus is a help to planning the intensity of seeding operations through the estimation of optimal stocking densities (OSD). At a "quadrat scale", Defeo (1996a) showed that highest densities of recruits were never coincident with highest densities of older clams. Maximum densities of recruits per sample core were observed during 1984 and 1985, when they reached between 4 000 and 5 000 ind·m^{-2}; during the same period, maximum adult densities were between 400 and 600 ind·m^{-2} but not in the same samples where the maximum recruit densities were recorded. These values of adult density, which correspond to the period of active fishery, were far below the maxima recorded after the fishery had been closed; in 1989 they reached 800-900 ind·m^{-2}. It is notable that when adult densities were at least 300 ind·m^{-2}, recruitment was almost absent in the same sample. The negative relation between recruit and adult densities for all years combined is shown in Figure 6.9. The line which defines the upper limits of the relationship represents maximum adult densities for varying levels of maximum recruitment; below the line, the lower values represent a wide range of suboptimal environmental conditions (Maller, 1990). The upper boundary mainly reflected higher densities of recruits during the years 1984-1985, and those of adults inhibiting recruitment during the experiment, i.e. in 1989 (Figure 6.9a). This "envelope" is linear in this case, but could take different forms (e.g. a monotonically decreasing exponential model). The form of this upper boundary should be taken into account when defining the appropriate combination of adult and recruitment densities in a stock-rebuilding experiment.

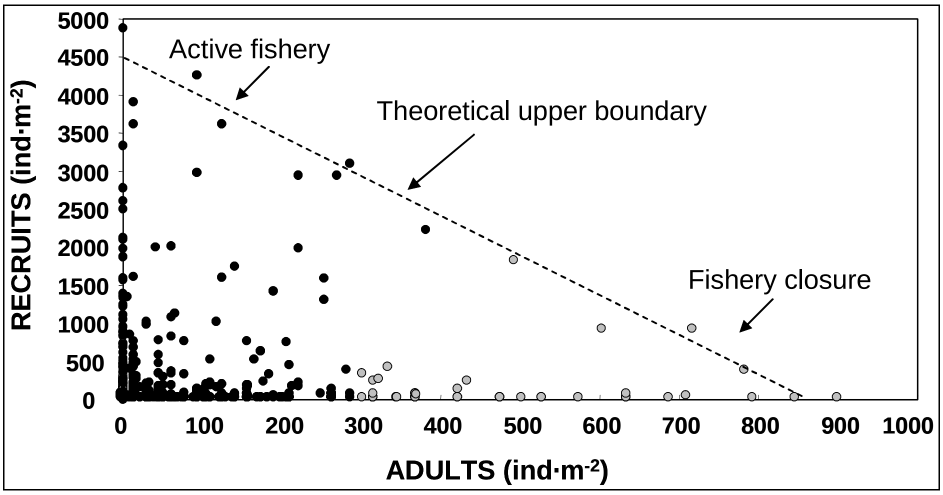

Figure 6.9 Scatter diagram of yellow clam recruit density plotted against adult density in each quadrat, for the months when recruitment peaked: see the difference between recruitment densities observed between 1983 and 1988 (●) under low adult densities and high extracting levels (1983-1987), and in 1989-1990 (○), as a result of the closure of the fishery. The dotted line defines the upper limit of the "envelope" between stock and recruitment, representing maximum recruitment densities for varying levels of maximum adult densities determined following Blackburn, Lawton and Perry (1992: see text for details).

Parsons and Dadswell (1992) found an inverse relationship between growth (shell height, meat weight, and whole weight) and stocking density in juvenile giant scallops, *Placopecten magellanicus*. This could affect OSD estimates, which was also dependent on the overall cultivation strategy type of grow-out technique, and the optimal timing of transfer from the pearl nets. Fréchette, Bergeron and Gagnon (1996) presented a method for estimating OSD via the analysis of the relationship between yield (biomass, B) and population density (N) at harvest, using a B-N diagram (BND). The analysis provided by the authors differs from the usual approach in aquaculture, in which yield is expressed as a function of initial population density, and B and N are analysed separately. Both methods allow estimation of OSD. The BND potentially allows (Fréchette, Bergeron and Gagnon, 1996): (1) assessment of the relative importance of competition-dependent and competition-independent mortality factors; (2) estimation of approximate OSD and maximum yield by extrapolation of results from short-term experiments; and (3) identification of the nature of the factor regulating competition-dependent mortality. They also compared the classical and BND methods using data from mussels grown in suspension cultures, and found that mortality patterns were the same for all stocking densities, and that competition-dependent mortality occurred only at high density. In an experiment designed to test the effect of spat origin (stock effect) on commercial yield, the classical approach suggested that there were no differences in yield and survival, despite differences in growth rate. The biomass-density approach (BN), however, showed that yield was constrained by self-thinning, not by intrinsic properties of the stocks. The BN approach, unlike the classical approach, yielded results consistent with state-of-the-art commercial practice and general knowledge about the stocks tested (Fréchette, Bergeron and Gagnon, 1996). Rheault and Rice (1996) showed that doubling the stocking density from 2.5 to 5.0 kg of oysters *Crassostrea virginica* per bag resulted in a 20 percent decrease in both the condition index and the growth rate (percent increase in weight). These observations may assist commercial growers determine optimal stocking density for their aquaculture grow-out systems. The variation in food concentration superimposed on the tidal current oscillation leads to massive changes in food flux and the degree of local resource competition.

Fréchette and Bacher (1998) noted that estimating physiological rates of the blue mussel *Mytilus edulis* in the field as a central part of carrying capacity studies. They also presented a strategy for estimating site-specific physiological rates based on the modelling of a reference growth experiment at a standard site. Growth of mussels was modelled as a function of population density to obtain estimates of biomass-density and production-density curves for the reference experiment. The authors stressed that these curves provide much of the information usually required for managing cultured populations. They concluded that combining the modelling of reference experiments in this way with particle transport models, may prove useful for assessing optimal stocking density in situations where intensive field work programs are not possible.

Intertidal mussels usually form complex multilayered matrices with density-dependent effects on survival and growth, and self-thinning scaling between biomass (B) and density (D) is expected. Guiñez and Castilla (1999) develop a tridimensional model of space-driven self-thinning that in addition to BN, explicitly includes the degree of packing of the mussels, measured as the number of layers (L). The model BNL could be considered as a generalized one in the sense that it encompasses previous bidimensional models (BN) of self-thinning (e.g. Fréchette and Lefaivre, 1990, 1995; Fréchette and Bacher, 1998) as special cases, and enables comparisons between mono- and multilayered populations. Guiñez and Castilla (1999) contrasted the predictions of the bi- and tridimensional models using data obtained from *Perumytilus purpuratus* mussel beds on the rocky shores of central Chile monitored during a 28-mo period. The B-N-L model suggested that density dependence is much more frequent than hitherto indicated by bidimensional models. The authors also applied their space-driven tridimensional model to other species where spatial overlapping configurations occur, such as the case of tunicate population of *Pyura praeputialis* in the Antofagasta Bay, northern Chile (Guiñez and Castilla, 2001).

