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Review

Aquaculture effects on environmental and public welfare – The case of Mediterranean mariculture

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ABSTRACT

Aquatic farming has been considered, during the last decades, as the fastest growing food production industry powered by governmental and technological impulsion. Compensation for fisheries decline, creation of new jobs and source of financial windfall are the most important benefits. However, similar to most of the human food-production activities, aquaculture raised several issues related to the environmental welfare and consumer safety. An effort to record the aquaculture-environment and -human safety interactions with regard to the Mediterranean mariculture, is attempted herein. We focused on this geographical area due to its individualities in both the hydrological and physicochemical characteristics and the forms of aquaculture activities. The cage farming of euryhaline marine fish species and more recently of bluefin tuna and mollusk farming are the dominating aquaculture activities. The impacts of these activities to the environment, through wastes offloads, introduction of alien species, genetic interactions, disease transfer, release of chemicals, use of wild recourses, alterations of coastal habitats and disturbance of wildlife, are analytically considered. Also the consumer safety issues related to the farming are assessed, including generation of antibiotic-resistant microorganisms, contaminants transferred to humans though food chain and other hazards from consumption of aquacultured items. Within these, the major literature findings are critically examined and suggestions for scientific areas that need further development are made. The major tasks for future aquaculture development in this region are: (i) to ensure sustainability and (ii) to balance the risks to public or environmental health with the substantial economical benefits. In regard with monitoring, tools must be created or adapted to predict the environmental costs and estimate consumer impact. At a canonistic and legal basis, the establishment of appropriate legal guidelines and common policies from all countries involved should be mandatory.

Contents

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1. Introduction

1.1. Characteristics of the Mediterranean marine aquaculture

Aquaculture seems to be the fastest growing animal-food production fuelled by governmental support, evolved technological assistance and pessimistic forecasts for fisheries production. Its accelerated growth is reflected by the fact that its contribution to world supplies of aquatic products has increased from 1 million tonnes per year in the early 1950s, to a production of 55.1 million tonnes in 2009 (FAO, 2006).

Contemporary Mediterranean mariculture was initiated in the 1980's mainly by the culture of European sea bass (*Dicentrarchus labrax*), gilthead sea bream (*Sparus aurata*) and shellfish. During the 1990s, Atlantic bluefin tuna (*Thunnus thynnus*) rearing and the farming of other sparids were introduced. More recently other species, mainly from Scienidae and Caragidae families entered the industry with the most promising including meagre (*Argyrosomous regious*) and greater amberjack (*Seriola dumerilii*). The last three decades following the global trend, Mediterranean mariculture has experienced a rapid growth. In particular, while total production accounted for 90 000 tonnes in 1985, the 2007 figures have grown to 436 401 tonnes for marine fish and 174 385 tonnes for molluscs (CAQ-GFCM, 2008, 2009; FAO, 2008; Barazi-Yeroulanos, 2010).

Several finfish marine species have been cultured in the Mediterranean, the most commercially important still represented by gilthead sea bream and European sea bass (see production by species in Fig. 1). Atlantic bluefin tuna rearing has now reached a considerable production and according to FAO (FAO, 2006), world tuna production increased almost 15 times during the past decade with Mediterranean ranching reaching the same figures. Meagre has emerged also as a promising candidate with predicted accelerated production for the following years. The farming of sharpsnout sea bream (*Diplodus puntazzo*) and other sparids have not been accompanied with the anticipated success. With regard to shellfish, production is mainly represented by the European mussel (*Mytilus galloprovincialis*), the Japanese carpet shell (*Ruditapes philippinarum*), the European flat oyster (*Ostrea edulis*) and the Pacific oyster (*Crassostrea gigas*) (CAQ-GFCM, 2008). Culture of aquatic plants and crustaceans remains low.

To cope with the continuous increase in production, Mediterranean finfish mariculture has moved from small land-based operations to large enterprises located along the coastline and more recently to off shore sites. A part of shellfish intense production has been also shifted from coastal lagoon locations to offshore areas. A consideration of the uniqueness of Mediterranean to sustain increased mariculture productivity should be mandatory. This semiclosed basin is characterized by several peculiarities such as low trophic potential, high sea water temperatures, phosphorus-limit primary production, low sedimentation rate and high maritime traffic (CIESM, 2007).

Almost all south European and north African countries display aquaculture activities with France, Spain, Italy, Turkey and Greece

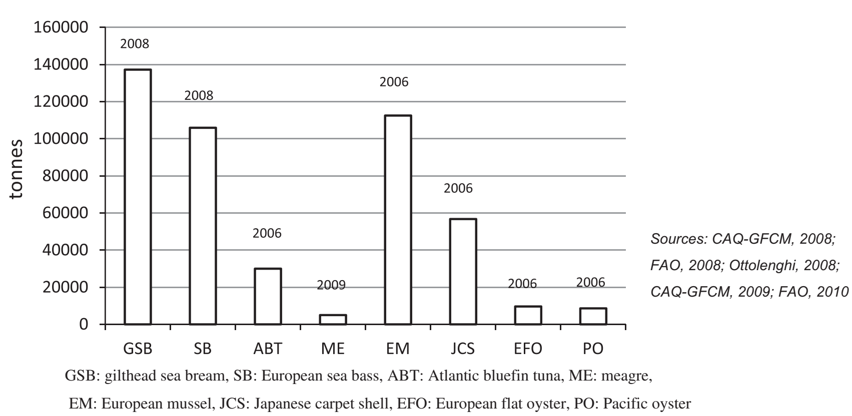

Fig. 1. Marine aquaculture production of the main species in the Mediterranean.

being the main producers in the marine environment. The mosaic of this industry is complicated by major structural differences of the participating countries. There is particularly a substantial variation in the state of aquaculture development as well as in the individual socio-economic and cultural aspects of these countries. Mediterranean EU countries have relatively common legislation that affects the different aspects of aquaculture which may, however, vary from related regulations established in north African countries (Hough, 2000). Thus, the potential impact of current aquaculture practices on human and environmental health may vary among these regions.

As in other parts of the world, the accelerated growth of aquaculture in the Mediterranean seemed in some occasions to have proceeded ahead of adequate environmental and human welfare, social acceptance and market adaptability. In particular, the industry recently suffers a characteristic local cyclicality (Barazi-Yeroulanos, 2010) and faces a set of constrains, such as recurrent unsuitability between production and demand, interaction and space competition with other coastal zone users, possible environmental impacts and consumers' negative perceptions for the product quality (CIESM, 2007).

This review summarizes available published literature, findings of European or national projects, web data, stakeholder meetings, grey literature and anecdotal reports aiming to provide a general overview of aquaculture impact on the environmental and public health in Mediterranean mariculture mainly with reference to finfish farming. Recommendations for management solutions, measures to minimize impacts and future priorities are also discussed.

2. Environmental impact

2.1. Genetic interactions

One of main environmental hazards associated with aquaculture activities in the aquatic environment are the potential genetic interactions between escaped farmed organisms and wild fish communities (Grigorakis, 2010). Also less importantly, spontaneous spawning of caged fish may result to release of gametes and thus genetic contamination in the souroundin environment of aquatic farms (Dimitriou et al., 2007).

Fish escapes from fish farming sites is an inevitable phenomenon resulting from human errors during routine handling, mechanical failures and damages caused by adverse weather conditions or aquatic predators such as seals and dolphins attempting to tear the nets. In the Mediterranean aquaculture, there is no current estimation for the actual size of the fish escapes. Besides escapes, another human action that results to cultured-wild organisms' interaction is the deliberate restocking of lagoons with culture-originated fish. Although this is not a systematic practice in the Mediterranean area, there have been incidences where restocking of lagoons resulted in consequences on both genetic variability and stability of the populations (Rossi et al., 2009).

Farmed fish originated from wild stocks supposedly will initially have the same genetic adaptability for the environment compared to wild stock. However, farmed fish species are genetically manipulated through breeding regimes and owning to domestication attain commercially desirable attributes such as high growth rates, disease resistance, altered aggression, adaptation to high stocking densities. Such breeding programs have been also mentioned for Mediterranean gilthead sea bream and European sea bass (Thorland et al., 2006; Dupont-Nivet et al., 2008; Antonello et al., 2009; Navarro et al., 2009).

The rapid development of cage farming in the Mediterranean has raised questions as to the genetic impact of farm-originated fish on natural populations. These objections are furthermore justified considering that it is a common practice in Mediterranean hatcheries to breed gilthead sea bream from Atlantic broodstocks mainly because of their prominent growth performance when compared to Mediterranean-originated gilthead sea bream (Miggiano et al., 2005). The potential dangers from breeding of escaped farmed fish with native wild populations, are the outbreeding depression (generation of less-fit hybrid populations) and the inbreeding depression (generation of large numbers of related genotypes derived from a small proportion of the broodstock) (Ward, 2006).

Various studies have been focused on finding genetic differentiations in gilthead sea bream populations. Youngson et al. (2001), based on allozyme and molecular data, indicated a substantial gene flow among gilthead sea bream across the eastern Mediterranean to the Azores. However, other studies showed opposing results. Genetic differentiation has been found between various wild populations of gilthead sea bream in the Mediterranean (Ben Slimen et al., 2004; Rossi et al., 2006; Chaoui et al., 2009). In a study by De Innocentiis et al. (2005) who investigated the geographical origin of Italian gilthead sea bream breeders, concerns were expressed owing to heterogeneity and non-locality of the broodstock in case of escapees or establishment of restocking programmes.

Miggiano et al. (2005), applying genetic fingerprinting in specimens from wild populations and from two different hatchery broodstocks, one of Atlantic and one of Mediterranean origin, found genetic differentiation among the three populations. The method that the previous authors developed, using genetic tagging by amplified fragment length polymorphism (AFLP) and microsatellite markers was indicated to be a useful tool for assessing cultured gilthead bream escapees and their impact on the wild populations, but with the presumption that extensive genetic tagging in Mediterranean broodstocks will take place. Karaiskou et al. (2009), using similar tools, demonstrated genetic differentiation between wild and cultured Greek populations of gilthead sea bream. Alarcon et al. (2004) in a study representing the first large-scale population genetic analysis, using allozyme, microsatellite and mitochondrial DNA, found a higher level of polymorphism in microsatellites. A decrease in variability has been observed in the same study, in cultivated populations when compared to wild ones; however, the size of this variability difference is, according to the former authors, not sufficient to document inbreeding depression effects. They suggested that no evidence occurs for significant genetic flow between cultivated and wild populations, since high differentiation between them was observed.

For European sea bass, Youngson et al. (2001) concluded that there is evidence for three distinctive populations, an Atlantic, a western and an eastern Mediterranean stock. They also claimed that most of aquaculture populations, all over Mediterranean, initially derived from Atlantic-originated fish and from mixed Atlantic and western Mediterranean fish. The potential interactions between cultured and wild European sea bass can be attributed in addition to escapes, also to the unintentional release of viable gametes from farmed populations (Youngson et al., 2001). Allegrucci et al. (1997), using allozyme analysis, found a significant degree of geographic genetic variation, being more profound between the Portuguese and the Mediterranean populations. In a study using microsatellite loci, Bahri-Sfar et al. (2000) reported two separate clusters differentiating the populations of eastern and western Mediterranean. In a latter study, the same authors (Bahri-Sfar et al., 2005), among 15 aquacultured populations, indicated only one revealing significant reduction of genetic variability. Therefore, it was revealed that these stocks are largely outbred and open to wild fishes. According to the later study, the use of occidental fingerlings to seed eastern Mediterranean broodstocks since the early 1980s, seriously questions the biological mechanisms explaining the maintenance of fish from occidental origin within an oriental context for two or three generations. The aforementioned data imply that, unlike gilthead sea bream that shows a high polymorphism and no clear structured pattern, European sea bass indicates a genetic structuring, and thus higher outbreeding possibilities (Ward, 2006).

