

AN ECOSYSTEM-BASED APPROACH TO STUDY TWO
DOLPHIN POPULATIONS AROUND THE ISLAND OF
KALAMOS, IONIAN SEA, GREECE

by

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Abstract

In the northeastern Ionian Sea, two populations of dolphins, the short-beaked common dolphin (*Delphinus delphis*) and the common bottlenose dolphin (*Tursiops truncatus*), have been studied since 1993 by the Tethys Research Institute, Italy. Results show a low density of bottlenose dolphins, and a rapid decline in common dolphin numbers and encounter rates. I constructed an ecosystem model using *Ecopath* with *Ecosim* for the northeastern Ionian Sea to explain the two different abundance trends of these species of dolphins, and the trophic interactions with their prey and with the fisheries. The *Ecopath* model was built for the year 1964, adding data on biomass, P/B, Q/B and diet for each functional group of the ecosystem, fisheries landings and discards. *Ecosim* was used to generate simulated trajectories to fit with the 'observed' trends (e.g., biomass, bycatch, CPUE and catches) for the most important groups of the ecosystem. The results of the fitting underline a clear decline of common dolphins caused by reduced prey availability. In particular, sardine and anchovy stocks, the main prey of common dolphins, have decreased sharply since the late 1970s due to intensive fishing pressure in the area until the end of 1990s. On the other hand, the population of bottlenose dolphins has increased through time; a positive correlation was observed between this species of dolphin and the development of the fish farm industry in the study area. The *Ecopath* model suggested that this increase could be attributed to an increase in productivity in the waters that surround the fish cages in the study area. *Ecosim* was used to simulate three fishing policy scenarios within the study area. Three different fishing closures were investigated: the closure of purse seiners; the closure of the industrial fisheries (purse seiners, trawlers and beach seiners); and the closure of the entire fishing fleets. According to *Ecosim* predictions, sardines and anchovies would benefit from all three management strategies. Common dolphins showed a significant increase only when the entire study area was closed to fishing. The creation of MPAs was explored using *Ecospace*. In particular, two different scenarios were evaluated: a MPA1 that closed the entire area to fisheries, and a MPA2 that allowed the small and big artisanal fisheries to operate. The results obtained using this spatial model agreed with similar scenarios simulated in

Ecosim. The application of *Ecopath* with *Ecosim* and *Ecospace* appeared to be a useful tool for understanding this marine ecosystem. The models underlined the management actions needed to restore and protect common dolphins, bottlenose dolphins and other target species. In particular, measures to reduce overfishing, stop illegal fishing activities and to enforce existing laws are needed. Implementing marine protected area seems to be an effective management measure to ensure prey survival and sustain marine predators such as common dolphins.

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CHAPTER 1: INTRODUCTION

1.1 Fisheries in the Mediterranean

In the Mediterranean Sea, the exploitation of marine resources has a long history. Evidence of fishing activities has been found and documented since the time of the Roman and Greek empires: writers like Polybius in his '*Histories*' 264 BC, Oppian with the '*Halieutica*' 180 BC and Pliny the Elder with the '*Natural History*' 77 AD well described various types of fishing such as the use of nets, spears, tridents, and various traps (Radcliffe, 1921; Farrugio et al., 1993).

Despite the fact that the art of fishing began several thousand years ago, fisheries research and fisheries management was only developed quite recently (after World War II), and most extensively so in the north-western countries of the Mediterranean Sea (Farrugio et al., 1993). During this period, it emerged that statistical reports of catches and effort were, and still often are unreliable, and actual levels are mostly underestimated, since an important part of the Mediterranean landings go directly to public markets and regional auctions, and this part is often not included in the official records. Moreover, there is still little control with the fleets especially with regard to their unregulated net sizes, catches of undersized fishes, and on the depth of their fishing grounds (Farrugio et al., 1993; EC, 2003).

On the other hand, it has been recently shown that during the last 50 years, while the fishing effort has rapidly been increasing, the Mediterranean catch rates have kept on declining, facing the same 'fishing down' impact on the marine ecosystems, which has been observed throughout the world (Grainger and Garcia, 1996; Pauly and Palomares, 2000; Stergiou and Koulouris, 2000).

1.2 Marine mammals-fisheries interactions

Interactions with fisheries are considered one of the major threats affecting marine mammals, especially at the stock level (Northridge, 1991; DeMaster et al., 2001; Crespo and Hall, 2002; Lavigne, 2003; Marsh et al., 2003; Reeves et al., 2003; Kaschner, 2004). The interactions between marine mammals and fisheries may fall into two categories:

- *Operational* or *direct* in which marine mammals interact with fishing gear;
- *Ecological* or *indirect* in which marine mammals and fisheries compete for the same food resources.

Operational interactions can be negative for either the fishery or the marine mammal. In the first case, marine mammals may damage the fishing gear by removing the fish caught in the net, causing economic loss for the fishers; on the other hand marine mammals can be trapped in the nets and perish by this entanglement (Northridge and Hoffman, 1999; DeMaster et al., 2001; Goldsworthy et al., 2003; Marsh et al., 2003). Examples of negative operational interactions have been observed throughout the world. Steller sea lions (*Eumetopias jubatus*), California sea lions (*Zalophus californianus*) and harbor seals (*Phoca vitulina*), for example, have caused substantial damage to American fisheries (from Alaska to California) by damaging nets or by removing fish, especially salmon from fishing gears (Fraker and Mate, 1999). In South Africa, Cape fur seals (*Arctocephalus pusillus*) were responsible for the removal of small pelagic fish and damage to purse seiners nets (Wickens, 1995).

Cases in which marine mammals can be negatively influenced by fishing operations have also been observed. From 1960 to 1972, for example, in the eastern tropical Pacific Ocean, four million dolphins, in particular, the northeastern offshore spotted dolphins *S. attenuata attenuata*, the coastal spotted dolphins *S. attenuata graffmani*, and the eastern spinner dolphins *S. longirostris orientalis*, have been killed, trapped in purse seiner nets. This mass mortality, induced the decline of these dolphin stocks and declaring them 'depleted' under the USA Marine Mammal Protection Act (MMPA) (Wade et al., 2007). The Vaquita (*Phocoena sinus*) is another clear example of operational interaction with fisheries in the Gulf of California: here this endemic species is in critical danger of extinction because of bycatch in gillnet fisheries (D'Agrosa et al., 2000).

Ecological interactions are, instead, more difficult to assess because of the complexity of the marine ecosystems: the increase or the decline of marine mammals populations or fisheries may have a negative or positive impact on the fish stock availability. In most cases, however, these interactions are seen as detrimental to either marine mammals or fisheries (Northridge and Hoffman, 1999). The phenomenon of overfishing can be seen as an 'ecological' interaction

in which fisheries and marine mammals are both negatively affected (Christensen, 1996; Northridge and Hoffman, 1999; DeMaster et al., 2001; Crespo and Hall, 2002; Plaganyi and Butterworth, 2002; Goldsworthy et al., 2003). Studies conducted in 1997 in the south-western Atlantic, for example, have shown that coastal fisheries have severely depleted certain sciaenid species, which are also the main prey of the river dolphin, Franciscana (*Pontoporia blainvillei*), causing a clear shift in the diet of this species (Secchi et al., 1997). Another example is along the coast of the western Gulf of Alaska, the Aleutian Islands, where the fishing pressure on Atka mackerel may have contributed to the decline of the Steller sea lions (*Eumetopias jubatus*) from the late 1970s to the late 1990s (Guénette et al., 2006).

1.2.1 Interactions fisheries-marine mammals in the Mediterranean Sea

In the Mediterranean Sea, marine mammals and fisheries have been known to interact since antiquity. In most of the cases, incidental entanglement in fishing gears and direct killing are considered the major threats to marine mammal populations (Northridge, 1991; Reeves et al., 2001; Bearzi, 2002; Tudela, 2004). In 1991, for example, the Italian driftnet fisheries, which mainly catch swordfish and albacore, were reported to have captured 1682 cetaceans, with striped dolphins (*Stenella coeruleoalba*), sperm whales, pilot whales and Risso's dolphins being the most impacted (Di Natale, 1995). Bycatch caused by driftnet fisheries have been observed also in the Aegean Sea, the Alboran Sea and in the Balearic Islands impacting a variety of cetaceans species such as common, bottlenose and striped dolphins, sperm whales, pilot whales and fin whales (Northridge, 1991; Tudela, 2004).

Moreover, in the early 1980s in Italian and Spanish waters, the longline fisheries killed, due to entanglement, several cetaceans such as, striped dolphins, false killer whales, Risso's dolphins, sperm and fin whales (Duguy, 1983). It has also been observed that in the same areas, the purse seine fishery can cause bycatch of small dolphins like common, striped and bottlenose dolphins (UNEP/IUCN 1994; University of Barcelona, 1995). Deliberate killings mainly of the endangered Mediterranean monk seals have been reported in

different areas of the Mediterranean Sea between the early 1960s to the end of 1980s. Turkey, Greece, the Balearic Islands, Tunisia, Algeria and Morocco were the main involved areas (Avella, 1979; Berkes et al., 1979; Avella, 1987; Panou et al., 1993). Intentional killing normally occurs because local fishers blame the monk seal to be the major cause of their loss of income (e.g. through damage of fishing gears and decreases in fish catch).

The reduction of food resources mainly caused by overfishing may be another threat causing the decline of Mediterranean marine mammals populations (Perrin, 1988; Reeves and Leatherwood, 1994; UNEP/IUCN, 1994; Bearzi, 2002). As mentioned in section 1.2, even if this type of interaction is very hard to assess, especially because of the complexity of the marine ecosystem, evidence indicating that overfishing may impact food supply for marine mammals has been seen in different areas of the Mediterranean Sea, mainly for small cetaceans.

Common dolphin is one of the cetacean species that competes most with fisheries (purse seiners) for fish resources, particularly small pelagic prey, like sardines and anchovies. Overfishing, for example, is considered one of the reasons why the common dolphin has disappeared from the northern Adriatic Sea (Bearzi et al., 2004a). Still in the Adriatic Sea and more precisely in Kvarneric, a study conducted between 1991-1994 revealed that 80% of common bottlenose dolphin behaviour largely consisted of feeding-related activities, suggesting high variability and depletion of Kvarneric fish resources (Bearzi et al., 1999). Moreover, in 1990 and 1994, mass mortality of common dolphins occurred along the coast of the Black Sea. Results from analysis of the carcasses showed that most of these animals were emaciated. This event occurred concurrently with the rapid decline of the European anchovy and European sprat stocks, the main prey of Black Sea common dolphin (Birkun, 2002).

1.2 Marine Protected Areas

Nowadays, it is commonly thought that marine reserves or marine protected areas (MPAs) are a useful tool for the management of fisheries and also for the conservation of marine ecosystems (Roberts and Polunin, 1993; Hall, 1998; Walters et al., 1999; Watson et al., 2000; Pitcher, 2001; Tudela, 2004). The

creation of MPAs, in the long term, may rebuild marine biodiversity by conserving habitats, fish stocks and top predators such as marine mammals, and may also allow the increase of fisheries yields in the surrounding areas (Agardy, 1997; Stergiou et al., 1997; Pitcher, 2001; Russ and Alcala, 2004; Russ et al., 2004; Pauly et al., 2005).

Different sizes and locations of MPAs, as well as the dispersal rate of marine organisms are probably the key parameters that influence the success of marine reserves (Halpern, 2003; Shanks et al., 2003; Kai and Shirakihara, 2005; Laurel and Bradbury, 2006). Two other important aspects for the MPAs success are the social and economic impacts of MPAs on the coastal communities (Guénette et al., 2000). These authors, have shown how coastal areas like the Great Barrier Reef Marine Park in Australia and US Florida Keys National Marine Sanctuary, have succeed thanks in part to community involvement.

In the Mediterranean, the establishment of the first MPA started in France in the middle of the 1970s, followed by Italy and Spain in the early 1980s. Most of these MPAs were created initially to preserve large sedentary species, such as rocky littoral fish, seagrass fish and invertebrate populations, which previously had been overexploited. The results of these studies have shown that, by eliminating or decreasing fishing activities, the key species may increase (biomass or numbers) rapidly (Goñi et al., 2000). Unfortunately most of the fisheries target species are mobile and only a few studies have shown how these species may be restored by limiting or stopping fishing activities in their distribution area (Garcia and Demetropoulos, 1986; Guidetti, 2006). Because of the complexity of assessing MPAs and their impact on marine ecosystem and on fisheries, ecosystem modeling has been proposed as a tool to design and implement MPAs and to evaluate the critical aspects mentioned above (Walters, 2000).

1.3 Ecosystem based-approach modeling

Since the early 1950s, fisheries scientists have given advice for the management of fisheries resources based on single species assessments (Pauly et al., 2002). Even if this approach is still in use, scientists are more and more conscious of the need to use ecosystems-based approaches to better understand

the complex dynamics of marine ecosystems, and to better manage it (Pauly and Christensen, 1996; Trites et al., 1999; Mace, 2001; Pauly and Christensen, 2002; Pauly et al., 2002; Smith et al., 2007).

Ecosystem modeling is considered a useful tool, mostly because it can assess the trophic interactions of an ecosystem, the direct or indirect impacts of human activities, such as fisheries, and the ecological perturbations that fisheries may cause on the trophic web. Moreover, it has been shown that this approach might be helpful in understanding the complex interactions between marine mammals, fisheries and the rest of the ecosystem (Earle, 1996; Mangel and Hofman, 1999; Trites et al., 1999). Ecosystem models, indeed, may quantify marine mammal food consumption and evaluate possible competition or overlap with fisheries, determining if the presence of one may be detrimental for the other one. Furthermore they can identify conflict areas and become an important management tool through creation of suitable placed marine protected areas (Trites et al., 1999; Kaschner et al., 2001).