7. GUIDELINES FOR CONDUCTING AND EVALUATING STOCK ENHANCEMENT PROGRAMMES

From what has been said up to now, it must be clear that an enhancement programme needs rigorous design, and requires not only in-depth knowledge about the life history of the species, but also an economic evaluation of the activity through the intertemporal flow of benefits and costs. While planning a restocking programme, the following topics should be considered:

7.1 Experimental design

Enhancement programmes need careful experimental design. Whatever the seeding technique selected, the appropriate scale of the experiment must be clearly defined according to the desired objectives (Sainsbury *et al.,* 1997).

7.1.1 Local scale

The wide variety of physical, environmental, biological, economic and social circumstances encountered in shellfish production requires that enhancement programmes should be site-specific. A good strategy is to design such experiments on relatively small spatial scales at first in order to allow a "hypothesis-falsifying" procedure to be followed (see McAllister and Peterman, 1992; Walters, 1997; Castilla and Defeo, 2001 and Section 3.5), incorporating control sites/replicates of e.g. selected spat densities.

In order to conduct enhancement experiments, each experimental unit must be adequately replicated in order that monitoring of growth, survival and production according to specific environmental and habitat characteristics, in such a way that estimates can be established within limits of statistical confidence. In this way, estimates of variability could also be used to evaluate the success of the experiment under uncertainty. Small-scale, replicated plots can be used to evaluate alternative scenarios (e.g. different stocking densities of the enhanced population) and effects of habitat quality (substratum, hydrodynamics). Basic ecological considerations, such as predator-prey interactions and the effects on the benthic community of massive transplantation/ seeding of organisms, should be analysed before extrapolating results to a larger scale.

If adequately replicated, enhanced pilot scale sub-populations in experimental plots can be used to evaluate the success of mixed management strategies within an adaptive framework. For example, small areas could be closed to fishing or even subjected to different intensities of fishing in order to assess the potential benefits both of a rotational management scheme and restocking with seeded juveniles (Brand *et al.,* 1991). If areas of similar productivity are considered, the experiment might be successful even on a short-term basis.

7.1.2 Large scale

The increasing demand for seafood places emphasis on large-scale, commercially oriented, technology and intensive enhancement programmes. Thus, results obtained on an experimental, local or pilot scale, must then be evaluated at larger scales (see May, 1994 for useful concepts relating ecological questions and spatial scales). For example, a large-scale transplantation of spat must consider the ecological implications of such a perturbation on conspecific organisms (e.g. the effects of “genetic contamination” by interbreeding of hatchery stock, which might be less adapted to the environment, with local stocks), and on the benthic community as a whole (see Castilla, 1988; Bailly, 1991; Brand *et al.,* 1991). Schiel (1993) gives one of the most useful examples of the experimental evaluation of commercial-scale enhancement of a shellfish population. He described a large-scale experiment in which growth/survival of seeded abalone *Haliotis iris* was assessed at a range of sites.

As in any ecological experiment, it is difficult to trace a rigorous sampling design in large-scale enhancement operations for the following reasons:

- The varying nature of each site (habitat quality) precludes the definition of adequate control areas and replicates. In many cases areas are so large that replication is almost impossible.

- Difficulties occur in filtering out the effects of a restocking programme from natural fluctuations of the stock or alternative management initiatives. Concomitant changes in fishing effort could also confound the results of the seeding process.

- Stock enhancement is a long-term goal and must be evaluated accordingly. However, it is difficult to wait for a commercially harvestable size to evaluate success. Some projections, based on knowledge obtained from short-term experiments, might help overcome this obstacle.

The experimental design of enhancement exercises therefore requires careful attention to metapopulation dynamics and recruitment processes. In this vein, the existence of a metapopulation offers an opportunity to perform large-scale enhancement experiments in order to evaluate the capacity of the species to restock population subunits previously depleted by fishing or other disturbance (such as red tide outbreaks). Transplanting could be particularly useful where metapopulations have clearly defined "source" or "sink" characteristics (Shepherd and Brown, 1993). In order to conduct active enhancement of a shellfish metapopulation, and define harvest refuges serving as sources of individuals for replenishment or transplanting, information on the early stages of the life cycle is critical. Information on habitat quality or adult density alone is not enough to assure a higher probability of success, and Lipcius, Stockhausen and Eggleston (2001) discounted determining the site of the reserve by chance without information on transport processes of larvae.

Shepherd and Brown (1993) provided a preliminary example on how to apply metapopulation theory to South Australian abalone stocks; the first requirement being to define the complex of substock units. They integrated within this the concept of a "minimum viable population" in order to develop a cost-effective management framework for such a complex of stock units. This and other studies on metapopulations are used here to define a tentative guideline on how to apply metapopulation theory for the purposes of experimental enhancement of shellfish productivity. The main steps could be summarized as follows:

1. Define subareas by extension and number, according to the intrinsic characteristics of the resource and the fishery (i.e. by scale of aggregation of the resource and behaviour and subregional access rights of the fishers). Mapping the fishing grounds and stock abundance should precede the design of a system for acquiring information on the spatial dynamics of settlement over the long-term (see Caddy and Garcia, 1986). Subareas should be easy to identify for fishers and researchers, and should facilitate the collection of spatially accurate information (Cabrera and Defeo, 2001).

2. Estimate the times of settlement and recruitment. Identify potential sources of larvae and discern between source and sink areas. Care should be taken to evaluate larval connectivity

between the discrete areas already defined. Monitor local recruitment of postlarvae and the degree of replenishment of different grounds. It is important to consider here the duration of the larval dispersal stage: those with a shorter larval development period may be more suitable candidates for hatchery rearing and subsequent seeding.

3. Identify key environmental variables, notably intensity and direction of currents that could explain prevailing larval dispersal and settlement. As each ground may be exposed to a distinct regime of primary production, nutrients, food availability, predation and disturbance, these between-ground differences should, if possible, be quantified.

4. Quantify spatial and temporal variations in density of recruits and adults (defined as sexually mature individuals), over a reasonable time frame and by site (e.g. source and sink areas). In each naturally seeded area, acquire information on resource users, local stock dynamics (growth and mortality). Accurate definition of spawning and recruitment timing is critical to providing fine-tuning of the appropriate timing for conducting stock enhancement programmes. Indeed, timing and durations of settlement of many species were often specific and quite short; especially in temperate latitudes (see e.g. Robinson and Tully, 2000). Thus, choosing the specific habitat and time of year for enhancement of a benthic species may be the key to success, and for most species, information on specific environmental requirements is generally lacking. Seasonal lows in abundance of previously established cohorts may represent the most suitable time for releasing juveniles in order to minimize inter- and intraspecific competition and predation, thus highlighting the value of careful ecological study of the potential release sites.

