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Impacts of Aquaculture and Mitigation Measures

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ABSTRACT

The role of aquaculture in world food production is increasing very fast, contributing with more than 40% for the total production of aquatic organisms. The general approach in modern aquaculture resembles much that of industrial agriculture and husbandry, with large energy subsidies and the usage of many chemicals in, predominantly, monoculture systems, with a large ecological footprint. In spite of the large body of regulation available worldwide, there are important ecologic, economic and social impacts in many countries as a result of aquaculture. In some cases, the anticipation of these impacts by local populations represents a negative feedback for aquaculture development. In the present work, a review of those impacts is presented, followed by a discussion of the carrying capacity concept, then by presenting some approaches and methods that may help planning aquaculture developments including the Drives Pressures States Impacts Responses framework, modelling and Decision Support Systems and, finally, by a synthesis of aquaculture related legislation worldwide. The analysis of a large number of works suggests that aquaculture management should be participated by local stakeholders and viewed within the context of other management approaches, such as Integrated Coastal Zone Management. This may allow for a better ecosystemic integration of aquaculture with other activities in line with Ecological Engineering concepts. Likely, there should be more investment in low-trophic level species to reduce aquaculture ecological footprint.

Keywords: carrying capacity, ecological aquaculture, ecological engineering
Abbreviations: BEP, Best Environmental Practices; **BMP**, Best Management Practices; **CC**, Carrying Capacity; **DSS**, Decision Support Systems; **DPSIR**, Drivers Pressures States Impacts Responses; **EIA**, Environmental Impact Assessment; **EEA**, Ecosystem Approach to Aquaculture; **GIS**, Geographic Information Systems; **GMO**, Genetically Modified Organisms; **ICZM**, Integrated Coastal Zone Management; **IWRM**, Integrated Water Resource Management; **HACCP**, Hazard Analysis and Critical Control Point; **RAS**, Recirculating Aquaculture Systems

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INTRODUCTION

The contribution of aquaculture to global production of aquatic organisms increased from c.a. 32%, in 2000, to 42%, in 2006, according to the FAO Fishery Statistical Collections (FAO, online). Following the same source, total aquaculture production increased over 18 times for the period 1997–2006, from 3,584,160 to 66,728,941 tonnes. Consi-

Received: 1 November, 2008. Accepted: 15 November, 2008.

Invited Review

dering the mentioned growth and that intensive aquaculture developed over the last years (Muir 2005), it is expected that its environmental impacts have also increased. These impacts as well as the sustainability of aquaculture were discussed in previous works (e.g. GESAMP 2001; SECRU 2002; Read and Fernandes 2003; GESAMP 2008). However, there are still several issues to clarify about how to guarantee aquaculture sustainability, giving the vagueness of the concept and the lack of a general paradigm to handle this problem that, together with social awareness, creates some negative-feedbacks to aquaculture development in regions where perceived costs outweigh the perceived benefits by local stakeholders (Gibbs 2009). Therefore, the main purpose of this work is to synthesize information and concepts that may be useful in defining a paradigm towards aquaculture sustainability.

This work is structured as follows: Much of what is known about environmental impacts of aquaculture is synthesised in a chapter about Aquaculture Environmental Impacts. In the following chapter, the carrying capacity concept and its application to aquaculture are discussed, together with methods for its quantification. In a chapter about aquaculture and environmental management, some approaches and tools that may help to manage aquaculture towards sustainability are presented. A review of available regulations on aquaculture management is given in the next chapter. Finally, some general conclusions are attempted.

Aquaculture industry seems to be following the same steps as agriculture: from traditional polyculture systems with low energy subsidies, to intensive monocultures with high energy inputs and biotechnological innovation. These high energy inputs are in the form of trophic energy, such as sun light and fish food, and auxiliary energy, such as renewable and non-renewable energy sources, to maintain production operations. In many countries, there is a strong investment in the production of carnivore species that comprise more than 30% of world aquaculture production in monetary terms (Primavera 2006), implying a relatively small efficiency in the conversion of primary production, though several steps of the food web, and a large ecological footprint. Furthermore, high intensive aquaculture systems require more pharmaceuticals and other chemicals to protect organisms from disease.

Perhaps one of the main problems in aquaculture, as well as in other human activities, is the apparent difficulty of people to think holistically, especially in industrialized societies, where compartmentalization is frequently equated with efficiency. Due to this limitation, local developments are planned without much consideration about integration with other activities, leading to the production of wastes that may represent an environmental problem when, if otherwise planned, could serve as raw materials for another activity. If some sort of integrated management is applied, such as Integrated Coastal Zone Management, with considerations about the spatial distribution of different activities, to guarantee proper access to resources by all stakeholders but, without consideration of material and energy fluxes related to different activities, there may still be sustainability problems. Therefore, traditional Chinese aquaculture-agriculture-husbandry-waste treatment systems may serve as a good example of empirical yet, holistic approaches, to be incorporated in modern developments but in tight interaction with scientific methods, well in line with the principles of Ecological Engineering (e.g. Yan and Ma 1997; Mitsch 1997).

Aquaculture may be important to alleviate poverty by generating food, employment and wealth if a more equitable distribution of its benefits is assumed. Its environmental impacts should be assessed at a larger scale than the farm scale, due to the cumulative effects of several farm operations in the same area and their combination with other human activities (GESAMP 2001). However, if aquaculture development is planned in isolation from other activities, if it implies changes in resource ownership, preventing local people from having access to resources, and it leads to concentration of wealth in a few people, its environment, economic and social effects are unsustainable.

AQUACULTURE ENVIRONMENTAL IMPACTS

Aquaculture units can generate considerable amounts of wastes/effluents containing a variety of substances such as, particulate material (mainly resulting from uneaten feed and faecal material), dissolved metabolites (from excretion via gills and kidneys), and various forms of chemicals (e.g. therapeutants, fertilizers, heavy metals), with undesirable environmental consequences (Wu 1995; Kelly *et al.* 1996; Deb 1998; Tovar *et al.* 2000a, 2000b; Pearson and Black 2001; Páez-Osuna 2001a, 2001b; Read and Fernandes 2003). The environmental impact resulting from particulate and dissolved organic and inorganic material (**Table 1**) is particularly important because these compounds are directly discharged into the environment affecting both the water column and the sediment compartment (Dalsgaard and Krause-Jensen 2006; Holmer *et al.* 2007). The magnitude of these impacts depends mainly on farm location, species, culture type, stocking densities, food digestibility, and on other husbandry factors such as feeding practices and disease status (Wu 1995).

The meteorological (e.g. wind patterns), hydrographical (e.g. bathymetry, currents, tidal regime, wave action, sedimentation rates) and geomorphological characteristics of aquaculture sites (Kempf *et al.* 2002; Nordvarg and Hakanson 2002; Kalantzi and Karakassis 2006; Rodriguez-Gallego *et al.* 2008), strongly influence the fate of any type of waste released into the water column. For instance, high-energy environments, well swept by bottom currents, are usually less affected by the impacts of waste material than low-energy environments, most likely due to the contribution of hydrodynamics to the dissipation and dispersion of exogenous material (Klaoudatos *et al.* 2006). Furthermore, re-suspension periodically re-exposes superficial sediments and waste products to oxygen, enhancing organic matter decomposition (Burdige 2006). Conversely, in shallow waters or in restricted exchange environments (e.g. semi-enclosed estuaries, bays or fjords) with weak bottom currents, there is a higher risk of particulate organic matter and nutrients to increase locally (Wallin and Hakanson 1991), causing not only the degradation of water quality but also severe negative impacts on benthic assemblages.

Effluents from intensive production systems, with a large feed input, typically have greater negative impacts than effluents from semi-intensive or extensive systems with little or no feed addition (Kautsky *et al.* 2000; Páez-Osuna 2001a; Banas *et al.* 2008). However, the economic viability of these systems, relying mostly on natural food, is usually compromised by their limited capacity to control environmental and husbandry factors (e.g. nutrition, predators and disease agents), and by their low productivity. To turn aquaculture into a more productive activity with improved profit margins, farmers worldwide have been intensifying production (World Bank 2006). As stocking densities increase, the systems increasingly require higher water volumes, use of feeds and chemicals, which substantially increase organic and inorganic loadings. For example, the ecological footprint of semi-intensive tilapia production systems is relatively low (approximately equal to the farm area) compared to intensive systems that require an area up to 10000 times higher than the farm area (Folke *et al.* 1998). The higher the degree of artificiality, more likely is the occurrence of environmental damages because recycling processes and their respective feedback mechanisms vaguely resemble natural systems (Kautsky *et al.* 2000; Banas *et al.* 2008).

Species cultured in intensive systems, usually high-trophic level species, have a higher ecological footprint than those producing low-trophic level species, as omnivorous or herbivorous fish (e.g. catfish, tilapia) (**Table 1**). Carnivore species require high-proteic manufactured feeds, releasing substantial amounts of wastes that are not easily assimilated

Table 1 Amounts (kg per ton of product) of Total Suspended Solids (TSS), Biochemical Oxygen Demand (BOD), Particulate Organic Matter (POM), Nitrogen (N) and Phosphorus (P) discharged from different aquaculture units.

Species	Culture method	TSS	BOD	POM	N	P	Reference
		kg per ton of product					
Finfish	Marine cage farming				61- 132	2.2 - 95	Enell and Ackefors 1991; Islam 2005
Gilthead-seabream (*Sparus aurata*)	Marine cage farming	7038 - 9105	235	843 - 1009	190	28	Jambrina 1995; Barbato *et al.* 1996; Tovar *et al.* 2000b
Octopus (*Octopus vulgaris*)	Marine cage farming				111	37	Mazón *et al.* 2007
Salmonids	Freshwater cage farming	474 - 4015	285 - 990		71	11	Beveridge *et al.* 1991; Kelly *et al.* 1996
Catfish (*Ictalurus punctatus*)	Freshwater systems				9.2	0.57	Schwartz and Boyd 1994
Rainbow trout (*Oncorhynchus mykiss*)	Freshwater systems	640	129 - 551			22	Holby and Hall 1991; Boaventura *et al.* 1997
Shrimps	Semi-intensive earth ponds	715 - 9105	235	257 - 918	29 - 48	2.6 - 4.6	Páez-Osuna *et al.* 1997; Biao and Kaijin 2007; Casillas-Hérnandez *et al.* 2007

by the environment (Karakassis *et al.* 2000; Choo 2001; Páez-Osuna 2001a; Pearson and Black 2001; King and Pushchak 2008). For instance, a study carried out by Folke *et al.* (1998) revealed that Atlantic salmon marine cage farming requires an ecosystem area 40000 to 50000 times higher than the farm area. However, as feed technology improves and higher feed conversion rates (FCR) are attained, the footprint of intensive carnivore production is likely to decrease (Black 2001). An additional factor contributing to the high ecological footprint of carnivorous aquaculture is the use of the so-called "trash fish" (i.e. fish unfit to human consumption) for the production of pelleted diets, which consumes a large quantity of natural resources (Black 2001).

The most environmentally benign production systems are probably those cultivating species from the base of the food web, like seaweeds or filter-feeders (Crawford *et al.* 2003). However, even these systems may have a relevant ecological footprint, depending on the location, farm dimension and stocking densities (Folke *et al.* 1998; Black 2001; World Bank 2006). For instance, large amounts of biodeposits (e.g. bivalves' faeces and pseudo-faeces) may induce changes on benthic processes and benthic communities (Buschmann *et al.* 1996; Kaiser 2001; SECRU 2002; Watson-Capps and Mann 2005), with consequences for the entire ecosystem.

Aquaculture systems combining species from different trophic levels (e.g. fish-shellfish or fish-seaweeds polyculture) or integrated with other activities like agriculture or waste treatment may significantly lower the environmental impacts of aquaculture because nutrients and organic matter are recycled within the system (Buschmann *et al.* 1996; World Bank 2006).

Organic matter enrichment

The immediate effects of particulate organic matter released from aquaculture operations include the stimulation of phytoplankton and bacterial development, which reduces the penetration of light into the water column, subsequently affecting benthic flora (Páez-Osuna 2001a; Ruiz *et al.* 2001; Watson-Capps and Mann 2005; Pérez *et al.* 2008). However, in oligotrophic systems such as the Mediterranean Sea, aquaculture impacts on the water column are minimal, presenting only localized or no effects on most water quality parameters (Maldonado *et al.* 2005). These findings are generally attributed to fast dilution (Pitta *et al.* 2006) and high nutrient recycling rates within the food web (Machias *et al.* 2004). Particulate organic loading also contributes to long term changes in the benthic environment (Gowen and Bradbury 1987; Wu 1995; Karakassis *et al.* 1998; Holmer *et al.* 2005; Klaoudatos *et al.* 2006).