1.5 Study area: the northeastern Ionian Sea

This study describes the marine ecosystem of the northeastern Ionian Sea, situated in the Western part of Greece. The area covers approximately 1021 km² of sea surface; it includes the islands of Meganisi, Kalamos and Kastos and it is delimited by the mainland of Greece, the Lefkada island and the northern tips of Kefalonia and Ithaca (see Fig.1.1)

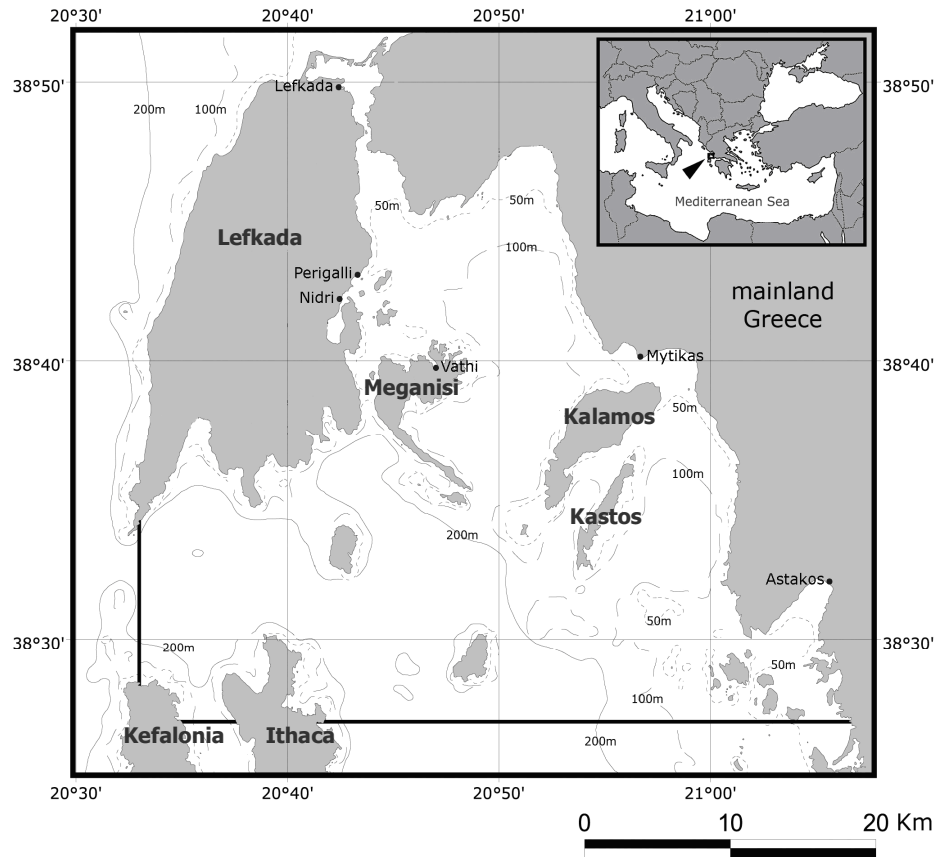


Fig.1.1. The study area.

The NE Ionian Sea is influenced by four main water masses: the near surface water mass also called North Atlantic Water (NAW), Ionian Surface Water (ISW), Levantine Intermediate Water (LIW) and Deep Water (DW) of the Eastern Mediterranean. The layer of the NAW consists mainly of water saturated in oxygen, low in salinity and very poor in nutrients; it extends from the surface to about 60 m. It spreads into the Eastern Mediterranean through the Sicily Strait and it is normally found in the East part of Greece. In winter, it also occurs in NE Ionian Sea. During the summer, the Ionian Surface Water flows in the area: ISW differentiates from the NAW because it is saltier and warmer (Malanotte-Rizzoli et al., 1998). LIW occupies the subsurface layers (80-150m) and it is characterized by higher salinities and nutrients, while DW has colder and more uniform water that extends from the lower part of the LIW down to the bottom layer (Boussoulenga et al., 1990; Psyllidou-Giouranovits et al., 1994; Malanotte-Rizzoli et al., 1998; Ramfos et al., 2005). Moreover, the presence of cyclonic and

anticyclonic gyres influences the oxygen and nutrient pattern of the Northeastern Ionian Sea (Souvermezoglou et al., 1992).

Most of the area is quite shallow, ranging in depth approximately between 100 to 200 meters. The sediments are mainly covered by seagrass meadows (*Posidonia oceanica* and *Cymodocea nodosa*) and consist of mud (silt-clay) in areas deeper than 50 meters, while at shallower depths (less than 50 meters) the sediments are primarily sandy (Haritonidis and Tsekos, 1976; Zenetos et al., 1997). A study conducted by Casotti et al. (2003) shows that this area is highly oligotrophic and may induce stability with low seasonality. Values of chlorophyll *a*, nutrients and particulate organic carbon were among the lowest registered in the Mediterranean coastal areas (Pitta et al., 1998).

1.6 The cetaceans of the Ionian Sea

Among the 19 cetaceans species that can be found in the Mediterranean basin, only six are present in the Ionian Sea: bottlenose dolphin (*Tursiops truncatus*); short-beaked common dolphin (*Delphinus delphis*); striped dolphin (*Stenella coeruleoalba*); Risso's dolphin (*Grampus griseus*); Cuvier's beaked whale (*Ziphius cavirostris*), and fin whale (*Balaenoptera physalus*) (Frantzis et al., 2003). The first attempt to study the cetacean species of the NE Ionian Sea started in 1991; the result of this survey have indicated that common dolphins and bottlenose dolphins are the two main cetacean species that regularly inhabit these coastal waters (Politi et al., 1994).

1.6.1 Key species: the short-beaked common dolphin

1.6.1.1 Description and distribution

The short-beaked common dolphin, *Delphinus delphis*, is a small dolphin that belongs to the Delphinidae family. The adult body length and body weight normally vary depending on the geographical areas in which the population live. In the Mediterranean Sea, the length ranges between 170 and 200 cm and with a body mass of approximately 90 kg (Notarbartolo di Sciara and Demma, 1997). Its distribution is worldwide: it can be found in temperate, tropical, and subtropical seas and occupy near-shore coastal waters as well as pelagic habitats (Heyning

and Perrin, 1994). It shows a strong preference for areas with warm, saline surface waters (Notarbartolo di Sciara and Demma, 1997; Reeves et al., 2002).

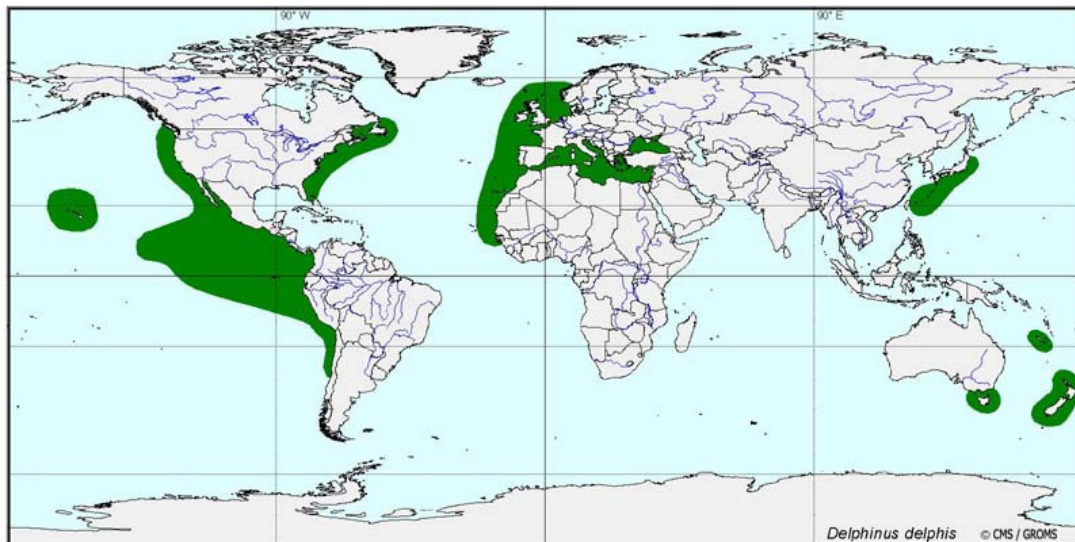


Fig.1.2. Distribution of the short-beaked common dolphin, *Delphinus delphis* (modified from Perrin 2002; Copyright: The Convention on the Conservation of Migratory Species of Wild Animals/ GROMS).

In the western Mediterranean Sea, hundred of individuals are occasionally sighted in the Alborà Sea and in the Gulf of Vera (Cañadas et al., 2002). Observations have been made along the coasts of Algeria and Tunisia (Boutiba, 1994; Boutiba and Abdelghani, 1995) and close to the Island of Malta (Vella, 1998, 1999). Sightings of isolated groups are also made around the western coasts of Sardinia and Corsica (Notarbartolo di Sciara et al., 1993; Lauriano and Notarbartolo di Sciara, 1995; Forcada and Hammond, 1998) and off the island of Ischia (Mussi et al., 2002). In the Eastern Mediterranean Sea, the species regularly occurs in the Aegean Sea (Frantzis, 1996; Carpentieri et al., 1999; Casale et al., 1999; Frantzis et al., 2001), in the Eastern Ionian Sea, especially around the island of Kalamos (Politi et al., 1998; Politi and Bearzi, 2004) and in the Gulf of Corinth (Frantzis and Herzing, 2002). Its presence is considered rare off the Israeli coastline (Bearzi et al., 2003).

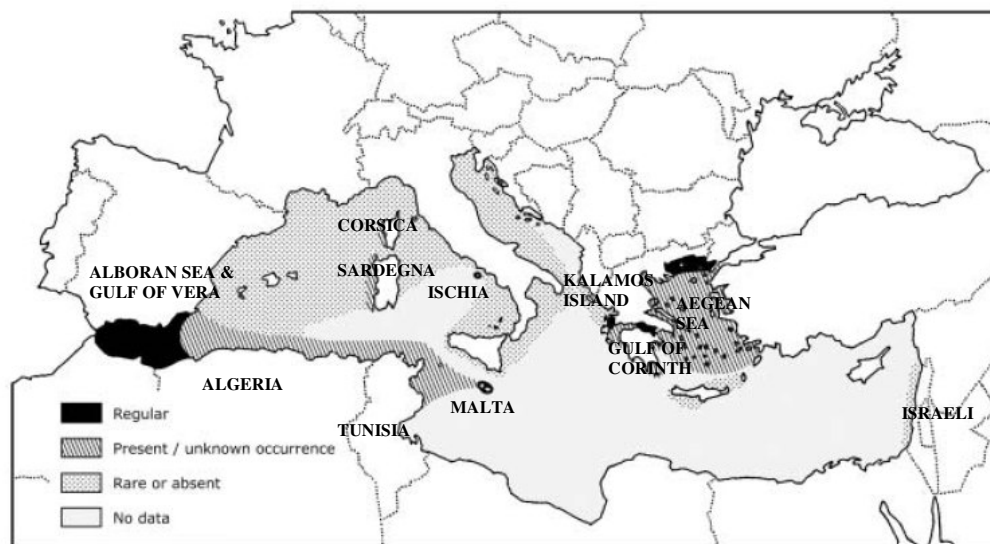


Fig.1.3. Approximate distribution and relative density map of short-beaked common dolphins in the Mediterranean Sea (modified from Bearzi et al., 2003).

1.6.1.2 Diet

The diet of the common dolphin consists mainly of mesopelagic and epipelagic fish and also of cephalopods and crustaceans. Analysis of the stomach contents from Mediterranean and other countries of the world have revealed the preference especially for schooling fish like sardines (*Sardina pilchardus*) and anchovies (*Engraulis encrasicolus*), and for squids (Loliginidae and Sepiolidae) (Sekiguchi et al., 1992; Orsi-Relini and Relini, 1993; Silva, 1999; Reid et al., 2003; Agazzi et al., 2004; Murphy, 2004; Pusineri et al., 2004).

1.6.1.3 Interaction with fisheries

The common dolphin is one of the most common marine mammal by-catch species for different types of fisheries; this is due mainly because of its preference for prey that are commercially important and targeted by fisheries. In 1988 for example, an estimated 16,189 common dolphins were killed in the eastern tropical Pacific by tuna purse-seine fishery (Evans, 1994). Incidental catches in pelagic trawl have been also reported in the Bay of Biscay, Western Approaches and Celtic Sea (Tregenza and Collet, 1998). Furthermore, research conducted between 1992 and 1993 off the Bay of Biscay, has shown that

on average 1.7 common dolphins have been caught per trip by the French driftnetters targetting albacore tuna (Goujon, 1996).

On the other hand, in the Mediterranean Sea, the level of interactions between fisheries and common dolphins are poorly known, probably as a consequence of their low abundance (Bearzi et al., 2003). However, a few reports have demonstrated that these interactions exist in the Mediterranean and are a serious threat for this species (Reyes, 1991; Silvani et al., 1999; Notarbartolo di Sciara and Bearzi, 2002).

1.6.1.4 Conservation status

In the 1996 IUCN Red List of Threatened Animals, the world population of short-beaked common dolphin was listed as lower risk 'conservation dependent' (Baillie and Groombridge, 1996). This was due to the fact that most of the world populations were still abundant (Gaskin, 1992; Wade and Gerrodette, 1993; LeDuc, 2002).