5. Compare growth and mortality information from fished and unfished grounds, in order to isolate effects of density and fishing intensity from those induced by environmental gradients in habitat quality. Growth and mortality patterns should be quantified through time (e.g. under different densities) and in space (e.g. by fishing grounds or LPs) in order to evaluate variations in density-dependent processes and habitat quality. Growth rates of transplanted/seeded individuals must be compared to those of the natural stock under different densities. Estimates of age-specific natural mortality (see Chapter 2) are particularly useful for detecting these critical periods when natural mortality from predation or other causes drops sharply from high values for spat to older animals. In order to have some idea of growth rates and development times from egg to mature adult, some information on environmental factors is critical, and it will be useful to keep time series of relevant water temperatures and wind conditions (e.g. Caddy, 1979c, Botsford, 2001). If development is protracted, and there are high rates of density-dependent mortality (e.g. cannibalism), then culture will be labour-intensive and economically prohibitive. Biometric relationships such as length vs. total weight/muscle weight should be determined to predict the expected meat yield from a mean individual size or age, thus allowing some economic projections for cultivation times.

6. If fishing rights are assigned geographically, quantify spatial variations in fishing intensity using e.g. a composite production modelling approach that includes simultaneous levels of production and fishing intensity from areas with variable intensities of harvesting but a similar basic ecosystem.

7. Estimate appropriate target and limit reference points (*sensu* Caddy and Mahon, 1995) for each LP. Complementary management strategies, such as catch quotas, number of fishers allowed to fish/area of ground, and minimum individual harvestable sizes, should also be

agreed upon. A minimum viable population and optimal fishing mortality or harvest rate should be based on simple yield simulations from known growth and mortality rates, or empirically using the composite production modelling approach.

8. Depending on the inherent spatial characteristics of the metapopulation (see Shepherd and Brown, 1993) and prevailing fishing intensity, an effort should be made to identify existing spawning refugia and nursery areas (for motile species: Herrenkind *et al.,* 1997). In essence, the refugia should be large enough to diminish the risk of stock collapse despite prolonged recruitment failure in LPs due to adverse environmental conditions. Spatio-temporal variability in abundance of stock, larvae and subsequent recruitment, as well as in the prevailing hydrographic regimes, should be considered when evaluating number and/or size of refugia or other spatially explicit management tool (e.g. MPAs). The dimensions of the area protected should be large enough for stock rebuilding purposes within and outside its boundaries.

9. Genetic factors must be taken into account. Classically, it has often been assumed that past enhancement programmes have been successful if populations appear to have been restored in their area of introduction. Testing allele frequency and mtDNA in hatchery stock, and comparing it with that from supposedly successful transplants may however paint a different picture. Thus, Burton and Tegner (2000) found that a red abalone population in California planted in 1979 which supposedly supported the fishery there during the 1980s, resembled other robust natural populations in the region in its genotypic frequencies, and showed no genetic signature of the broodstock used in the transplants. Although the test was not considered conclusive, it does not suggest discarding the previous generalization that hatchery outplants of abalone attempted to date appear to have been unsuccessful. One of the problems of cultivating shellfish for transplanting was illustrated by a genotyping of individual abalone larvae produced in a hatchery (Selvamani, Degnan and Degnan, 2001). Despite attempts to normalize the share of sperm from a number of males used to fertilize eggs in culture, DNA markers revealed that a single father fertilized almost all eggs reaching larval stage. This suggests the need for highly controlled breeding practices to ensure that the genetic diversity of the broodstock for out-planting to the field is maintained. Evidence from finfish culture has already warned of the dangers that repeated enhancement using a narrow genotype will adversely affect species resilience over the long term, and the same message evidently applies to abalone and other invertebrates produced in culture.

7.2 Technical feasibility

While it is easy to import a technology from elsewhere, in many cases enhancement programmes fail when technical problems substantially increase processing costs and lead to serious economic losses. As already mentioned, high costs of production of spat, and high predation on them when released onto the grounds, are critical factors. Other technical problems mentioned are environmental impacts due to seasonally extreme conditions (e.g. ice cover in high latitude waters, summer hypoxia in shallow tropical waters and lagoons, or heavy wave action) and processing constraints (the byssus of *Mytilus edulis* clogs the sorting and cleaning machinery). These kinds of technical problems may lead to a drop in production or compromise enhancement programmes (Kristensen and Hoffmann, 1991). Technical feasibility in rearing larvae, juveniles or adults may also constitute major bottlenecks in enhancement activities. Progress therefore requires a combination of technical applications in the methodology, and ecological acumen (see Peterson, Summerson and Luettich Jr., 1996 for a test of alternative transplantation techniques in scallops).

Facilitation of collection of sufficient numbers of larvae and juveniles from the wild for on-growing is of utmost importance: the timing of collector placement, and collector design, are essential to maximize seed collection during peak settlement periods. A key problem in enhancement programmes is the precise timing of release of juveniles or spat into the wild, in order to minimise mortality rates and costs. Release time should ideally be set after the critical stage in the life history has passed, where this is characterized for example by specialized diet or susceptibility to predation. In general, the longer the rearing time before release, the higher probability of survival. However, this increases production costs, so a trade-off between ecological and economic factors will need to be made in determining the optimum individual size for restocking (Tegner, 1989; Larkin, 1991).

The capacity to rear larval stages through to commercial size before releasing, and thus the appropriate duration of rearing for ranching is critical, i.e. whether the specimens are to be released in a recent post-settlement stage, or as adults, must be evaluated. Economic considerations are critical here, as well as ecological issues (competition, predation). For example, if some early stage is especially vulnerable to predation, it would be better to collect and release juveniles after this critical stage, particularly if natural mortality declines above a given size.

Another constraint may arise when shipping organisms to the transplanted sites. Schiel (1993) found that the greatest stress to transplanted abalone *Haliotis iris* occurred in packing and transport, and here the density per shipping tube needs to be carefully evaluated. The same author reported mortality rates of up to 47 percent in tubes where juveniles of *Haliotis iris* were packed at densities of 1000/tube. Many fragile organisms must be transported in aerated seawater and released at sea at well-defined experimental sites to assure the success of the operation. Peterson, Summerson and Luettich Jr. (1996) evaluated the success of alternative transplant methods for adult bay scallops *Argopecten irradians*, using five different sets of environmental conditions for a 6-h time of transfer from the source to the destination site. They found negligible mortality rates during travel times of up to four h with high flow speed and therefore high oxygen concentration, and this minimized the risks of stress and mortality from handling and transport.

Once in the wild, estimations of survival through recapture rates provide a way of monitoring success. Controlled release onto shellfish habitat, microtagging, and the development of a monitoring plan, represent three important methodological aspects directed at evaluating the technical feasibility of an enhancement programme. If microtagged, the precise location of recaptured animals is needed to evaluate the extent of local movement and the capacity of the seeded animals to restock target or adjacent areas (Addison and Bannister, 1994).

7.3 Economic feasibility

The economic significance of enhancement programmes has still not been fully evaluated, perhaps due to the difficulties in estimating total profits and costs derived from the seeding activity. One economic bottleneck derives from high hatchery costs: these programmes can be prohibitively expensive even for high unit-value resources (Addison and Bannister, 1994). One way to reduce total costs is by releasing juveniles at an early stage. However, this in turn could increase mortality rates after release, because of higher predation rates and susceptibility to environmental variations. This clearly constitutes a bioeconomic trade-off, and must be evaluated accordingly. The maximum cost that could be justified for evaluating new enhancement practices, should be in proportion to the expected benefits and impacts.