Data for other Mediterranean-cultured fish species are very limited. Indications occur, for instance, about genetic differentiation between Atlantic and Mediterranean populations of sharpsnout sea bream (Bargelloni et al., 2005). However, there is no clear picture for the extend of interbreeding due to aquaculture, although the high values of genetic variation within, and the low values among the studied populations, so far indicated no inbreeding effects, and thus a proper hatchery management (Pereira et al., 2010).

In conclusion, data concerning interbreeding between wild and cultured fish in the Mediterranean, are variable and sometimes contradictory and this fact has been recently stated (Porta et al., 2010). This is mainly due to the different methodological approaches in the various studies, but also maybe because of the delayed beginning of the monitoring when compared to the long existence of interbreeding and the pre-existed genetic changes in wild populations. The differential efficiency of the various genetic tools used may also have contributed to the data variability. Overall, however, there seems to be enough evidence to suggest that there is genetic impact to wild populations induced by escapees from Mediterranean farming sites.

2.2. Aquaculture-mediated species invasions

Aquaculture, and primarily mollusc culture, is responsible for introduction of many new alien species in ecosystems. In a recent review, Galil (2009) recorded 573 metazoan species that are exotic to the Mediterranean. The 10% of these introductions was aquaculture-mediated, while an additional 2% was arrived by aquaculture and secondarily spread by vessels (Galil, 2009; Zenetos et al., 2009). In total, according to the former, there are 48 algae species, 2 polychaeta, 8 crustacean species, and 15 molluscs introduced through aquaculture (Table 1). Most of these species are of Indo-Pacific origin.

Aquaculture seems to be the primer mechanism for macrophytes invasion in the Mediterranean (Galil, 2009). It has also been the primer reason of intentionally introduced alien shellfish species. The Portugese oyster (*Crassostrea angulata*) was introduced in the 19th century from the Atlantic coast in the French Mediterranean coast area and subsequently in many locations in the Mediterranean (Galil, 2000). The Pacific oyster was introduced from Japan in the Mediterranean coast of France by the late 1960s, in Italian lagoons a decade later and subsequently in the rest of the Mediterranean (Zibrowius, 1992; Galil, 2000). The Sydney rock oyster (*Saccostrea commercialis*) was transferred in the mid 1980s in Venice lagoon from New South Wales, while the Manila clam (*R. philippinarum*) was introduced in France in late 1970s, in the Venice lagoon in 1983 and in the rest of Italy during 1985 (Galil, 2000).

There seems to be a strong differentiation in eastern Mediterranean where, contrary to the rest of the region, mariculture as alien species introduction method contributed only a 3% of the total species present (Galil, 2009). This has been well documented in the case of macrophytes in Greece, where aquaculture plays a minor role in macrophytes introduction (16%), unlike the western Mediterranean where aquaculture accounts for 73% of the macrophyte introductions (Tsiamis et al., 2008). The most profound explanation for this differentiation is the fact that mariculture of non-indigenous shellfish species predominate in northern Mediterranean lagoons in France and Italy, and facilitated the invasion of correlated macrophytes (Galil, 2000; Mineur et al., 2007). Mariculture in the Mediterranean, besides being an alien species introduction mechanism, has also been mentioned to offer ecological niches for alien fish species that have been introduced by other methods (Golani, 2004).

Maybe the most studied ecosystem case, in aspects of alien species invasion, is that of Thau lagoon in France which represents a traditional shellfish farming area with production zones covering the 1/5 of the lagoon area and a total production reaching 15 000 tones of oysters and mussels (Mesnage et al., 2007). Most of the studies about Thau lagoon refer to algae invasive species and their mediation through oyster aquaculture (Table 2). The impacts of alien invasive species have been also recorded for some lagoons in the north Adriatic such as the Venice lagoon (Sfriso and Curiel, 2007), the Sacca di Goro lagoon in the Po river delta (Mistri, 2002, 2004a, 2004b; Munari, 2008), the south Adriatic lagoon of Lesina (Nonnis Marzano et al., 2003) and the Rio Padrogiano delta in Sardinia (Munari, 2008).

The ecological impact of these invaders is very variable. Perhaps the most pronounced ecological impact is that of tropical alga *Caulerpa taxifolia*, also known as the killer alga. It was accidentally introduced by aquaria outflow in Monaco (Galil, 2000). This species is known to have toxic impact to other macroalgae, to have a high ecological fitness and is known to have colonized large areas in the littoral zone along the French and Italian Riviera (Boudouresque and Verlaque, 2002; Streftaris and Zenetos, 2006). Its toxic effects are due to the production of terpenoids (Jung et al., 2002) that have been shown to be synthesized in higher concentrations in the Mediterranean than in tropical waters (Guerriero et al., 1992).

Other aquaculture mediated alien species have been shown to induce negative environmental effects. Aquaculture itself can, in many cases, be impacted by the invasive species. Some of the macrophytes alien species, significantly contribute to the fouling communities in shellfish aquaculture (Streftaris and Zenetos, 2006). Among these *Antithamnion nipponicum*, *Grateloupia turuturu* and *Polysiphonia morrowii* are also aquaculture mediated. Furthermore, *Undaria pinnatifida* a brown algae, was introduced for culture purposes, but being spread in the Mediterranean and sprawling over the shellfish beds, it obstructs light and inhibits shellfish growth (Verlaque, 1994). *Sargassum myticum* was introduced in France and Southern Spain with oyster culture and it covered the artificial substrates in high rates and inhibited the development of native algae; it also produces problems in navigation by entanglement in the ship propellers and water pipes. The veined rapa whelk (*Rapana venosa*) is a mollusc that is an active predator of epifaunal bivalves and poses a serious threat to cultivated and wild populations of oysters and mussels (Zenetos et al., 2003).

Among the alien aquaculture-mediated crustaceans, the mud crab (*Dyspanopeus sayi*) feeds on bivalves and seems to cause effects in local clam farming in North Andriatic Sea (Mistri, 2004b; Streftaris and Zenetos, 2006). The copepod *Myticola orientalis*, an intestinal parasite, has been found to produce serious damage to the indigenous European flat oyster and to the blue mussel (*Mytilus edulis*) (Galil, 2000; Streftaris and Zenetos, 2006). *Myicola ostreae* is also an alien parasitic copepod, that infests the gill of oysters, but no significant effect in the Mediterranean has been mentioned so far according to Streftaris and Zenetos (2006). The invasive copepod species *Acartia tonsa* has caused the disappearance of *Acartia margalefi*, in northern Adriatic lagoons due to its high tolerance to anoxic and sulphidic conditions (Sei et al., 1996; Sei and Ferrari, 2006).

However, there are cases when invaders may also have positive effects on the recipient environment. Such an example is the epizootic *Crepidula fornicata*, of N. American Atlantic origin, also known as slipper limpet, introduced with oysters in the Mediterranean. The slipper limpet, on one hand, impedes the native blue mussel (Thieltges et al., 2006), and enhances silting and modification of faunal assemblages (De Montaudouin et al., 1999). On the other hand, however, it may also provide protection against starfish pre-

Table 1
Reported exotic species introduced by aquaculture activities in the Mediterranean. (*Sources*: Zibrowius, 1992; Simonini, 2002; Zenetos et al., 2005; Savivi and Occhipinti-Ambrogi, 2006; Sei and Ferrari, 2006; IUCN, 2007; Galil, 2009; Zenetos et al., 2009).

Exotic organism		Origin	Region of introduction	Remarks
Macrophytes				
Forty-eight algae species (29 phodophyceae, 13 phaeophyceae, 6 chlorophyceae)				
Annelida (polychaeta)				
Neanthes willeyi	O	Southern Ocean	Egypt	Possibly through aquaculture
Ophryotrocha japonica	O	Pacific	Italy	
Hydroides dianthus	O	Atlantic	Greece	
Hydroides diramphus	O	Circumtropical		
Crustacea				
Acartia tonsa (copepoda)				Possibly through aquaculture
Caprella scaura	O	W. Indian	Italy	
Dyspanopeus sayi	PR	NW Atlantic	Italy	
Elminius modestus (cirripedia)	F	Pacific	France	
Fenneropennaeus meguiensis	V	Indo-Pacific	Turkey	
Marsupenaeus japonicus	V	Indo-Pacific	All over Mediterranean	
Myicola ostreae (copepoda)	P	NW Pacific	France	
Mytilicola orientalis (copepoda)	P	NW Pacific	France	
Paracerceis sculpta	O	NE Pacific	Italy	
Rhithropanopeus harrisii	O	NW Atlantic	Italy, France, Tunisia, Malta, Libya	
Scyllarus caparti	O	E Atlantic	Italy	
Molluscs				
Chromodoris quadricolor		Indian	Turkey, Italy, Tunisia, Malta, Libya	
Crepidula fornicata	O	NW Atlantic	Greece	By shipping related to oyster/mussel aquaculture
			France, Sicily, Tunisia, Malta, Libya	
Gibbula cineraria		E Atlantic	Tunisia, Malta, Libya	
Haminoea callidegenita		NE Pacific, E Atlantic	Italy	
Rapana venosa	PR	Japan	N. Aegean	Possibly transported with products of marine farming
			N. Adriatic	
Crassostrea gigas	A	Indo-Pacific	Greece	Intentionally introduced for Mediterranean aquaculture
			All over Mediterranean	
Anadara inflata		NW Pacific	Turkey	
Mercenaria mercenaria	A	NW Atlantic	Italy, France	
Musculista senhousia		W. Pacific	Egypt, Israel, Italy, France	
Mya arenaria	A		Greece	
			Italy, France	
Pinctada margaritifera		Indo-Pacific	Egypt, Israel, Italy	
Petricola pholadiformis	A			
Pinctada radiata (Pearl oyster)	A	Indo-Pacific	Aegean islands, Evoikos (Greece)	Intentionally introduced for Mediterranean aquaculture
			All over Mediterranean	
Ruditapes philippinarum	A	Pacific	Turkey, Italy, France	Introduced in the 1980s for aquaculture
Saccostrea commercialis	A	SW Pacific	Italy	
Xenostrobus securis		Pacific	Italy, France	
Osteichtyes				
Abudefduf vaigiensis		Indo-Pacific	Israel, Italy	
Mugil soiuy	A	W Pacific	Aegean (Greece, Turkey)	
Liza hematocheilus		NW Pacific	Turkey, Greece	
Pagrus major	A	NW Pacific	East Adriatic	Intentionally introduced for Mediterranean aquaculture
Platax teira		Indo-Pacific	Turkey	
Sciaenops ocellatus	A	W Atlantic	Israel	Introduced in 1990s for Mediterranean aquaculture

A: aquacultured organism, F: fouling organism, P: pathogen, PR: predator of aquacultured organisms, V: organism with commercial value O: other organism.

dation and additionally serve as a sink for infectious trematodes, and therefore have beneficial effects to the mussels (Thieltges et al., 2006). Additionally, this organism seems to cause a phytoplankton shift from toxic flagellates to diatoms (Thieltges et al., 2006).

Musculista senhousia, the alien green mussel of western Pacific origin, has been reported to have a gross positive effect on the benthic community. In specific, it has been positively correlated with mussel abundance, it plays the role of secondary substratum, thus enhancing the environmental structural complexity and it aggregates living organisms thus interacting with other invertebrates (Munari, 2008).