On the other hand, in the Mediterranean Sea, the conservation of this dolphin had started to be of concern in the early 1970s. This was because, common dolphin, which used to be one of the most 'common' cetacean species of the Mediterranean Sea, had faced a drastic decline (Bearzi et al., 2003). The area around the Island of Kalamos, in the Northeastern Ionian Sea, for example, has been included by the Greek Ministry of the Environment in the Natura 2000 network as 'Site of Community importance' under the 9243 EEC 'Habitats' directive. This was decided based on the presence of one of the last communities of common dolphins in the Mediterranean Sea (Frantzis, 1996).

Despite this designation, common dolphins, which at the beginning of the study were abundant in the area, have undergone a substantial decline at least since 1997 (Bearzi et al., 2005). As a consequence, in 2002, the *Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and the Contiguous Atlantic Area* (ACCOBAMS) underlined the importance of the Kalamos area to develop pilot management actions for the preservation of common dolphin critical habitats. In 2003, the *IUCN Red List of Threatened Animals* listed the Mediterranean common dolphin 'subpopulation' as "endangered based on

criterion A2 which refers to a 50% decline in abundance over the last three generations” (Bearzi et al., 2003). Unfortunately, even if there has been an increase in public awareness about dolphin conservation in several areas of the Mediterranean, very little has been done to understand the causes that contributed to common dolphin’s regional decline (Bearzi et al., 2003).

1.6.2 Key species: the common bottlenose dolphin

1.6.2.1 Description and distribution

The common bottlenose dolphin, *Tursiops truncatus*, is one of the better known cetaceans and it belongs, as the common dolphin, to the Delphinidae family. As for the common dolphin, its dimensions may vary depending on the geographic area in which it lives; in the Mediterranean Sea the adult mean length is around 2.5 up to a maximum of 4 meters. The body weight can be between 250-350 kg (Notarbartolo di Sciara and Demma, 1997). The bottlenose dolphin is one of the cetaceans species that is most adaptable to a wide variety of habitats: it can be found in pelagic waters like in the Eastern Tropical Pacific and around the Faroe Islands, but normally lives in coastal water (Notarbartolo di Sciara et al., 1993; Bearzi et al., 1997; Notarbartolo di Sciara and Demma, 1997). It is abundant in tropical, subtropical and temperate water of the world and it’s absent only in the very cold waters of the globe. It is rare in the Baltic Sea, while it is found in all the Mediterranean Basin and the Black Seas (Notarbartolo di Sciara and Demma, 1997; Wells and Scott, 1999).

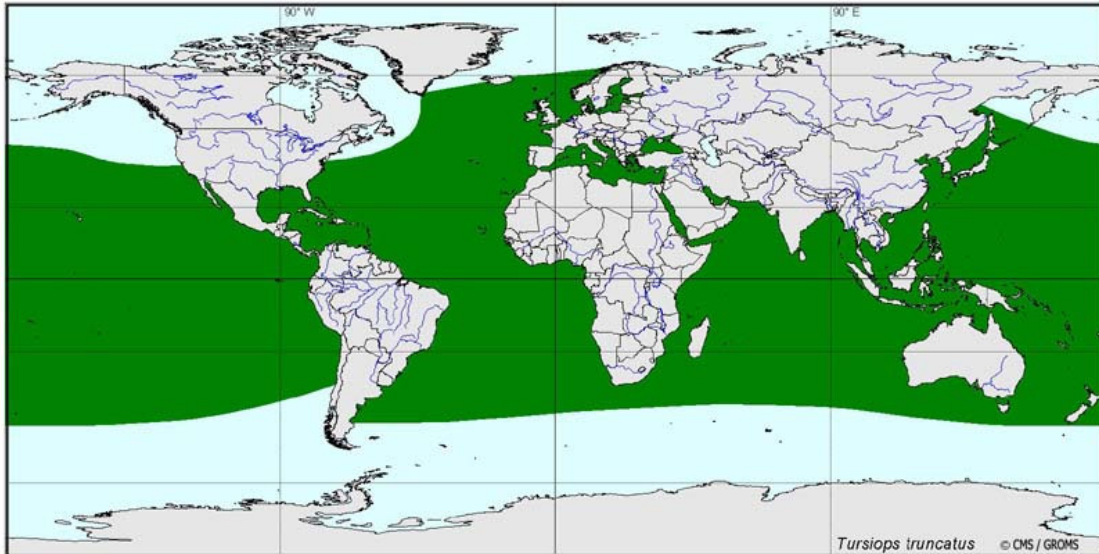


Fig.1.4. Distribution of the common bottlenose dolphin, *Tursiops truncatus*, (modified from Wells and Scott, 2002; Copyright: The Convention on the Conservation of Migratory Species of Wild Animals/ GROMS).

1.6.2.2 Diet

The bottlenose dolphin has opportunistic feeding habits; this means that it can adapt its diet depending on the availability of prey (Leatherwood, 1975; Barros and Odell, 1990; Bearzi et al., 1999). It feeds on hake, flathead mullets, European congers, anchovies, sardines, mackerel, red mullets but also a wide variety of cephalopods, crustaceans and benthic invertebrates (Notarbartolo di Sciara and Demma, 1997; Carwardine et al., 1998; Wells and Scott, 2002).

1.6.2.3 Interaction with fisheries

Most of the interactions that affect the bycatch level of bottlenose dolphins have increased through time all around the world with the development of the fisheries (Wells and Scott, 2002). In North Carolina, for example, the interactions between bottlenose dolphins and gillnets are still very common (Read et al., 2003). In China, this kind of fisheries keeps on killing hundreds of individuals (Yang, 1999), while in Australia, the Taiwan gillnet fishery incidentally catch more than 2000 individuals per year (Northridge, 1991).

In the Mediterranean, the interactions are mostly associated with longlines, gillnets, driftnets, trawlers and occasionally purse seiners (IWC, 1994; UNEP/IUCN, 1994; Reeves et al., 2003). In Spain, especially in the Balearic Islands, studies conducted between 1992 and 1995 have shown that gillnet fisheries targeting red mullets and cuttlefish, have every year caught around 30 bottlenose dolphins (Silvani et al., 1992). Several strandings happened between 1986 and 1999 along the Italian coasts, and were caused by indirect bycatch in driftnet fisheries (Di Natale, 1983; Cagnolaro and Notarbartolo di Sciara, 1992).

1.6.2.4 Conservation status

Even if bottlenose dolphins are still quite abundant throughout the world, its preference for coastal habitats make this dolphin particularly sensitive to human impacts, notably through fisheries and industrialization. Due to these human activities, for example, the bottlenose dolphin has started to decline in areas such as in the San Diego Bay and the Bay of Biscay, inducing every year more gaps in its distribution (Wells and Scott, 2002). The species is listed as “Data deficient” by the IUCN Red List due to the lack or “inadequate information to make a direct, or indirect, assessment of its risk of extinction based on its distribution and/or population status” (IUCN, 2007). In the Mediterranean, the populations of bottlenose dolphins are in decline and they have been proposed for classification as “vulnerable” based on IUCN Red List criteria (Reeves and Notarbartolo di Sciara, 2006); the *Mediterranean Action Plan* of UNEP (*United Nations Environmental Program*) recommends the adoption of rigorous measures for the conservation of this cetacean species.

1.7 The fisheries of the northeastern Ionian Sea

The marine fisheries landings of Greece have been recorded since 1964 by the National Statistical Service of Hellas (NSSH Bulletins, 1964-2003). The Greek waters have been separated into 18 statistical fishing subareas (see fig. 1.4) (Stergiou et al., 2007).

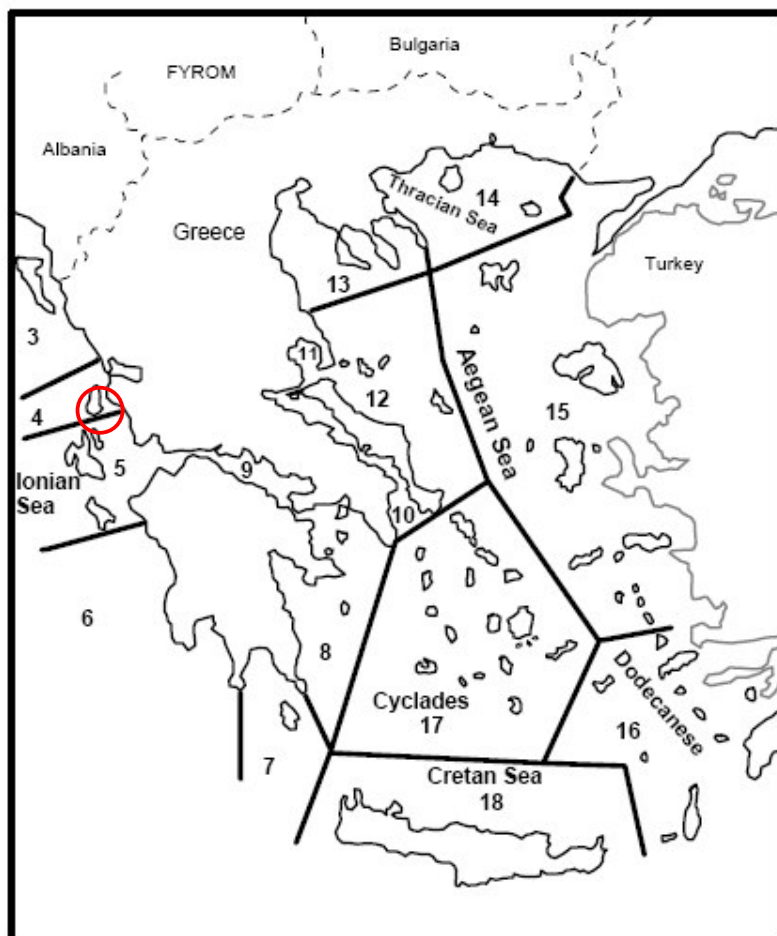


Fig 1.5. The 18 statistical fishing subareas (modified from Stergiou et al., 2007). The circle near the left edge (area 4) shows the present study area.

The fishing area that will be taken into account in this study is Area 4 (see Fig. 1.5). The commercial fisheries of the study area include: trawlers, purse seiners, beach seiners and artisanal boats (that include longliners, and trammel netters) operating along the coasts. About 70 species of fish, cephalopods and crustaceans are fished commercially in the area, while only a few constitute the most important target of the fishing fleets: sardine (*Sardina pilchardus*); anchovy (*Engraulis encrasicolus*); horse mackerel (*Trachurus mediterraneus*); bonito (*Sarda sarda*); bogue (*Boops boops*); pickerel (*Spicara smaris*); hake (*Merluccius merluccius*); red mullet (*Mullus barbatus*) and striped red mullet (*Mullus surmuletus*) (Tsikliras et al., 2007).

Fisheries management regulations and conservation policy are under the authority of the Ministry of Agriculture (Papaconstantinou, 2005). Measures to regulate the fisheries issues have been in place since the early 1960s. For example, Royal Decree 666/66 has been established to control trawl and purse seine fishery licensing. The license is given to trawl vessels with > 14 meters of length, maximum outer width of 5 meters, > 2 meters of height, capacity of > 28 GWT and engine with > 80 HP. For the purse seiners, the vessels need to have > 11 meters of length, maximum outer width of 4.5 meters, > 1.1 meters of height and engine with > 40 HP. Furthermore, the spatio-temporal activity of the trawl and purse seine fisheries is under national and European regulations (EU Regulation, 1967/2006). Trawling is prohibited from May to the end of September. Generally, the European Union has established a 3 nautical miles (n.m.) distance from the coastline as the main limit to trawling, but this depends on the limits of the 50 meters depth zone. If the latter spreads beyond the 3 n.m., then the 3 n.m. is the valid restriction, but when the 50 meters zone is inside the 3 n.m. zone then the restrictions are determined by the depth zone. The minimum mesh size of the cod-end is 40 mm (from knot to knot).

For purse seiners, the fishery is prohibited from December to the end of February. In addition, day/night purse-seine fishery is prohibited at distances of less than 100 m from the coastline, less than 1000 m from fish farms, and less than 500 m from barrier traps. Day/night purse-seine fishery is also prohibited at depths less than 30 meters, during full moon (two days before and two days after full moon). The minimum mesh size of the net is 14 mm (from knot to knot). Regarding the beach seiners, the Greek legislation allows them to operate from October to March. Fishing is allowed from one hour before sunrise until one hour after sunset. The activity of vessel is restricted to a distance of about 1 nautical mile along the coast and on smooth bottoms only. During hauling the vessel must remain at a distance of less than 70 m from the shoreline. The mesh size is at least 16 mm (full mesh). Trammel net and longline fisheries do not have general regulations. The only measures specified by the Council Regulation 1967/2006 are: trammel netters cannot carry on board and use more than 5,000 meters of bottom nets and the height of the bottom nets is restricted to 4 meters. Longline fleets cannot carry on board and use more than 7,000 meters of bottom longlines

and more than 60 km of floating longlines (http://ec.europa.eu/fisheries/publications/factsheets/legal_texts/section_d.pdf).

Despite the presence of different management/conservation measures, it has been observed that in all Greece, demersal and pelagic fisheries resources have been overfished and that the abundance of the target species has been drastically diminished by an order of magnitude between 1989 and 1993 (Stergiou et al., 1997; Stergiou and Koulouris, 2000). Due to the inadequacy of these regulations, new management actions are required for the preservation of the marine ecosystem and for the sustainability of the Greek fisheries (Stergiou et al., 1997).

1.8 Objectives and aim of the study

The objective of this study is to investigate the trophic interactions of common dolphins and bottlenose dolphins with the other components of the marine ecosystem and with human activities, using the computer program *Ecopath* with *Ecosim*. In particular, the model will be used to assess the interactions between the local fisheries and common dolphin population, and to explain why these dolphins have been declining for the past 10 years. The *Ecopath* model will be constructed for the year 1964; secondly, *Ecosim* will be used to fit time series data trends (biomass, CPUE and catches of the main key species) with predicted trajectories. Furthermore, fishing policy scenarios and MPAs will be evaluated and implemented for a better conservation of the marine ecosystem, especially for common and bottlenose dolphins.