Hilborn (1998) reviewed the economic performance of nine marine stock enhancement projects for fish, turtle and lobsters involving restocking. He noted that no project evaluated showed clear evidence of a resulting increase in abundance as a result, but then few were planned in such a way that success criteria or economic performance could be evaluated. His suggestions were that systematic marking of released individuals would help establish survival and population enhancement, with explicit control areas incorporated in a proper statistical design, and subject to prior peer review by experts. The economics of stocking should be compared with other approaches such as habitat protection or improved management of the wild stock, and in this connection, evaluating the various benefits to the stock and ecosystem of protected areas requires close consideration (see e.g. Dixon and Sherman, 1991).

Bioeconomic analysis must be defined at the planning stage of the enhancement programme and must be specific to a local (among grounds) and regional (among countries) basis. Especially the former is crucial for benthic species with marked spatial variations in carrying capacity, recruitment, and growth and mortality rates, which constitute input variables affecting the economic analysis of stock enhancement. Moreover, some economic inputs might differ on a regional scale (e.g. opportunity costs of labour and capital) and thus economic analyses must not be overgeneralized. An analysis of marketing is also needed, because the choice of the species to be enhanced will be guided by demand/supply market laws. Different product types (whole weight, muscle weight), and the corresponding unit prices must be also included in the economic analysis, according to variations in the local/international demand.

In enhancement operations, relatively short sampling periods are used to estimate abundance, growth and survival rates through time (Schiel, 1993). Thus, economic projections should be employed to estimate the net present value (*NPV*) of the enhancement activity: abundance, growth and survival estimates derived from the short-term project must be extrapolated to the period at which organisms will be available to harvesting. An enhancement programme will be economically efficient if it maximizes the *NPV* of the yield obtained, which could be estimated as:

$$NPV = \sum_{i=1}^{t} \frac{TR_t - TC_t}{(1 + d)^t}$$

where *TR* and *TC* are respectively the total revenues and costs in time *t*, and *d* is the discount rate. Total revenues are obtained by multiplying the unit price of the products (e.g. whole weight, muscle weight, shells) by the estimated catch according to specific growth and survival rates. Total costs in each year are mainly based on costs of rearing individuals through a selected “seeding stage”. The discount rate *d* considers the future value of the money invested.

An increasing discount rate diminishes the value of any future yield. Even though traditional values should approach an interest rate of *ca.* five percent, discount rates could generally takes higher values (up to ten percent), mainly as a result of uncertainty about future yields derived from the enhancement activity. As in fisheries, there are high uncertainty levels about changes in costs and prices, stock magnitude, growth and survival rates, and the prevailing economic and market regional situation. Therefore, *d* will tend to increase still further due to a probable expectation about falling prices or rapid depletion of the enhanced stock, implying that high exploitation rates in the short-term will be preferred over a long-term sustainable goal. This is particularly important in open-access regimes, in which free-rider behaviour and externalities commonly occur (Seijo, Defeo and Salas, 1998).

The high variability and difficulties in the estimation of some inputs (supply of spats, recapture, survival and growth rates and economic variables), together with a costly and low intensity of sampling through time, add uncertainty to the estimation of the *NPV*. Thus, different sources of uncertainty should be included when estimating the economic feasibility of the enhancement operation, e.g. different scenarios of growth and survival rates, and prices and costs could be used as inputs to estimate benefit and costs and the corresponding *NPV* of the enhancement plan. Moreover, different *d* values should be used to reflect dissimilar intertemporal preferences in resource use (e.g. different minimum harvestable sizes according to market demand).

After estimating uncertainty in the input variables, a criterion for choice among estimates is needed to provide options to a decision-making body. This could be done through decision theory. Decision analysis applied in fisheries (Hilborn, Pickitch and Francis, 1993; Hilborn and Peterman, 1996) considers alternative bioeconomic states of the fishery with the corresponding probabilities of occurrence, as a function of some possible policy actions. In this context, Bayesian inference allows the simultaneous consideration of multiple hypotheses and the integration of different types of information from many sources, reflecting scientific judgement as well as existing empirical data. Decision analysis could also be used to incorporate the above estimates of uncertainty into choices of enhancement actions. Data gathered in surveys conducted over experimental grounds on which enhancement programmes are taking place, as well as life history parameters derived from these data, could be used to provide a formal assessment of the enhanced stock, and in such cases the Bayesian approach is robust for estimating parameters, despite concerns over possible data outliers and mis-specification of priors (Millar, 2002; Myers *et al.*, 2002).

In the above context, a decision table could be built on the basis of alternative enhancement actions and alternative hypotheses erected about parameter values and their corresponding probabilities of occurrence (Table 7.1). For example, high, medium and low levels of stocking densities ("alternative enhancement decisions") could be evaluated as a function of different hypotheses about resource performance (alternative scenarios of growth/survival rates; time needed to reach the minimum legal size, etc).

In some cases “experience may be insufficient for decision makers to be willing to attach numerical (cardinal) probabilities to the possible outcomes (states of nature)” (Schmid, 1989). Thus, decision tables could be created to account for different alternative system states (columns) and the possible decisions (rows), left with probabilities missing. The likelihood of outcomes could then be ranked only ordinally, and thus decision-makers could make a choice under uncertainty by expressing their subjective judgement about likelihoods in directional and qualitative terms. Schmid (1989) proposed three criteria for dealing with uncertainty and to guide decisions, without the need for explicit statements as to the probabilities of alternative parameter values: Maximin, Minimax and Maximax. These criteria vary according to degree of precaution. The Maximin criterion is a risk-averse approach that leads to selecting the maximum of all minimum outcomes. The Minimax Regret criterion is a less cautious approach that selects the minimum of the maximum regret, defined as the difference between the real benefit and the one that could have been obtained if the correct decision had been taken. Finally, the Maximax criterion is the most optimistic, in that it selects the highest outcome within the decision table.

Once the table is built, the *NPV* of the activity is estimated for each combination of the enhancement actions and parameter values. These criteria were successfully applied in fisheries management (FAO, 1995; Pérez and Defeo, 1996; Seijo, Defeo and Salas, 1998; Defeo and Seijo, 1999) and could be easily adapted to enhancement problems. The reader must refer to the papers above-mentioned for a detailed application of these criteria.

Table 7.1 Key elements of a hypothetical decision table directed at evaluating alternative enhancement options. S1 is a hypothesis that implies a lower level of individual growth rate or a higher survival than S2 and S3. D1 to D3 represent alternative decisions concerning stocking densities. p values represent the probabilities of alternative hypotheses being true, and O_{ij} represent the relative value of the outcome of a given stocking density i as applied to a given growth/survival rate j. O_{ij} values could be regarded as representing net revenues obtained by each enhancement option. Finally, V1 to V3 represent the expected values of each action across all alternative hypotheses. A variance term might be added to each expected V value (after FAO, 1995; Hilborn and Peterman, 1996; Defeo and Seijo, 1999).