Conclusively, a large number of alien species have been related to aquaculture activities in the Mediterranean, but comparing to other methods of introduction, aquaculture accounts only for a small proportion (10%) of the total species invasions. However, as shown herein, the impacts of these aquaculture-mediated species are not negligible.

Table 2
Existing literature on aquaculture-mediated alien species introduction in Thau lagoon (France).

Reference	Remarks
Pérez et al. (1981)	Biology of of Japanese brown algae (*Undaria pinnatifida*)
Riouall (1985)	Accidental introduction of *Sphaerotrichia divaricata* (Rhodophyta) and *Chondra filum* (Phaeophyta)
Floch et al. (1991)	Ecology of Japanese brown algae (*Undaria pinnatifida*)
Grizel and Heral (1991)	Introduction of *Undaria pinnatifida* and *Laminaria japonica* with the Pacific oyster (*Crassostrea gigas*)
Peters et al. (1993)	Genetic origin of alien species *Sphaerotrichia divaricata* (Rhodophyta)
Guelorget et al. (1994)	Structure and organization benthic fauna (macrofauna–meiofauna)
Verlaque (1994)	Origin of macroalgae and environmental consequences
Gerbal and Verlaque (1995)	Macrophyte benthos in the soft substrate and related environmental factors.
Verlaque and Latala (1996)	Accidental introduction of *Chondrus giganteus* (Rhodophyta)
Lamy et al. (1998)	Macrophyte ecology of oyster breeding tables – hard substrate
Verlaque (2001)	A checklist of macroalgae based on both literature records and collections
Verlaque (2002)	Ecology of alien species *Dasya sessilis* (Rhodophyta)
Verlaque et al. (2002)	Ecology of alien species *Ulva pertusa* (Chlorophyta)
Mouillot et al. (2005)	Taxonomic diversity indices for macrophyte community
Verlaque et al. (2005)	Genetic analysis of five alien *Grateloupia* (Rhodophyta) species
Vincent et al. (2006)	Relationship between introduced species and indigenous macrophytes
Mineur et al. (2007)	Experimental assessment of oyster transfers as a vector for macroalgal introductions

Table 3
Wild fish assemblages near fish farms in the Mediterranean.

Aquacultured species	Mediterranean geographic area – location	Number of wild fish species	Most abundant fish species	References
European sea bass and gilthead bream	Spanish coasts (SW)	26	Black sea horse mackerel (*Trachurus mediterraneus*), pompano (*Trachinotus ovatus*), saddle bream (*Oblada melanura*), European barracuda (*Sphyraena sphyraena*), Mugilidae	Dempster et al. (2005)
	Spanish coasts (SW)– open sea	32 (17 taxa)	Black sea horse mackerel, bogue (*Boops boops*), round sardinella, pompano, bluefish (*Pomatomus saltatrix*), flathead grey mullet	Fernandez-Jover et al. (2008)
	Aegean sea (E)	34 finfish and 4 invertebrates	Bogue, flathead grey mullet, blackspot sea bream (*Pagellus bogaraveo*), salema (*Sarpa salpa*), common two-banded sea bream (*Diplodus vulgaris*), annular sea bream (*Diplodus annularis*), striped sea bream (*Lithognathus mormyrus*), gilthead sea bream	Akyol and Ertosluk (2010)
Atlantic bluefin tuna	Adriatic sea	22	Bogue, garfish (*Belone belone*)	Segvic Bubic et al. (2011)

2.3. Transfer of diseases

Marine aquaculture enterprises in Mediterranean area are dominated by caged open systems and thus, the direct interaction between farmed and wild organisms is unavoidable. It has been documented that apart from molluscs and other non-fish scavengers, which represent a considerable part of the resident biota beneath aquatic farms, numerous wild fish species are aggregated in the vicinity of fish cages located at specific Mediterranean locations (Table 3). Moreover, their abundances, which may persist for long periods in the vicinity of sea cages, are enormously increased when compared to their control habitats (Dempster et al., 2004). Consequently, the possible pathogen flow from farmed to wild organisms and vice versa is inevitable.

It is generally unrealistic to determine the effects of farm-originated pathogens on local wild fish populations without the use of exhaustive research. Such attempt should include at least a full investigation on the numerous wild fish species that reside around farms and their related parasitic fauna (Fernandez-Jover et al., 2010). The possible significance of such interaction between farmed and wild populations has not been investigated in depth in Mediterranean region. Some recent studies were unable to provide full evidence for cross infection of pathogen from farmed to wild fish in Mediterranean (Mladineo et al., 2009; Fernandez-Jover et al., 2010). On the contrary, Mladineo and Marsic-Lucic (2007) suspected pathogen switching from wild sparids to caged gilthead sea bream. Earlier studies and unpublished reports (reviewed by Vatsos et al. (2007)), in most occasions have simply isolated common aquaculture pathogens in wild fish species, while actual evidence for pathogen cross-contagion at both directions have been only limited in Israel (Zlotkin et al., 1998; Diamant et al., 2000, 2005, 2007; Ucko et al., 2002; Ucko and Colorni, 2005).

There are mostly anecdotal reports on the isolation of important bacterial pathogens affecting Mediterranean farmed fish including *Listonella* (*Vibrio*) *anguillarum* and other pathogenic *Vibrio* spp., *Photobacterium damselae* ssp. *piscicida* and *Tenacibaculum maritimum*, from different wild fish species (Vatsos et al., 2007). Some published bibliography exists elsewhere (Snieszko et al., 1964; Burke and Rodgers, 1981; Muroga et al., 1984; Chen et al., 1995; Frances et al., 1997; Hoi et al., 1998; Ferguson et al., 2000; Evans et al., 2002), in Mediterranean area (Yiagnisis et al., 2007) or Red Sea (Diamant et al., 2007).

Overall, there are few documented examples of bacteria exchange between farmed and wild populations. These studies regard *Streptococcus* spp. (Zlotkin et al., 1998; Colorni et al., 2002; Kvitt and Colorni, 2004; Diamant et al., 2007) and *Mycobacterium marinum* (Diamant et al., 2000; Ucko et al., 2002; Ucko and Colorni, 2005).

Isolation of epitheliocystis-like organisms, lymphocystis virus and important parasites such as *Enteromyxum leei* (and other myxosporeans), *Ceratothoa oestroides*, *Amyloodinium ocellatum* and *Cryptocaryon irritans* in wild Mediterranean inhabitants have been notified in unpublished reports (Vatsos et al., 2007) and to a lesser extent in published works (Faisal and Imam, 1990) but they lack demonstration of transfer with few exceptions (Diamant et al., 2005; Mladineo and Marsic-Lucic, 2007). A possible transfer of the myxosporean *Kudoa iwatai* from wild to farmed fish was suggested by Diamant et al. (2005). Similarly, Mladineo and Marsic-Lucic (2007) suspected switch of the monogenean *Lamello-*

discus elegans from wild fish, to farmed gilthead sea bream. Papapanagiotou et al. (1999) attributed farmed European sea bass losses associated with the presence of the isopode *Emetha audouini,* in wild fish populations, but no parallel examination of wild fish was carried out.

Encephalopathy and retinopathy disease caused by nodavirus is associated with serious mortalities in established and new farmed finfish species in the Mediterranean and has raised a great amount of debate as far as the disease spread from farming activities to wild populations is concerned (Rigos and Katharios, 2010). Wild species harboring the pathogen have been blamed and vice versa but there is no adequate evidence to support either statement. Occasional losses among different wild Serranidae species have been noticed during the last two decades in eastern Mediterranean by scientists and sport fishermen even in remote sites located at a great distance from farming facilities. The possible impact between wild and reared population is not fully understood, though wild species appeared to represent relatively a minor concern for infection of cultured fish, while wild stocks may be at higher risk when exposed to the pathogen in the vicinity of infected farming sites.

Within Atlantic bluefin tuna there is no evidence of disease interactions between wild and farmed fish (Mladineo, 2007). Feeding of local fresh or frozen fish to tuna is considered by some to be a risk for disease emergence. Moreover, when the domestic baitfish catch is insufficient, frozen fish are imported and administered to tuna. The systematic dumping into the Mediterranean marine ecosystems of thousands of tonnes of exotic whole fish constitutes a first order environmental threat due to the significant risk of spreading allochtonous diseases to the native fish populations (WWF, 2005) Interestingly, Jones et al. (1997) suspected that two outbreaks of herpesvirus mass mortalities of native pilchard (*Sardinops sagax*) in south Australia were possibly introduced by imported frozen pilchards given to reared tuna (Mladineo, 2007).

The majority of shellfish farming operations is also characterized as open systems and thus pathogen exchange is possible between wild and farmed organisms. More importantly, bivalves as filter feeders may accumulate microbes that may be pathogenic for other species including fish. As vaccination and drug administration via diets is impossible in these animals, prevention and treatment schedules are basically impossible in shellfish stocks. While there has been some evidence for pathogen transfer mainly due to co-existence of the same pathogen (e.g.: Haemocytic infectious virus in Pacific cap oyster, *Vibrio tapetis* in Japanese carpet shell, *Bonamia ostreae* in European flat oyster) in farmed and wild shellfish along the Atlantic cost (reviewed by Mortensen et al. (2007)), in Mediterranean region such data is mainly limited to *B. ostreae* infections to European flat oyster (Pichot, 2002). The disease was actually pre-existed in farmed stocks before being spread into the wild populations. Some suspicions for possible transfer of *Marteilia* spp. from farmed to wild individuals of European flat oyster and European mussel exists but the required evidence is missing (Mortensen et al., 2007).

Overall, as in the other aquaculture regions, the interaction between farmed and wild stocks in terms of pathogen exchange are poorly understood in Mediterranean and there are conflicting views regarding the actual impact on the environment. Given the small amount of available research it can be hypothesized that fish communities seemed not to be seriously affected by the influence of coastal sea-cage aquaculture. There is however, some evidence for the existence of parasite and species specificity of impact. A continued monitoring associated with extensive research will be necessary to accomplish a good understanding of the disease interactions between farmed and wild fish stocks in the Mediterranean region.

2.4. Loading of organic wastes

Feed waste products can be derived from non-ingested feed, non-digested feed components and fish excretions. Uneaten feed is one of the most important factors causing organic and nutrient loading in the vicinity of aquatic farms. The amount of uneaten feed loss in a farming site mainly relies on the skills and experience of the personnel, as well as to the feeding management (automatic feeders vs. hand feeding, use of low pollution diets). Feed wastage can range from 1% to 38% in different types of intensive fish farming (Wu, 1995), while in euryhaline fish species that dominate Mediterranean aquaculture, respective loss is lower (2–9%) (Dorsat, 2001).

A considerable fraction of many feed components such as protein, fat, carbohydrates and other nutrients such as phosphorus (P) which are highly present in the feed, cannot pass the intestinal barriers during digestion and thus are rejected via faecal excretions by the fish. Moreover, in gilthead sea bream it has been estimated that 17%, 52%, 31% and 39% of nitrogen (N), P, organic matter and dry matter, respectively are excreted in faeces (Lupatsch and Kissil, 1998). Fish excretions (kidney and gills) can introduce soluble compounds in the environment. Ammonia (NH_3) is the end product of protein metabolism, is excreted via the gills and represents the major component of N excretion. The remaining is in the form of mainly urea, creatinin and other excretions. CO_2 is also excreted by the gill (accounts for 50% of carbon loss) but has limited impact on the environment (Dorsat, 2001).