CHAPTER 2: METHODOLOGY

2.1 The input parameters

An *Ecopath with Ecosim* model was constructed for the NE Ionian Sea to represent the year 1964. This year was chosen because of the information on catches time series that was available from 1964 to 2003. 22 ‘functional groups’ were considered in the model: among these, there were 3 marine mammals, 1 sea turtle, 1 sea bird, 8 fishes, 5 invertebrates, and 2 primary producer groups. For each group, four input parameters were estimated: biomass, production per unit of biomass (P/B), consumption per unit of biomass (Q/B) and diet composition (Table 3.1 and Table 3.2).

In particular, biomass of each functional group was obtained from observed data or estimated from a population reconstruction; its unit is t of wet weight per km². The P/B ratio was estimated as instantaneous total mortality, Z (Allen, 1971). Therefore it was calculated as fishing mortality (F) plus natural mortality (M) for commercially exploited stocks, while it was set to natural mortality for non- commercial stocks. Its unit is per year. The Q/B ratio is defined as the food intake by a group during a certain period of time (here: a year) divided by its biomass. When it was possible, the consumption of each group was obtained through field studies; otherwise it was estimated from empirical equations. For all finfish, Q/B was calculated using the following empirical equation (Christensen et al., 2005):

$$\text{Log (Q/B)} = 7.964 - 0.204 \cdot \log W_{\infty} - 1.965 \cdot T' + 0.083 \cdot A + 0.532 \cdot h + 0.398 \cdot d$$

where W_{∞} is the asymptotic weight (g), T' is the annual temperature of the water, expressed using $T' = 1000/\text{Kelvin}$ (Kelvin = °C + 273.15), A is the aspect ratio expressed as (h^2/s) where h is the height and s is the surface area of the caudal fin, h is a dummy variable expressing food type (1 for herbivores, and 0 for detritivores and carnivores), and d is a dummy variable also expressing food type (1 for detritivores, and 0 for herbivores and carnivores) (Christensen et al., 2005). The diet matrix was constructed using either observed data or, where this was not possible, through diet data obtained from the literature for the same

species in similar ecosystems. A description of the functional groups and the estimation of their respective input parameters are presented in the follow section.

2.2 Functional group description

2.2.1 Short-beaked common dolphins (*Delphinus delphis*)

Common dolphins have been studied in the NE Ionian Sea by the Tethys Research Institute, an Italian NGO, since 1993. Population assessments were estimated by Tethys for the years 1995-2006. In 1995, the number of individuals that were photo identified was 120; however, during the study period, a substantial decline in the encounter rate and dolphin numbers was observed (Bearzi et al., 2005, 2006).

The biomass for 1964 was calculated using the population reconstruction applied by Christensen (2006). The equation is a logistic growth model:

$$N_{t+1} = N_t + r \cdot N_t (1 - N_t/k) - C_t$$

where r is the intrinsic rate of growth, N_t is the abundance at the time t , k is the carrying capacity and C_t is the by-catch. In particular, for C_t , different values were given to obtain an N_t sequence that would better fit the observed abundance. One run was done by setting the by-catch value equal to 1, a second run with a value equal to 3 and the last run with the value of 6. The intrinsic rate parameter r (0.09 year^{-1}) was estimated using a life history table model from Barlow and Boveng (1991). The best carrying capacity (k) estimate for the year 1964 found by the model was 240 individuals with a by-catch of 6 individuals per year. Therefore, assuming a mean body weight of 80 kg (Cagnolaro et al., 1983), the average annual biomass was 19.2 t or $0.019 \text{ t} \cdot \text{km}^{-2}$. The P/B was assumed to be equal to M (natural mortality) and it was calculated through a life history table model (Barlow and Boveng, 1991) that estimate survivorship and mortality according to the longevity of a given species. The value found was 0.09 year^{-1} . This value has been consequently modified to 0.05 year^{-1} to account for reduction in reproduction under stress (Chapman and Reiss, 1999) and to fit to time series data. Consumption per unit of biomass was found by using an empirical equation applied by Hunt et al. (2000) to describe energy requirement:

$$E = a \cdot W^{0.75}$$

where E is the energy requirement per day ($\text{kcal} \cdot \text{day}^{-1}$), W the mean body weight (kg) and a is a coefficient varying with the group of mammals ($a=320$ for otariids, 200 for phocids, 192 for mysticetes, 317 for odontocetes, and 320 for sea otters). The coefficient of 0.75, as often used in general for mammals, was subsequently changed to 0.714 following Hunter (2005) who estimated a more precise coefficient for marine mammals. The energy requirement of common dolphins was then divided by the average prey kcal content (1339.4 kcal per kg of prey), which gave a Q/B value of 24.67 year^{-1} .

The diet composition was obtained from stomach contents in the study area and in other neritic ecosystems (Boutiba and Abdelghani, 1995; Silva, 1999; Bearzi, 2000; Pusineri et al., 2004), from fish scale sampling during surface feeding activities in the study area (Agazzi and Bearzi, 2005), and from direct observation on feeding behaviour in the study area and in other Mediterranean coastal waters (Bearzi et al., 2005; Cañadas, 2006). The diet of an adult population is mainly dominated by sardines (50%), anchovies (30%) other pelagics (10%) such as blue whiting and *Trachurus* species, other demersal (5%) such as gobies, and cephalopods (5%). A total of five individuals have been reported by-caught in fishing gears and killed in 2005 (Bearzi, 2006).

2.2.2 Common bottlenose dolphins (*Tursiops truncatus*)

Together with common dolphins, bottlenose dolphins have been studied in the area since 1993. Even in this case, the population assessment has been conducted by Tethys for the years 1995-2006. Results of the study have shown a low density of this community in the area (Bearzi et al., 2005, 2006). Evidence of malnutrition have, however, been observed in high proportion among photo-identified individuals, suggesting a lack of nutrition due to scarcity of fish (Politi et al., 2000; Bearzi et al., 2005). The biomass for the year 1964 was not calculated, as for common dolphins, with the population reconstruction because of the stability shown by the community in the area. Instead, I based my calculation on interviews with fishers that I conducted in 2006.

I gathered qualitative information on population abundance changes, fisheries interactions, and spatial movements. Marine mammal pictures were shown to each interviewee. In particular, the questions were focused on whether the abundance of bottlenose dolphins had increased, remained the same or decreased during their fishing career. To quantify this trend, I asked if the number of bottlenose dolphins were two times more in 1960s than 2006, the same in both years or two times less than 2006. Almost all my interviewees agreed that the population of bottlenose dolphins in the 1960s was double the abundance of the year 2006. Therefore, assuming a mean body weight of 300 kg (Notarbartolo di Sciara and Demma, 1997), the average annual biomass was 13.2 t or $0.013 \text{ t} \cdot \text{km}^{-2}$. P/B and Q/B were estimated, as for common dolphins, through life history table modeling and the energy requirement equation. The P/B value obtained was 0.08 year^{-1} .

The energy requirement of bottlenose dolphins was divided by the average prey kcal content (1074.73 kcal per kg of prey), which gave a Q/B value of 21.07 year^{-1} . Information on bottlenose dolphin diet was taken from the literature (Blanco et al., 2001); moreover, Bearzi et al. (2004b) has also observed an increase of dolphin foraging behaviour alongside the fish farm cages present in the study area. This opportunistic feeding behaviour has been incorporated into the model. The bottlenose dolphin diet is mainly composed of hake (43.9%), other demersals (37.5%) such as European conger, red bandfish, mullus sp., other pelagics (5.1%) such as *Thachurus* sp. and common Pandora, sardines (2.4%), anchovies (6.7%), piscivores with pens (1%) cephalopods (3.1%) and crustaceans (0.3%). In 2005, one bottlenose dolphin was found dead with the flukes cut off by a knife suggesting a by-catch event (Bearzi, 2006).

2.2.3 Mediterranean monk seal (*Monachus monachus*)

Historically, the Mediterranean monk seal (*Monachus monachus*) was a common marine mammal distributed throughout the Mediterranean Sea, the Black Sea, the Atlantic Ocean, and the northwest coast of Africa (Panou et al., 1993; Adamantopoulou et al., 1999; Androukaki et al., 1999). Nowadays it is one of the most endangered marine mammals species of the world, and it is threatened with extinction (IUCN, 1984; IUCN/UNEP, 1988). The total world

monk seal number is estimated to be between 400 and 500 individuals, and about half of them inhabit the Greek coastal waters (Reijnders et al., 1993). Fisheries interactions and loss of habitat have been considered the major causes of its decline (Jacobs and Panou, 1988; Panou et al., 1993; Glain et al., 2001).

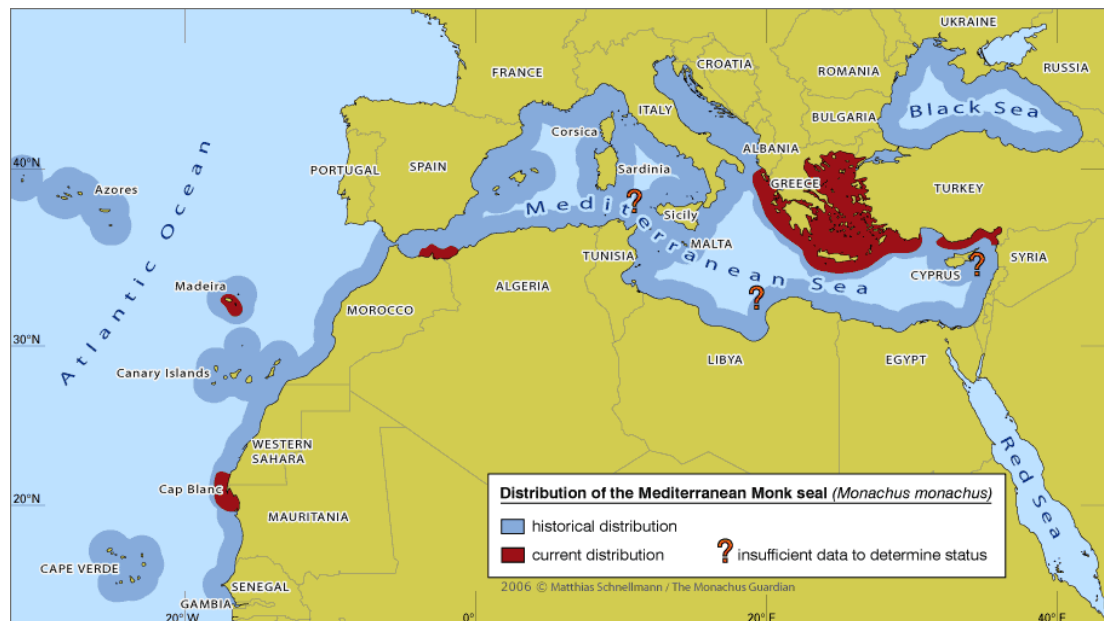


Fig.2.1. Distribution of the Mediterranean monk seal. The blue areas indicate the historical distribution while the red areas are representing the current presence of this marine mammal. The question marks shown in the southern part of the Mediterranean basin indicate insufficient data (map taken from The Monachus Guardian © Matthias Schnellmann).

Today, the Mediterranean monk seal population in the NE Ionian Sea is at a low density (Panou et al., 1993). Data on population assessment were available for the years 1977, 1986 and 1987 (Goedicke, 1981; Panou et al., 1993). The biomass for 1964 was calculated using the population reconstruction model. The result suggests that approximately 50 individuals were inhabiting the area in that period of time. Therefore, assuming a mean body weight of 300 kg (Boulva 1979; Marchessaux 1989), the average annual biomass was 15 t or $0.015 \text{ t} \cdot \text{km}^{-2}$. The life history table model and the energy requirement equation were used to estimate P/B (0.12 year^{-1}) and Q/B (12.59 year^{-1}). The average prey kcal content for the monk seal caloric consumption was 1134.35 kcal per kg of prey. Mediterranean monk seal feeds opportunistically on a large number of prey

(Jacobs and Panou, 1988; Cebrian et al., 1990; Scoullos et al., 1994; Boutiba and Abdelghani, 1997; Salman et al., 2001), in my model, I assumed a diet composed of demersal fish (45%), including mullets and bogues, pelagic fish (20%), such as sea bream species, cephalopods (25%) and crustaceans (10%). Between 1963 and 1987, Panou et al. (1993) reported the death of 25 monk seals, mainly due to deliberate killings and incidental by-catch. In 1989, 3 other individuals, 2 adults and a pup, were killed for the same reason (Cebrian and Vlachoutsikou, 1992).

2.2.4 Sea birds

This group includes Cory's shearwater (*Calonectris diomedea*); Yellow legged gull (*Larus cachinnans*); Lesser black-backed gull (*Larus fuscus*); Gull-billed tern (*Sterna nilotica*); Sandwich tern (*Sterna sandvicensis*) and Great Cormorant (*Phalacrocorax carbo*). The identification of these species of sea birds was possible thanks to photographs taken by Tethys Research Institute in the study area.