Alternative stocking densities	**Alternative hypotheses about parameter values (**e.g. **growth)**			**Expected values**
	H1	**H2**	**H3**	
	P1	**p2**	**p3**	
D1 (50 ind·m^{-2})	O_{11}	O_{21}	O_{31}	V1
D2 (100 ind·m^{-2})	O_{12}	O_{22}	O_{32}	V2
D3 (150 ind·m^{-2})	O_{13}	O_{23}	O_{33}	V3

Given the high uncertainty usually found in the majority of the parameters of an enhancement model, a precautionary approach must be considered suitable for evaluating the economic feasibility of the operation. Thus, not only the lower levels of the confidence intervals of the parameters should be used as inputs to estimates of the *NPV*, but also the criterion that gives a cautious approach (e.g. Maxim*In* Defeo and Seijo, 1999).

7.4 Evaluating the success of enhancement exercises

Enhancement practices have been applied to protect, maintain or improve shellfish populations. Because of the increasing number of enhancement programmes around the world, a scientific approach to evaluate their effectiveness in stock rebuilding is essential. However, the extent to which stock enhancement programmes contributed to natural populations of shellfishes has not been adequately assessed. Indeed, even though intuitively attractive, restocking programmes have been pursued with little evaluation of their success or failure (Addison and Bannister, 1994). Some reasons arise from: (a) the absence of biological knowledge of the species; (b) the lack of definition of clear objectives from the beginning of the planning stage; (c) experimental inadequacies resulting from an undefined methodological framework; and (d) technical problems associated with the supply, maintenance and rearing of spat (Cowx, 1994).

The following steps summarize the information provided earlier in this document and could be considered when assessing success of any enhancement plan:

(1) ***Determine the initial number and size structure of seeded organisms, together with the sites of placement. If possible, mark or differentiate them from wild animals. Use control, unseeded sites for comparative purposes. Characterize each site as precisely as possible***.

In order to evaluate the success of any seeding experiment, seeded animals should be microtagged (Wickins, Beard and Jones, 1986), thus permitting the identification of hatchery-reared animals during subsequent field sampling and monitoring of commercial landings. It is commonly difficult to discern whether hatchery-reared animals have survived in addition to, or at the expense of, natural stock. However, Schiel (1993) suggested an effective and cheap means of tagging abalone indelibly by allowing abalones to feed sequentially on different algae before releasing. A continuous switching between algae generates alternating bands of different colours on the apex of the shell that can be seen for several years. At least this is applicable to abalone stocks.

A target density should ideally be estimated for seeding. It will be based on previously acquired knowledge about the SRR and carrying capacity of the system. A range of sites and, if possible, densities at each site, should be used to test hypotheses related to habitat quality and variations in the carrying capacity in each habitat. Data must be interpreted quantitatively in order to assess among-site variations in growth and survival rates and the success of active restocking. Unseeded sites should be useful controls for comparative purposes. Some measures of the effectiveness of the restocking process should also be quantified. Site-specific survival and individual growth rates of released animals, from the beginning of the seeding process to the time at which the individuals become available to harvesting, could be used for this purpose.

(2) ***Estimate abundance variations through successive and periodic sampling. Estimate survival and individual growth rates and compare them with those of the wild stock***.

Length-frequency analysis should be clearly the best way to provide estimates of growth and survival rates. Overall growth and mortality patterns must be compared to those of unseeded sites and also among seeded sites. ANOVA procedures should be useful for this purpose.

As mentioned above, an enhancement plan is essentially long-term. However, it will be difficult to carry on sampling for years in order to estimate population dynamics features (growth, survival) until the harvestable size is reached. This is almost impossible for long-lived shellfish. Thus, projections of growth and mortality rates must be done from the seeding stage to the length at which organisms reach the minimum legal size.

(3) ***Estimate the number of microtagged organisms that survived to the harvestable size (biological samplings) and the relative contribution of the enhancement operation to the global landings from the whole area (by sampling landings and markets)***.

The success of stock enhancement programmes should be evaluated by field sampling (target fishing close to the release sites) and by monitoring fishery landings. Stock enhancement, if effective, can be detected from the concomitant increase of fishing yields reported by fishers' logbooks. Concerning this important issue, Kitada, Taga and Kishino (1992, and references therein) reviewed four groups of methods for estimating of the effectiveness of enhancement programmes on the basis of tag recoveries, which can be summarized as follows:

(a) <u>Estimation of total recoveries from fishers' reports</u>. Tag recoveries are intuitively attractive because of low costs of acquiring information (Crowe, Dobson and Lee, 2001). However, the proportion of recaptured animals tends to underestimate the survival rate of seeds and the consequent measure of effectiveness of the enhancement programme, for several reasons (Addison and Bannister, 1994):

- A substantial part of the catch escapes monitoring.

- Landings away from the release site, and thus with low probability of recapture, tend not to be monitored.

- Estimates of abundance from tagging are based on some marked to unmarked ratio. However, dissimilar behaviour between marked and unmarked animals (survival rates of the former group tend to be lower), together with a generally low chance of recapture because of low percentages of marked animals, usually lead to underestimates of abundance (Hilborn and Walters, 1992).

These limitations could be mitigated by intensive sampling in the field and of landings (Kitada, Taga and Kishino, 1992). Bannister and Pawson (1991) showed that microtagged *Homarus americanus* in field samples at releasing sites constituted up to 50 percent of the catch on specific days and ten percent over a season, including egg-bearing females. This fact unambiguously shows that hatchery-reared animals survived to maturity and contributed to the breeding stock. However, scientific results concerning this point for most examples are usually either nonexistent or inconclusive.

(b) Correlation between annual number of fingerlings and the corresponding landing weight. This method could be an option for shellfish with short life spans and relatively stable recruitment rates. However, recruitment tends to be highly variable and not related to the amount of the parental stock but to show environmentally driven fluctuations in early life stages. Even should the above assumption be valid, it is difficult to discern between increasing landings as the response to the enhancement programme, or as a result of natural recruitment. The situation is complicated when several age-classes are contributing to spawning.

(c) Prediction of recoveries by calculating yield per release based on the catch equation and simulation models. This method is a complement to the others mentioned, because the recovery rate of shellfish released is not taken into account. Once this measure is quantified, simulation models could be performed to evaluate the effectiveness of restocking.

(d) Sampling surveys of commercial landings and fish markets. Kitada, Taga and Kishino (1992) suggested that a proper estimate of recovery could not always be obtained by these three groups of methods. They proposed a two-stage sampling survey of markets of cooperative associations of fishers (primary sample unit) and landing days (secondary sample unit) to estimate the success of enhancement programmes. Measures of effectiveness included the ratio of marked animals in the landings and recovery rates. These estimates were then used to evaluate the economic feasibility of the programme.