In a monitoring of four Mediterranean fish farms producing European sea bass and gilthead sea bream, it was found that the dominant of all waste forms was ammonium (NH_4), followed by particulate N (Holmer et al., 2008). Phosphorus on the other side, exhibited the highest release as fine particulates and less as dissolved phosphate (PO_4^{3-}). It has been estimated that the two aforementioned species release 1.5 times higher N and P than Atlantic salmon (*Salmo salar*) (Islam, 2005; Holmer et al., 2008). This underlines the serious impacts of farming these two species, dominating the Mediterranean aquaculture. Karakassis et al. (2005) in a previous study calculated that the overall N and P offload originated from finfish production in the Mediterranean accounts for a 5% of the total annual anthropogenic discharge in the area Thelong-term consequences can severely affect the Mediterranean biodiversity.

There is a vast amount of literature in regard with the organic wastes deriving from aquaculture and their environmental impact in the Mediterranean area. The existing literature also uses various tools to assess these environmental impacts, such as chemical analyses in the water and sediment, analyses of carbon and nitrogen stable isototopes, microbial indexes, biochemical ones (e.g. chlorophyll α), various macrobenthic ones and mathematical modelling. It is not within the scope of the present study to review this literature, but we will rather try to focus on the most important findings. However, there are some existing papers that mostly review the situation in the Mediterranean and incorporate the majority of the existing literature (Danovaro et al., 2003; Holmer et al., 2008; Claudet and Fraschetti, 2010).

In general, the impact in the water column, by finfish culture, is due to the resolved nutrients. Phytoplankton primary production is induced for at least 150 m downstream in the dominant current direction (Holmer et al., 2008). This stimulation of algal productivity may be attributed to the general Mediterranean oligotrophic conditions.

Most of the existing literature examines the influence of cage aquaculture in the soft sediment benthic communities (Mazzola et al., 2000; Mirto et al., 2000; La Rosa et al., 2001; Danovaro et al., 2003; Holmer et al., 2008). Regarding microbial benthos, it has been shown that bacterial and total microbial density increases significantly in impacted sediments, with prokaryotic auto-fluorescent cells increasing but eukaryotic autofluorescent cells decreasing, while the ratio of total bacterial to meiofaunal biomass increases 3–4 times in impacted areas (Danovaro et al., 2003). In general, the most profound effect of aquaculture wastes on benthic fauna is the reduction of species number (Mazzola et al., 2000; La Rosa et al., 2001) and the differentiation of species composition. Copepods, nematodes and polychaeta dominated the meiofauna of the affected zone (Mazzola et al., 2000; Mirto et al., 2000; La Rosa et al., 2001). Also a distinctive increase in individuals' size was observed for certain species (nematodes) (Mirto et al., 2000; La Rosa et al., 2001).

The development of fouling communities was assessed for two Mediterranean sites, one in Crete, Greece and one in Piran, Slovenia, in order to evaluate the impact of caged mariculture in the sublittoral epibiota (Cook et al., 2006). The epibenthic communities developed, are generally composed by suspension feeders and macroalgae. Caged mariculture, through the organic offloads, enhances the fouling communities biomass particularly in oligotrophic regions and may have a greater community-structure impact in regions of low dispersion, that characterize the Mediterranean.

One of the most studied Mediterranean benthic habitats for the impact of aquaculture is the meadows of Mediterranean tapeweed (*Posidonia oceanic*) (Delgado et al., 1999; Holmer et al., 2008; Apostolaki et al., 2009). This organic waste biomarker is of particular importance because Mediterranean tapeweed is an endemic Mediterranean seagrass that is widely spread in the coastlines. It plays a critical role in the coastal ecosystem, because it consists itself as a major food source for benthic animals, offers shelter from predators, serves as a nursery area for many fish species, has a structure role enhancing habitat complexity by supporting epiphytes and epifauna and modifies the sediment texture and the hydrodynamics of the area (Duarte, 2002). Various and complex effects of fish farms have been observed in Mediterranean tapeweed. The sedimentation of organic material is mostly influential by altering biogeochemical sediment processes. It was found to modify the microbial activity, therefore the nutrient regeneration, and to lead to sediment sulfide increase and subsequently to sulfide invasion into sea grasses and declining of Mediterranean tapeweed. Also additional farming impacts are those of epiphytes loading increase and in some cases increase of sea urchins grazing the meadows. The end-results are increased mortality and reduced recruitment. Furthermore, another result of organic discharge, is the nutrients loss observed from the meadows (Apostolaki et al., 2009). Holmer et al. (2008) suggested that with current environmental conditions shoots decrease by half within 3–26 months and meadow close to the fish cages disappear within 5–11 years.

An extensive study that included six Mediterranean sites, four of them producing finfish such as gilthead sea bream, European sea bass and Atlantic bluefin tuna, one producing mussels and one raising both finfish and molluscs, assessed variable benthic indexes to evaluate the organic offload effects (Borja et al., 2009). The former authors used Partial Redundancy Analysis to explain the macrofaunal variability (individual abundance, species richness, diversity, AZTI's Marine Biotic Index and Infaunal Trophic Index). Three groups of environmental variables were singled out: "hydrography" (depth, distance from farm and average current speed) which explained 11.5% of the variance, "sediment" (Eh and percentages of silt and total organic matter) explained 5.4%, and "cages" (total production time and capacity), explained 15.2% of the total macrofaunal variability. The interactions of these three groups, justified a further 21% of the variability.

Comparing the impact of aquaculture on different sediments, it was found differential response of muddy/silt sediments and coarse ones to organic enrichment, with the former habitat type being more susceptible (Papageorgiou et al., 2010). On the contrary, the coarse sediments showed better reaction to the farm wastes, often exhibiting existence of diversified communities with high abundance and biomass. Thus, they are characterized as more "safe", at least in the context of macrofaunal and geochemical monitoring (Apostolaki et al., 2007; Papageorgiou et al., 2010). This raises questions on the politics of moving sea cages offshore, where, in the Mediterranean, the sediments compose of silt.

In a recent study, Sanz-Lázaro et al. (2011) discovered the influence of Mediterranean sea farming, not only in physico-chemical and biological aspects, but also from a trophic point of view, affecting the grazers and the balance of the trophic groups. The impact on the benthic community was found to extend further to the sedimentation zone. Thus, the former authors concluded that taking into account data obtained only from waste dispersion rates leads to underestimation of the fish farm impact.

It should be also noted that effects may differentiate based on the type of the affected environment, with lagoons and coastal environments being more complex environments, supporting higher biodiversity, but also more susceptible to fish farming wastes (Mantzavrakos et al., 2007; Munari et al., 2010).

The general environmental implications related to off shore fish farming are discussed in detail in the recent review of Holmer (2010). Ruiz et al. (2010) clearly indicated that waste from offshore farms can have large areas of influence (over kilometers), thus affecting Mediterranean tapeweed meadows in remote near-shore locations. Of course, these effects are far from sufficient to produce significant meadow alterations, but changes in the abundance of epiphytes and possibly in grazers activity have been attributed to this farm-originated nutrient induction. This also raises objections to the claim that transport of aquaculture facilities into off-shore areas is a strategy that clearly eliminates environmental risk.

Although the vast majority of literature refer to European sea bass and gilthead sea bream farming, recently also the Atlantic bluefin tuna farming effects related to wastes were investigated through monitoring (Vezzulli et al., 2008; Aksu et al., 2010). The main difference in tuna fattening is that it is based on bait fish instead of commercial diets. Although someone would expect a greater environmental impact in such farming activity due to higher reared biomass, high feed conversion ratios, higher dissolved nitrogen and total phosphorus outflow (Aguado-Gimenez and Garcia-Garcia, 2005; Aguado-Gimenez et al., 2006; Vezzulli et al., 2008) studies have shown that organic impact is of a lesser degree (Vezzulli et al., 2008; Aksu et al., 2010). This has been attributed to the controlled feeding, the currents and the location of the cages (off shore exposed sites, high depths, strong hydrodynamic regimes).

Beyond the impacts of organic off-load in water column and sediment, the ability of the environment to recover after cessation of aquaculture has been also studied. Rapid initiation recovery process has been mentioned, but far from complete even after a cessation period of several months (Mazzola et al., 2000; La Rosa et al., 2001; Pereira et al., 2004) and also site dependable (Pereira et al., 2004; Sanz-Lázaro et al., 2011).

Less studied than the cage-culture impacts, is the effect of shellfish culture on the Mediterranean environment. Shellfish are filter-feeders and their potential organic pollution effects are limited to their excretion-originated biodeposition. Mesnage et al. (2007) and Dedieu et al. (2007) have related the increased organic matter in the sediment and altered oxygen fluxes in a French Mediterranean lagoon (Thau), with the activity of oyster farming. In another lagoon in NW Mediterranean, the Salses-Leucate lagoon, with allowance of permanent exchange of sea water, Carlier et al. (2009) indicated that oyster biodeposition did not affect the trophic pathway of the underlying benthic macrofauna, and they explained that as a result of the weak benthic–pelagic coupling occurring in this specific lagoon.

For the coastal area of Tyrrhenian Sea, mussel farming was shown to enhance local microbial assemblages and particularly cyanobacteria, while other organisms such as picoeukaryotic cells, turbellarian, ostracod and kinorhynch are strongly decreased (Mirto et al., 2000). Obviously, comparisons between fish farming and mussel farming clearly showed that the latter induced a considerably lower benthic community disturbance (Mirto et al., 2000).

The important aggregation of mucus due to bivalve biotic functions has also been studied in the NW Mediterranean coastal area (Cornello et al., 2005). Sedimentation, accumulation and subsequent degradation of high amounts of mucous material can result to benthic hypoxia. However, the former scientists showed no direct effect form mucous aggregates in the soft-bottom macro-zoobenthos, although mucus showed some adverse affects in the mussel growth.

All previous studies refer to coastal areas. In a study of the impacts of suspended bivalve culture on open-sea (Fabi et al., 2009), there were indications of only minimal detrimental effects on the zoobenthic communities.

Important work has been done in respect to the hypothesis that integrated multi-trophic aquaculture can be ecologically beneficial. According to this, including simultaneous farming of low trophic level organisms, like bivalves, together with the fish farms, would act as an ecological engineering practice. In specifically, through the recycling of the fish waste by the low trophic level species, the fish waste impacts would be reduced and the total productivity would enhance as a result of the gained biomass (Troell et al., 2003). Sarà et al. (2009) confirmed the above hypothesis by showing enhanced growth of mussels close to fish farms and enhanced mussel fouling. However, another recent study (Navarrete-Mier et al., 2011) contradicts these findings, by showing that proximity to the fish farm did not enhance bivalve growth, that there was no relationship between the main input of fish farm organic matter and the trophic behavior of the bivalves, and that neither European flat oyster nor European mussel actually incorporated the fish farming wastes. Thus, the former scientists suggested that polyculture of fin fish and bivalves does not seem to constitute an appropriate tool for the reduction of fish farm environmental impact.

The aquaculture-originated organic offloads have been proved to cause non-negligible environmental effects. These seem to be in general localized, although the total impact is dependent on many different factors, such as the aquaculture production characteristics (cultured species, capacity, feeding management, etc.), the water column hydrological conditions (depth, currents, temperatures, etc.) and the nature of substrate (fine, coarse) and impacted community and the site (open sea, coastal area).

2.5. Release of chemicals

2.5.1. Metals

Various metals are naturally present in the aquatic environment via geochemical processes and anthropogenic industrial sources, where they can be accumulated along the food chain. The vicinity of cage farms and particularly the sediment can be further enriched with trace metals derived from unutilized feed, fish excreta and antifouling chemicals used to prevent development of fouling organisms in the nets (Basaran et al., 2010). It is well accepted that the main source of metal pollution in sediments under fish cages are fish diets which are supplemented with various metals to fulfill perceived mineral requirements (Sapkota et al., 2008).