Unfortunately no publications were available for sea birds in the NE Ionian Sea; therefore, data on population estimates, mortality and consumption rates were taken from similar ecosystems or from direct observations. In particular, personal communication with Vasiliki S. Karpouzi (Fisheries Centre, University of British Columbia), combined with questionnaire to fishers provided a biomass estimate of $0.0031 \text{ t}\cdot\text{km}^{-2}$. The P/B entered for this group was set as equal to 4.61 year^{-1} based on data determined for a model of the Adriatic Sea (Coll et al., 2007). The consumption rate calculated by Karpouzi et al. (2007) of 105.43 year^{-1} was used for these species. Diet composition was taken from the literature (Fasola et al., 1989; Fasola and Bogliani, 1990; Bogliani et al., 1992; Grieco, 1994; Oro, 1996; Granadeiro et al., 1998): in particular, they feed mainly on sardines (30%), anchovies (30%), other pelagics (5%) such as *Atherina boyeri*, *Oblada melanura*, *Spicara* sp., other demersals (4%) such as *Dentex dentex*, *Dicentrarchus labrax*, *Diplodus annularis*, *D. vulgaris*, *Pagellus* sp., *Pagrus pagrus*, *Sciaena umbra*, *Sparus aurata*, cephalopods (5%), crustaceans (2%), discards (15%) and prey that are not part of the considered marine ecosystem, (e.g., terrestrial prey) (9%). Data have been modified to satisfy mass balance in the model.

2.2.5 Loggerhead turtle (*Caretta caretta*)

The Loggerhead turtle, *Caretta caretta*, is the most common marine turtle in the Mediterranean Sea and the Greek Ionian Islands, in particular Zakynthos and Kefalonia, are considered one of the most important nesting areas for this species (Panou et al., 1999; Margaritoulis, 2000). The Loggerhead turtle is a migratory species; postnesting movements of turtles tagged in Greece showed a wide range of dispersion in the Eastern Mediterranean, with preference mainly for the Adriatic, North Ionian and Aegean Sea (Margaritoulis, 2000).

Until 1994, it was listed as “vulnerable” in the IUCN Red List and became “endangered” in 1996 (Marine Turtle Specialist Group, 1996). During the past twenty years, Greek conservation projects have focused their attention mainly on the protection of nesting beaches and on the monitoring of nesting populations (Panou et al., 1999; Margaritoulis, 2000; Margaritoulis and Rees, 2003; Margaritoulis, 2005). Very little is known about abundance estimates, survival and consumption rates for the loggerhead turtle in the Mediterranean Sea.

For the aim of this study, I calculated the sea turtle biomass from the number of nests found in Kefalonia for the years 1984-2005 (Katelios Group, 2005). It is important to state that the biomass value obtained is underestimated due to the lack of information on other nesting sites present in the study area. The methodology that I used follows the same criteria applied by the Turtle Expert Working Group (1998) for the loggerhead turtle abundance estimate. On average, loggerhead adult females reproduce every 2.5 years (Richardson and Richardson, 1982; Hays, 2000; Broderick et al., 2002; Katelios Group, 2005) and deposit, on each migration, an average of 3.5 nests (Katelios Group, 2005) with approximately 100 eggs per nest (Turtle Expert Working Group, 1998; Katelios Group, 2005).

Therefore, to estimate the population number of loggerhead, I divided the number of observed nests by 3.5 nests/season (in this case 2.5 yrs) to obtain the number of adult females that nest every season. Then, assuming that the number of males in the population is at a 1:1 ratio with the number of females (Turtle Expert Working Group, 1998), and that on average, the weight of an adult turtle is 68 kg (EuroTurtle, 2006), the biomass for the 1984 was $0.0034 \text{ t}\cdot\text{km}^{-2}$. The 1964

biomass was determined using the same methodology applied for bottlenose dolphins and therefore, multiplying by a factor of 2 the biomass of 1984. Mortality rate was calculated from studies on loggerhead survival probabilities in different areas of the world (Bjorndal et al., 2003; Mazaris et al., 2005; Casale et al., 2007). Z was estimated for all 4 age classes (adult, subadult, benthic and pelagic juveniles). The annual average value obtained was 0.19 year^{-1} . For the consumption rate, a value of 2.54 year^{-1} was used (Coll et al., 2006). Sea turtle diet was mainly composed of crustaceans (4%), benthic invertebrates (52%), zooplankton (17%), discards (19%) and detritus (7%) (Houghton et al., 2000; Tomás et al., 2001; Revelles et al., 2007).

2.2.6 Tuna

The two main commercial species of tuna that inhabit the NE Ionian Sea are the bluefin tuna (*Thunnus thynnus*) and albacore tuna (*Thunnus alalunga*). The first one is the largest of the tunas, reaching over 600 kg and 4 m of length and living for up to 30 years. It is a great migratory species, with a speed that is approximately 40 km per hour and it can dive to a depth of 1 kilometer. Its spawning area is principally the Mediterranean Sea with highest concentration in the Balearic Island, the Tyrrhenian Sea, and the central Mediterranean (ICCAT, 2003). Bluefin tuna has been known and fished since ancient times (as described by, e.g., Homer, Herodotus and Aristotle), and at present it is highly depleted with the Mediterranean population being at high risk of extinction (ICCAT, 1997).

The albacore tuna is, on the other hand, one of the smallest tuna species. Its maximum length is estimated to be approximately 130 cm, with a weight of 60 kg and a life span of about 15 years (ICCAT, 2005). Independent spawning areas, different morphometrics, growth rates and larvae distribution make the Mediterranean albacore stock independent from the northern and southern Atlantic populations (ICCAT, 2005).

The main Greek fisheries that target tuna are the purse seiners and the longliners. Unfortunately little is known about Greek tuna catch composition and fishing effort (ICCAT, 2003, 2005). Reported landings of bluefin tuna in the NE Ionian Sea start in 1970, with an estimate of 9.9 t; no data are available for the

albacore tuna (Tsikliras et al., 2007). Skipjack tuna (*Katsuwonus pelamis*) and Little tunny (*Euthynnus alletteratus*) are also caught with respectively 2.21 t (1964) and 0.08 t (1970). Biomass has been obtained from a stock assessment done by ICCAT for the area of Greece (ICCAT 2003). The total biomass was calculated as 0.04 t·km⁻². For the fishing mortality F , I used an average value estimated by ICCAT (2003) while natural mortality M was calculated through the empirical model of Pauly (1980):

$$M = K^{0.65} \cdot L_{\infty}^{-0.279} \cdot T_c^{0.463}$$

where M is the natural mortality (year⁻¹), K is the curvature parameter of the von Bertalanffy Growth Function (VBGF) (year⁻¹), L_{∞} is the asymptotic length (cm) and T_c is the mean water temperature (°C). The P/B was 0.2 year⁻¹; Q/B (3.22 year⁻¹) was determined using the empirical equation mentioned above. All the required parameters used in these two equations were obtained from *Fishbase* (www.fishbase.org) (Froese and Pauly, 1995). The diet of tuna has been determined through literature information (Orsi-Relini et al., 1998; Stergiou and Karpouzi, 2002; *Fishbase*): it is principally made of sardines (20%), anchovies (30%), other pelagics such as European sprat, bluefish and Atlantic mackerel (10%), cephalopods (25%) and crustaceans (15%). Data have been modified to balance the model.

2.2.7 Swordfish

Swordfish, *Xiphias gladius*, is a large pelagic marine fish, highly migratory and distributed worldwide. Its size can reach 4.5 meters of length (fork length) and 650 kg (Tserpes et al., 2006; Damalas et al., 2007). Genetic studies have confirmed that the Mediterranean swordfish population form a unique stock with independent growth and reproductive characteristics from the adjacent Atlantic Ocean (Tserpes et al., 2001; ICCAT, 2004; Tserpes et al., 2006).

In the Mediterranean, the most important spawning areas are the ones around the Balearic Islands, the southern and central Tyrrhenian Sea, the Ionian Sea, and the Strait of Messina. Juveniles can be found in the entire Mediterranean basin but often they concentrate along the coasts (ICCAT, 2004). Recent studies conducted by ICCAT (2004) have shown how catch rates in

different areas of the Mediterranean (Italy, Greece, Spain), did not present any particular trend over time, suggesting that the level of exploitation of swordfish may be sustainable. Unfortunately, lack of historical data on catches per unit effort and landings, does not permit determination of the current Mediterranean swordfish population relative to its unexploited stock, precluding, therefore, any future stock assessments (ICCAT, 2004).

The Greek swordfish fishery developed in the early 1970s (De Metrio et al., 1988) and nowadays extends from the Eastern Ionian Sea to the Levantine Sea. The fishing gear that is mainly used to catch swordfish is the longline net; the fishing season starts in February and closes in September (Tserpes et al., 2006). The first reported catch for the NE Ionian Sea, is in 1970 with $0.007 \text{ t}\cdot\text{km}^{-2}$ (Tsikliras et al., 2007). The 1964 biomass was calculated from a reconstruction population model; the value obtained was $0.06 \text{ t}\cdot\text{km}^{-2}$. The total mortality used was 0.38 year^{-1} (from Megalofonou et al., 1987, for the year 1986-1987); this value was then reduced to 0.30 year^{-1} to take into account variations of fishing mortality for the year 1964 and to fit the model to time series data. Q/B (4.4 year^{-1}) was calculated through the empirical equation shown above. The value was then modified to balance the model. Swordfish feeds mainly on sardines (19%), anchovies (46%), other pelagics such as European sprat, mackerels and seabreams, (9%), hake (1%), other demersal species such as bogue (1%), cephalopods (17%) and crustaceans (7%) (Peristeraki and Tserpes, 2001; Stergiou and Karpouzi 2002; Salman, 2004; Peristeraki et al., 2005). Diet data were changed to balance the model.

2.2.8 Sardines

The species of sardines considered in this model are European pilchard (*Sardina pilchardus*) and round sardinella (*Sardinella aurita*). The first is one of the most important small-sized pelagic fish of Greece, mostly harvested for human consumption; the second is mainly used in the canning industry or as a bait for other pelagic fisheries targeting bluefin tuna and swordfish (Voulgaridou and Stergiou, 2003; Tsikliras et al., 2005). The total length of a sardine can reach 18 cm, but varies depending on geographic area; the total weight reached is

approximately 54 g (Voulgaridou and Stergiou, 2003, Tsikliras et al., 2005, *Fishbase*).

Sardines are multiple spawners that can release several batches of eggs within a reproductive season (Somarakis et al., 2001); eggs and larvae are generally distributed in inshore waters at a depth of 40 to 90 m (Somarakis et al., 2006). The spawning season occurs between November and April and in the Ionian Sea the reproductive activity last for 2-4 months (Voulgaridou and Stergiou, 2003; Somarakis et al., 2006). Moreover, Somarakis et al. (2006) has demonstrated, using a daily egg production method, that the coastal waters of NE Ionian Sea are a good refuge for spawning females and are favourable nursery areas for their eggs and larvae (Giannoulaki et al., 2001; Somarakis et al., 2006).

Sardines dominate the mean total reported landings of Greece with 14.1% for the period 1989-2003. They are mainly caught by purse seiners, constituting 25.6% of their total catch, beach seiners with a 11.7% and small artisanal fisheries (e.g. trammel nets) with a 9.2% (Stergiou et al., 2007). In the NE Ionian Sea the mean total catch for the year 1964 was $0.19 \text{ t}\cdot\text{km}^{-2}$ (Tsikliras et al., 2007). The stock biomass ($2.1 \text{ t}\cdot\text{km}^{-2}$) was determined through the reconstruction population model; P/B was calculated using the following equation (Beverton and Holt, 1957):

$$Z = K \cdot (L_{\infty} - L) / (L - L')$$

where L_{∞} is the asymptotic length, K is the VBGF curvature parameter, L is the mean length in the population, and L' represents the mean length at first capture. Required values were taken from Stergiou et al. (1997) and *Fishbase*. P/B was 1.55 year^{-1} . This value was calculated using data from the years 1983-1984; it was, subsequently decreased to 0.7 year^{-1} for the year 1964 to better fit the biomass curve. Q/B was 9.84 year^{-1} . It was slightly decreased to balance the model. Sardine diet is mainly composed of zooplankton (80%) and phytoplankton (20%) (Stergiou and Karpouzi, 2002; Cunha et al., 2005; Somarakis et al., 2006).

2.2.9 Anchovy

The European anchovy, *Engraulis encrasicolus*, is abundant in the entire Mediterranean basin. It can reach a maximum length of 14 cm and a maximum weight of 18 g. As for sardine, European anchovy is a multiple spawner; studies conducted since 1972 have shown a high concentration of spawning stock biomass, eggs and larvae in the inshore waters of the NE Ionian Sea, especially around Kalamos island (Yannopoulos and Yannopoulos, 1976; Machias et al., 2001a; Somarakis et al., 2002).

Together with the European pilchard, it constitutes the most important commercial pelagic fish of Greece (Stergiou et al., 1997; Nikoloudakis et al., 2000). For the period 1989-2003, the European anchovy dominated the mean total reported landings of Greece with 14.6%. It is mainly caught by purse seine with 32.7% of the total catch, and in a smaller proportion by small scale artisanal fishery with only 4% of the total catch (Stergiou et al., 2007). In the NE Ionian Sea the mean total catch for the year 1964 was $0.13 \text{ t} \cdot \text{km}^{-2}$ (Tsikliras et al., 2007). The stock biomass ($1.2 \text{ t} \cdot \text{km}^{-2}$) was determined using the population reconstruction model; for the total mortality (1.3 year^{-1}) I applied the same method as used for sardine. Parameters were obtained from *Fishbase* and Stergiou et al. (1997). This value was decreased to 0.8 year^{-1} to better fit the biomass trends. Consumption over biomass was 11.66 year^{-1} . It was modified successively to balance the model. The diet was taken from the literature. Zooplankton is its only prey (100%) (Tudela and Palomera, 1995; Stergiou and Karpouzi, 2002).