(4) ***Perform an economic analysis of the activity through the estimation of the net present value of the intertemporal flow of benefits and costs. Use different discount rates to reflect dissimilar intertemporal preferences of society in resource use. Identify some possible bottlenecks that might have to be mitigated in order to reduce costs.***

As detailed earlier in this Chapter, the economic success of any restocking programme must be assessed to evaluate its commercial viability. To this end, costs (variable and fixed) and economic revenues must be carefully estimated in order to have indicators as to the feasibility of the operation. Simple spreadsheet methods incorporating life history parameters have commonly been used for calculating mortality and growth of fish populations (see e.g. Sparre and Venema, 1992) using for example the Thompson and Bell procedure (Ricker, 1975). This approach has been employed for modelling abalone populations (see Sanders and Beinssen, 1998 and De Waal and Cook, 2001), who have extended it to incorporate a cost-benefit analysis. The economic effectiveness of a seeding operation under different conditions of survival, growth, labour costs and product sale prices can be investigated. De Waal and Cook (2001) show that ranching shellfish is only likely to be economically viable where mortality is not excessive and survival rate increases with age, which of course is generally the case (Caddy, 1991, 1996).

(5) ***Estimate uncertainty in the main inputs of the enhancement model,*** i.e. ***from growth and survival rates to unit prices and costs. Employ for this purpose alternative hypotheses for parameter values to predict outcomes from alternative enhancement (***e.g. ***stocking densities) strategies in a decision analysis.***

Uncertainty and risk analyses must be conducted to evaluate the bioeconomic feasibility of a stock enhancement programme. For example, the profit from a stock enhancement programme for a flatfish (the example is valid also for shellfish), as estimated by Kitada, Taga and Kishino (1992) was US$ 63 000, but the 95 percent confidence interval ranged between unprofitable and profitable [- US$ 4 000 to US$ 151 000].

Given the high variability in outcomes, a precautionary approach should be used to minimize risks. Some specific Reference Points (Caddy and Mahon, 1995; FAO, 1995, 1996) should be used as targets. In this specific case, Reference Points are not necessarily those derived from classic surplus production and yield per recruit models, and conventionally used to manage fisheries (e.g. *MSY*, F_{MAX}, etc). Such models assume that recruitment is constant and rarely include input for recruitment variability, which can be one of the main sources of variability in invertebrate populations (Conan, 1986; Caddy, 1989b). In this specific case, variable stocking densities should be included as an option.

(6) ***Try to reduce uncertainty in input variables by achieving as accurate biological and economic data as possible as a result of a rigorous experimental design. Focus research on improving the performance of different enhancement strategies. Develop methods for optimizing the monitoring system***.

Post-stocking evaluation has been largely neglected in enhancement programmes (Cowx, 1994). An enhancement programme requires explicit specification of the information needed to achieve enhancement objectives, taking into account all the processes (e.g. growth, mortality, prices, and market demand) required to ensure that these needs are met. Periodic evaluation and revision of the data collection and the results achieved is necessary. This should aid in reducing uncertainty in key variables, which in turn will affect the *NPV* from the activity. The evaluation should assess the long-term benefits of alternative stocking practices and regimes, and attempt to identify those factors precluding enhancement success.

8. MANAGEMENT OF ENHANCEMENT AND USER RIGHTS

8.1 The social context of stock enhancement

The implementation of stock enhancement as a management strategy requires a review of who has access to the resource, and if this has not yet been done, an allocation of rights. Indeed, stock enhancement initiatives are a waste of time if not complemented by additional management strategies directed to sustaining the activity over time. If the fishery is under an open access system, it is not clear how the biological and economic benefits of enhancement can be properly realized.

The social context is the key to success for local fishery restoration, if inshore fishery management programmes are to succeed. Local municipal control of shellfisheries is a common phenomenon in the USA, and has given rise to a diversity of shellfish cultivation techniques, some of which have been described in this report. The history of a shellfish management programme as described by MacFarlane (1998) on Cape Cod, Massachussetts, is of interest. This evolved from relaying native oysters, to the use of hatchery-raised seed and several approaches to nursery culture: bottom culture, raft culture, a municipal hatchery, a land-based upweller system, tidal upweller, and floating trays. The programme always operated under ongoing financial restrictions and changing political and social factors. The management priority was primarily on the high survival of spat rather than fast growth, and the most successful approach was found to be a land-based upweller system which provided 1 million seed/year with 95 percent survival. Subsequent survival in the field after relaying was determined mainly by the environmental conditions at the time of planting.

A further example is provided by MacFarlane (1996) of a local socially driven programme which arose from concern about declining stocks of municipally managed shellfish species. In this case, deterioration of water quality and habitat forced the local town council to address the causes of environmental degradation through instituting a water-quality task force, with terms of reference to recommend changes in land-use practices. This led to a drainage remediation programme, and resulted in the reopening of a shellfish ground after a 12 year closure for reasons of public health. The issues mentioned that adversely affected shellfish quality and hence enhancement, were nutrient runoff, groundwater, flushing rate of bays, and contamination associated with proliferation of private docks in the public tidelands, as well as the effects of beach dynamics and the erosion control mechanisms installed.

One conclusion that leads to a more specific and appropriate use of coastal areas with minimal negative interaction, is to ensure that user rights are specified for specific subareas of the coastal area within a realistic map, preferably specified within a GIS (Geographical Information System) (Taconet and Bensch, 2000; Manson and Die, 2001). This can become an essential basis for consideration by the local and regional authorities of suitable areas where exclusive user rights can permit cost-effective stock restoration. In many developing countries the question of creation of sources of employment is politically important, and often fisheries have been the “employment of last resort” for impoverished peoples. Removing this option, following the logic of restricted entry beloved of economists of developed countries, can have serious consequences on the quality of life and diet for the rural poor. It is this aspect that often leads to reluctance to allow exclusive user rights to coastal populations, but the best alternative seems to be to ensure that these rights are delegated to the community for reallocation to community members of specific areas/resources for stock enhancement.

8.2 Legislation and ownership

The elucidation of ownership is a critical issue in enhancement programmes. Free-rider behaviour under unrestricted access is a common feature in coastal shellfisheries (Seijo, 1993; Shepherd and Rodda, 2001). Enhancement of high-value shellfish stocks under a specified ownership regime will discourage unproductive investment (in time or money) by groups of fishers which will tend to absorb an important share of the enhancement benefits. Thus, methods of restricting access to the enhanced stock must be introduced, together with some legislation to protect rights of fishing of those persons or organizations that invested in the enhancement programme (Castilla, 1994; Addison and Bannister, 1994; Castilla and Defeo, 2001). This issue of limiting access to the fishery is still controversial: Mahon *et al.* (2003) stressed that, whereas some fishers recognize that the most efficient way to control sea urchin harvesting in Barbados is by limiting the number of fishers, the majority are of the view that no one should be prevented from harvesting, and that overfishing should be controlled by adjusting the length of the fishing season.

Institutional changes are needed in support of enhancement schemes, based on an adequate legislation that must recognize the concept of ownership and adequate use rights to protect investments (Bailly, 1991; Troadec, 1991; Bannister and Pawson, 1991; Castilla, 1994). This topic has been considered as a necessary condition for any enhancement programme to succeed (Larkin, 1991). It becomes critical to identify those who pay the hatchery costs in order to assign them the corresponding benefits. Thus, some legislation directed to protect the rights of authorized fishers will be required to ensure that only those who invested or who were responsible for the enhancement programme can benefit from the increased stocks. If a private company or fishery cooperative releases spat, there must be some confidence as to the benefits that will be obtained. This consideration increases in importance with the increasing scarcity and cost of catching wild shellfish, which in turn makes stock enhancement procedures economically attractive. If the resource is under open access, there is no basis for any private investment in stock enhancement, and little or no return to government from doing so.