Heavy metals such as zinc (Zn), copper (Cu), iron (Fe), cadmium (Cd), lead (Pb) and nickel (Ni) have been measured in the water column and sediment beneath cage sites from eastern Mediterranean (Belias et al., 2003; Basaran et al., 2010). Other studies have reported accumulations of heavy metals in sediments accounted for aquaculture activities outside the Mediterranean (Chou et al., 2002; Mendiguchia et al., 2006; Sutherland et al., 2007). For the occurrence of particular metals such Fe and Zn in the sediment, a correlation with organic matter accumulation has been demonstrated (Basaran et al., 2010).

To our knowledge, no EU regulations are available regarding metal concentration in sediments. Basaran et al. (2010) compared the mineral concentrations found to their study with the guideline values recommended by the Canadian Council of Ministers of the Environment (CCME, 1995). Concerns were only limited to Cu concentrations in some sampling stations which were estimated to be between guideline values and the probable effect level. Nevertheless, the authors suggest that based on their findings metals concentrations in the sediments have yet to reach dangerous levels for the aquatic environment. However, further investigation is required to obtain a complete picture before drawing final conclusions.

2.5.2. Organochlorine compounds

Organochlorine compounds originated from fish feeds such as polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs) and dichlorodiphenyltrichloroethane (DDT) have been found in sediments beneath salmon farms (Hellou et al., 2005). These findings highlight the potential danger for their accumulation in wildlife. The health implications associated with the accidental consumption of these unwanted compounds are discussed later in this review. Additional work is urgently needed to gain a thorough understanding of the possible presence of halogenated compounds in sediments below cage farms in the Mediterranean and their possible bioaccumulation in primary and secondary aquacultured items.

2.5.3. Antibacterial agents

Antibacterial therapy still remains the last resort to combat bacterial fish infections at least for bacterial pathogens for which no effective prevention exists. Despite optimization of therapeutic strategies, significant quantities of drugs may be released into the vicinity of the fish farms from different routes (Rigos and Troisi, 2005). This is via non-ingested medicated pellets, unabsorbed drug in the feaces, unprocessed parent drug from renal and gill excretions and drug metabolites following renal and faecal excretion (Rigos et al., 2004). For example, using information from absorption studies of oxytetracycline (OTC), one of the most commonly used drugs, 27–74% of total orally administered dosage to euryhaline farmed fish species may be passed unabsorbed from fish to the environment via faecal loss (Rigos et al., 1999, 2002). Similarly, it has been estimated that total antibacterial loss into the surrounding environment from all pathways may reach as high as 75% of the administered dietary dosage in farmed fish (Richardson and Bowron, 1985; Halling-Sørensen et al., 1998; Lalumera et al., 2004).

Antibacterial drugs used in aquaculture have been shown to persist in water and sediments in the vicinity of euryhaline fish farms, sometimes long after their use has ceased (Jacobsen and Berglind, 1988; Capone et al., 1996). The persistence of drugs released from fish farms prolongs their life in the water column and sediment and allows them to enter the aquatic food chain leading to contamination of non-target organisms. Upon release into water, abiotic degradation of antibacterials, mainly by photolysis can take several days or weeks (Oka et al., 1989; Samuelsen, 1989). Antibacterial drugs, particularly OTC, are easily partitioned into organic matter of aquatic sediments and suspended particulates (Rabolle and Spliid, 2000). Several antibacterial drugs are relatively resistant to biodegradation in cold water surface sediments (Hektoen et al., 1995). Fortunately, degradation of these compounds tends to be more extensive in Mediterranean euryhaline environments where temperature, salinity and light intensity are higher than in cold water mariculture, and thus degradation is higher (Oka et al., 1989; Samuelsen, 1989; Doi and Stoskopf, 2000). However, determination of abiotic and biological degradation rates for antibacterials, specifically under warm-water euryhaline conditions, is necessary to fully comprehend their overall persistence in sediments and water around Mediterranean fish farms.

One of the major consequences of drug pollution from aquatic farms is the development of antibiotic resistance in fish pathogens, which reduces their therapeutic value for fish therapy and is extensively discussed later on. Data on the toxic doses of antibacterials for aquatic wildlife are available only for few species, and limited to short term test data not providing information on long-term exposure toxicity of antibiotics. Furthermore, the influence of higher water temperature, salinity and light availability found around coastal Mediterranean fish farms on ecological effects of antibacterials has not been studied yet. In the case of antibacterial phytotoxicity, cyanobacteria are an order of magnitude more sensitive to toxic effects of these drugs than eukaryotic green algae (Holten Lutzhoft et al., 1999; Halling-Sørensen, 2000) and moreover, higher plants may be less sensitive to the toxicity of antibiotics than prokaryotic algae (Migliore et al., 1997).

Given the limited available toxicity data, it would appear that water and sediment drug levels beneath and around fish farms reported to date, are unlikely to be adequate for causing effects in wildlife (Rigos and Troisi, 2005). For example, even the maximum reported concentration of flumequine (0.6 $\mu g\ kg^{-1}$ d.w.) in sediments from the vicinity of Italian fish farms reported by Lalumera et al. (2004) is well below the median effective concentration (EC50) value for of flumequine analyzed for brine shrimp (Migliore et al., 1997). However, the possible non-lethal effects of antibacterial agents with low exposure over time on aquatic biota should be further appreciated.

2.5.4. Other chemicals

Previously a list of the various chemicals used in EU aquaculture was discussed by Costello et al. (2001). According to this review no anaesthetic or antiparasitic compound is registered for aquaculture use in the EU Mediterranean countries. An updated literature search seemed to verify this fact (www.emea.europa.eu). There is no available data on the possible contamination of water and sediments beneath Mediterranean fish cages with illegal pharmaceuticals.

2.6. Use of wild resources for aquaculture needs

One of the major issues in the aquaculture sustainability is the use of wild resources for the production of farmed organisms. This includes the use of wild fish populations as stocks for aquaculture purposes, the utilization of wild fish for fish meal and fish oil production and the wild fisheries direct use for cultured fish feeding. Beveridge (2001) suggested that marine fish farming is more dependent on wild stocks due to their more complex life circles when compared to freshwater fish. In general, new species culture is initially wild resources-dependent when initiated due to the lack of know-how and the inability for completing the cycle. As soon as the know-how for a certain species culture develops, the dependence on wild recourses decreases. Also, the achievement of better feeding techniques, of more species-adopted feeds, and thus better growth and utilization indexes, leads to reduced fishmeal and fish oil utilization for the respective farming activities.

2.6.1. Capture of wild stocks for rearing

The ideal way of aquaculture operation should compensate or even enhance wild populations and strengthen the maintenance of biodiversity. Nevertheless, aquaculture in some occasions seemed to increase the pressure already existed by capture fisheries on wild stocks inhabited the Mediterranean. For example, capture-based aquaculture activities such as shellfish farming have totally been depended for decades on the collection of spat and juveniles from the wild. Finfish broodstocks are continuously renewed and enhanced from wild stocks for hatchery needs. Due to the relative inability of reproduction in captivity, finfish seeds for certain new marine species adopted for farming at pilot scale such as greater amberjack have been collected from the environment in some areas, although recent technologies for maturation, spawning and weaning have been developed (Papadakis et al., 2008).

Probably the most serious concerns have been issued for Atlantic bluefin tuna stocks, which are continuously depleted for rearing in the Mediterranean area. Although considerable progress has been made in tuna propagation in captivity, its complete cycle production has not yet been accomplished and consequently its rearing is continuously based on regular collection of large or small specimens for either fattening (rearing for 1–7 months: Spain and other countries) or farming (rearing for >20 months: Croatia) (International Commission for the Conservation of Atlantic Tuna – ICCAT; http://www.iccat.es/ffb.htm). This has put a lot of pressure on wild stocks and the obvious environmental implications of these activities have already emerged and criticized (Simard et al., 2008). The fact that the Mediterranean basin is one of the two spawning grounds of Atlantic bluefin tuna globally has also strengthen the debate (Ottolenghi, 2008). FAO (2005) described the farming of Atlantic bluefin tuna in the Mediterranean Sea as an activity clearly overlapping with fisheries, and due to the continuous spawning stock decline, a possible collapse of the wild population was evident (Ottolenghi, 2008). Consequently, in the context of sustainable increase of tuna farming, all attempts to achieve a real regional management of this key Mediterranean fish resource have resulted in failure, leading recently to reconsider the banning of tuna trade ban (Europeanvoice.com 2010).

There is also pressure into the wild fry of the Mediterranean population from in land farming operations. Most of the flathead grey mullet (*Mugil cephalus*) fry used in commercial landed aquaculture, representing over 100 000 tonnes yearly production, is almost relying upon collections from the wild (FAO, 2006). The same is also evident, for decades, for European eel (*Anguilla anguilla*) farming. Eel represents a case for which direct interactions between aquaculture and fisheries exist. It is well known that the main limiting factor for eel aquaculture is the necessity of capturing wild seed. Induced spawning of this species is to be considered out of reach, at least for the short term, despite the amount of research dedicated on the reproductive biology of eels (Ciccotti, 2005). In general, a discouraging trend evidenced for global yields coincides with the apparent decline in recruitment reported in Mediterranean region (Ciccotti, 2002).

Therefore, as a principle for sustainable growth, the stocking of aquaculture farms should not affect the natural status of wild populations. It is preferable that organisms that are to be raised in aquaculture farms should have been produced in artificial environments. For this purpose, research on reproduction and hatchery programs should be further encouraged to accomplish full domestication of particular species. Although considerable progress has been made in this regard, the economically feasible "closed cycle" production of some candidate species with great farming potential has not yet been achieved. Nevertheless, in special circumstances, where scientific control is applied, using wild seeds for aquaculture purposes could be in some occasions a sustainable alternative (Ottolenghi et al., 2004).

2.6.2. Depletion of fish stocks used as processed fish feed

The total proportion of world fish production that is used for fish meal and fish oil production was 20.8 million tonnes according to the latest official reports, accounting for the year 2008 (FAO, 2010). The produced fishmeal and fish oil, utilized as feeds for aquaculture purposes in the Mediterranean, mainly derive from the Pacific coast of South America and additionally from the north-eastern Atlantic and the North Sea (IUCN, 2007). The main species that are processed into fish meal and fish oil are anchovy (*Engraulis ringens*), jack mackerel (*Trachurus symmetricus*), Atlantic horse mackerel (*Trachurus trachurus*), lesser sand eel (*Ammodytes tobianus*), sprat (*Clupea sprattus*), blue whiting (*Micromesistius poutassou*), capelin (*Mallotus villosus*) and herring (*Clupea harengus*) (Tuominen and Esmark, 2003; FAO, 2010).

There are no exact data for the fish meal and fish oil quantities that are consumed within the Mediterranean aquaculture. However, a rough estimation of the total consumption of these two raw materials for Mediterranean aquaculture production can be made. Based on the production facts for the basic Mediterranean fish species (Fig. 1), the average calculated feed conversion ratio (FCR) of 2:1 and the typical inclusion of fish meal and fish oil in the diet (Barazi-Yeroulanos, 2010), the total annual use for the dietary needs of the main Mediterranean farmed marine fish species is calculated to be as high as 150 000 and 100 000 tonnes of fish meal and fish oil, respectively (Fig. 2). Considering these high values it is understandable why the use of wild fish to feed farmed aquatic animals is a global issue guiding recent research efforts to discover alternative resources usually from terrestrial plants and marine sustainable resources (Naylor et al., 2009). A vast amount of literature exists on this subject, but it is not within the present scope to review it. These sustainable alternatives appeared to be effective to some degree with questioning results with regard to overall performance of the fed fish (Grigorakis, 2010).