2.2.10 Other pelagics

This group includes a number of commercial pelagic species of the Greek fisheries such as common dentex (*Dentex dentex*); annular seabream (*Diplodus annularis*); blue whiting (*Micromesistius poutassou*); flathead mullet (*Mugil cephalus*); saddled seabream (*Oblada melanura*); common Pandora (*Pagellus erythrinus*); common seabream (*Pagrus pagrus*); Atlantic bonito (*Sarda sarda*); salema (*Sarpa salpa*); chub mackerel (*Scomber japonicus*); Atlantic mackerel (*Scomber scombrus*); greater amberjack (*Seriola dumerili*); picarel (*Spicara*

smaris); black seabream (*Spondyllosoma cantharus*); European sprat (*Sprattus sprattus*); Mediterranean horse mackerel (*Trachurus mediterraneus*); Atlantic horse mackerel (*Trachurus trachurus*) (Papaconstantinou, 1986; Stergiou and Papaconstantinou, 1987; Tsikliras et al., 2007).

Among these, flathead mullet, chub mackerel, picarel and Mediterranean horse mackerel dominated the fisheries catches with respectively 2.4%, 4.2%, 5.6% and 6.4% of the total catch for the period 1982-2003 (Stergiou et al., 2007). In 1964 the mean reported landing was $0.39 \text{ t}\cdot\text{km}^{-2}$ (Tsikliras et al., 2007). The stock biomass obtained with the reconstruction model was $1.95 \text{ t}\cdot\text{km}^{-2}$; P/B was estimated by taking into account the different mortalities of these pelagic fish. The resulting average was 1.95 year^{-1} . This estimate was then modified to balance the model. Q/B was 7.70 year^{-1} . The parameters were taken from the literature (Papaconstantinou et al., 1988b; Papaconstantinou and Petrakis, 1989; Stergiou et al., 1997; Stergiou and Moutopoulos, 2001; Abaunza et al., 2003; Ragonese et al., 2004; Chilari et al., 2006; Zaboukas et al., 2006) and from *Fishbase*. The diet of other pelagics was mainly extrapolated from the literature (Daan, 1989; Papaconstantinou and Caragitsou, 1989; Gonçalves and Erzini, 1998; Ticina et al., 2000; Stergiou and Karpouzi, 2002; Olaso et al., 2005). I assumed 5% sardines, 5% anchovies, 5% other pelagics, 5% other demersals, 49% benthic invertebrates and 35% zooplankton. This diet has been adjusted to balance the model.

2.2.11 Hake

The hake, *Merluccius merluccius*, is a very common and commercially important demersal fish that inhabits the Greek seas (Vassilopoulou and Papaconstantinou, 1988; Papaconstantinou and Stergiou, 1995). In the Ionian Sea, hake can reach a maximum length of 72 cm and a maximum weight of 3.3 kg (Papaconstantinou and Stergiou, 1995). They spawn in very deep waters that range between 75 and 250 meters (Vassilopoulou and Papaconstantinou, 1988).

In Greece, hake represents 3.5% of the mean total landing; trawling is the main fishery, landing 10% of the total catch (Stergiou et al., 2007). In 1964 the total landing of the NE Ionian Sea was $0.022 \text{ t}\cdot\text{km}^{-2}$ (Tsikliras et al., 2007). Stock

biomass was available for the period 1994-1999 (Orsi-Relini et al., 2002); the 1964 biomass ($0.18 \text{ t}\cdot\text{km}^{-2}$) was estimated with the reconstruction model. P/B was obtained using a total mortality of 0.82 year^{-1} (Papaconstantinou and Stergiou, 1995); this value has been modified to balance the model. Q/B (3.26 year^{-1}) was determined using parameters found in the literature (Papaconstantinou and Stergiou, 1995); this value has been modified to balance the model. Diet is mostly composed of sardines (22%), anchovies (15%), other pelagics (13%) such as *Spicara smaris* and *Trachurus* sp., other demersals (7%) such as *Callionymus maculatus* and gurnard *Lepidotrigla cavillone*, cephalopods (4%), crustaceans (8%) benthic invertebrates (32%) (Papaconstantinou and Caragitsou, 1987).

2.2.12 Other demersals

This group includes the major commercial demersal species caught in Greece. In particular, I considered bogue (*Boops boops*); gurnard (*Lepidotrigla cavillone*); European conger (*Conger conger*); European seabass (*Dicentrarchus labrax*); white seabream (*Diplodus sargus sargus*); Dusky grouper (*Epinephelus marginatus*); Blackbelly rosefish (*Helicolenus dactylopterus dactylopterus*); anglerfish (*Lophius piscatorius*); red mullet (*Mullus barbatus*) grey mullet (*Mullus surmuletus*); wreckfish (*Polyprion americanus*); turbot (*Psetta maxima*); thornback ray (*Raja clavata*); common sole (*Solea solea*); gilthead seabream (*Sparus aurata*); Scorpaenidae and Serranidae (Papaconstantinou, 1986; Tsikliras et al., 2007).

Among the many species, red mullet and bogue dominated the fisheries catches during 1982-2003 with respectively 2.1% and 7.0% of the total catch (Stergiou et al., 2007). In 1964, in the NE Ionian Sea, the mean reported landing for this group was $0.15 \text{ t}\cdot\text{km}^{-2}$ (Tsikliras et al., 2007). The stock biomass obtained with the reconstruction model was $0.9 \text{ t}\cdot\text{km}^{-2}$; P/B was estimated by averaging all the different mortalities. The result was 1.35 year^{-1} . Q/B was 7.425 year^{-1} . The parameters were taken from the literature (Tsimenides and Ondrias, 1980; Papaconstantinou et al., 1988a; Stergiou et al., 1997; Stergiou and Moutopoulos, 2001) and from *Fishbase*. Their diet was mainly extrapolated from the literature (Papaconstantinou and Caragitsou, 1989; Papaconstantinou et al., 1989;

Labropoulou and Papadopoulou-Smith, 1999; Stergiou and Karpouzi, 2002). I assumed 1% sardines, 1% anchovies, 4% other pelagics, 2% other demersals, 10% crustaceans, 80% benthic invertebrates, 1% zooplankton and 1% detritus. This diet has been adjusted to balance the model.

2.2.13 Pen-associated planktivores

This group was inserted in the model to take into consideration the diet of bottlenose dolphin and its opportunistic feeding behaviour. Different studies have demonstrated that fish farms have a positive effect on wild fish abundance especially in the vicinity of the fish cages (Dempster et al., 2002; Machias et al., 2004, 2005; Giannoulaki et al., 2005). It has been demonstrated, indeed, that sea-cage fish farms act as 'super-FADs' (FAD: fish aggregation device) attracting a large variety of fish (Dempster et al., 2002) because they can feed on large food pellets lost through the cage, and/or particulate organic matter (POM) of broken pellets and faeces (Dempster et al., 2002; Giannoulaki et al., 2005; Machias et al., 2005).

In Greece, the number of marine fish farms has increased rapidly during the period 1985-2002. In particular, European sea bass (*Dicentrarchus labrax*) and the gilthead sea bream (*Sparus aurata*) dominate Greek aquaculture, with respectively 48% and 50% of total production. The finfish aquaculture mainly consists of floating cages along the coastal waters (Conides and Kevrekidis, 2005). In the study area, the marine fish culture has also developed rapidly through the study period. Initially it was mainly composed of European sea bass (*Dicentrarchus labrax*), and gilthead sea bream (*Sparus aurata*). Recently, however, white seabream (*Diplodus sargus sargus*), brown meagre (*Sciaena umbra*), and common Pandora (*Pagellus erythrinus*) were added (Conides and Kevrekidis, 2005).

For my model, I decided to incorporate time series data of the number of cages in the study area to better understand and explain the biomass curve of the bottlenose dolphins. In particular, the number of cages of the study area was estimated through high-definition satellite images taken from GoogleEarth (2007). No information were available on time series data for the NE Ionian Sea; therefore, this information was taken for the whole of Greece for the period 1990-

2002 (Conides and Kevrekidis, 2005). Using Conides and Kevrekidis data, I extrapolated the increase in number of cages to the study area, thus obtaining the following trend (see fig.2.2). In absence of available data for the years between 2002 and 2006, I assumed a constant value from the year 2002 to the year 2007.

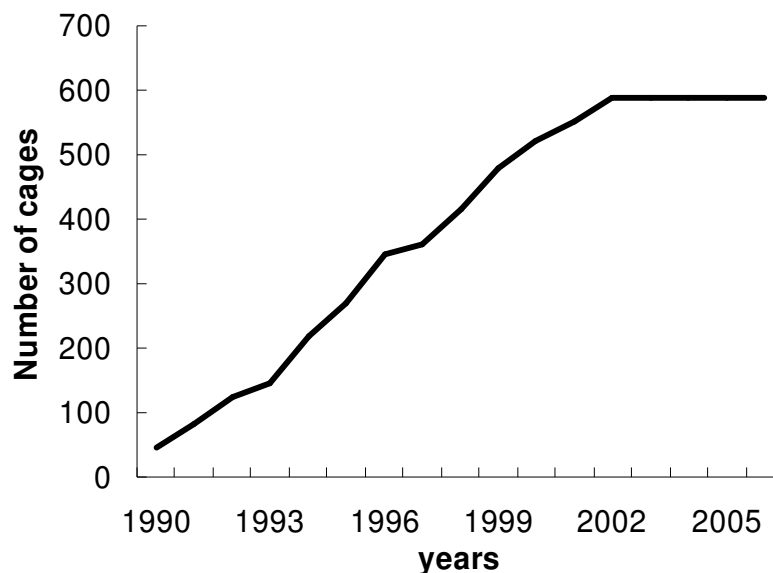


Fig.2.2. Trends in fish farm number of cages for the NE Ionian Sea.

The 1964 biomass for this functional group was assumed to be very low, due to the small number of fish farms in the area. Thus, the biomass was set to $0.002 \text{ t}\cdot\text{km}^{-2}$; P/B (1.275 year^{-1}) and Q/B (7.5625 year^{-1}) were calculated by averaging the data obtained previously for pelagic and demersal fish. Furthermore, these planktivores were forced to feed only the zooplankton (90%) and phytoplankton (10%) originated in the vicinity of the farm pens.

2.2.14 Cephalopods

This group considered most of the commercial cephalopods caught in Greece: various squids of the Loliginidae and Ommastrepidae families; the common cuttlefish (*Sepia officinalis*); common octopus (*Octopus vulgaris*) and other members of the family Octopodidae (Tsikliras et al., 2007).

In 1964 the mean reported catch was $0.017 \text{ t}\cdot\text{km}^{-2}$ (Tsikliras et al., 2007). Biomass information was available for the years 1999 and 2000 (Lefkaditou et al.,

2003; Politou et al., 2003). The 1964 stock biomass was estimated using the population reconstruction model: $0.4 \text{ t}\cdot\text{km}^{-2}$ was the calculated value; P/B (3.3 year^{-1}) and Q/B (7.0 year^{-1}) were determined using general values for octopi and squids. Q/B was calculated using the daily feeding rate equation used by Rodhouse and Nigmatullin (1996):

$$FR = 0.0683 + 0.0474 W$$

where FR ($\text{g}\cdot\text{day}^{-1}$) is the feeding rate and W is the body mass. The parameters were taken from the literature (Guerra, 1979; Rodhouse and Nigmatullin, 1996) and from *Cephbase* (www.cephbase.org). Cephalopods feed mostly on sardines (20%), anchovies (10%), other pelagics (5%), other demersals (6%), cephalopods (3%), crustaceans (6%) and benthic invertebrates (50%). (Rodhouse and Nigmatullin, 1996; *Cephbase*). This diet has been adjusted to balance the model.

2.2.15 Crustaceans

The main commercial crustaceans considered in the model were the European lobster (*Hommarus gammarus*); the caramote prawn (*Penaeus kerathurus*); the Mediterranean shore crab (*Carcinus aestuarii*); the Norway lobster (*Nephrops norvegicus*) and miscellaneous small crustaceans (Tsikliras et al., 2007).

The mean reported landing for the year 1964, for the NE Ionian Sea was $0.011 \text{ t}\cdot\text{km}^{-2}$ (Tsikliras et al., 2007). Data on biomass was available for the year 2000 (D'Onghia et al., 2003; Politou et al., 2003). The 1964 biomass was $0.05 \text{ t}\cdot\text{km}^{-2}$ that was then increased to balance the model. The total mortality (1.35 year^{-1}) was an average value of all the Z values found for each species of crustaceans (Abelló et al., 2001, 2002; Conides et al., 2006). Q/B (6.5 year^{-1}) was taken from the literature (Cartes and Maynou, 1998; Maynou and Cartes, 1998). The diet was composed of other demersals (5%), cephalopods (1%), crustaceans (12%), benthic invertebrates (80%), zooplankton (1%) and discards (1%) (Cartes, 1995; Sarda' and Cartes, 1997; Cartes and Maynou, 1998; Maynou and Cartes, 1998; Kapiris, 2004). Diet data have been adjusted to balance the model.

2.2.16 Benthic invertebrates

This group considered a variety of organisms, including polychaetes, molluscs, echinoderms and other minor benthic groups. Several studies have been conducted in the study area to determine the composition and the abundance of these invertebrates (Pancucci and Zenetos, 1985; Zenetos, 1993; Zenetos et al., 1997).

Values of dry weight biomass were obtained from Zenetos (1993). The mean annual biomass was converted into wet weight following the conversion factors in Cauffope' and Heymans (2005), resulting in an estimate of $28.39 \text{ t}\cdot\text{km}^{-2}$. For P/B (2.5 year^{-1}) and Q/B (22.0 year^{-1}), I used the same parameters as given in Brey (2001). The diet of this group consists of zooplankton (0.6%), phytoplankton (5%), discards (0.001%) and detritus (94%) (Brey, 2001).