On reaching the bottom, biodeposits may be incorporated into the sediment or re-suspended by bottom currents (Jones *et al.* 2001) that disperse them further away from the discharge point. With the continuous deposition of organic matter, microbial activity is enhanced and sediments become reduced due to an increase in oxygen consumption (Giles *et al.* 2006; Belias *et al.* 2007; Holmer and Frederiksen 2007). When the oxygen demand caused by the input of organic matter exceeds the oxygen mixing rate from overlying waters, sediments become anoxic and anaerobic processes dominate (SECRU 2002; Holmer and Frederiksen 2007). Microbiological processes such as denitrification, nitrate, manganese, iron and sulphate reductions, and methanogenesis prevail (Pearson and Black 2001), whilst aerobic respiration and nitrification processes are inhibited by sulphide (Deb 1998). The outcome of these reactions is the production of toxic gases (e.g. ammonia, methane and hydrogen sulphide) and the development of hypoxia in the water column (SECRU 2002).

Changes in the physical and chemical characteristics of sediments generally have strong adverse impacts on the structure of benthic communities (Naylor *et al.* 2000; Pearson and Black 2001; Kelly and Elberizon 2001; Páez-Osuna 2001a; Nordvarg and Hakanson 2002; Edgar *et al.* 2005; Watson-Capps and Mann 2005; Klaoudatos *et al.* 2006; Rodriguez-Gallego *et al.* 2008). Although initially the diversity and biomass of benthic fauna increases, mostly due to the expansion of opportunistic species (e.g. small annelid and nematode worms) and the immigration of other species, the continuous organic matter input will promote anoxia of the deeper sediment layers leading to the elimination of larger and deeper burrowing long-lived forms and subsequently to a decrease in biodiversity (Kelly and Elberizon 2001; Pearson and Black 2001; Edgar *et al.* 2005; Felsing *et al.* 2005; Klaoudatos *et al.* 2006). The increasing sediment oxygen demand will eventually bring anoxia into the lower levels of the water column, originating the appearance of an azoic zone (Tovar *et al.* 2000a; Ruiz *et al.* 2001; Kelly and Elberizon 2001; Pearson and Black 2001; Read and Fernandes 2003; Edgar *et al.* 2005; Gyllenhamman and Hakanson 2005; Watson-Capps and Mann 2005).

The impacts of aquaculture on benthic primary producers, particularly on seagrass communities, have been widely reported (Ruiz *et al.* 2001; Pérez *et al.* 2008). The combined effects of light attenuation, mainly due to the shade effect of aquaculture structures and high concentrations of suspended solids, with the accumulation of organic wastes on bottom sediments, significantly reduces the density of seagrass meadows, such as *Posidonia oceanica* (Cancemi *et al.* 2003; Pérez *et al.* 2008). Bottom sediment enrichment may also increase epiphytic growth and herbivore pressure, limiting the seagrasses photosynthetic activity (Ruiz *et al.* 2001). Moreover, the decomposition of organic matter increases porewater nutrient availability and sulphide concentrations in the root zone, which negatively affects seagrasses health and survival (Pérez *et al.* 2008).

Changes on the benthic compartment may affect trophic relations and energy transfer along the aquatic food webs (Wu 1995; Deb 1998; Karakassis *et al.* 2000; Tovar *et al.*

2000a; Kelly and Elberizon 2001; Pearson and Black 2001; Read and Fernandes 2003; Felsing *et al.* 2005; Gyllenhamman and Hakanson 2005; King and Pushchak 2008). For instance, studies carried out in marine cage farms revealed that the organic wastes released from aquaculture operations constitute an additional food source for wild fish living in the vicinity of the culture site, making fish to congregate locally (Pearson and Black 2001; Machias *et al.* 2004; Gyllenhamman and Hakanson 2005). The reduction of the fishing pressure and the refuge/protection provided by aquaculture structures (Pearson and Black 2001; Machias *et al.* 2004) may additionally contribute for wild fish assemblages. Although the magnitude of these bottom environmental impacts depends on several factors such as, culture type, stocking densities and cultivated species (Wu 1995; Kempf *et al.* 2002; Kalantzi and Karakassis 2006), in general, the major negative effects are found in the farm area and in its immediate vicinity, decreasing with greater distance from farming operations (Karakassis *et al.* 1998; Pearson and Black 2001; Kaiser 2001; Cromey *et al.* 2002; Felsing *et al.* 2005).

Nutrient enrichment

Inputs of inorganic compounds (e.g. ammonia, nitrates, nitrites and phosphates) through organic matter breakdown, animal excretion and pond fertilization may also have potentially hazardous effects on the surrounding environment (Wu 1995; Buschmann *et al.* 1996; Deb 1998; Tovar *et al.* 2000a, 2000b; Páez-Osuna 2001a; Pearson and Black 2001; Read and Fernandes 2003; Biao and Kaijin 2007; Pérez *et al.* 2008; Rodriguez-Gallego *et al.* 2008). Most of the undesirable ecological consequences related to the excessive nutrient availability from aquaculture discharges (**Table 1**) are related to eutrophication, and include, for example, hypernutrification and the depletion of dissolved oxygen that cause the deterioration of water quality (Tovar *et al.* 2000a; Read and Fernandes 2003). Nutrient loadings also contribute to the pool of plant nutrients in aquatic systems, stimulating the growth of primary producers (Read and Fernandes 2003; Biao and Kaijin 2007) and even changing the structure and composition of these key communities (SECRU 2002).

Should nutrient enrichment coincide with certain physical conditions, and other, poorly understood factors, there may be a growth of toxic phytoplankton species, leading to the formation of Harmful Algal Blooms, HAB (Biao and Kaijin 2007; King and Pushchak 2008). For example, reports of HAB of *Chattonella marina*, presumably, caused by effluent discharges from shrimp farms were documented alongshore the north of the Yellow Sea in 1993 and 1995 (Biao and Kaijin 2007). Toxic phytoplankton blooms may produce different types of toxins (e.g. DSP- diarrheic shellfish poisoning, PSP - paralytic shellfish poisoning, and ASD - amnesiac shellfish disease), that often cause shellfish poisoning and the mortality of benthic fauna and wild/farmed fish, thereby threatening the economic viability of aquaculture activities (Pearson and Black 2001; Read and Fernandes 2003; Gyllenhamman and Hakanson 2005).

Although the potential for eutrophication appears unlikely to marine cage farming due to the dilution effect of seawater (Wu 1995; Pearson and Black 2001), the possibility of localized eutrophication in areas of poor flushing cannot be excluded (Wu 1995; Pearson and Black 2001). In terms of restricted exchange areas, such as coastal lagoons and estuaries, excessive nutrient availability may affect the ecosystem productivity (OAERRE 2001) and in some cases, negatively affect the aquaculture activity itself (Deb 1998; Páez-Osuna 2001b).

Chemical contamination

The overuse and misuse of chemicals in aquaculture operations is also a reason for apprehension due to the pollution and contamination effects that it may have on the aquatic environment. Chemicals used in aquaculture operations may be categorised as: 1) feed additives (e.g. vitamins, pigments, minerals, and hormones), 2) disinfectants (e.g. bleach, malachite green) and pesticides (e.g. molluscicides and piscicides), 3) liming materials, 4) metals (e.g. antifoulants) and 5) veterinary medicines, including antibiotics, anaesthetics, parasiticides, and vaccines (Read and Fernandes 2003) used to control external and internal parasites or microbial infections (Costello *et al.* 2001). Other biological products, such as, organic matter decomposers (e.g. bacteria and enzyme preparations) are also used (Gräslund and Bengtsson 2001).

The application of these chemicals is mainly dependent on the culture system. For instance, while semi-intensive shrimp farms require a minimal use of chemicals, mostly fertilizers and liming materials (Boyd and Massaut 1999; Choo 2001; Gräslund and Bengtsson 2001), as shrimp production is intensified, management becomes more problematic, and the number and diversity of chemical compounds largely increases (Gräslund and Bengtsson 2001). Intensive pond culture also requires a higher diversity of chemicals when compared to cage systems, which mostly use disinfectants, antifoulants and veterinary medicines (Kelly and Elberizon 2001; Read and Fernandes 2003).

The main environmental risks associated with the use of chemical compounds relate to: i) deterioration of water quality, ii) interference on biogeochemical processes, iii) direct toxicity to wild fauna and flora, iv) development of resistance by pathogenic organisms, and v) reduction of the prophylactic efficiency of therapeutants (Costello *et al.* 2001). The improper use of chemical compounds may also affect the safety of the aquaculture products, constituting a threat to human health (Choo 2001, Islam *et al.* 2004).

Since many of the chemicals used in aquaculture were not originally developed for this industry, their effects on the aquatic environment are not fully known. Examples of the specific environmental effects of some of these chemicals may be found in the sections below:

Feed additives

Vitamins are often added to artificial feeds, particularly in marine fish farming, but are not commonly used in the tropics and sub-tropics regions (Wu 1995; Choo 2001). The exception seems to be intensive shrimp farming, where vitamins C and E have been used to reduce malformations of the carapace or disorders of the gills and, to enhance disease resistance (Gräslund and Bengtsson 2001). Even though the environmental effects of vitamins are poorly known (Wu 1995), some vitamins such as biotin and vitamin B_{12} have been shown to stimulate growth of some phytoplankton species (e.g. *Gymnodinium aureoles*, *Heterosigma akashiwo* and *Chrysochromulina polylepis*) and may be implicated in their toxicity (Gowen and Bradbury 1987; Honjo 1993). Impacts on bottom sediments should only be relevant in extreme anoxic conditions since the half-lives of vitamins are very short in aerobic conditions (Samuelsen 1989).

Disinfectants and pesticides

Chemical disinfectants used for eliminating pathogens, control phytoplankton or to treat bottom sediments (GESAMP 1997; Gräslund and Bengtsson 2001), can have severe negative effects on the environment. For example, during the oxidation of organic compounds, the hypochlorite ions of bleaching powder, used as a disinfectant in shrimp farms, are quickly transformed into trihalomethanes (THMs) and other by-products with a potentially toxic and carcinogenic effect on aquatic organisms (Gräslund and Bengtsson 2001; Biao and Kaijin 2007). Pesticides usually used to kill organisms such as algae, fungi, snails and fish (respectively algicides, fungicides, molluscicides and piscicides) may also cause toxicity in non-target species and affect biodiversity (Gräslund and Bengtsson 2001; World Bank 2006).

Liming materials

Even though the environmental impacts of liming materials, commonly used to neutralize acidic conditions and as disinfectants during the preparation of shrimp ponds, are considered to be minimal (Boyd and Massaut 1999), an excessive use of these substances may stimulate the growth and abundance of some phytoplanktonic species (e.g. cyanophytes) and restrain the propagation of others (e.g. diatoms), and thereby affect the entire aquatic food chain (Biao and Kaijin 2007).

Metals

Metals present in artificial feeds (e.g. copper, zinc, cobalt, cadmium, lead and mercury), antifouling agents (e.g. organotins, copper-based antifoulants and sometimes TBT) and in therapeutants represent a risk, both to the environment and to the aquatic living organisms (Mendiguchia *et al.* 2006). For example, the use of heavy metal-based antifoulants in farm structures (e.g. cages, pens or nets) is particularly noxious to benthic communities because their residues are highly persistent in bottom sediments. Still, the toxicity and bioavailability of these contaminants strongly depends on their degradation, accumulation, and dispersion rates (Gräslund and Bengtsson 2001), and on the biogeochemical characteristics of farm sediments. While in organically enriched sediments, with a high biological oxygen demand and negative redox potential, metals like copper and zinc are less likely to become biologically available (SECRU 2002), in highly disturbed sediments the remobilisation of these metals is facilitated (SECRU 2002). Although, further research is required on the environmental effects of these compounds, particularly for long-term and multiple-source impacts (Mendiguchia *et al.* 2006), the environmental risk of antifoulants in marine environments is considered to be low if used according to the regulatory guidelines (Carbonell *et al.* 1998).

Veterinary medicines

The use of medicines, and in particular of antibiotics, in prophylactic treatments or to cure specific infections (GESAMP 1997), poses a diversity of risks not only to the environment, but also to wild/cultivated species and to human health (Biao and Kajin 2007). The main ecological concerns over the use of these compounds relate to: i) the accumulation of antibiotic residues in farmed species and in nontarget species, ii) the development of resistant pathogens in farmed species and an increased susceptibility to infection iii) the impacts on bottom sediments, particularly, on the structure and activity of microbial communities, and on the development of antibiotic-resistance by some bacterial strains and finally over iv) its effects on human health due to the development of drug resistance in human pathogens and transfer of resistance from target animal pathogens to human pathogens (GESAMP 1997; Choo 2001; Gräslund and Bengtsson 2001; SECRU 2002; Read and Fernandes 2003; Islam *et al.* 2004; Biao and Kaijin 2007). Antimicrobial compounds (e.g. amoxicillin, oxytetracycline and terramycin) often administered as feed additives to control diseases outbreaks (Wu 1995), readily associate with particulate material and accumulate on bottom sediments. Although antibiotics are not likely to accumulate in oxic environments due to their short half-lives (Wu 1995), residues are often found in organically enriched farm sediments usually on a restricted area. The environmental risk of antibiotics is thought to further decline in the future due to the development of vaccines (SECRU 2002; Gräslund and Bengtsson 2001).