The scale and objectives of stock enhancement are related to the entry or person who receives the allocation of rights. If it is intended to enhance the stock "for the public good", the scale of operation should probably be larger than in a strict private context. In the former case, some efforts must be made to legislate criteria for allocation of rights so as to favour those who wish to participate in enhancement of the fishery, and we may suggest that evidence of adequate funding set aside for the purpose would be one criterion for participation. Conflicts of interest might occur between different groups of resource users, as well as between fishers and other marine activities (see Bannister and Pawson, 1991). In cases of private hatcheries, the scale of enhancement should be smaller and restricted to those areas with specific rights of access. Commercial fishing licenses might be required for this purpose. Despite the above considerations, problems related to the allocation of space or catch between investing and non-investing groups are likely to remain.

Castilla (1994) illustrated a successful example of institutionalization of management practices in the Chilean small-scale benthic shellfisheries, notably those based on a mixed scheme. This included enhancement, together with allocation of rights through fishery preserves or concessions. The Chilean artisanal fishery activity is developed along more than 4 000 km of coastline, including 200 small coastal villages, coves or “caletas”. After an increasing period of landings, which peaked in 1991 with *ca.* 150 000 t, many Chilean shellfishes were overexploited. In 1991 a new Chilean law was approved, and incorporated the concept of

"Maritime Destination", a management area for benthic resources accessed only by duly organized artisanal communities pertaining to each cove. Access to these areas by community members is free of cost, and based on an agreement on management and exploitation plans between fishers and the fishery authority. The management plan is periodically reviewed according to specific rules established in the legislation. The marine concessions were used to evaluate the recovering capacity of some shellfishes in the absence of fishing *i.e.* passive or natural restocking. Alternatively, the local community can subject these grounds to specific enhancement activities, including the granting of permits to use seed collectors. Thus, efficient management practices were accompanied by some sort of ownership, by which the artisanal community defended their grounds and promoted stock enhancement as in agriculture. Although there are several “similar” examples in other benthic shellfisheries, cultural perceptions, legal, political and economic factors, degree of knowledge about the resource and even the geography of each coastline are different in each case, and thus there is not a magic rule to apply to provide successful enhancement results (Castilla and Defeo, 2001).

The sessile or sedentary nature of shellfishes favours the allocation of TURFs to individuals or groups on specific grounds. However, shellfish have marked spatial variations in abundance, so that some rationale must be found to allocate ownership or access rights as a function of the relative productivity of each area. Priority might be given to those fishers with longer activity in the fishery. Grounds might be transferable according to the performance of each fisher, which could be evaluated on a communal basis. An example of this is given by Seijo (1993) for the collectively managed spiny lobster fishery of Punta Allen, Mexico. This isolated coastal community in the Yucatan Peninsula is a collective voluntary organization that performs informal privatization of fishing grounds to sustain resource rent over time. The temporary (renting) or permanent (selling) transfer of individual rights to fishing grounds involves simple artisanal transactions: a specific payment is made according to ground size and its perceived relative productivity in previous years. Permanent transfer of fishing grounds between cooperative members may include monetary payments and/or barter transactions. A variety of penalties imposed by strong community rules and self-policing strategies assure a relatively stable development of the community. Stock enhancement in this context is promoted through the use of artificial habitats or “sombras”, so that each fishing ground can be subjected to a variety of enhancement initiatives as a result of a community-based management scheme (see also Miller, 1994 and references therein). Similar concepts were proposed by Brand *et al.* (1991) for the pectinid fisheries of the Isle of Man: the success of large-scale transplantations of spat depends on the voluntary cooperation of the local community (see example below). However, enforcement becomes more difficult as the number of fishers, landing sites, and regulated species, increases. The success of the earlier examples basically relies on the relative isolation of the local communities and the restricted scale of the territorial permit, which in turn favours the implementation of self-policing strategies and voluntary cooperative action to avoid the infringement of rules and free-rider behaviour (Seijo, 1993).

Jamieson (1986) explained the rationale behind fishery regulations on invertebrates in British Columbia, Canada, classifying them by management concern (Table 8.1), which illustrates the many and varied practical, theoretical, and administrative issues that require attention from fisheries scientists in a varied invertebrate fishery:

8.3 Co-management

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). In contrast, minimum management controls need to be evaluated periodically to ensure that the privileged group is making socially acceptable use of the resource. In this context, effective control could be achieved through the joint management by fishers and government, *i.e.* co-management. Here, resource users must ideally be incorporated at various levels into management decision-making through active consultation within those bodies responsible for management. Moreover, the local community should be authorized to enforce and assure (through internal rules and self-policing strategies) that management tools (gear regulations, quotas, closed seasons/areas, harvest limits) are being respected, and free rider behaviour minimized or avoided. Hanna (1994) briefly documented the case of the Maine soft-shell clam *Mya arenaria* as a typical example of co-management, in which the State and the coastal towns share the control of management. "The local communities with approved shellfish conservation programmes are authorized to design and implement management plans which set harvest limits, establish open and closed areas, establish the rules of access and enforce regulations" (Hanna, 1994: p. 234). This is critical when an active enhancement of productivity is projected.

Table 8.1 Rationales behind invertebrate fisheries regulations: the British Columbia example (from Jamieson, 1986).

	Management concern	**Management measures**	**Species**
1	Conservation	- Area quotas and seasonal closures - Gear restrictions	- Abalone, geoduck, shrimp (trawl), sea-urchin - All species
2	Allocation	- Vessel quotas	- Abalone
3	Stability of return	- Minimum size limit - Limited entry - Area quota - Seasonal closure	- Abalone, intertidal clams, crabs, sea urchin - Abalone, geoducks, horse clams, shrimp (trawl) - Geoduck - Prawn, shrimp (trap)
4	Conflict over grounds/ resources	- Area closures - Quotas - Seasonal closures	- Shrimp (trawl and trap), euphausids - Euphausids - Euphausids
5	Processing economics	- Seasonal closures	- Crab, sea urchin
6	Social factors	- Closed areas - Human health closures	- Abalone, clams octopus, crabs - Horse clams, intertidal clams, goose barnacles
7	Administration	- Closed areas - Fishing log completion - Research study areas - License requirements	- Abalone - Abalone, geoduck, shrimp (trawl and trap), octopus, goose barnacles, euphausids, sea cucumber, sea urchin - Geoduck, shrimp trawl, sea cucumber - (Almost) all species

While it is generally accepted that "co-management is an effective means of minimizing conflict in fisheries management and recirculating the benefits of effective management back into the local communities" (Noble, 2000), the development of this strategic institutional structure (*sensu* Orensanz and Jamieson, 1998) has been slowed by institutional constraints. Institutions are important prerequisites to effective co-management, and form the substrate from which decisions are made and collective action is taken. In a 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, but also add modern elements of fisheries management. Thus, once this strategic institutional arrangement is in place, additional, risk-averse, precautionary management schemes could be gradually introduced (Castilla and Defeo, 2001).