2.2.17 Zooplankton and pen-associated zooplankton

Copepods, cladocerans, appendicularians and chaetognats are the most abundant groups of zooplankton in the NE Ionian Sea, with an abundance that decreases with depth between 50 to 200 meters (Moraitou-Apostolopoulou and Vournazou, 1978; Rottini and Fabris, 1978; Pagou et al., 1989; Stergiou et al., 1997; Siokou-Frangou et al., 2005; Ramfos et al., 2006).

However, studies have demonstrated that the abundance of zooplankton, in these oligotrophic coastal waters, is one of the lowest recorded in the Mediterranean (Stergiou et al., 1997; Siokou-Frangou et al., 2005; Ramfos et al., 2006). Values of dry weight biomass were available for the year 2000, 1998, 1988, 1987 (Siokou-Frangou and Papathanassiou, 1989; Panayotidis et al., 1994; Ramfos et al., 2006). The 1987 estimate was taken as representative for my model; I transformed this value to wet weight following the conversion factors shown in Cauffope' and Heymans (2005). The resulting value was $3.7 \text{ t}\cdot\text{km}^{-2}$. P/B was estimated (31 year^{-1}) using zooplankton production found by Christou (1991); this value was subsequently reduced to balance the model. For Q/B (40 year^{-1}), no data was available for the Hellenic Sea, so I took a mean value used for the Adriatic model (Coll et al., 2006). This value was then reduced to balance the model. The diet consists of phytoplankton (90%) and zooplankton (10%).

Pen-associated zooplankton is a functional group that was mainly created to increase the productivity around fish farms. The biomass for 1964 was set to a very low value, $0.001 \text{ t} \cdot \text{km}^{-2}$ and then slightly changed to balance the model; P/B and Q/B were the same as calculated for the previous zooplankton group. Pen-associated zooplankton was forced to feed on only the pen-associated zooplankton (10%) and pen-associated phytoplankton (90%).

2.2.18 Phytoplankton and pen-associated phytoplankton

As already mentioned in Chapter 1, the productivity of the NE Ionian Sea is very low and, in particular, phytoplankton abundance of the area is among the lowest in Greece (Pagou et al., 1989; Pagou and Gotsis-Skretas, 1990; Gotsis-Skretas et al., 1993; Panayotidis et al., 1994; Stergiou et al., 1997; Pitta et al., 1998). Dinoflagellates, in particular *Gymnodium* sp., are the main species that can be found in these coastal waters, especially in summer and spring (Pagou et al., 1989; Pagou and Gotsis-Skretas, 1990; Gotsis-Skretas et al., 1993; Stergiou et al., 1997). An abundance estimate (6220 cells/l) was obtained from Pagou and Gotsis-Skretas (1989); this value was converted into $\text{t} \cdot \text{km}^{-2}$, using the following steps. First I calculated the cell carbon content of *Gymnodium* sp. as a function of a cell volume through the equation used by Mullin (1966):

$$\text{Log}_{10}C = 0,76 \cdot \text{log}_{10}V - 0,29$$

where C is the cell carbon in picograms and V is the cell volumes in μm^3 ; the volume was obtained from Banse (1976). The value (in pgC/cell) was multiplied by the biomass (cells/l) in order to get a biomass in gC/l. By assuming that 1gC is equal to 9g of wet weight (Pauly and Christensen 1995), the calculated biomass was $2.33 \text{ t} \cdot \text{km}^{-2}$. The mean primary production for my marine ecosystem was obtained for the years 1998-2002 (Inland and Marine Waters Unit (IMW), Institute for Environment & Sustainability, EU Joint Research Center (JRC), Ispra, Italy). The 1998 data ($378.96 \text{ t} \cdot \text{km}^{-2} \text{ year}^{-1}$) was divided by the biomass estimate giving a P/B of 162.65 year^{-1} . This value was then increased to balance the model.

Pen-associated phytoplankton, as mentioned earlier, was added into the model to increase the production around the fish farms. The biomass for 1964 was set to a very low value, $0.001 \text{ t} \cdot \text{km}^{-2}$ and P/B and Q/B were the same as calculated for the previous phytoplankton group.

2.3 Fisheries parameters

Fisheries information was obtained from direct observations conducted since November 2006 by Tethys Research Institute in the study area and from estimated parameters for the entire Greece (Stergiou et al., 2007; Tsikliras et al., 2007). A total of 266 fishing boats were observed of which 11 were purse seiners, 7 were trawlers, 24 were beach seiners, 30 were longliners, and 194 trammel net boats.

The mean annual landings for the year 2007, for purse seiners ($1231 \text{ t} \cdot \text{year}^{-1}$), beach seiners ($512 \text{ t} \cdot \text{year}^{-1}$) and trawlers ($418 \text{ t} \cdot \text{year}^{-1}$) were calculated by multiplying the observed mean catch value (Tethys unpublished data) with the mean number of days at sea (Kapadagakis et al., 2001) and with the number of observed boats. The species composition of the mean landings per fishery was taken from direct observations (Tethys, unpublished data) and from Stergiou et al. (2007). Unfortunately, no landing data were recorded in the study area for longline and trammel net fisheries. Therefore, the mean catch value and the mean number of days at sea were assumed to be equal to the one obtained for the entire Ionian Sea by Kapadagakis et al. (2001).

In order to take into consideration different landings sizes from different boat sizes, trammel nets and longline fleets were split into two categories. Small artisanal boats with a length, that ranged between 4 and 7 m, and large artisanal boats with a length greater than 7 meters. Moreover, I assumed one fisher per boat for the small boats and two fishers for the big artisanal ones.

Based on these assumptions, the mean total landings for the coastal boats (small boats and big boats) was $1244 \text{ t} \cdot \text{year}^{-1}$; furthermore, the mean estimated landings/fisher ratio was equal to 3.48 t. This result agrees with the value obtained by Stergiou et al. (2007) for the whole of Greece, where the landings/fisher ratio for the coastal boats was 3.42 t. The fisheries landings

calculated for the year 2007 shows that purse seiners catch the most with 36% of the total catch, followed by trammel netters 31%, beach seiners 15%, trawlers 12% and longliners 6%. The same percentage was used for the year 1964 with the mean total landings estimated by Tsikliras et al. (2007).

As mentioned in Chapter 1, the mean total landings were taken from the statistical fishing subarea n.4 (Fig.1.4). The mean total landings of this subarea included the study area and also Amvrakikos Gulf. Amvrakikos is a semi-enclosed gulf, situated north of the study area that, because of its eutrophic features, differs completely from oligotrophic ecosystems such as the Kalamos area. In order to separate the landings of these two different ecosystems, I eliminated from the reported landings, species that occurred mainly in brackish waters, (e.g., lagoons, gulfs) using information gathered from *Fishbase*. Secondly, I assumed that until 1990, in Amvrakikos, species like European pilchard, caramote prawn, red mullet and white seabream represented 40% of total landings of subarea n.4, while European seabass and gilthead seabream was a 10%, and all other marine species 5%. After 1990, a severe depletion has occurred for the target species (European pilchard, caramote prawn, and red mullet), constituting, therefore, only 10% of total catches; the main part of landings consisted of European seabass, gilthead seabream and white seabream with 40% (D. Moutopoulos, pers. comm.).

Fishing effort was not available for this study area; therefore, I extrapolated a time series fishing effort expressed as horse power ($\text{HP} \cdot \text{year}^{-1}$) for each type of fishery, using data from the Ionian Sea from Kapadagakis et al. (2001) for the period 1996-2000 and from Stergiou et al. (2007) for Greece for the period 1964-1995 and 2001-2003. The resulting trends are shown in Fig. 2.3 and Fig. 2.4.

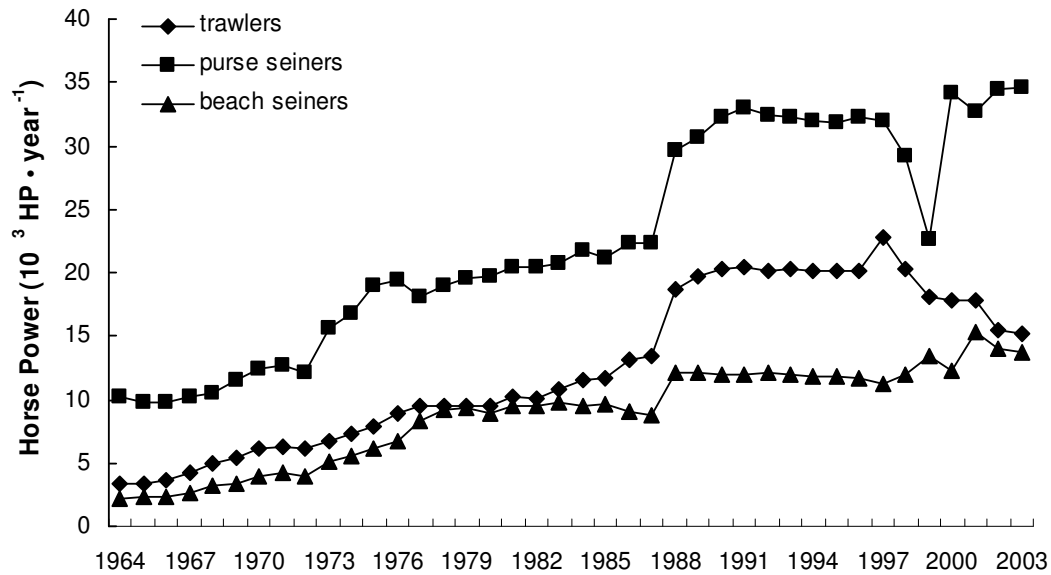


Fig.2.3. Fishing effort ($10^3 \text{ HP} \cdot \text{year}^{-1}$) for trawlers, purse seiners and beach seiners.

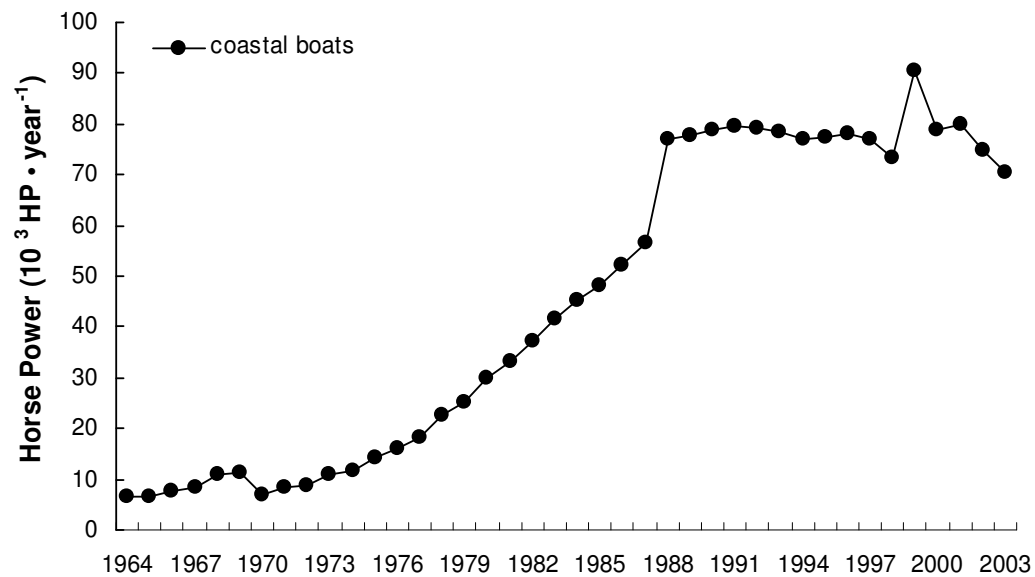


Fig.2.4. Fishing effort ($10^3 \text{ HP} \cdot \text{year}^{-1}$) for coastal boats.

Moreover, in 2006 I conducted, with the approval of the University of British Columbia Ethical Review Committee (see Appendix B), a series of interviews with fishers within the study area. In particular, 30 fishers, of different class ages, were picked randomly in six different cities: Astakos, Mytika, Lefkada, Vathi, Perigiali and Nidri (see Fig.1.1). The questionnaires were designed to

gather qualitative information especially on population abundance changes, fisheries activities and spatial movements. The questions were mainly focused on whether the abundance of certain species had increased, remained the same or decreased during the fishing careers of the interviewees. Marine mammals, birds and fish pictures were shown to each interviewee. A nautical map was also used for spatial fishing fleets movements.

2.3.1 Discards

Discard rates and discard species composition were taken from different sources: in particular, trawlers discard 39% of the total catch (Machias et al., 2001b), purse seiners 14% (Tsimenides et al., 1995), beach seiners 28% (Stergiou et al., 1996), netters and longliners 9.8% (Tzanatos et al. 2007). These percentages were used for the calculation of discard rates for the 1964 model. $0.1346 \text{ t} \cdot \text{km}^{-2}$ was the calculated value.

2.4 Ecopath with Ecosim

Among all, *Ecopath with Ecosim* (EwE) is the most used ecosystem model throughout the world and the number of publications that use it has increased exponentially in the last decade (Christensen and Walters, 2005). *Ecopath with Ecosim* is a software developed at the Fisheries Centre of the University of British Columbia that describes the marine species within the ecosystems, their trophic interactions and the fishing pressure on a particular ecosystem (Christensen and Pauly, 1992, 1993, 1995).

Ecopath was created by Polovina (1984) and subsequently improved by Christensen and Pauly (1992) and Walters et al. (1997). It is a mass-balanced model that provides a static description of an ecosystem at a precise period in time (Walters et al., 1997). EwE can describe all the principal autotrophs and heterotrophs species individually or by aggregating them into functional groups (species with similar ecotrophic role) and inserting data on their biomass, consumption, production and efficiency; furthermore, fishing activities are included by adding data on landings, discards and bycatch as well as bioeconomic parameters (i.e., value and cost).