The parasiticides used to treat sea lice, the most common parasite in Atlantic salmon (*Salmo salar*) farms, are commonly administrated in bath treatments (e.g. cypermethrin) or as feed additives (e.g. teflubenzuron or diflubenzuron). Independently from the type of treatment, sediments usually act as a sink for these medicines, which may lead to potential negative impacts on benthic organisms, such as invertebrate crustaceans (SECRU 2002), with further repercussions to the local food webs. Even though, sea lice medicines are generally considered to have low to moderate environmental risk (SECRU 2002), the dispersion, fate and cumulative effects from multiple treatments remain unknown and require further investigation.

Spread of parasites and diseases

The dissemination of parasites and diseases from farmed species to wild stocks, principally through water, escapees or diseased seed (Nash 2005), constitutes an important constraint to the sustainability of the aquaculture industry, not only from the ecological point of view but also from the economical perspective because it affects the investors confidence, the commercialization of aquatic products and the profit margins (Choo 2001; Kaiser 2001; Pearson and Black 2001; Subasinghe and Phillips 2002). Even though this was usually considered a localized problem in the past, with the expansion and globalization of the aquaculture industry, pathogens and parasites restricted to one region are now rapidly spreading over the world. For instance, the introduction of postlarvae and broodstock from areas affected by the White Spot Syndrome Virus and Taura Syndrome Virus caused mass mortalities in a wide range of shrimp species in Asia and Latin America countries (Choo 2001). Wild salmon and sea trout cultivated in marine cage farms are also thought to be at risk due to the spread of infective larval sea lice from salmon farms (SECRU 2002). The level of risk for disease or parasites transfer is usually difficult to quantify not only because hosts may carry pathogenic organisms without showing any symptoms but also because a wide range of parasitic worms, pathogenic bacteria (*Salmonella*, *Eschericia*, *Vibrio*, and others) and viruses are already present in natural waters, being common to both wild and cultured species. Many of these pathogenic organisms may also be introduced by other human activities besides aquaculture, like livestock, human waste and aquatic products transportation (SECRU 2002). Besides the environmental risks, the propagation of parasites and diseases also constitutes a risk to human health although it can be minimised or even completely eliminated, through the implementation of strict sanitary and food safety regulations (e.g. HACCP) to commercial aquaculture (World Bank 2006).

Habitat destruction and modification

The loss or degradation of habitats, in particular of coastal habitats such as mangrove systems and other wetlands (seagrass meadows, saltmarshes, coastal lagoons, estuaries) is one of major adverse impacts of aquaculture (Wu 1995; Deb 1998; Naylor *et al.* 2000; Black 2001; Páez-Osuna 2001b; Ruiz *et al.* 2001; Pérez *et al.* 2008). Studies carried out in marine cage farms on the Mediterranean coastline reported the destruction/degradation of *Posidonia oceanica* meadows, as a consequence of the high organic and nutrient loading from fish farming activities. Conversion of mangrove forests into shrimp farms (Deb 1998; Choo 2001; Páez-Osuna 2001b) has mainly caused the loss of feeding, nursery, shelter and spawning grounds for a wide variety of marine and terrestrial animals (Ruiz *et al.* 2001; Pérez *et al.* 2008), and the loss of natural protection against floods, storms and hurricanes (Deb 1998; Choo 2001; Páez-Osuna 2001b). Coastal lowlands, such as mangroves and saltmarshes, play a significant role in shore protection by deflecting and reducing the energy of water masses, and by being important routes of water discharge (Deb 1998; Choo 2001; Páez-Osuna 2001b). The construction of channels and dikes for inland aquaculture has also irreversibly altered the hydrological conditions (e.g. water discharge rates and sediment loads) of many coastal systems and the shore geomorphology (Deb 1998; Primavera 2006). Habitat modification caused by bivalve farming during harvesting or the preparation of cultivation grounds (usually by addition of gravel, sand and protecting nets), may additionally change the sedimentary processes and the biogeochemistry of farming sites. This disruption of bottom communities (e.g. benthic fauna or seagrasses) may have negative consequences for the higher trophic levels, for example, by affecting the feeding behaviour of wadding birds and of marine mammals (Kaiser 2001; Watson-Capps and Mann 2005). Other potentially adverse impacts on marine mammals include for example, the death or injury through entanglement in gear, habitat displacement, and disruption of migration pathways, especially for large cetaceans (Watson-Capps and Mann 2005).

Introduction of new species and new genetic varieties

The deliberate or inadvertent introduction of new species or genetic varieties should be a key aspect when assessing the environmental impacts of aquaculture. The main impacts of introductions fall into two categories: i) ecological, including biological and genetic effects, and ii) socio-economic (cf. – Socio-economic impacts), that can be interrelated. Despite providing significant social and economic benefits (e.g. supply of animal protein and disease control), the use of exotic species may also seriously affect ecosystem functioning. The main negative ecological impacts resulting from the introduction of new species and genetic varieties include: i) loss of biodiversity, due to direct biological interactions such as predation and competition; ii) loss of genetic diversity in wild populations, mainly due to breeding of alien organisms with local strains or species; iii) transmission or spread of diseases to which indigenous species are more vulnerable; iv) and habitat modification (Black 2001). A case reporting the hazards of species introductions is that of the Nile perch in Lake Victoria, which became the dominant species of the lake's fauna. Even though the introduction of Nile perch generally provided relevant economic benefits for some entrepreneurs (may be not so for the population depending directly on lake biodiversity), the arrival of the invasive water hyacinth blocked waterways and the access to riparian villages and fishing grounds, causing major economic losses (World Bank 2006). Whirling disease, a virus infection that affects rainbow trout, was introduced in North America through the importation of European brown trout that was immune to the virus (World Bank 2006). Other vectors for species introduction include for example the ships ballast water or the faeces and digestive tracks of commercialised bivalves, which may transport the resting cysts of toxic phytoplanktonic species and of seaweeds species (Kaiser 2001).

The release of cultivated organisms to the natural environment, either by accident or natural catastrophes, not only poses a risk for the structure of wild populations but also to the regional economies (Youngson *et al.* 2001; Read and Fernandes 2003). Most of the negative ecological impacts resulting from the interaction between cultivated and wild species result from the genetic interaction of wild organisms with their aquaculture conspecifics. The genetic impacts of escapes on wild populations are a complex subject, but the fundamental problem rests on the genetic differences between wild and farmed species (Kapuscinski and Brister 2001). As part of the evolutionary strategy, wild species possess higher genetic diversity both within and between populations (SECRU 2002). Escapees that survive and spread to spawning grounds can interbreed with wild organisms (Kapuscinski and Brister 2001; SECRU 2002; Naylor *et al.* 2005), posing two types of hazards: firstly, outbreeding depression (i.e. loss of fitness in the offspring) that mainly reduce the survival fitness and efficiency of wild organisms and secondly, the homogenization of genetic differences which increases the vulnerability of individuals to environmental changes, and compromise the sustainability of wild populations (Kapuscinski and Brister 2001). Even though domesticated species, such as the farmed Atlantic salmon, are generally less fit for survival and breeding (mainly due to a lower ability to participate in breeding and to a poorer quality and quantity of gametes), when a substantial proportion of escapees secure matings with wild fish, outbreeding depression may cause the decline of wild populations (Kapuscinski and Brister 2001; SECRU 2002; Naylor *et al.* 2005) due to the loss of environmental adaptive genotypes which determine the species success. These risks are greater for small populations that are already threatened and, whenever genetic modified organisms (GMOs) are used. The growing development of GMOs to increase the quantity and quality of aquatic products may seriously jeopardize the genetic integrity of wild stocks and the ecosystems functioning (Spreij 2004).

Harvest of wild stocks as feed or seed/broodstock to aquaculture operations

The depletion of wild resources and biodiversity to produce animal feeds or to supply seed/broodstock to aquaculture can cause significant damages to aquatic ecosystems (Deb 1998; Choo 2001; Kaiser 2001; Páez-Osuna 2001b). Fish species of low commercial value (e.g. Japanese anchovy and chub mackerel) are mainly targeted to be processed into feeds for carnivorous fish, or as supplements for other species, like for example, shrimp, tilapia and milkfish (Black 2001). The use of this so-called "trash fish" puts even more pressure on the already overexploited wild fish stocks. The broad collection of wild seed (e.g. of eel, grouper, yellowtail, and tuna aquaculture) and broodstock for aquaculture purposes also contributes to the decline of natural populations. The collection of wild shrimp and shellfish seed is particularly environmentally-damaging because not only it threatens the wild stocks of target species (e.g. by affecting species recruitment) but also affects the stocks of other living resources (other shrimp species, macrozooplankton, finfish and shellfish juveniles and larvae) that are indiscriminately killed. This reduces the food availability for other organisms such as aquatic birds, reptiles and mammals linked through the trophic web, and may subsequently increase their mortality at the same time that it reduces their breeding success (Choo 2001). Harvest of wild species may also cause genetic degradation of native populations and the destruction and modification of natural habitats, causing further disturbances on the aquatic food web (Deb 1998; Primavera 1998; Islam *et al.* 2004; World Bank 2006). This activity is particularly dangerous for heavily fished species and for species with low reproductive capacities (World Bank 2006), but probably as long as the production of broodstock in captivity remains costly, the purchase of wild spawners will continue, causing environmental damages in ecosystems around the world (Nash 2005; World Bank 2006).

Socio-economic impacts

Despite the negative impacts that it might have on the environment, aquaculture may also provide important socio-economic benefits. For instance, aquaculture is foreseen to become the major source of animal protein (Naylor *et al.* 2000; Sugiura *et al.* 2006; World Bank 2006).

The commercialisation of aquaculture products is also an important source of incomes (Biao *et al.* 2004; Primavera 2006) and largely contributes to the countries economic development (**Table 2**). For instance, since 1970, the

Table 2 Top ten aquaculture producer countries in 2006 and their respective aquaculture revenues.

Country	10E+6 tons	%	10E+9$	%
China	34.4	67.7	38.4	48.8
India	3.12	6.05	3.43	4.36
Vietnam	1.66	3.20	3.32	4.21
Thailand	1.39	2.68	2.22	2.81
Indonesia	1.29	2.50	2.25	2.86
Bangladesh	0.892	1.73	1.36	1.73
Chile	0.802	1.55	4.43	5.62
Japan	0.734	1.42	3.10	3.93
Norway	0.708	1.37	2.72	3.45
Philippines	0.623	1.21	0.981	1.25

Source: FAO Fishstat, ftp://ftp.fao.org/fi/stat/summary/default.htm aquaculture sector has increased at an average annual rate of 10.4% in developing countries (World Bank 2006) while in developed countries it grew on average 4% per year. The trade of aquaculture products is particularly important for developing countries and to low profit-food deficit countries (e.g. Bangladesh, Indonesia, Vietnam) because it considerably increases their revenues. Besides contributing to the development of national economies, aquaculture has also allowed the stabilization and strengthen of populations from remote regions or marginalised social groups (mainly in Asia and Africa), by increasing rural development and reducing poverty and hunger (Black 2001; World Bank 2006). Aquaculture production may also contribute to the reduction of fish prices, at the same time that it increases the access to fish products by poor households. An example of pro-poor aquaculture has been implemented in Asia, where it was developed under two models: one in which commercial opportunities have been opened for enterprises, and the other consisting in using public support to generate enough critical mass for smallholders. The enterprise model not only generated growth and employment in poor regions where alternative employment is scarce as also increased the stability of local communities (Black 2001). For example, this sector employs more than 12 million people in China, Indonesia, and Bangladesh alone (FAO 2007). Many of these people are rural dwellers and some, such as wild shrimp seed collectors, are among the poorest and most marginalized (Deb 1998). On the other hand, public support extended profit opportunities to smallholders in China, Vietnam, and Bangladesh mainly by combining a supportive policy (e.g. microcredit) with the dissemination of knowledge on proven technologies (e.g. polyculture). This strategy has also proven to be an effective mean of targeting the landless poor (e.g. rice farmers) mainly by improving their livelihoods (World Bank 2006; FAO 2007). A surplus in households may turn into a social benefit because it improves the nutritional state of poor populations and provides an opportunity to invest in education.

Other social benefits provided by aquaculture include for example, women empowerment. In Bangladesh and Vietnam, more than 50 percent of workers in seed collection, fish markets and processing plants are women, and although salaries of these workers are still quite low ($1–$3 per day), they are significantly higher than salaries earned in agricultural activities (World Bank 2006). In the Mekong delta aquaculture has also contributed to a decrease in urban migration by young women and prevented women from being forced into prostitution, reducing the risks of spreading sexual diseases (FAO 2007).