2.6.3. Direct use of fisheries catches for feeding of cultured fish

While occasionally broodstock of marine farmed fish in the Mediterranean are fed with locally-caught bait fish, the largest amount of fisheries catches is used for the rearing of Atlantic bluefin tuna. During captivity, farmed tuna are fed on an exclusive basis, large quantities of a variety of whole bait fish including sardine (*Sardinella aurita*), pilchard (*Sardina pilchardus*), round sardinella (*S. aurita*), herring (*C. harengus*), mackerel (*Scomber japonicus*), bogue (*Boops boops*) and squid (*Illex* sp.) (Ottolenghi, 2008). Mediterranean countries involved in Atlantic bluefin tuna ranching obtain bait fish from locally fished stocks or from imports stocks from outside the region, with the latter usually representing the largest proportion of the fish used by the industry (Ottolenghi, 2008).

It has been estimated that approximately 225 000 tonnes of bait fish were used as diet for 25 000 tonnes of captive Atlantic bluefin tuna in Mediterranean during 2004, using an average feeding period of 180 d and a daily feeding ration of 5% b. w. (WWF, 2005). The high volume of baitfish required to sustain tuna farming stressed an urgent need to further develop artificial tuna diets. These diets will support a more controllable feeding strategy and eliminate the direct impact on fisheries stock (Ottolenghi et al., 2004). It is however important to note that regardless the high production costs, artificial feeds for tuna have been recently produced by fish feed industries, but the Japanese market, the main consumer of farmed Atlantic bluefin tuna, showed reduced acceptability for this product considering possible adverse effects on taste of raw meet (Ottolenghi, 2008). This negative consumer perception has made also farmers more reluctant to the adoption of feeding strategies including pellets.

2.7. Other ecological interactions

The aquaculture installations and the aquaculture farms operations can also drive other ecological alterations. These can be due to prey-predation relationships of cultured species with wild ones, due to the aggregation of various wild species in the area, but also due to alterations of water and general habitats characteristics due to the physical presence of aquaculture installations. There are indications that also other ecological characteristics may differ between wild and cultured fish, such as the swimming ability and therefore the ecological performance, as shown for European sea bass (Handelsman et al., 2010).

2.7.1. Prey predation interactions

The floating sea cages dominate the Mediterranean as an aquaculture method. There are particular references about predators attracted by the cultured organisms or the wild fish populations around the farms in the Mediterranean. These predators include finfish, like bluefish (*Pomatomus saltatrix*) (Sanchez-Jerez et al., 2008) monk seals (*Monachus monachus*) (Guclusoy and Savas, 2003), dolphins (Diaz Lopez and Bernal Shirai, 2007) and seabirds (Melotti et al., 1993). They are quite often trapped in aquaculture nets, annoyed due to the existence of anti-predator measures, or, even killed by the farmers sometimes. Aquaculture has been proved to have also indirect impacts on the behavior of predator animals and social structure of their communities (Diaz Lopez and Bernal Shirai, 2007). The predators are indirectly forced to attack the installations, due to the scarcity of their natural preys as a result of human activities (even aquaculture itself), and directly forced out of their food source due to the anti-predator measures. This impact to the natural predators can have a dramatic dimension for the Mediterranean ecosystem, considering that some of these predator species, like the monk seal, are endangered species (Grigorakis, 2010).

2.7.2. Wild fish aggregations

There is a confirmed interaction between sea cages and wild fisheries in the Mediterranean (Dempster et al., 2002). In this

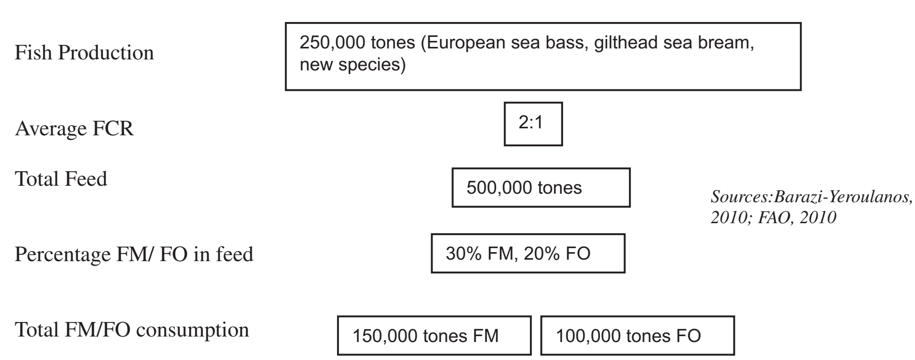

Fig. 2. Estimated fish meal (FM) and fish oil (FO) consumption for Mediterranean fin fish mariculture.

oligotrophic region, aquaculture enterprises lead to a significant increase in wild fisheries landings in the respective areas, where farms operate (Machias et al., 2006; Akyol and Ertosluk, 2010). This has been possibly attributed in addition to an ideal nursery ground also to either direct consumption of lost pellets by wild fish, or to the nutrients-derived increase in primary production, or to the direct transfer of nutrients in the food web (Machias et al., 2006). Aggregation of wild fish communities in the vicinity of aquatic farms in Mediterranean can be variable (Table 3). Furthermore, sea cages act are also nursery habitats for various fish species, thus facilitating the settlement of their juveniles (Fernandez-Jover et al., 2009). The aggregated wild fish have been found to consume a large proportion of the total sedimentation nutrients, accounting for up to 80% in the Mediterranean (Vita et al., 2004), consequently facilitating a reduction of the impact on benthic systems by reducing nutrients sedimentation (Sanz-Lázaro et al., 2011). However, wild fish have been found to increase the nitrogen and carbon loads into the water column through feacal leaching, thus affecting the pelagic system and the wastes dispersion (Fernandez-Jover et al., 2007a). Additionally, due to the reduced waste sedimentation, some effects of the wild fish to the benthos have been noticed. In a very recent study in an open sea Mediterranean fish farm, Sanz-Lázaro et al. (2011) found that wild fish-driven low sedimentation rates reduced in turn the effects of wild fish in the benthos, thus allowing benthos changes. Since these impacts are not quite clear yet, there should be further investigation on them.

Further to the ecological effects of the wild fish aggregation near the sea cages, there is an impact on the biology and the physiology of wild fish themselves. Thus, they become farm effluents feeders by feeding on commercial pellets and subsequently altering their fat deposition and their fatty acid composition (Fernandez-Jover et al., 2007b).

Besides the obvious effects by the wild fish aggregations near the cages, it has been shown that fish farms operating in oligotrophic areas like the Mediterranean may have a positive influence to the wild fish assemblages, extending beyond those species that directly relate to the fish cages, increasing the fisheries resources without reducing total biodiversity (Machias et al., 2004).

2.7.3. Habitat alteration

Although little is known about the exact impacts, direct changes in natural environment due to the presence of aquaculture installations have been mentioned. These refer to aspects of water movement reduction which also leads in changes of water physicochemical parameters (Neofitou and Klaoudatos, 2008). The Mediterranean fish and shellfish farming, utilizing water-based production systems, in general, do not pose important threat in terms of habitat alterations (Beveridge, 2001). However, some adverse affects have been mentioned, in aspects of important wild life habitat losses. Beveridge (2001) referred to the examples of mussel farming in the vicinity of an ecologically-protected area in Greece, and of coastal salt-pond ("salinas") utilization in France and Spain, that have led to disturbance of wildlife and to loss of feeding habitats.

On the other side, there is a long existence of extensive culture forms for European sea bass, sea breams, flathead grey mullet and European eel with no reported adverse affects. On the contrary, these lagoon systems mostly based in Italy and Greece have been used as valuable feeding and breeding habitats for wild life.

Conclusively, the implications of aquaculture to wild organisms, by aggregations, of either predators or other various fish species, and by direct habitat alterations can vary, they are not necessarily negative, but are far from negligible.

3. Public safety

3.1. Development and transfer of antibacterial resistance

As in other animal production sectors, antibacterial agents are used in aquaculture mainly to treat bacterial diseases. Antibacterials may not be always used in a responsible manner in aquaculture due to the urgency of the farmer's response to an outbreak. This may often result to an ill-informed decision-making based on a rushed diagnosis and possible use of inappropriate drugs and dosing regimens.

Legislation enforcing the use of chemotherapeutics varies among countries throughout the world (Serrano, 2005). International organizations, including the Food and Agriculture Organization (FAO), the World Health Organization (WHO), the Inter national Office of Epizootics (OIE) and more particularly, regulatory bodies in Europe as the Committee for Veterinary Medicinal Products (CVMP), have all raised issues associated with heavy, improper and illegal use of antibacterials in animal production sectors, with particular concern for potential risks to public health.

One of the most hazardous consequences of improper drug usage in aquaculture medicine is the development of antibiotic resistance in fish pathogens which initially reduces their therapeutic value for fish therapy. The occurrence of resistant bacterial populations in sediments beneath and around marine fish farms is well documented (Samuelsen et al., 1992; Kerry et al., 1994, 1996; Capone et al., 1996; Alderman and Hastings, 1998; Angulo, 1999; Schmidt et al., 2000; Miranda and Zemelman, 2002; Furushita et al., 2005; Serrano, 2005). In Mediterranean region, however, studies investigating bacterial resistance under marine cages are very limited (Chelossi et al., 2003; Giraud et al., 2006). Chelossi et al. (2003) reported high incidences of quinolone, tetracycline and penicillin – resistant bacterial populations in sediments from the vicinity of Ligurian Sea coastal fish farms. Giraud et al. (2006) demonstrated a selection towards oxolinic acid resistance in the intestines of fish subjected to treatment, while no clear evidence of resistance process was apparent either in bivalves or in sediments.

Bacterial pathogens resistant to drugs (*Vibrio* spp., *Pseudomonas* spp. and *Aeromonas* spp.) have also been isolated in addition to diseased farmed fish in Mediterranean region (Bakopoulos et al., 1995; Zorilla et al., 2003; Smith and Christofilogiannis, 2007), from wild fish species captured near fish farms outside of the Mediterranean (Ervik et al., 1994; Miranda and Zemelman, 2001; Castro-Escarpulli et al., 2003; Kim et al., 2004). A high incidence of resistant bacteria was also reported in molluscs collected from areas close to a fish farm (Samuelsen et al., 1992). The above findings reveal that wild inhabitants of farming sites subjected to medication can also act as a reservoir of bacterial resistance.

Development and transfer of bacterial resistance as a result of antibacterial use in aquaculture has received a great amount of criticism addressed in numerous reviews (Smith et al., 1994; Alderman and Hastings, 1998; Angulo, 1999, 2000; Angulo et al., 2004; Cabello, 2004, 2006; Sapkota et al., 2008; Rigos et al., 2010). Growing concerns are related to the fact that the use of antibacterial drugs is likely to select for resistance bacteria that are not fish pathogens, representing thus a direct pathway of resistance from aquatic practices to humans via improperly cooked aquaculture items. Antimicrobial-resistant *Salmonella* strains have been actually detected in imported foods, primarily of seafood origin (Zhao et al., 2003).