Much attention has focused around co-management as a process for realizing effective fisheries management. In the light of the current dangerous state of many shellfish resources, a reasonable attitude to conduct enhancement initiatives is to "close the fisheries management cycle" (see Chapter 2) by involving the fishers communities in designing stock-rebuilding programmes. Adopting the traffic light approach (*sensu* Caddy, 2002) to management potentially restores to the local communities the necessary range of data for informed decision-making, and more control over their traditional fishing grounds (Castilla, 1994; Hanna, 1994). The absence of co-management practices supported by appropriate legislation, and guided by reliable data is a critical factor that has led to the collapse of coastal small-scale benthic fisheries around the world. Scientists and policy-makers must learn from the various forms of community-based management followed for centuries by traditional fisher communities, and not assume that traditional approaches must be discarded, as opposed to updated. Frequently this is the opposite approach to that followed by fishery management bodies over the past 30-40 years. Local communities need to agree on appropriate responses when an increasing number of indices move beyond their LRPs into a "red" category (Annex I), which justifies closure of fisheries for stock-rebuilding purposes. Once this agreement is achieved, local fishers must participate actively in the implementation and control and surveillance activities, and the management measures needed to restore stocks to health. They should know what indices, what values of indices, and why, should lead to prompt action by stakeholders. Thus, co-management of fisheries is likely to provide the context for applying traffic light control systems, since top-down management approaches arguably have not worked (Castilla and Defeo, 2001). The fruitful interaction among fishers, policy makers, scientists, extension workers and politicians should provide a comprehensive course of action in scope, including cooperation in setting up easily understandable and reactive mechanisms to respond to overfishing indicators (Caddy, 2002).

Castilla and Defeo (2001) concluded 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 authors also highlighted the advantages of institutionalizing co-management procedures for stock-rebuilding purposes. The most important factors supporting this statement concern the development of enhancement programmes, and were summarized by the authors as follows:

1. A comparatively reduced scale of fishing operations and well-defined boundaries for each management sub-unit is required. Whenever possible, the scale of the management unit should ideally be that corresponding to the range of activities of the local fishing community,

thus facilitating the application of co-management, as demonstrated by the successful Chilean examples documented in this text.

2. Allocation of institutionalized co-ownership authority and responsibility to fishers in shellfish management decisions and actions concerning stock enhancement programmes needs to be explicit (Pinkerton, 1994; Gimbel, 1994; Pomeroy and Williams, 1994; Mahon *et al.,* 2003). Shellfish co-management needs to be institutionalized within a legal framework including well-defined fishers' rights, responsibilities and a clear identification of the community role in the management process. Participation of fishers will improve shellfish management: and the perception of ownership by fishers is the most important focal point determining co-management success (Castilla, 1994). Informal government recognition is not enough for allocation of TURFs or other fishing rights and ad hoc implementation of co-management systems. Several examples which included the voluntary participation of the fishers in enforcing regulations became unsuccessful years later, due to changing management policies (Defeo, 2003). Fishers felt themselves unprotected under an uncertain management environment, and changed their long-term, "sustainable" perspective on the fishery to a short-term, profit-maximizing behavior. The legitimacy of co-management and the perception of ownership by fishers should override or constrain expectations of the benefits to be derived from shellfish extraction. The assignment of fishing grounds to well-defined groups of fishers represents the recognition of the role of local communities in conservation and management.

3. Communal ownership encourages cooperation among fishers and improves surveillance of regulations, and reduces information and enforcement costs. Well organized fisher communities take good care of their assigned fishing grounds by preventing illegal extractions. This has had major repercussions on yields, product quality (individual sizes far above the minimum legal size permitted) and economic returns (Castilla, 1997). In some cases, 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). Together, these may significantly increase yields from enhanced stocks. Thus, fishers play an outstanding role in the implementation and surveillance of regulations, and the reduction of enforcement costs. This is of utmost importance, because it has been widely documented that, at least in developing countries, operational and quota-based management measures are extremely difficult to enforce and are beyond the finances of most management agencies. Moreover, reliable information flowing from fishers to scientists implies lower monitoring and enforcement costs, and provides fine-grained signals about resource status, which allows spatially explicit management measures (e.g. ground closures) to be established. Implementation of regulatory measures in a co-management context provides an 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).

4. Improvement of the quality and quantity of fishery information results from cooperation and improved information flow. Cooperation among scientists, fishers and managers exponentially increases the quality and quantity of fishery information, with clear management connotations (McCay, 1989), reducing the misreporting and uncertainties inherent to stock assessment. 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 has a synergistic effect on the acquisition of

knowledge on the dynamics of the stock and the fishery (Pinkerton, 1994, 1999; Jentoft, McCay and Wilson, 1998).

5. Existence of community fishery traditions needs to be conserved. Fisher communities that have taken the responsibility for managing coastal shellfish resources, often build upon old or traditional roots (Castilla, 1994; Johannes, 1998). Ancient collective organizations often found in coastal shellfisheries include strong community rules and voluntary self-policing tools. Small groups with clearly defined members and leadership encourage cooperation, and promote the identification and exclusion of non-contributing users. Thus, trust among fishers and group cohesion is necessary conditions to improving co-management (Pomeroy and Williams, 1994).

6. Allocation of TURFs has proved an effective tool where geographically restricted harvesting occurs. When accompanied by co-management, allocation of TURFs ameliorates the weaknesses of enforcement regulations, diminishing information and enforcement costs (Mantjoro, 1996). In these cases, fishers play an outstanding role in the implementation and surveillance of regulations, improving the status of shellfisheries, increasing abundance, individual sizes of the specimens and the economic benefits derived from the enhanced stocks (Seijo, 1993; Castilla, 1994, González, 1996; Castilla *et al.*, 1998). Given the current state of most benthic shellfish stocks around the world and the continuing declining trend or collapse of many resources, effective management is likely to be a hybrid of traditional and modern arrangements. The community may allocate extraction quotas, access rules and self-policing strategies among fishers, 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: Castilla and Defeo, 2001). The local relevance of a given mix of management strategies will depend on the kind of resources to be enhanced and managed, and the nature of the ecosystem inhabited by the species. Some pros and cons of different management schemes in shellfish populations are discussed in Chapter 2.

7. Co-management improves the results of enhancement experiments and the application of spatially explicit management tools (e.g. reproductive refugia, rotation of grounds, natural re-stocking). Management experiments without the help and advice of fishers are nonsense. The joint venture into enhancement experiments between fishers, scientists and managers promotes a better understanding of the biology of shellfish stocks and leads to adequate administration of wild resources and/or habitats for conservation and management. Experimental management units (e.g. involving TURFs), with dissimilar effort levels in each, could be the subject of a rigorous experimental design in which the spatial and temporal coupling of operational management tools (i.e. management redundancy) could be evaluated through specific “area-season windows” (Caddy, 1999a) to consolidate a sustainable management framework for shellfish. Participation of fishers is of critical importance in assuring unbiased reporting of results and implementation of an up to date information flow from fishers to scientists, as well as in enforcing regulations through their participation throughout the enhancement experiment.