Although responsible aquaculture can provide significant economic benefits, uncontrolled and irresponsible aquaculture operations can cause a wide range of negative socio-economical impacts, particularly when the ecosystem functioning is radically altered and the resources that support other human activities are affected. For instance, pandemics outbreaks have devastated shrimp farming in many producing countries (Deb 1998). Other adverse effects result from the introduction of new species. For example, the introduction of the golden apple snail into Asian countries, mainly with the purpose of developing an export industry, resulted in high damages to rice farmers, since this snail consumed large quantities of paddy-rice (World Bank 2006). The import of crayfish and oysters from North America also destroyed the European crayfish and oyster industries mainly due to the introduction of pathogens hosted on the imported organisms (World Bank 2006). Conversely, in Chile, the introduction of the Pacific and Atlantic salmon in the 1970s turned into an economic benefit, since the country is now the world's leader in salmon production. Tilapia, a group of species originating in Africa, is also cultured worldwide and provides income and high-quality protein to many rural areas, especially in developing countries.

The inexistence of an ecosystemic approach for the management of the aquaculture industry, often lead to conflicts over common resources such as land and water. For instance, the conversion of mangrove forests into commercial shrimp farms led to the loss of forest products and fisheries (Primavera 2006), affecting principally the poor populations. The conversion of residential, agriculture (rice and pastures) and common lands in Asian countries (Thailand, Bangladesh and Philippines) has also raised serious conflicts between agriculture and shrimp farmers (Deb 1998; Choo 2001; Primavera 2006). Conflicts over water use are particularly frequent because aquaculture effluents may contaminate the water used by other aquaculture units downstream (Deb 1998; Gräslund and Bengtsson 2001). On the other hand, aquaculture itself may be subjected to water contamination due to urban waste and agricultural pollution. Saltwater intrusion caused by aquaculture activities, either from the percolation of water discharged from brackish/marine cultivation ponds or from active pumping of groundwater, has also several negative socio-economic repercussions, including, for example, the loss of agricultural crops, land subsidence, decrease in fish production or the occurrence of freshwater crisis that cause gastrointestinal diseases (Deb 1998; Choo 2001; Páez-Osuna 2001b). Other negative impacts resulting from the massive introduction of aquaculture structures (ponds, cages, or rafts) include the blocked access to coastal resources, navigational hazards, privatisation of public lands and waterways, and fisheries decline (Primavera 2006). Conflicts over common resources generally lead to serious social problems and even in some cases, to human rights abuse (World Bank 2006).

An ecosystem approach to the management of the aquaculture industry is therefore crucial for its sustainability. Letting aquaculture development proceed irresponsibly or taking only partial approaches to its management incurs a risk that the negative impacts may counteract any benefits from aquaculture or that it will not produce the expected benefits.

THE CONCEPT OF CARRYING CAPACITY IN AQUACULTURE

In a broad sense, carrying capacity (CC) may be defined as the capacity of a natural or man-made system to hold a certain pressure without driving its structure and function above Limits of Acceptable Change (LAC) (Duarte 2003). Whilst this general and simple definition may be appropriate as a first approach, it is important to apply the concept to some specific areas and to develop more precise definitions. CC may be defined within the scope of any activity implying some sort of environmental, social or economical impact. The LAC concept has long been used in tourism management (e.g. Wearing and Neil 1999). The goal is to be able to use natural and man-made ecosystems without compromising their capacity to continue providing the goods and services that people need. The definition of LACs is not straightforward, because though some of these limits may be defined on a quantitative way, others are rather subjective and depend on people's perception about the environment. For example, water quality parameters may be used to establish quantitative limits on aquaculture outflows to prevent ecosystem degradation. However, it may be more arguable to establish limits in relation to scenic or habitat quality (GESAMP 2001).

The concept of CC is a central theme in aquaculture and it may be related to the amount of natural resources available for aquaculture operations, such as food and space, the services provided by natural ecosystems, such as organic matter mineralization and nutrient cycling, or the economic yield of aquaculture and its economic and social effects. When CC is exceed, negative-feedbacks affect aquaculture operations and may result in yield losses.

Policy makers must take management decisions that may affect the sustainability of natural resources. Having at hand the relevant CC indicators, is the way to prevent them from taking decisions that will jeopardize options for future usages. Whenever possible, these indicators should be quantitative, such as the area that may be allocated for aquaculture, the standing stock of fish that may be kept in a fish culture area, etc.

Given the multiple exploitation possibilities of aquatic ecosystems and their synergic effects, it is clear that CC must be accessed for different activities taking into account their interactions. For example, if a coastal zone is used for sewage dispersal, it's CC for aquaculture may be limited, because not all areas will have the necessary water quality for aquaculture and also because the impact of the sewage outfall may limit ecosystems resilience to assimilate organic loads from aquaculture leases. These complex set of interactions between different uses and the ambiguities of resource ownership leads to the idea of including aquaculture within the framework of Integrated Coastal Zone Management (GESAMP 2001). Concerning inland aquaculture, similar integrated approaches are needed integrating other activities such as agriculture, tourism, nature conservation, etc.

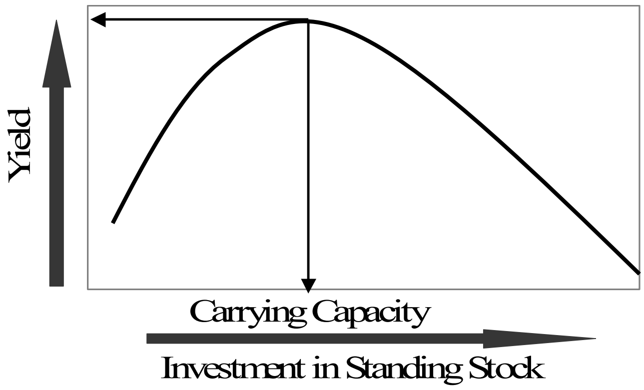

Fig. 1 The parabolic relationship between stock and yield in bivalve culture. Carrying capacity increases with stock up to a point above which individual growth is severely compromised due to food limitation.

CC categories and definitions

The CC definition and classification defined by Inglis *et al.* (2000), adopted by McKindsey *et al.* (2006) and adapted by Gibbs (2009), regarding coastal aquaculture development, was followed in the present work for aquaculture in general:

(i) physical CC – the total area of farms that can be accommodated in the available physical space;
(ii) production CC – the stocking density of cultured organisms at which harvests are maximized;
(iii) ecological CC – the stocking or farm density which causes unacceptable ecological impacts;
(iv) economic CC – the biomass that investors are willing to establish and maintain;
(v) social CC – the level of farm development that causes unacceptable social impacts or that community is willing to allow.

Some of the above categories are defined differently by different authors. For example, according to Jiang and Gibbs (2005), production CC is the theoretical maximum culture that could be supported in an embayment. Alternatively, production CC was defined as the maximum sustainable yield of culture that can be produced within a region, whereas ecological CC was defined as the level of culture that can be supported without leading to significant changes to ecological processes, species, populations or communities in the growing environment (Gibbs 2007). Therefore, in defining production CC, most authors choose to express it as a stock measure (e.g. Carver and Mallet 1990; Bacher *et al.* 1998; Smaal and Héral 1998; Inglies *et al.* 2000; Jiang and Gibbs 2005), whereas others define it as an yield measure (e.g. Gibbs 2007). Therefore, it is important to agree on some common measurements for the sake of comparability within and across different aquaculture areas. Since stock and yield are related, although differently in different aquaculture areas, and since stock is easier to regulate, perhaps it is the most straightforward way to quantify CC.

CC and limiting factors

The CC categories above reflect some of the most common limiting factors for aquaculture development. However, it must be emphasised that, in most instances, these categories are interlinked. In the case of physical CC, space may be limiting due to the lack of sheltered areas and to other competing uses such as sewage dispersal, harbour activities, fisheries, tourism, nature conservation and water availability (in the case of inland aquaculture). For example, the Southeast Asia's seas are under several threatens – 11% of coral reefs collapsed, whilst 80% face risks, mangroves – one of the most threatened tropical environments (Valiela *et al*. 2001) – have lost 70% of their cover, seagrass beds' loss ranged from 20 to 60%, urbanization is predicted to increase and there are tens of pollution hot spots (PEMSEA 2003). Whilst urbanization and resulting pollution may limit geographically aquaculture development, aquaculture itself has been one of the reasons for mangrove destruction in most tropical countries (Primavera 2006) – in relation to ecological CC. In the case of Thailand, a ban on mangrove destruction in the early 1990s was followed by a shift from salt water to low-salinity inland shrimp farming, leading to competition for soil resources between rice and shrimp farmers and to soil salinization (GESAMP 2008). Competition between farmers may be a result of overcoming social CC.

Regarding production CC, limiting factors depend mostly on the culture type. In the case of extensive and semi-intensive cultures, stocks may be limited by food availability and water quality. A typical example of extensive systems, fully dependent on natural food (phytoplankton and organic detritus) is the cultivation of bivalve suspension-feeders. Both the quantity and the quality of these food items are important for bivalve growth (Bayne 1992; Hawkins *et al.* 1998). Production CC for bivalve cultivation depends on the renewal rate of available food. Suspension feeders have a remarkable capacity to filter the water column such that they are food limited at higher culture density. Therefore, water residence times and phytoplankton doubling times may limit CC (Dame and Prins 1998).

The relationship between bivalve production and bivalve standing stock is parabolic (**Fig. 1**), as demonstrated by the theoretical model described in Bacher *et al.* (1998) and the results of Ferreira *et al.* (1998) and Duarte *et al.* (2003). There is an initial increase in production, but as available space becomes filled up with stock, individual bivalve growth rate is depressed and mortality increases due to several factors associated with overcrowding. The overall result of these effects is a strong reduction of harvest yields above a certain stock threshold.

In semi-intensive and intensive systems, production CC may be limited by water quality, namely, by dissolved oxygen (DO) in some fish farms (Shin and Wu 2003). On the other hand, release of faeces, uneaten food and excreta may increase biochemical oxygen demand (BOD) and nutrient concentrations that may overtake limits defined for ecological CC. According to Sarà (2007), available literature data on the effects of aquaculture leases on water quality present a convincing evidence for increases in ammonium, nitrite and nitrate and, to a lesser degree, dissolved phosphorus, in comparison to non-aquaculture sites. These "aquaculture effects" are most noticeable in sheltered water bodies with high residence times.

Another important limitation for bivalve production in coastal areas is Harmful Algal Blooms (HAB) that may cause bivalve contamination and mortality by harmful toxins (Hágaret *et al.* 2007) (cf. – Aquaculture environmental impacts - Nutrient enrichment). In most areas of the world bivalves are monitored for the occurrence of several toxins to prevent their commercialization.

In a recent paper, Gibbs (2009) discusses the role of social barriers to the establishment of aquaculture activities in suitable areas. According to this author, local stakeholders tend to be more environmentally conscious and demanding strong evidence about the environmental and economic sustainability of aquaculture development. This attitude is related to their perception that aquaculture benefits are diffused among the community and state, while costs are internalized locally, especially in coastal regions where recreational and amenity values are high.

Methods for determining CC

Physical CC may be analysed and estimated from physical, chemical and biological data, with the help of a Geographical Information System (GIS). These data may include geographic descriptors, sediment and vegetation types, depth, meteorology, hydrography, water quality, land use, etc. The interception of layers with this data types helps selecting areas that may potentially be used for different aquaculture types. For example, sensitive habitats may be excluded, as well as contaminated or other areas, where land use, management plans or political boundaries are not compatible with aquaculture development.

GIS may also be used to help assessing economic and social CC, if it contains information on relevant descriptors. For example, areas that are used for some other economic activities or where local stakeholders have a strong opposition to aquaculture developments may be excluded, reducing social conflicts.

Production and ecological CC may be approached at several spatial scales, such as the scale of the cultivation unit (farms, rafts, etc.) and the ecosystem scale. The former is directly relevant to farmers, whereas the latter is relevant for ecosystem management (Duarte 2003). In accordance to this, aquaculture leases produce “near-field” and “far-field” effects – the latter result from the cumulative effects of the former at the ecosystem scale. This scale may be easy to define in the case of estuaries, bays and fjords but more difficult for open coastal areas (Andersen *et al.* 2006).

Following the last authors, if the scale of the farm is large in comparison with the ecosystem scale, more important impacts are expected than in the opposite situation. Therefore, the definition of ecosystem boundaries is critical in evaluating aquaculture impacts. One possible approach is the analysis of impacts from the farmer scale to progressively larger scales, until they are no longer relevant. Such an approach is hardly achieved without a mathematical model (see below).