It is unfavorable that, although limited in number, some antibacterials commonly used in aquaculture therapy are also employed in human medicine (Serrano, 2005). Inevitably, the use of antibacterials in the aquaculture industry can favor the occurrence of antibacterial resistance in other therapeutic fields. Antibacterial resistance emerges through mutations in bacterial DNA or via horizontal gene transfer mechanisms including conjugation with other bacteria, transduction with bacteriophage and the uptake of free DNA via transformation (Casas et al., 2005). It has been indicated that dissemination of antibacterial resistance may be facilitated by the horizontal transfer of R plasmids between related and diverse bacteria (Wantanabe et al., 1971; Aoki, 1988). Unfortunately, these resistance genes have no phylogenetic, geographical or ecological barriers (OIE/FAO/WHO, 2004). Thus, fish pathogens which may have developed acquired resistance as a result of drug usage and the continuous presence of residual levels in the fish body, can act as a host for resistance genes that can be transferred to human pathogens even moving from one producing country to another (OIE/FAO/WHO, 2004). This is considered as an indirect transfer of resistance from aquaculture to human medicine caused by horizontal gene transfer and has been demonstrated also *in vitro*. For example, plasmids carrying resistance determinants have been transferred from fish to human pathogens including *Vibrio cholerae* (Aoki, 1988), *Vibrio parahemolyticus* (Nakajima et al., 1983) and *Escherichia coli* (Son et al., 1997). Interestingly, Rhodes et al. (2000) provided evidence to claim that aquaculture and human compartments (hospital effluents) may behave as a single interactive environment, supported by the fact that related drug resistance-encoding plasmids were disseminated between different *Aeromonas* species and *E. coli* and between the human and aquaculture environments in distinct geographical locations. Research focused on bacterial resistance determination and possible transfer in consumers is also lacking in Mediterranean aquaculture. The need for surveillance targeting bacterial resistance is urgent to ensure safety of aquaculture products.

3.2. Possible contaminants in aquacultured fish items

3.2.1. Antibacterials

An extensive overview of the registered antibacterial agents for aquaculture purposes in European Mediterranean countries have been given by Costello et al. (2001) and Rigos and Troisi (2005). It is therefore not in the scope of the present review to further elaborate on the list of legal drugs used in Mediterranean aquaculture but rather to focus on their possible implications.

Guidelines such as the acceptable daily intake (ADI), maximum residue levels (MRL) and withdrawal times have been established by EU regulatory bodies to protect the consumer from side effects resulting from improper use of antibacterials and other chemicals. Although most Mediterranean countries are legally forced to perform continued inspection efforts for both exported and imported aquaculture items with reference to unwanted residues, relative published information for local products is missing, though such data possibly exists on national recording systems.

The presence of residual antibacterials in commercialized farmed aquatic products and the associated risks to the consumer have been the subject of several investigations (Johnston and Santillo, 2002; Cabello, 2003, 2004; Angulo et al., 2004; Hastein et al., 2006; Rigos et al., 2010). The potential accumulation of drug residues in the aquatic food chain leading to direct or indirect exposure of biota around farm sites represents another hidden threat. Secondary farmed items including mainly harvested scavengers and wild fish destined for human consumption, have been found to be contaminated with antibacterial drugs to levels beyond the safety factor (Bjorklund et al., 1990, 1991; Ervik et al., 1994; Capone et al., 1996; Coyne et al., 1997). For example, the above studies revealed that concentrations of various drugs commonly used in aquatic farms such as OTC, oxolinic acid and flumequine ranged from 0.1 to 12.5 $\mu g\ g^{-1}$ in tissues of wild fauna (mussels, crabs, fish) inhabiting the adjacent areas of farming sites outside the Mediterranean.

In terms of human health, low-level exposures to antibiotic residues present in food are not likely to cause acute toxic effects among the general public (Jones et al., 2004), except in the cases of allergic reactions which are generally difficult to monitor and consequently underestimated. Chronic effects which are more likely are also largely unstudied. The risks of chronic exposure to antibacterial residues in human organism, with particular reference to intestinal microflora, have been stressed by several authorities (FDA, 1993; CVMP, 1995).

Residual drugs in aquacultured products may be degraded by cooking procedures to some degree (Dehai et al., 1996; Uno et al., 2006) and inactivated to a variable extent in the intestines by decomposition of bacterial enzymes faecal components. However, the possibility of inducing alteration of the human intestinal flora and disruption of the colonization barrier remains (Edlund and Nord, 1993; Cerniglia and Kotarski, 1999). Side effects of residual drugs on human intestinal microflora can be obviously more apparent during therapeutic applications where targeted tissue levels are high; however, prolonged accidental exposure to antibacterial residues may also disturb the equilibrium of the gut microflora. The associated consequences include changed population density and composition, altered enzyme activity for the metabolism of endogenous and exogenous substances and impaired colonization resistance (Cerniglia and Kotarski, 1999). Increased susceptibility to infection by exogenous potential bacterial pathogens, colonization of new resistant pathogenic bacteria and outgrowth of indigenous opportunistic inhabitants represent the most serious implications (Nord and Edlund, 1991; Cabello, 2004).

Continued monitoring for drug residues in aquacultured items should be obligatory by national authorities in Mediterranean region coupled with formal education of fish producers and obligatory implementation of associated food safety procedures.

3.2.2. Organochlorine compounds

Farmed organisms can be contaminated by potential exposure to various organochlorine compounds resulting in most of the cases from the consumption of feed. These compounds are mainly represented by dioxins (polychlorinated dibenzodioxins: PCDDs and polychlorinated dibenzofurans: PCDFs), PCBs, PAHs, hexachlorobenzenes (HCBs), polybrominated diphenyl ethers (PBDEs) and DDT. Their health implications associated with potential human exposure have been extensively criticized (International Agency for Research on Cancer, 1997; Darnerud et al., 2001; Faroon et al., 2001; Pompa et al., 2003; Steenland et al., 2004; Gomara et al., 2005; WHO, 2006). Organochlorines as lipophilic compounds are highly persistent contaminants which are widespread in the environment and may bioaccumulate in the food chain. They are among the primary concerns of the European public on account of their serious and multiple environmental and and human health effects. Actually, dietary intake of marine animals has been claimed as the most important entrance of these compounds to consumers. The higher fat content of aquacultured organisms (Grigorakis, 2007) indicates that farmed-raised fish have a larger reservoir for absorption of lipophilic environmental contaminants compared to their wild counterparts.

Several studies have documented the occurrence of various organochlorines in aquaculture items by providing useful comparisons between wild and farmed fish counterparts in the Mediterranean (Table 4). The pertinent literature has actually provided evidence that farmed fish species including European sea bass (Antunes and Gil, 2004; Lo Turco et al., 2007) and white sea bream (*Diplodus sargus*) (Ferreira et al., 2008) are more contaminated than wild fish. On the contrary, farmed gilthead sea bream from the Spanish Mediterranean Coast appeared to be less polluted than wild counterparts probably due to the low contamination level of the used diet (Serrano et al., 2008; Blanes et al., 2009). Contradicting comparative data have been reported for Atlantic bluefin tuna from Mediterranean (Vizzini et al., 2010).

In particular, Antunes and Gil (2004) and Lo Turco et al. (2007) reported considerably higher levels of total PCBs and DDTs in the edible tissue of farmed European sea bass from Portugal and Italy, respectively, compared to wild-caught counterparts. The total PCBs and DDTs levels were measured to be higher in muscle fat of various sizes of farmed white sea bream compared to wild fish (Ferreira et al., 2008). On the contrary, the same organochlorines appeared to be lower in the muscle of farmed vs. wild gilthead sea bream (Serrano et al., 2008). In farmed Atlantic bluefin tuna muscle, total PCBs and *p.p′* DDE (DDT metabolite) concentrations expressed relative to lipid content, reached lower values compared to those measured in wild tuna while HCB levels were higher in farmed individuals (Vizzini et al., 2010).

Not surprisingly, fish diets have been blamed as the main source of organochlorine pollution in farmed fish tissues possibly due to the high concentrations found in the raw materials used (Easton et al., 2002; Antunes and Gil, 2004; Hites et al., 2004; Carlson and Hites, 2005). For example, in Atlantic salmon diet, concentrations of DDT, PCBs and PBDEs were measured to be 36, 66 and 10.9 ng g^{-1}, respectively (Easton et al., 2002; Hites et al., 2004) while gilthead sea bream diets appeared to be less contaminated regarding PCBs (<7.9 ng g^{-1}) (Serrano et al., 2008).

EU regulation has not yet set specific limits in terms of absolute concentrations allowable for organochlorine compounds for fish (Vizzini et al., 2010). Regarding dioxins and PCBs however, the EU Scientific Committee on Food adopted opinions (OSCF, 2000, 2001) on dioxins and dioxin-like PCBs in food establishing a tolerable weekly intake (TWI) of 14 pg World Health Organisation toxic equivalent (WHO-TEQ) kg^{-1} body weight for both substances. Importantly, Commission Regulation (2006) has also determined that each congener of dioxins or dioxin-like PCBs exhibits a different level of toxicity and thus, in order to be able to sum up the toxicity of these different congeners, the concept of toxic equivalency factors (TEFs) has been established to facilitate risk assessment and regulatory control. This means that the analytical results relating to all the individual dioxin and dioxin-like PCB congeners of toxicological concern are expressed in terms of a quantifiable unit, namely toxic equivalent (TEQ). Vizzini et al. (2010) and Padula et al. (2008) compared the concentrations of the organochlorine compounds found in farmed Atlantic and southern Australian bluefin tuna (*Thunnus maccoyii*) respectively, with the maximum residue values existing elsewhere including Japan (Ministry of Health, Labour and Welfare; 500, 3000 and 100 ng g^{-1} w.w. for PCBs, p,p-DDE and HCB, respectively), Australia (Australian Competent Authority, Food Standards Australia, New Zealand; 500, 1000 and 10 ng g^{-1} w.w. for PCBs, p,p-DDE and HCB, respectively) and USA (FDA; 2 μg g^{-1} w.w. for PCBs). According to these regulatory limits, the concentrations of organochlorine compounds found in the edible tissues of farmed fish in the pertinent literature seemed to be within the safety margin. Nevertheless, the establishment of specific limits for residual organochlorine substances in the edible tissues of aquatic farmed organisms should be a priority at EU level.

3.2.3. Metals

Where cage farming exists, in addition to the natural presence of metals in the aquatic environment from geochemical and anthropogenic processes, excessive sources of metals may be resulted from metal (Cu) based antifoulants used periodically to protect the nets from fouling. Fish diets are also enriched with various metals including Cu, Fe, Znmanganese (Mn), cobalt (Co), arsenic (As), magnesium (Mg) and selenium (Se) to complete mineral requirements of farmed fish (CIESM, 2007). The European Parliament and Council has however established regulations which govern the use of various additive metals in animal nutrition. Moreover, several Directives have been set by EU to determine the acceptable levels of some metals in edible tissues.

Table 4
Concentrations (ng g^{-1}) of organochlorine compounds in the edible tissues of various farmed fish vs. wild counterparts in the Mediterranean.

Species	PCBs	HCBs	DDTs	p,p′-DDEs	References
Gilthead sea bream					
Farmed	3–39[a]		1.4–18[b]		Serrano et al. (2008)
Wild	11–23[a]		7.6–19[b]		
European sea bass					
Farmed	5.3–59.7		0.2–39		Antunes and Gil (2004) Lo Turco et al. (2007)
Wild	1.1–13		0.1–5.4		
White sea bream					
Farmed	101–736[c]		34–103[c]		Ferreira et al. (2008)
Wild	111[c]		20[c]		
Atlantic bluefin tuna					
Farmed	1917[c]	16[c]		2540[c]	Vizzini et al. (2010)
Wild	2572[c]	31.6[c]		2923[c]	

PCBs: polychlorinated biphenyls, HCBs: hexachlorobenzenes, DDTs: dichlorodiphenyltrichloroethanes, DDEs: dichlorodiphenylethylenes (DDT metabolite).
[a] Red muscle.
[b] Red and white muscle.
[c] Lipid based.

The adverse implications associated with human exposure to heavy metals are diverse and mainly include neurotoxic and carcinogenic effects (Sapkota et al., 2008). Generally, unclear findings have been reported from studies presenting comparisons in metal concentrations of edible tissues between farmed and wild fish.

In Mediterranean, farmed Atlantic bluefin tuna was found to contain less concentration of zinc Zn, mercury (Hg), and Cd but higher levels of Cu, Pb and As compared to wild tuna (Vizzini et al., 2010). Concentrations of Cd, Pb and Hg in farmed Atlantic bluefin tuna muscle were always below the acceptable levels (European Directive 466/2001: 0.10, 0.20 and 1 μg g^{-1} w.w., respectively). The European Community has not yet established limits for Cu and Zn in the edible parts of fish (Vizzini et al., 2010), however, farmed tunas showed concentrations that were far below the limits proposed by Australia (Cu: 10 μg g^{-1}; Zn: 150 μg g^{-1}) and Canada (Cu: 100 μg g^{-1}; Zn: 100 μg g^{-1}) (Vizzini et al., 2010).

In gilthead sea bream, lower levels of all metals examined where found in farmed vs. wild fish (Minganti et al., 2010). In particular, concentration of total and organic Hg, Cu, Fe, Mn, Zn and As were measured to be 0.12 and 0.10, 1.3, 10.3, 0.5, 15.9 and 4.9 μg g^{-1} in farmed gilthead sea bream while corresponding values or wild fish were 0.54 and 0.54, 1.6, 14.4, 0.5, 18.2 and 31.6 μg g^{-1}. Foran et al. (2004) reported higher levels of As in farmed Atlantic salmon, while concentrations of cobalt (Co), Cu and Cd were found to be higher in wild fish. Urena et al. (2007) reported no significant differences in tissue concentrations of various metals between wild-caught and farm-raised European eel. Overall, the studies that include relative comparisons between farmed and wild fish provide inadequate evidence to claim that the consumption of farmed fish may lead to higher exposures to selected metals in comparison with wild conspesifics.

3.2.4. Other chemicals

Other agrochemicals including antiparasitic pesticides, antifungals, anaesthetics and disinfectants may be also illegally used in aquaculture activities. The majority of them are suspected carcinogens and mutagens and are therefore banned from use in aquaculture in most geographic regions including the European Union. There is no available data on the possible contamination of aquaculture items originated from Mediterranean mariculture with illegal pharmaceuticals, antifoulings or agrochemicals used in plant raw materials.

3.3. Potential hazards from consumption of shellfish

A full discussion of the potential threats resulting from consumption of cultured shellfish is beyond the scope of this review. However, a brief presentation of the main hazards which should be of high concern for the public is given below.

Due to the fact that shellfish are often consumed raw or slightly cooked and they are eaten including their gastrointestinal tract, they are generally classified as a high-risk for human consumers by health authorities worldwide. These organisms are filter feeders extracting particles which may include serious pathogenic organisms such as bacteria, viruses, apicoplexan parasites and phytoplankton-originated toxins from the surrounding environment. In cases where farmed shellfish are collected from polluted waters, the presence of pathogenic bacteria including *Clostridium* sp., *Salmonella* spp., *Shigella* sp. and *Vibrio* spp. which may induce serious disorders to consumers is not unlikely (Jackson and Ogburn, 1999). Viral agents have been also implicated in several large food poisoning outbreaks associated with shellfish. Investigation of such outbreaks has often demonstrated sewage contamination of the shellfish growing area, and human enteric viruses such as Noroviruses (Norwalk virus) and hepatitis A have been mostly pointed as aetiological agents (Kingsley and Richards, 2003; Doyle et al., 2004; Greening and McCoubrey, 2010).

Due to the fact that farmed shellfish may accumulate protozoan oocysts, their consumption has implicated in parasitic infections such as *Giardia* sp. in European mussel (Gomez-Couso et al., 2005), *Giardia* sp. and *Cryptosporidium palvum* in European mussel (Giangaspero et al., 2009) and *C. palvum* in Japanese carpet shell (Freire et al., 2001). Another considerable health threat involved in eating contaminated raw farmed shellfish is the presence of *Toxoplasma gondii* in Pacific oysters from Italy (Putignani et al., 2011).

Some algae are able produce biotoxins which during algal blooms can be accumulated in shellfish to levels which can be lethal to humans. It is unfortunate that these toxins may not be degraded during cooking. Incidences of poisoning related to marine algal toxins include paralytic shellfish poisoning (PSP), neurotoxic shellfish poisoning (NSP), diarrhetic shellfish poisoning (DSP) and amnesic shellfish poisoning (ASP) (Ciminiello and Fattorusso, 2006). Actually, a DSP episode related to the consumption of European mussel was recently recorded in Greece (Economou et al., 2007).

Shellfish are subject to depuration procedures in order to reduce the likelihood of transmitting infectious agents to consumers. Depuration has been demonstrated to successfully reduce bacterial indicators in farmed European mussels from Galicia (Martinez et al., 2009) but not Noroviruses contamination in cultured European mussels and Pacific oysters (Savini et al., 2009). Consequently, shellfish associated disease outbreaks have occurred around the world including the Mediterranean (Savini et al., 2009), despite the operation of quality assurance programs in case of farming originated items. Depurated and non-depurated shellfish have both been implicated in foodborne disease outbreaks, however it is difficult to assess the significance of many of these outbreaks in terms of failure of sanitation programs or depuration procedures (Jackson and Ogburn, 1999).

Apart from the continuing need to control pollution of water bodies around on shore shellfish farms, additional procedures which would promote improved quality assurance procedures in the shellfish industry include the implementation of Hazard Analysis Critical Control Point (HACCP)-based food safety plans, formal compulsory training of shellfish producers in sanitary control measures and the development of a labelling system to increase customer perception. Moving to off shore farming has been an effective strategy to achieve high water quality in shellfish farming eliminating also the possibility of industrial and anthropogenic impacts.

4. Conclusions, recommendations and future priorities

Aquaculture as a primary food production sector must remain an overriding priority and therefore should continue to growth in all geographical areas including the Mediterranean. However, further aquaculture development in this region must ensure sustainability and balance the risks to public or environmental health with the substantial economical benefits. Side effects related to Mediterranean mariculture are obviously issues with high importance and priority, but also with distinctive nature. Respective research and released data appear well-addressed in some aspects but scarce in others.

The degree of genetic pollution resulting from the interactions between wild and escaped cultured populations is not clear concerning the traditional species. This is because of long pre-existed culture of particular species and therefore established interactions for decades and also due to the variability of existing tools for genetic assessments often providing contradicting information. The development of novel genetic tools with high validity to understand the actual status of both cultivated and wild populations, should be one of the research priorities concerning this area. Future research should also emphasize to the new introduced species for aquaculture, so appropriate monitoring and understanding of potential genetic interactions could be realized and proper genetic management be applied. The later will also provide *ex ante* information for the situation in the traditional species like gilthead sea bream and European sea bass. Additionally, all selective breeding programs must also take into account the ecological capacity of the organisms and overall, culture of fully domesticated and endemic animals should be rather encouraged.

The aquaculture-mediated species invasions account only for a small proportion of the total invasions, but their effects are not negligible. The adverse effects of these invasions can be minimized if correct management takes place, native species are preferred for farming when feasible and continuous monitoring occurs. Introduction of allien species for culture should include all precautions and consider the appropriate institutional rules to ensure correct actions. Total quality management schemes and good production practices are good frames for controlling unwanted invasions.

Due to the low research effort devoted to this aspect, the impacts of farmed fish on wild stocks in terms of pathogen exchange are not very clear in the Mediterranean and moreover there are conflicting views regarding the overall effects. Although there is some evidence for the existence of parasitic transfer between farmed and wild populations it can be concluded that wild fish stocks are not seriously affected by the influence of coastal sea-cage aquaculture. A continued monitoring associated with extensive research will be necessary to accomplish a good understanding of the disease interactions between farmed and wild fish stocks in the Mediterranean region. Maintaining high hygiene levels and adopting modern management practises with advanced preventive strategies will critically aid to eliminate disease in the interactive environment containing both the farmed and wild fish communities.

Organic pollution caused by aquaculture is one aspect which has received a lot of attention, both in terms of social concern and conducted research. The impacts of organic effluents are variable. Aquaculture should not be daemonized but special attention should always take place in order to avoid creating negative public image. Particular emphasis should be given in good manufacturing practices and systematic monitoring of the environment. New practices, like off-shore farming and polycultures that include low trophic level species, offer additional solutions and seem to be more efficient than classic techniques in aspects of organic pollution.

However, research data indicate that they should also not be idealized against the classic techniques, and that further research and understanding can give a better picture for these alternatives.

As far as the release of various pollutants in the vicinity of marine farms in the Mediterranean is concerned, little attention has been given with regard to the presence of metals and antibacterials, while research on the existence of organochlorine compounds and registered or banned chemicals is almost neglected. New research focused on the above aspects is therefore necessary. Possibly the establishment of EU regulations regarding the concentration of at least the most hazardous pollutants in the sediments will help to better control this type of pollution.

One of the most important issues for the sustainable development of aquaculture is the sustainability of the source of cultivated fish or shellfish. As worldwide fisheries stocks and their supporting ecosystems are in a fragile state, the growing importance of aquaculture production should not increase the pressure already exercised by capture fisheries on wild stocks. Rather the opposite, aquaculture should be a way to relieve this pressure on wild stocks and foster the maintenance of biodiversity. It is preferable that species aimed to be raised in aquaculture farms should have been produced in artificial environments. For this goal, research should be further encouraged to accomplish the full domestication by closing the life cycle of particular species. In cases where the later aim cannot be fully acheived, the capture of specimens to be used as broodstock or as starting farmed populations should ideally not distort wild stocks.

The utilization of fish meal and fish oil originated from wild fish is a universal problem not limited in the Mediterranean. Alternative resources of terrestrial plant and marine origin have attracted attention by researchers and industry (Naylor et al., 2009). Sustainable alternatives can solve the problem up to a degree, but they can also raise other ethical issues related to animal and environmental welfare and relatively lower the performance and nutritional value of fish (Grigorakis, 2010). Considering the high volume of baitfish needed to feed tuna there is an urgent need to further invest on on-going research to develop practical and effective artificial diets able to support a more controllable feeding strategy and eliminate the direct impact on fisheries stock.

As regards the influences of aquaculture on wild biota such as aggregations of wild animals and changes in the habitats, these may be variable in both qualitative and quantitative terms. This variability has to do with the farming type, the aquaculture design and characteristics and the receiving environment characteristics. Since the outcome of these interactions is not always negative, the aim should be optimization of these interactions, rather than minimization of them. In practical terms, this should mean individualized assessments in each case that include environmental impact assessments, hydrodynamic and ecological assessments of the existing sites or potential aquaculture development sites. Under a broader perspective, also characterization of certain ecosystems as compatible or incompatible with certain aquaculture activities, should be considered.

With respect to public safety in the Mediterranean mariculture there is an obvious need for surveillance since virtually no published studies concerning the potential transfer of antibacterial resistance via aquaculture items are available. Similarly, inspection data on the possible levels of contaminants including antibacterials and various chemicals are almost lacking at the scientific level. The establishment of HACCP procedures with regard to hazardous substances and pathogens is more than vital and should be an obligatory prerequisite for the operation of production units concerning both fish and shellfish farming. Continuous training and education of mariculture producers in sanitary control measures and the development of a labelling system to increase customer perception would promote improved quality assurance procedures.

In terms of required legislation at the EU level, the determination of legal absolute concentrations allowable for organochlorine compounds in fish tissues is more than urgent and thus should a future legislative priority.

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