One important point here is that whatever method is used to estimate aquaculture impacts or CC, it should allow resolving scales smaller than the ecosystem scale. The rationale beyond this statement is discussed in Duarte *et al.* (2005) in relation to bivalve culture, but concepts may be extended to other culture types. The general idea is that if CC is evaluated at a scale larger than the farm scale, “farm effects” are diluted over a relatively larger area. For example, in the case of bivalve suspension-feeders, food limitation may be underestimated, since local food depletion is ignored, with the result of overestimating production CC. Ecological CC may also be over estimated, since excreta from cultivated organisms are “diluted” over a larger area.

Ideally, the smaller scale resolved should be small enough for water residence time to be lower than the time needed for significant changes to occur in any chemical or biological factors related with CC. When this condition holds, water properties do not change much across the scale considered. Current speed measurements or a hydrodynamic model may be used to determine the mentioned smaller scale.

In **Fig. 2**, a practical example of the above concepts is presented (for details see Duarte *et al.* 2003, 2005) regarding Sungo bay (People’s Republic of China). This bay is extensively used for kelp and bivalve culture. If a whole system bivalve production CC is estimated from water residence time, phytoplankton doubling time and bivalve clearance time (the time it takes for the bivalves to filter the water in the bay), as described by Dame and Prins (1998), the obtained result suggests that bivalve density may be doubled within the ecosystem (from c.a. 44000 to c.a. 88000 tonnes). In fact, Nunes *et al.* (2003), using a zero dimensional bay ecosystem model obtained even larger production CC estimates. On the other hand, Duarte *et al.* (2003), using a two dimensional hydrodynamic-biogeochemical model, with a finite-difference grid of 500 m resolution (**Fig. 2**) – inline with considerations above on the need to resolve scales smaller than the ecosystem scale - obtained much lower CC estimates. Given average current velocities in Sungo Bay, water residence time within the 500 X 500 m grid cells depicted in **Fig. 2**, is smaller than the time bivalves need to filtrate the water within the cells, considering their large densities within cultivated areas (Duarte *et al.* 2005).

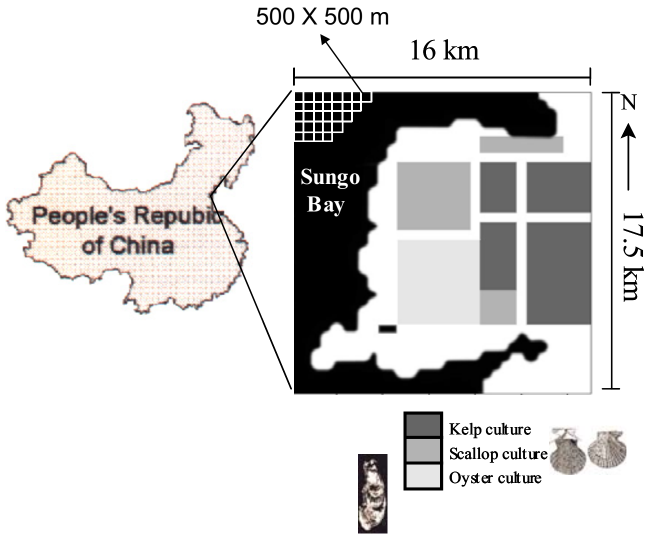

Fig. 2 Areas cultivated in Sungo Bay since 1999 with kelps (*Laminaria japonica*), oysters (*Crassostrea gigas*) and scallops (*Chlamys farreri*), including part of a two dimensional model grid (upper left corner), for which the spatial step is 500 m. Adapted from Duarte *et al.* (2003).

Considering the complex feedbacks between cultivated species and environmental variables, the cumulative effects of many aquaculture activities and the various dimensions of CC, an ecosystem model is necessary for a description of the problem. However, any model is just a pale description of the real system with many limitations, as discussed by Gibbs (2009). Whenever there are no available data and models for a reliable estimate of CC, an adaptive approach should be used by being conservative, according to the precautionary principle, monitoring relevant variables and processes over time and being able to make any adjustments to avoid permanent damage to natural and man-made systems. In fact, this adaptive approach should be followed even in the presence of detailed data and validated models, due to the limitations mentioned above (Gibbs 2009). The confrontation of model results and observations allows model improvements over time, as more knowledge is accumulated about the ecosystem under study. Furthermore, uncertainties associated with model parameters and results may help defining sampling strategies and experiments to fill the gaps.

A model capable of predicting production and ecological CC should include a transport and a biogeochemical sub-model. Ideally, it should also include a thermodynamic sub-model, for water temperature calculations, and biological sub-models for relevant species or species groups. The transport sub-model should be able to predict current speeds and water mixing (or simply to read and return current speed time series measured or obtained with another model) and calculate the transport of dissolved substances and particles. It may be forced by wind, river flows, tidal height variability at sea boundaries, etc. The biogeochemical sub-

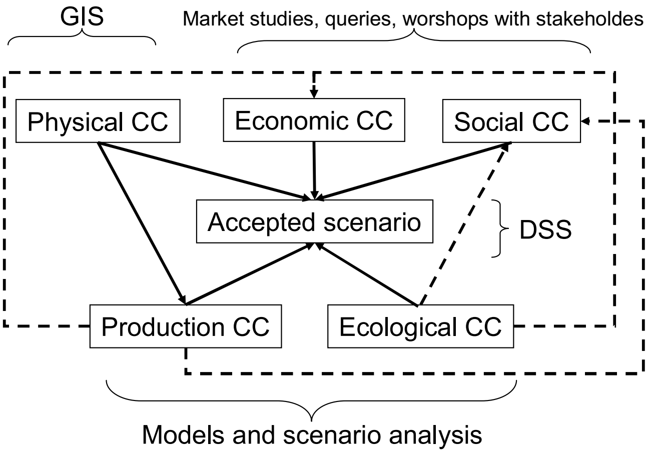

Fig. 3 Diagram showing the interactions and feedbacks among different carrying capacity categories towards an accepted aquaculture scenario, and the tools used for physical, production and ecologic CC and for the scenario selection. Continuous lines show direct influences of CC over the accepted scenario or other CC category. Physical CC limits production CC. Dashed lines showing feedbacks from production and ecological CC to economic and social CC.

model should reproduce biogeochemical cycles of elements that are most likely to become limiting, such as phosphorus and nitrogen, that may limit primary production of phytoplankton and cultivated plants, oxygen, that may limit fish survival, etc. This sub-model should include a pelagic and a benthic compartment, especially when water-sediment interactions are more important, as in shallow water ecosystems. The biological sub-models should simulate growth, production and biological interactions of most relevant species or species groups. It should also simulate nutrient production/consumption and link these with the biogeochemical sub-model. The spatial resolution of the model should follow considerations above. For some examples see Duarte *et al.* (2003, 2007), Ferreira *et al.* (2007), Grant *et al.* (2007), and Shin and Wu (2003). For a review of recent CC models see McKindsey *et al.* (2006).

The above sub-models should be forced with time series obtained at their boundaries (e.g. river or sea boundaries) for the simulated water column variables. It is also important to have time series of meteorological data on: solar radiation, air temperature, wind speed, and relative humidity. For very large areas, it may be necessary to nest more detailed models within the grid of larger scale models, with the latter providing boundary conditions for the former.

Having a model to estimate production and ecological CC it is then necessary to simulate several aquaculture scenarios regarding density of organisms, their geographical distribution and different rearing techniques, for example. The analysis of obtained results concerning predicted production and water quality variables may then be used to evaluate the different scenarios. Typically, an increase in production leads to changes in water quality variables and deciding whether these are acceptable or not, depends on the availability of some criteria. For example, Duarte *et al.* (2007) simulated water and sediment quality as a function of bivalve density in Ria Formosa (Portugal) and compared scenarios on the basis of bivalve production and water quality using the IFREMER water and sediment classification scheme (e.g. Austoni *et al.* 2004).

Ideally, a Decision Support System (DSS) should be used, integrating also economic and social descriptors (for an example see Pereira *et al.* 2007). It is important to involve local stakeholders in the decision process. At this point, economic and social CC may be revised by stakeholders, since obtained results may change their initial perspectives (**Fig. 3**). It may also be necessary to try other scenarios and iteratively reach a good solution.

AQUACULTURE AND ENVIRONMENTAL MANAGEMENT TOWARDS SUSTAINABLE DEVELOPMENT

In the next paragraphs, some possible mitigation measures and methodological approaches are suggested to reduce and anticipate, respectively, aquaculture impacts. Management aspects that may help reducing the direct ecological impacts of aquaculture leases are discussed in sub-chapter about Mitigation measures. The advantages of Ecological Aquaculture are discussed in sub-chapter Ecoaquaculture. Remaining sub-chapters present methodological approaches do help stakeholders and decision makers defining potential problems of aquaculture developments and deciding on alternative scenarios.

Mitigation measures

The sustainable development of the aquaculture industry depends largely on the preservation of natural resources and on ecosystem CC (Read *et al.* 2001). The adoption of an ecosystem approach to aquaculture (EEA) is probably the way to overcome the problems related to its increasing growth and intensification, in particular those associated with the use and allocation of common resources. The implementation of an EAA requires a partnership among aquaculture organizations (e.g. producers associations), governmental agencies (e.g. fisheries administration, rural, urban and industrial development organizations) and the public sector (e.g. NGO's), for the development of appropriate regulatory frameworks and efficient enforcement mechanisms. As an alternative to legal frameworks, the aquaculture industry has developed self-regulation instruments, such as Codes of Conduct (e.g. the FAO Code of Conduct for Aquaculture Practices and the International Aquatic Animal Health Code) and Codes of Practice (cf. – Legislation), to ensure the sustainable development of the activity. Compliance to the norms and principles defined in these codes may also contribute for the minimisation of the negative impacts of aquaculture. At the farm and at the ecosystem levels, an efficient use of Environmental Impact Assessment (EIA) or other decision-making tools (e.g. Decision Support Systems during the planning phase of aquaculture operations together with the implementation of mitigation measures (e.g. environmental monitoring) for activities that already exist, may also contribute to a more environmentally-friendly activity. Some of the important decisions that can be made are mainly related to site selection, species selection (exotic *versus* native), definition of stocking densities and proper farming systems or technologies and, on the socio-economic relevance of aquaculture projects (Read and Fernandes 2003). During the operational phase of aquaculture units, specific proactive measures may also be adopted to safeguard the ecosystems integrity. Some of these mitigation measures are presented in the following paragraphs.

Interference in biogeochemical processes

Given that the impacts on bottom sediments are the most obvious form of pollution resulting from aquaculture activities, the reduction of the amount of wastes and effluents released into the environment is crucial for avoiding that the ecological CC is exceeded (Giles *et al.* 2006). The effects of organic and inorganic waste discharges can be significantly reduced by careful site selection. The specific hydrographic conditions (hydrodynamics, water residence time, and tidal regime), topography, geography and the ecological CC of the receiving body (Buschmann *et al.* 1996; Pearson and Black 2001; Choo 2001; Gräslund and Bengtsson 2001; Primavera 2006), strongly influence the behaviour of all type of wastes released into the water column. For instance, the impacts of wastes discharges from marine cage farming may be minimised by avoiding regions of restricted water exchange, such as enclosed bays or fjords (Pearson and Black 2001). Site rotation allows the seabed to return to normal conditions. Site selection is also crucial for managing the environmental impacts of shrimp farming since aquaculture units are usually established in mangrove areas and tidal wetlands, which in addition to their high ecological value are also characterised by acidic soils and high organic loadings, that may contribute to the deterioration of water quality and to disease outbreaks (Kongkeo 1997; Boyd and Clay 1998).

Organic sediment enrichment can also cause severe environmental impacts if the scale of the farm operation is not suitable for the aquaculture site, i.e., if organic and nutrient loadings are above the ecological CC of the water body. Hence, the limitation of stocking densities may contribute to a significant reduction in the amount of wastes released into the environment, particularly in sensitive habitats, such as mangrove systems and salt marshes (Buschmann *et al.* 1996; Kautsky *et al.* 2000; Gräslund and Bengtsson 2001; Páez-Osuna 2001a; Primavera 2006).

Improving of feeding husbandry techniques (e.g. meal timing or methods for feed supply) and of feed formulation may also be an effective strategy for reducing organic loadings and to prevent the hypernutrification of aquatic systems (Buschmann *et al.* 1996; Páez-Osuna *et al.* 1998; Páez-Osuna 2001a; Pearson and Black 2001). In marine cage farms or pens, the installation of feeding devices with hydrosensors that detect the reduction of fish activity or the use of acoustic feed detectors to reduce the loss of feed pellets, may prevent overfeeding and excessive waste production (Pearson and Black 2001). Other mitigation measures for open systems include for example, the use of settling devices for collection of faecal pellets and food wastes under the cages and the use of pumps for the dispersion of solid elements (Gowen and Bradbury 1987; Buschmann *et al.* 1996). Improvement of feed pellet technology, either by increasing the stability of feeds or reducing its sinking rates may also be a way to maximise the amount of feed ingested, and thereby to minimise waste production (Choo 2001; World Bank 2006). The development of appropriate feeds (with optimal protein/energy ratio) for each species and respective developmental stages further reduces the organic and inorganic loadings to the environment. Since energy requirements can generally be satisfied by lipids and carbohydrates, diets with a higher content of these compounds, increase protein retention and improve feed conversion rates (World Bank 2006). Feeds with high FCRs, like the ones currently used by the Atlantic salmon industry (FCR = 1:1.1, i.e., approximately 1 kg aquatic product per kg of feed), not only reduce the amount of nutrients (nitrogen and phosphorus) released into the environment as also minimise the costs with feeds, since protein is mainly used for body tissue construction (Black 2001; Choo 2001; World Bank 2006). The use of formulated artificial feed instead of “trash fish” in shrimp and carnivorous finfish culture is also desirable not only in terms of its nutritional value and supply but also in terms of waste loadings (World Bank 2006). Furthermore, aquacultures activities depending on these resources are particularly vulnerable to collapse since a reduction in fisheries, will most likely increase feed prices and consequently cause a loss of profits (Black 2001; World Bank 2006).

Interference with the life cycles of wild species

Water-related best management practices (BMPs) may also minimise the risks associated with the introduction and dissemination of viruses and other pathogens (Kongkeo 1997). Recirculating Aquaculture Systems (RAS) systems in particular, not only reduce the possibility of pathogen introduction in freshwater systems as may be an alternative method for the production of healthy seed for marine aquaculture systems (Gutierrez-Wing and Malone 2006). The compliance to other BMPs related to environmental control, as for example careful species selection, limitation of stocking densities and use of proper feeds to avoid deterioration of water quality, or to disease prevention and/or control BMPs like the use of effective vaccines or other prophylactic agents (e.g. probiotics), use of approved medicines and development of disease free strains by selective breeding (Dunham *et al.* 2001; Primavera 2006; World Bank 2006), may also mitigate the negative environmental impacts of aquaculture. Diseases spread through trade and transboundary movements can also be managed by veterinary control or strict regulations for the movement of living aquatic organisms (either eggs, seeds, juveniles or adults) and by the use of certified disease-free organisms (Argue *et al.* 2002; SEACASE 2007). Other measures such as the implementation of environmental programmes, e.g. the Hazard Analysis and Critical Control Point (HACCP) method may also minimise the deleterious effects of disease transmission, and ensure the safety of aquatic products. The reduction of disease incidence is a key aspect for the environmental sustainability of aquaculture because not only it reduces the use of chemicals (e.g. antibiotics) and the requirements for land and water, as also improves the efficiency and viability of the farming activity (Hulata 2001; Argue *et al.* 2002).

As intensification progresses and new species are cultured, seed-based aquaculture is likely to expand, and thereby every effort should be made to reduce the dependence on wild seed. Control/regulation of wild seed by-catch through the establishment of suitable sites, periods, catch efforts, and the production of commercial hatchery postlarvae (Páez-Osuna 2001b; World Bank 2006), may minimise the interference of seed/broodstock harvest in the life cycle of wild species and potential adverse effects on the ecosystems food-webs. These measures should be accompanied by alternatives to minimise the social-economic effects of the reduction of wild seed collection in traditional aquaculture systems and in particular to low livelihood farmers.

Impacts of introduction of new species or genetic varieties

Some of the negative environmental impacts associated with the introduction of new species and new genetic varieties, including the loss of ecosystem integrity and genetic diversity, may be avoided or substantially mitigated through the effective implementation of the existing Codes of Practice and guidelines on this issue. Risk assessment and the application of preventive measures to species introductions (World Bank 2006), namely quarantine systems and cooperation between neighbouring countries before introducing non-native species into transboundary aquatic ecosystems, may also contribute to a responsible use of these species for aquaculture purposes (World Bank 2006). These limitations may easily be overcome by the use of RAS because farmed species are physically contained in these systems, eliminating the risk of escapes (Black 2001; Gutierrez-Wing and Malone 2006).

Degradation of genetic diversity

Another option to minimise the potential loss of genetic diversity due to the interaction of farmed and wild species is to ensure that escapees cannot breed. This is done successfully with rainbow trout by sterilising the females through the induction of a chromosomal abnormality called triploidy (SECRU 2002). Additional preventive measures proposed for cage aquaculture include the improvement of cage design, anchoring, net management, regulation of near-farm operations, deployment of fish cages at a safe distance from wild populations and the development of contingency plans in case of escapes, including for example the capture of escapees identified by genetic markers or tags (Pearson and Black 2001; SECRU 2002). Current methods to reduce Atlantic salmon escapes from cage farms also include the reduction of net damage from predators by using acoustic deterrents (SECRU 2002), however these method may negatively affect and even exclude marine species with high sensitivity to underwater acoustic noise, such as whales and dolphins (SECRU 2002). In restocking programmes to rebuild endangered species or depleted stocks, the utilisation of juveniles with minimal genetic divergence from their wild counterparts may minimise the loss of the species genetic pool (World Bank 2006). This can be achieved for example by using a large number of breeders and genetic markers (World Bank 2006).

Modification and/or destruction of habitats

The problem of the destruction and/or modification of ecosystem structure, function and services by aquaculture activities may be generally solved by effective EIA. In the case of existing aquacultures, specific mitigation measures including the creation of buffer zones may also prevent or minimise the impacts of aquaculture operations on natural habitats (Choo 2001; Páez-Osuna 2001b). For shrimp aquaculture it has been also suggested that the use of abandoned ponds to restore mangrove systems and halophyte crop, or the conversion of shrimp ponds into salt ponds or for cultivation of other species (e.g. shellfish and crabs) (Páez-Osuna 2001b; Primavera 2006), may not only turn into an ecological benefit but also into an economic benefit.

Ecoaquaculture

Integrated aquaculture systems, either polyculture (e.g. fish and mussels, fish and seaweeds) or integrated aquaculture-agriculture systems (e.g. rice – fish farming), has also been considered an efficient and environmentally sound strategy for recycling aquaculture wastes (Buschmann *et al.* 1996; Pearson and Black 2001; Choo 2001; Gräslund and Bengtsson 2001; Páez-Osuna 2001a; Primavera 2006). Examples of the efficiency of these systems can be found worldwide. For instance, filter-feeders (e.g. oysters, mussels) and economically important seaweeds (e.g. *Gracilaria*, kelp) cultured in the immediacy of finfish cages were proven to remove a significant part of the suspended organic matter and dissolved nutrients generated by cage aquaculture, alleviating waste loadings at the same time that it increase the farm productivity (Pearson and Black 2001). Polyculture with shellfish is particularly viable in eutrophic systems because these organisms can significantly reduce algal densities and nutrients loadings (Pearson and Black 2001), in a way that minimise the risks of eutrophication (cf. – Aquaculture environmental impacts). Coupling shrimp culture with bivalve molluscs and fish has also been considered (Sandifer and Hopkins 1996) a promising methodology to reduce the negative environmental effects resulting from the intensification of shrimp farming (Gräslund and Bengtsson 2001; Páez-Osuna 2001a; Biao *et al.* 2004; Primavera 2006). Another example of polyculture is the combined culture of the Chinese and Indian major carps in China, which has the added value that aquaculture wastes can be converted into agricultural wastes (World Bank 2006). Integrated aquaculture-agriculture practices are considered as an ecotechnology, particularly for inland aquaculture. For example, in Vietnam, the use of effluents from hybrid catfish aquaculture on rice farming was able to reduce 32% of total nitrogen and 24% of total P loadings (Lin and Yi 2003). Low-salinity effluents from inland shrimp farming were also used to irrigate melon crops in Brazil, and proved to be an efficient method for minimising the impacts of effluent discharges (Miranda *et al.* 2008). Integrated aquaculture-agriculture may also be used to remove nutrients from pond sediments (Lin and Yi 2003). According to these authors the use of rooted aquatic plants, such as lotus (*Nelumbo mucifera*) in semi-intensive cultures of tilapia (*Nile tilapia*) may remove up to 300 kg N and 43 kg P/ha/year. Besides its widely proven efficiency in removing aquaculture wastes, integrated aquaculture systems, may also reduce the risks of chemical contamination (Gifford *et al.* 2004; Primavera 2006). For instance, as aquaculture effluents naturally improve the fertilization of agriculture fields they reduce the use of environmentally damaging agriculture chemicals (e.g. pesticides, fertilizers), helping farmers to improve protein production and to ensure the economic viability of the activity (Lin and Yi 2003). Polyculture done with bivalves, that filter large volumes of water, may significantly lower the quantity of toxic contaminants released into the environment, acting as bioremediators of stressed coastal environments (Gifford *et al.* 2004). However, if human-consumed bivalves are involved carefully should be taken to avoid chemical and bacterial contamination (Gifford *et al.* 2004).

Another alternative to limit the impacts of effluents from pond aquaculture is the improvement of pond design. For example, ponds that are too shallow might be invaded by macrophytes, whereas in deeper ponds, the water may stratify, causing severe water quality problems, such as oxygen depletion (Boyd 1995). The creation of buffer ponds (e.g. constructed wetlands) has also been proposed as a remediation measure for shrimp farming since it promotes the sedimentation of organic matter and the removal of other pollutants associated with suspended solids before the water is released into the surrounding environment (Boyd and Clay 1998; Kautsky *et al.* 2000; Páez-Osuna 2001a; Primavera 2006). An example from the Red Sea, considered as the third-generation of shrimp farms, consists of circular ponds with central drainage, in which more than 50% of the water surface (including upstream buffer ponds and wastewater treatment ponds) is dedicated to water quality control (Páez-Osuna 2001b). Reduction or elimination of water exchange rates between shrimp ponds and the adjacent water bodies has also been proposed to minimise the adverse effects of effluents discharge (Kongkeo 1997; Boyd and Clay 1998; Páez-Osuna 2001a; Primavera 2006). Restricted water exchange rates will not only lower the risk for sudden changes in water quality parameters, as may minimise the risks of water contamination by saltwater intrusion because it reduces the needs for groundwater. Other measures to reduce or even avoid saltwater intrusion include the utilisation of pond liners and of pond effluents to grow terrestrial halophytes in conjunction with natural filters such as mangroves (Páez-Osuna 2001a; Primavera 2006).

Recirculating Aquaculture Systems (RAS) may also be considered as an ecotechnology. The use of these systems has proven to reduce the amount of effluents by a factor of 500-1000 (Chen *et al.* 1997; Timmons *et al.* 2001), mainly because more than 90% of the water is recycled within the system (Black 2001). Even though the use of RAS does not always result in the overall reduction of discharges but rather on a relocation of wastes (Piedrahita 2003), these systems may facilitate effluent treatment, and thereby minimise potential negative impacts on the environment. Besides requiring fewer water resources, RAS allow a better control over waste discharges and diseases and may prevent the loss of genetic biodiversity (Black 2001; Piedrahita 2003; Gutierrez-Wing and Malone 2006). Because there is no possibility of interactions with wild stocks, this technology also allows the diversification and domestication of farmed species (Black 2001; SECRU 2002; Gutierrez-Wing and Malone 2006) and the intensification of aquaculture operations without seriously damaging the environment, and may contribute to an increase in the productivity and profitability of the aquaculture industry (Black 2001). On the other hand, the use of this technology may have significant economic drawbacks mainly related to the high capital expenditure and running costs (e.g. energy and maintenance) that it involves and due to the increased risk of failure if the systems are not adapted (in terms of biological and engineering concepts) to the species requirements (Black 2001).

As aquaculture grows, it extends its demands on environmental resources, making it urgent to develop new regulations that ensure the transition of the sector to more responsible and environmentally friendly practices. Sound policies, regulatory frameworks, codes of practice and BMPs, including EIAs, physical planning, and economic instruments (World Bank 2006), are among the tools that can be used to reduce the ecological footprint of aquaculture operations and to ensure the sustainability of this activity. Since a substantial component in this footprint is related to wastes production and to the use of fish meal/oils for the production of pelleted diets, the improvement of diet formulations is fundamental for the minimisation of the aquaculture environmental impacts. The development of ecofeeds relies largely on a vast understanding of the nutritional physiology and biochemistry of the different cultivated species (World Bank 2006), from which results a selection of very digestible ingredients that facilitate nutrient assimilation and promote the increase of FCRs. High FCRs have been shown to maximise protein retention and minimise the amount of solid wastes and nutrient loadings resulting from undigested, unutilized and uneaten feeds (Black 2001; World Bank 2006). One of the current lines of investigations on ecofeeds consists for example, in the substitution of fishmeal protein from "trash fish" by a vegetable protein source (e.g. soya), in order to reduce the pressure on natural fisheries resources (Kaushik *et al.* 2004). However, vegetable substitutes often lack essential amino acids and fatty acids, which may constitute an impediment for the economic viability of aquaculture systems. Another constraint is the increasing consumer pressure so that these vegetable ingredients are GMO-free, i.e., not produced from genetically modified organisms (SEACASE 2007).

Given the necessity to ensure the safety of aquaculture products and the increasing consumers demand on food safety and welfare, the adoption of the environmentally friendly practices mentioned above becomes fundamental. The development of certification and ecolabeling schemes, attesting the character of the production processes and the quality of the products, may be an easy and efficient way to achieve the consumer perception and a mean to fulfil the market requirements and of adding value to aquaculture products (WorldBank 2006; SEACASE 2007).

Drivers, pressures, states, impacts and responses (DPSIR)

DPSIR is a causal framework for integrated environmental assessment, describing the interactions between society and the environment (UNEP/RIVM 1994, RIVM 1995). According to this framework, there is a chain of causal links with the following components: Driving forces, Pressures, States, Impacts and Responses. A Driving force results from a need, leading to activities that cause Pressures, affecting the state of the environment, causing Impacts that demand Responses from the society.

Table 3 is a possible example of an application of DPSIR to aquaculture development. It is important to have indicators to quantify each of the five DPSIR components, whenever possible. These indicators may be spatially resolved and integrated in a GIS. Suggested indicators for the example given in **Table 3** could be: Driver - area allocated for fish farms; Pressure – Fluxes of nutrients, organic matter and xenobiotics, and differences in drag related to the presence of aquaculture leases; State – Concentrations in the water and in the organisms (regarding xenobiotics); Impact – changes in described rates; Response – seaweed production, area of sediments where pumping takes place, proportion of leases reallocated and changes in fish density within the farms, respectively.

Implementing the DPSIR framework may be useful to synthesize those indicators that should be included in a GIS for physical CC assessment, as well as those aspects that should be accounted for in CC models, including scenarios to be analyzed (cf. – methods for determining CC). This framework may be used in more complex situations, when there are more drivers besides aquaculture.

Decision support systems

A DSS is an information system that may bring together databases, models and other information sources to help the decision-making process. Considering the multiple interactions between aquaculture systems and other uses of natural resources, an important point about any DSS is to define for whom it is intended. Different actors and stakeholders are important in the decision process. This is well in line with the Integrated Water Resource Management (IWRM) concept, where a balance is to be found between economic and environmental objectives, and where public participation is a key issue (Agnetis *et al.* 2002).

A DSS should allow stakeholders and decision makers to analyse different aquaculture scenarios using geographic and socio-economic data, and model results. These data should reflect best knowledge about several aspects of CC, discussed before (cf. – the concept of CC). The DSS should include a methodology to evaluate those scenarios on some quantitative way towards an informed final decision (Agnetis *et al.* 2002; Pereira *et al.* 2007).

For example, let's assume that several scenarios were purposed regarding increasing the number of fish cages in a particular ecosystem. After conducting a DPSIR analysis (cf. – Ecoaquaculture - Drivers, pressures, states, impacts and responses (DPSIR)) with stakeholders, decision-makers and scientists, potential shortcomings could be identified and used to define the responses needed from scenario analysis. Afterwards, an ecological model of the system under study could predict that increasing fish cages would increase fish production by a certain amount and decrease water quality (for example, though increases in ammonia concentrations and decreases in oxygen levels). An economic assessment of yields could reveal that the aquaculture income was not linearly related to fish production if market prices were not elastic. Therefore, at the end of the simulation process, several results regarding water quality, fish production and economic gains would have to be somehow weighted and compared. This could be done using the Analytic Hierarchical Process (AHP) methodology (Saaty 1980) as suggested in Agnetis *et al.* (2002) and obtaining a score for each scenario. This methodology allows for some subjectivity to be incorporated in the decision process, as a result of different sensitivities of stakeholders to environmental, economic and social aspects. For some examples see Agnetis *et al.* (2002), where this general approach was applied to several management scenarios (including aquaculture) for five coastal lagoons across southern Europe.

LEGISLATION

There is a growing understanding that the sustainability of the aquaculture industry depends largely on the definition of

Table 3 Drivers, Pressures, States, Impacts and Responses for a hypothetical aquaculture development.

Driver	Pressure	State	Impact	Response
Fish farming	Increased nutrient fluxes	Increased nutrient and organic matter concentrations	Increased phytoplankton biomass/eutrophication	Seaweeds production to remove excess nutrients
	Increased organic matter fluxes and oxygen	Decreased oxygen levels Accumulation of organic matter in the sediments	Higher mortality of benthic organisms/decreased benthic diversity	Bottom aeration
	Increased drag forces	Reduced flow-through and increased residence time	Increased sediment deposition	Reallocation to areas of more intense hydrodynamics
	Release of xenobiotics	Bioconcentration	Increased mortality of non-target species	Less intensive farming to reduce disease propagation

an overall policy with appropriate institutional, legal and management frameworks (FAO 2007). The regulation of the industry by state, civil society and/or private sector mechanisms is also crucial for a responsible development, constituting an effective instrument to protect not only the industry, the consumer, and other resource users, but also the environment (Spreij 2003).

Due to the pressure of aquaculture expansion, many countries are in the process of reviewing their legislation or even adopting new comprehensive regulatory frameworks in order to promote a sustainable industry. The development of modern aquaculture legislation should not only be based on the adoption of specific aquaculture texts, but also on the amendment and/or enactment of a number of other related laws, including land, water and environmental legislation, and further regulations on food safety and fish health (FAO 2007). Regarding the aquaculture environmental impacts, several international and national regulatory instruments have been created. At an international level, there are currently several Conventions that can directly influence the management and regulation of aquaculture impacts. For example, the Convention on Biological Diversity (CBD), the Cartagena Protocol on Biosafety, the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and others on marine pollution such as, the OSPAR Convention for the Protection of the Marine Environment of the North East Atlantic and the Helsinki Convention (HELCOM) for the Protection of the Marine Environment of the Baltic Sea Area (Davies 2001). The OSPAR convention is particularly important because it encloses several management practices that contribute to the preservation of the environment on one hand and to guarantee food safety on the other. Among the most important OSPAR recommendations is the PARCOM Recommendation, which defines Best Environmental Practices (BEP) for the reduction of inputs of potentially toxic chemicals from aquaculture units, including methods to reduce the use/release of toxic substances (e.g. antibiotics, parasiticides and antifoulants) and approval systems for fish medicines and monitoring of residues of chemicals in aquatic products (Read and Fernandes 2003). International Conventions are ratified by the signatory States through the implementation of national legislation and regulations. In addition to these Conventions, there are also several international accords with an indirect effect on the monitoring and regulation of the aquaculture industry (Read *et al.* 2001; World Bank 2006).

Because aquaculture units are generally located within national borders, most environmental impacts are managed and monitored by a wide variety of national instruments and arrangements. However, the majority of these frameworks are often applied in an inconsistent manner (Spreij 2003). Given that uncertain and inappropriate legislative arrangements may lead to potential conflicts over the allocation and sharing of natural resources (Spreij 2003), the adoption of an ecosystem approach to aquaculture (EEA) management becomes crucial for the sustainable development of the sector. This concept is gradually wining thought, with some countries already basing future aquaculture developments on integrated management plans (Spreij 2003). For instance, the Philippine Fisheries Code specifically declares that aquatic resources should be managed in a manner consistent with the concept of integrated coastal zone management (ICZM). Likewise, the Tasmanian Marine Farming Planning Act (1995) defines Marine Farming Development Plans that take into account the physical suitability of potential aquaculture units, current legislation and the need to minimize impacts on other users (Spreij 2003). Another example of EEA is found in the European Union (EU) legal framework that regulates and manages the environmental impacts of aquaculture by a series of Directives (**Table 4**) directly implicated in the integration of aquaculture within the management of the whole coastal zone (Read and Fernandes 2003). These Directives are ratified by EU Member States through the implementation of national legislation and regulations.

The way that legal frameworks control inadequate developments and promote responsible aquaculture varies significantly from country to country (**Tables 4**, **5** and **6**). While some countries have a well-defined and comprehensive legislation for the aquaculture sector (e.g. European Union, China, and Japan), others lack any specific legislation (e.g. Thailand, Vietnam) or have only included a statement in their national policy and development plans. The role of aquaculture is also often limited to a section in the basic fisheries legislation, lacking any criteria to set up and operate an aquaculture establishment (Spreij 2003). On the other hand, in some countries like Japan, the regulation of

Table 4 Basic legal frameworks regarding aquaculture environmental impacts in the European Union (EU).

	Basic legislation	General contents
European Union	Dangerous Substances Directive (76/464/EEC)	Requires farmers to reduce pollution by substances with deleterious effects on the aquatic environment and only authorises their release on the basis of Emission Limit Values (ELVs) to ensure compliance with Environmental Quality Objectives (EQOs).
	Quality of Shellfish Growing Waters Directive (79/923/EEC)	Concerns to the quality of shellfish waters in areas designated by the Member States as needing protection or improvement in order to contribute to the high quality of shellfish products directly edible by man; and defines guidelines and imperative values for shellfish flesh and shellfish waters.
	Shellfish Directive (91/492/EEC)	Defines the quality of shellfish waters and lays down the health conditions for the commercial production and the placing on the market of live bivalve molluscs, mussels and scallops.
	Environmental Impact Assessment (EIA) Directives	Defines how to assess the effects of certain public and private aquaculture projects on the environment, reinforcing the need for certain projects to undergo compulsory EIA, according to the scale and intensity of the activity and local conditions.
	Strategic Environmental Assessment Directive (2001/42/EC)	Ensures that the environmental consequences of certain plans and programmes are identified and assessed during their preparation and before their adoption.
	Species and Habitats Directive (92/43/EEC)	Concerns to the protection and conservation of natural habitats. The Directive takes into account the environment carrying capacity, i.e., it acknowledges that receiving areas can accept activities without undue effects. If aquaculture facilities are within these protected sites or adjacent to them and thus likely to affect such sites, then additional restrictions must be anticipated.
	Wild Birds Directive (79/409/EEC)	Concerns to the protection and conservation of wild bird populations.
	Water Framework Directive (2000/60/EC)	Concerns to the maintenance of the integrity of the ecosystem characteristics, and aims to protect natural biodiversity in matters related to biological contaminants, such as genetically modified or selected individuals from farmed stock.
	EC Directives affecting the marketing of veterinary medicinal products	Establishes Maximum Residue Limits (MRLs) and Marketing Authorisations (MAs) for chemicals administered to aquatic products.

Source: Read and Fernandes (2003).

Table 5 Basic legal frameworks regarding aquaculture environmental impacts in some Asian countries.

	Basic legislation	General contents
China	Fisheries Law (1986, amended in 2000)	Seeks to enhance the simultaneous development of aquaculture, fishing and processing.
	Land Administration Law and the Water Law (1988, amended in 2002)	Regulates the development, utilization, saving, protection, allocation and management of water resources.
	Environmental Protection Law (1989)	Defines the necessity to investigate and assess the environmental situation during the planning phase of aquaculture projects.
	Law on the Prevention and Control of Water Pollution (1984, as amended)	Aims to prevent and control pollution in freshwater bodies and groundwater.
	Marine Environment Protection Law (1982, as amended)	Deals with marine pollution, regulating the discharge of oils, oil mixtures, wastes and other harmful substances.
	Law on Animal Diseases (1997)	Describes the procedure to control and cease animal diseases, including quarantine measures.
Philippines	Philippine Environment Code (1988)	Defines all the measures dealing with the natural environment, including the management of air quality, water, land use, natural resources and waste.
	Philippine Fisheries Code (1998)	Provides for the development, management, conservation and utilization of fisheries and aquatic resources. Chapter II, Article III (Sections 45-57) of the Code deals specifically with aquaculture.
	Fisheries Administrative Order No. 214 (2001)	Establishes a Code of Practice to Aquaculture that include the general principles and guidelines for environmentally-sound design and operation of the aquaculture industry, including topics on: site selection/evaluation, farm design and construction, environmental impact studies, water usage, water discharge and sludge/effluent management, use of chemicals (e.g. potentially toxic pesticides and fertilizers), stock selection/stocking practices, introduction of exotic species/GMOs, feed use and management, and fish health management.
Japan	Fisheries Law (1949, as revised in 1962)	Deals in detail with several kinds of fishing rights and licenses for individuals and groups of persons.
	Fisheries Cooperative Association Law (1948, as amended)	Provides the legal framework for local Fisheries Cooperative Associations (FCAs) who bear the responsibility for a particular geographical area.
	Law to Ensure Sustainable Aquaculture Production (1999)	Seeks to prevent the self-induced environmental deterioration around fish farms and includes the basic guidelines to ensure sustainable aquaculture production.
	Basic Environmental Law (1993)	Provides the general principles for environmental protection.
	Environmental Impact Assessment Law (1997)	Ensures that EIAs are conducted properly and smoothly with respect to large-scale projects that could have a serious impact on the environment.
	Water Pollution Control Law (1970, as amended)	Seeks to prevent the pollution of water in public water areas by regulating the effluent discharged by factories or establishments in order to protect human health and to preserve the living environment.
	Law on the Protection of Fishery Resources (1996, as amended)	Aims to prevent the spread of fish disease through imports of marine animals for use in aquaculture or propagation of stocks.
	The Pharmaceutical Affairs Law (1960)	Regulates veterinary drugs and veterinary medical devices.
Vietnam	Fisheries Law	Incorporates aquaculture regulations, regulations on processing, trading, export and import of aquatic products, regulations on rewards and sanctions as well as regulations on clauses for implementation.
Thailand	Fisheries Act (1947, as amended in 1953 and 1985)	Principal legislative instrument dealing with fisheries and the cultivation of aquatic animals.
	Enhancement and Preservation of Natural Environmental Quality Act (1992)	Provides for the establishment of environmentally protected areas and pollution control areas, defines the principal requirements concerning water pollution and specifies the type and size of projects or activities that require an EIA.
	Drug Act (1967) and the Hazardous Substances Act (1992)	Regulate the use of drugs, including veterinary drugs, and chemical substances by creating a list of substances which are controlled and the quantities of these which may be used in particular applications.

Source: http://www.fao.org/fishery.

aquaculture is a unique and very complete process. The Japan's regulatory framework relies mainly on a self-regulatory system known as the Community-based Fisheries Management System. According to this system, an "aquaculture right" legally, granted by the fisheries cooperative associations (Fisheries Cooperative Association Law), protects aquaculture conducted in public waters. These associations are also responsible for ensuring fair allocation of lots, determining the types of structures that should be built, specifying the number of facilities, and setting maximum stocking densities (Spreij 2003).

Legislation specifically dealing with the environmental impacts of the aquaculture industry is particularly diverse (**Tables 4**, **5** and **6**) mostly due to the wide range of interests and institutions involved on the management of aquatic systems. In what concerns to chemical contamination, direct prohibition or restriction upon the use of specific chemicals that are harmful to the environment, may be generally found in basic environmental laws (e.g. China's Environmental Protection Law and the Law on the Prevention and Control of Water Pollution and, Japan's Water Pollution Control Law and Pharmaceutical Affairs Law) or in more specific acts related to the use of particular chemicals (e.g. the EU Directive on Dangerous Substances and Thailand's Drug and Hazardous Substances Acts). Another tool used to control/reduce environmental impacts from aquaculture discharges, include the creation of licensing system for wastewater discharge, as established by the Ecuador's Environmental Management Law and Chile's General Law of the Environment (**Table 6**). To prevent the hazards resulting from disease outbreaks and the spread of parasites many countries (e.g. EU, Japan, China, Australia, U.S.A. and Canada) have introduced legislation that prohibits and/or restricts the movement of fish and other aquatic organisms (Spreij 2004). Particularly remarkable for its comprehensive strategy is the Australian National Strategic Plan for Aquatic Animal Health, which outlines a national approach for the management of aquatic animal health, including quarantine, surveillance, monitoring and research programmes, as well as legislation, policies and jurisdiction (Spreij 2004). In what concerns to the environmental impacts resulting from the introduction and use of exotic or transgenic species, in most developed countries restrictive legislation has been adopted over the years (e.g. EU Water Framework Directive, Canada's National Code on Introductions and Transfers of Aquatic Organisms and the United States National

Table 6 Basic legal frameworks regarding aquaculture environmental impacts in North-American and Latin-American countries.

	Basic legislation	**General contents**
Canada	Federal Fisheries Act (1985)	Regulates the emission of licenses for aquaculture projects in areas of federal property, and licences for the importation into Canada and movement between provinces of live fish (salmonids), eggs, and dead, uneviscerated fish. Prohibits the harmful alteration, disruption or destruction of fish habitat and the deposit of deleterious substances.
	Canadian Environmental Assessment Act (1992)	Legislates the bases for federal environmental impact assessment, and aims to protect wild fisheries and the marine environment while maintaining safe marine navigation.
	National Code on Introductions and Transfers of Aquatic Organisms (2002)	Requires the submission of a detailed application for each introduction and transfer of aquatic organism, including the nature of the organism, the geographical area of the proposed introduction and the likely interaction with native species.
	Health of Animals Act (1990) and Regulations	Defines the protocols for prevention and control of diseases.
	Food and Drug Act (1985)	All veterinary drugs used in aquaculture must be licensed at the federal level by Health Canada (Bureau of Veterinary Drugs), under the Food and Drug Act (1985) Establishes the maximum residue limits (MRLs), administrative maximum residue limits (AMRLs) or residue limits for antibiotic drugs used in aquaculture.
U.S.A.	National Aquaculture Act	Establishes and implements a national aquaculture development plan.
	National Offshore Aquaculture Act (2005)	Establishes and implements a regulatory system for offshore aquaculture in the U.S. Economic Exclusive Zone.
	National Oceans Protection Act (2005)	Specifies the need to define national standards and regulations to protect native stocks and prevent parasites and disease transmission, genetic dilution of wild stocks, introduction of invasive species, and deterioration of habitat and water quality.
	Regional, State and Other Policy Actions, (e.g. California bill SB 245 and the Washington Fish and Wildlife Commission)	Defines policies for state waters: e.g. respectively, the restriction of aquaculture of salmon, exotic and transgenic fish in California waters and specific rules for the control and prevention of fish escapes.
Ecuador	General Regulation to the Fisheries and Fisheries Development Law (1974 as amended) revised in 1985.	Establishes the procedures to set up aquaculture facilities and deals with operational aspects of the activity (authorization system; environmental impact assessment; and use of veterinary drugs).
	Environmental Management Law	Ensures that authorizations for aquaculture in highlands exploiting groundwater sources, is subject to the submission of an Environment Impact Study. Optimizes the management of all types of waste matter. Regulates the movement of wild fauna, which applies also to aquatic resources and sets restrictions on the use of veterinary drugs in aquaculture activities.
	Animal Health Law	Regulates disease control but no specific rules were found on aquaculture.
Chile	Fisheries and Aquaculture Law (1989, as amended up to 2006)	Regulates the conservation of living aquatic resources. Title VI of the Law is dedicated to aquaculture.
	General Law on the Environment	Defines that the authorizations and concessions to conduct aquaculture are granted based on environmental permits.
	Regulation on the EIA System (1997, as modified in 2001)	Defines that EIA studies should include a description of the project, a plan of compliance with the relevant environmental legislation, an estimate of the environmental impact of the project, including possible risks, a mitigation, repair and/or compensation plan, and a monitoring plan of the main environmental variables for which the Environmental Impact Study is required.

Source: http://www.fao.org/fishery.

Oceans Protection Act), while in developing countries only few measures have been taken so far (e.g. the Philippines Administrative Order No. 214). Specific legislation on the use of genetic modified organisms (GMOs) is quite recent and is being implemented as these organisms become available for the consumers, mainly due to the necessity to guarantee public health and prevent environmental pollution. Regulations controlling the safety of GMOs are included for example on, the Chinese Safety Administration Regulation on Genetic Engineering (1993) and on the EU Water Framework Directive (2000/60/EC).

The most widespread regulatory tool for controlling the environmental impacts of aquaculture operations and preventing unsustainable developments is the obligation of governmental authorization before establishing and/or operating an aquaculture facility. These authorizations often require the submission of an Environmental Impact Assessment (EIA) study, depending on the farm size or the sensibility of the area involved. For instance, the Environmental Impact Assessment Regulation No. 25318 (2003) of Turkey legislation, defines that the use of EIA is compulsory in projects with an annual production greater than 1000 tonnes, while farms with a capacity between 30 and 1000 tonnes per year are only required to submit a preliminary EIA (Spreij 2004). If well applied, EIA may also be an effective tool for preventing the destruction and/or modification of natural habitats, in particular those with high ecological value like spawning, breeding and feeding grounds and major migration passages of aquatic species, as defined in the Environmental Protection Law of the People's Republic of China (Spreij 2004). A wide range of EIAs are currently employed in developed countries (e.g. EU countries, Japan), often with an impressive and exhaustive detail, and are increasingly being used in developing countries (e.g. Thailand, Vietnam, Ecuador, Chile). Still, in many other cases such procedures are simply not used, not sufficiently developed, or not implemented (FAO 2007). A typical EIA usually identifies and evaluates the magnitude of potential environmental impacts during all phases of farming activity, i.e. implementation phase, operational phase and deactivation phase. Moreover, it considers both positive and negative environmental, social and economic aspects, whether direct or indirect and at different time scales, i.e. short, medium and long term (SECRU 2002). The main aspects focused on EIA studies for aquaculture projects, include for example, local hydrography/water quality, topography and soil characteristics, benthic and water column conditions, ecology, sensitive sites and species, water-based human activities (e.g. navigation, anchorage, fisheries and other non-recreational water uses), waste management, landscape, noise, cultural heritage, socio-economics, accesses and recreation, traffic and transport (World Bank 2006). Frequently, EIAs also present environmental management measures to avoid, minimize and mitigate the expected negative impacts of farming activities (**Tables 4**, **5** and **6**). Despite their relevance, EIAs may have significant limitations. For instance, they cannot be applied to existing aquaculture farms because in most cases there is not sufficient information on the environment state prior to the establishment of farm units. Another drawback is that EIAs are frequently activity-oriented, and do not consider integrated planning, which is crucial for the sustainable use of common resources and the development of other human activities (Read and Fernandes 2003). Fortunately this is starting to change. For instance, in China, the Environmental Impact Assessment Law (2002) expands EIA requirements from individual construction projects to government planning (Spreij 2004). In developing countries, EIAs often fail to take into account the human and social aspects associated with the farming activity, particularly in what concerns to the poorest social strata (FAO 2007).

Due to the overlap of laws, regulations, government institutions and agencies involved in the management of the aquaculture sector, the implementation of enforcement mechanisms is usually complicated. Where the introduction of new legislation is difficult, or will cause excessive delay, other voluntary instruments, such as Codes of Conduct and Codes of Practice serve as an alternative to legal frameworks (Spreij 2004). While, Codes of Conduct comprise a set of general rules and principles that should lead to the responsible and sustained development of the industry, Codes of Practice provide proactive guidelines to avoid pollution and give recommendations on practices that optimise the management of aquaculture operations (Fernandes *et al.* 2002). In practice, the fulfilment of these codes requires the adoption of Best Management Practices (Best Environmental Practices and Best Available Technologies) in relation to several aquaculture aspects, such as, site selection (e.g. prohibition of mangrove exploitation), standards for feeds and seeds, definition of maximum production per area, use of chemicals and rules for disease control (Ervik *et al.* 1994; FAO 2007). These voluntary codes have been actively pursued on a worldwide scale, being established at different levels, internationally, nationally and at the industry level, as mechanisms of self-regulation. At an international level, the Food and Agriculture Organisation (FAO) adopted a Code of Conduct for Responsible Fisheries (1995), later expanded (in 1997) to include aquaculture development. Industry/organization codes, like for example the voluntary code of conduct of the Federation of European Aquaculture Producers (FEAP), have also been adopted to promote the sustainability of the aquaculture industry. Codes of Conduct and Codes of Practice were also specifically created for shrimp faming (e.g. in India, Malaysia, Sri Lanka, Philippines and Thailand), for salmon farmers of British Colombia and Australian prawn farmers (FAO 2007).

Although the modernization of aquaculture legal frameworks is progressing slowly, the main problem continues to be the lack of means (e.g. trained staff and budgets) for appropriate law enforcement. Thereby, in the future, other options such as financial support for environmental friendly locations and technologies or the application of the "user pays" principle should be explored to encourage farmers to make more efficient use of resources and to take full responsibility for the mitigation or minimisation of the environmental impacts caused by aquaculture operations (Spreij 2004; World Bank 2006).

CONCLUDING REMARKS

Considering all the aspects discussed in previous chapters, some conclusions may be synthesized as follows:

(i) Aquaculture management should be participated by relevant stakeholders and viewed within the context of management plans, including other activities with which it that may have positive and negative synergies;

(ii) Ideally, an ecosystemic approach in line with Ecological Engineering should be developed towards an "ecological aquaculture" to prevent going through the same mistakes as industrial agriculture and husbandry;

(iii) Low trophic level species should be preferred for a higher energy efficiency and low ecological footprint;

(iv) The Carrying Capacity concept is central to aquaculture sustainability in all its environmental, economic and social dimensions;

(v) There are several tools that may and should be used in aquaculture management and that have already been tested widely, such as GIS, the DPSIR framework, mathematical models and DSS;

The development and implementation of effective legal frameworks together with the adoption of environmentally friendly technologies/systems and the application of codes of conduct and best practices for responsible aquaculture may contribute to its environmental sustainability, by increasing economic returns while providing environmental protection.

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