2.4.1 Ecopath

The *Ecopath* model is based on two main equations. In the first one (see Eq.1), the biological production of a functional group is equal to the sum of fishing mortality, predation mortality, net migration, biomass accumulation, and other unexplained mortality.

$$(P/B)_i \cdot B_i = Y_i + B_j \cdot (Q/B)_j \cdot DC_{ji} + E_i + BA_i + (P/B)_i \cdot B_i (1 - EE_i) \quad \text{Eq.1.1}$$

where (P/B) is the production to biomass ratio for a certain functional group (i), B_i is the biomass of a group (i), Y_i the total fishery catch rate of group (i), $(Q/B)_j$ is the consumption to biomass ratio for the predator j, DC_{ji} is the proportion of the group (i) in the diet of predator (j), E_i is the net migration rate (emigration – immigration), BA_i is the biomass accumulation rate for the group (i), EE_i is the ecotrophic efficiency, and $(1 - EE_i)$ represents mortality other than predation and fishing. In the second equation (see Eq. 1.2), the consumption of a functional group is equal to the sum of production, respiration and unassimilated food.

$$\text{Consumption} = \text{production} + \text{respiration} + \text{unassimilated food} \quad \dots 2)$$

The implication of these two equations is that the model is mass-balanced; under this assumption, *Ecopath* uses and solves a system of linear equations (one for each functional group presents in the system) estimating missing parameters (Christensen and Pauly, 1995; Christensen and Walters, 2004). Therefore, the input parameters (B, P/B, Q/B, and DC) are entered first, and then the mass-balance in the model is ensured. To do so, the model is modified by adjusting the input parameters of those groups with $EE > 1$ (Christensen and Walters, 2004). The procedures used to balance the model are described in the following Chapter.

2.4.2 Ecosim

Ecosim provides temporal simulations using the initial parameters of the *Ecopath* master equation (eq.1.1). It works with a couple of differential equations to estimate biomass fluxes as follow:

$$dB_i/dt = g_i \sum_j Q_{ji} - \sum_j Q_{ij} + I_i - (M_i + F_i + e_i)B_i \quad \dots 3)$$

where dB_i/dt is the biomass growth rate of group (i) during the interval dt , g_i is the net growth efficiency (production/consumption ratio), I_i is the immigration rate, M_i and F_i are natural and fishing mortality rates of group (i), e_i is emigration rate (Walters et al., 1997; Christensen et al., 2005; Pauly et al., 2000; Pauly et al., 2002; Christensen and Walters, 2004). Another important aspect of *Ecosim* is its ability to describe the interactions between predators and prey by attributing a vulnerability term for each of these interactions. This vulnerability parameter shows the maximum increase in predation mortality a given predator can cause on a given prey. Low values of vulnerability (close to 1) mean that prey production determines the predation mortality (phenomenon also known as 'bottom-up' control) while high values of vulnerability, (e.g., 100) mean that predators biomass impact how much prey is consumed (top-down control) (Christensen and Walters, 2004).

After the model was balanced, *Ecosim* was used to run dynamic simulations starting from the 1964 *Ecopath* model. In order to do that, I used time series data for the period 1964-2006 considering, especially, biomass, bycatch, CPUE and catches for the most important groups of my ecosystem, while driving the model with effort time series. A table with time series data is shown in the Appendix. More precisely, *Ecosim* output scenarios were compared to the 'observed' time series data in order to get a goodness of fit. The goodness of fit is calculated by *Ecosim* as a weighted sum of squared differences (SS) between the log 'observed' and log 'predicted' data (Christensen et al., 2005). The fit that best represents the 'observed' data was chosen. The model was fitted by changing the vulnerabilities of certain prey and/or by searching for nutrient anomalies.

In order to evaluate the impact of environmental factors on the trophic organisms I used an automated procedure in *Ecosim* called 'forcing function'; in my case, the forcing function was applied to simulate a time series of nutrient loading within the ecosystem. In particular, this procedure allowed the estimation of changes in nutrient concentration which could impact differently the primary production rates for the producer groups. In other words, the nutrient anomalies could change the total amount of energy that enters in the system allowing a cascading up the food web that increases or decreases the food availability all

through the ecosystem (Preikshot, 2007). Nutrient anomalies were used in the model to improve the trajectories of the predicted time series of biomass, in particular for anchovy and sardine groups.

A 'trophic mediation function' was utilized in the model to assess the possibility that changes in mortality or in consumption of one organism may have indirect effects on other functional groups. More precisely, the mediation function was used to evaluate if the decline of common dolphins have had any indirect effect on the bottlenose dolphin population. Furthermore, to assess if bottlenose dolphins abundance was related to the increase of fish farm cages present in the study area, I have included in the model three other functional groups to represent fish farms, i.e., pen-associated planktivores, pen-associated zooplankton and pen-associated phytoplankton. The bottlenose dolphins were allowed to feed on the pen-associated planktivores and time series number of pens from 1990 to 2006 were added to the time series data as a driver.

Moreover, other parameters were changed to fit the model. First of all, the maximum relative feeding time was set to 10 for the marine mammals and to 2.0 (default value) for the other functional groups, considering that if the prey becomes more scarce, marine mammals will spend more time looking for their food resources (Christensen et al., 2005). The feeding time adjustment rate was set to 0.5 for marine mammals and sea birds and to 0 for the other groups. This is because marine mammals and sea birds may change their search feeding time as food availability varies.

Fishing mortality can be expressed as the product of catchability times fishing effort, where catchability represents the proportion of the stock removed by a unit of fishing effort. *Ecosim* default considers the fishing effort (f_0) equal to fishing mortality (F_0), assuming a catchability (q_0) equal to 1, where f_0 , F_0 , and q_0 are the baseline fishing effort, fishing mortality rate, and catchability provided by *Ecopath*. Therefore, if the stock size of a certain prey is very low, *Ecosim* allows to create a density-dependent catchability effect (Q_{\max}/q_0), setting the Q_{\max}/q_0 ratio with a value greater than 1.0 (Christensen et al., 2005). In my case, the density-dependent catchability was changed for sardines and anchovies to 6.0 and 3.0 respectively, considering that these small pelagic fish are schooling fish and their catchability is higher.

Furthermore, the model assumes density dependent predator-prey switching. This means that in models with prey switching, no prey population is severely reduced or become very abundant. Conversely, if the predator has a strong preference for a prey, the prey abundance will not be stable (Christensen et al., 2005). The switching power was changed to 2.0 for bottlenose dolphins and monk seals and left at 0 for the other species; this was done by taking into account the opportunist foraging behaviour of these two marine mammals that can adapt their diet depending on the availability of the prey. Fishing effort was used to drive the model; fishing closures were also added to the model to simulate their potential impact.

2.4.3 Ecospace

Ecospace is the spatial version of *Ecopath* and includes all the key parameters of *Ecosim*; it is represented by a grid map that is divided into a number of different habitats in which the functional groups and the fishing fleets are allocated. The distribution of each group depends on the preference for different habitat. Each cell runs a set of *Ecosim* calculations and is linked to the other cells by symmetrical movements (Christensen et al., 2005; Pauly et al., 2000; Pauly et al., 2002; Christensen and Walters, 2004).

Ecospace has also the capability to explore the role of marine protected areas (MPAs). It can be used to evaluate the optimum size and placement of MPAs, and decide when and which kind of fishing fleets to allow in the reserves. For this reason it has been recognized as a powerful tool for fisheries management and for conservation goals (Walters et al., 1999; Watson et al., 2000; Beattie et al., 2002). *Ecospace* was used with the purpose of evaluating the impact of MPAs for the restoration of the common dolphin population and for the conservation of the marine habitats in the area (e.g. fish spawning areas). The *Ecospace* model was run after having fit the model to time series data using *Ecosim*.

First, I created a grid map (my study area) divided into 30 x 30 cells (cell length=1 km), which was divided into a number of different habitats; in this case, depths ranging from 0-20 to 200-300 meters and fish farms. Then, I assigned

these habitats to each functional groups and each fishery. Dispersal rates were also given to each species: in particular, the big predators, common and bottlenose dolphins, monk seals, sea birds, sea turtles, tuna and swordfish, were set to 300 km/year, while a value of 30 km/year was given for the other groups. Moreover, in order to show a higher distribution of bottlenose dolphins in the vicinity of the fish farm pens, I created a comma-separated values file format (csv) with time series number of pens from 1990 to 2006, which were run in *Ecospace* as a driver. Then I sketched in the grid map two different MPAs, using, as size, the entire study area. By default, protected areas are closed to fishing for the entire year; in my model, I set the MPA1 closed to all fisheries for the entire year, while I decided to allow small and big artisanal fishery to operate in the MPA2. The results are shown in the following Chapter.

CHAPTER 3: RESULTS

In this chapter, the modifications of the original parameters used to balance the *Ecopath* model are shown and described. The estimated nutrient concentrations anomaly, simulated by *Ecosim*, from the 1964 to 2006 is compared to predicted primary production obtained for the NE Ionian Sea. The observed relative biomasses, CPUE and catches trends are compared with the predicted ones found by the *Ecosim* simulations. The best scenario was considered the one that presented the lowest sum of squared (SS) differences between the simulated and the observed trajectories. Moreover, in this section, fishing closures are evaluated and proposed; *Ecospace* maps and output for the main key species are described.

3.1 Balancing the 1964 model

In order to balance the 1964 model several modifications of the input data were required. In particular, the main adjustments were related to the diet composition matrix, which had higher uncertainty compared to the other input parameters. P/B and Q/B were also modified, to less extend, based on the availability of the data source. For example, in the sea birds diet, the predation on sardines, anchovies, other demersals and cephalopods was too high, and was decreased while the predation on discards and import, (e.g., terrestrial prey) was underestimated, and had to be increased. For tuna and swordfish diet, the proportion of sardines and other pelagics was too low, and was increased, whereas the proportion of anchovy and cephalopods had to be decreased. Moreover, the fraction of sardines, anchovies, other pelagics and other demersals consumed by other pelagics, other demersals and cephalopods was overestimated and decreased subsequently. In the diet of other demersals, cephalopod and crustaceans, the predation on all the prey had to be decreased, except for the zooplankton group that was increased for other demersal and crustacean, and for the crustacean and benthic invertebrates that were increased for the cephalopod.

On the other hand, the P/B values of swordfish, sardines, anchovies, other pelagics, zooplankton and phytoplankton were decreased to take into

consideration variations of fishing mortality from the 1980s (where most of the data were available) to the year 1964. Q/B values had to be slightly changed for several functional groups (e.g., swordfish, sardine, anchovies and hake). The crustaceans biomass was the only biomass that has been modified from the original input data; the value, indeed, was too low and had to be increased. The basic original *Ecopath* parameters are shown in the following tables (Table 3.1, 3.2, 3.3, 3.4) and are compared with the balanced one (Table 3.5, 3.6).

Table 3.1. Functional groups and their basic original *Ecopath* parameters for the northeastern Ionian Sea model.

Group no.	Functional group	B (t· km ⁻²)	P/B (year ⁻¹)	Q/B (year ⁻¹)
1	<i>Common dolphins</i>	0.019	0.09	24.67
2	<i>Bottlenose dolphins</i>	0.013	0.08	21.07
3	<i>Monk seals</i>	0.015	0.12	12.59
4	<i>Sea birds</i>	0.0031	4.6	105.43
5	<i>Sea turtles</i>	0.0069	0.19	2.54
6	<i>Tuna</i>	0.04	0.2	3.22
7	<i>Swordfish</i>	0.06	0.38	4.4
8	<i>Sardines</i>	2.1	1.55	9.84
9	<i>Anchovies</i>	1.2	1.3	11.66
10	<i>Other pelagics</i>	1.95	1.95	7.7
11	<i>Hake</i>	0.18	0.82	3.26
12	<i>Other demersals</i>	0.9	1.35	7.43
13	<i>Planktivores with pens</i>	0.002	1.275	7.563
14	<i>Cephalopods</i>	0.4	3.3	7.0
15	<i>Crustaceans</i>	0.05	1.35	6.5
16	<i>Benthic invertebrates</i>	28.39	2.5	22.0
17	<i>Zooplankton</i>	3.7	31.0	40.0
18	<i>Zooplankton with pens</i>	0.0013	31.0	40.0
19	<i>Phytoplankton</i>	2.33	162.65	—
20	<i>Phytoplankton with pens</i>	0.001	162.65	—
21	<i>Discards</i>	0.135	—	—
22	<i>Detritus</i>	69.87	—	—

Table 3.2. Functional groups and their diet matrix for the northeastern Ionian Sea model.

Prey/Predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
1 <i>Common d.</i>																		
2 <i>Bottlenose d.</i>																		
3 <i>Monk seal</i>																		
4 <i>Sea birds</i>																		
5 <i>Sea turtle</i>																		
6 <i>Tuna</i>																		
7 <i>Swordfish</i>																		
8 <i>Sardine</i>	0.50	0.02		0.30		0.20	0.19			0.05	0.17	0.01		0.20				
9 <i>Anchovy</i>	0.30	0.07		0.30		0.30	0.46			0.05	0.30	0.01		0.10				
10 <i>Other pelagics</i>	0.10	0.05	0.20	0.05		0.10	0.09			0.05	0.10	0.04		0.05				
11 <i>Hake</i>		0.44					0.01											
12 <i>Other demersals</i>	0.05	0.38	0.45	0.04			0.01			0.01	0.09	0.02		0.06	0.05			
13 <i>Planktivores/pens</i>		0.01	0.25								0.08							
14 <i>Cephalopod</i>	0.05	0.03	0.10	0.05		0.25	0.17				0.06			0.03	0.01			
15 <i>Crustacean</i>		$3 \cdot 10^{-3}$		0.02	0.04	0.15	0.07				0.20	0.10		0.06	0.12			
16 <i>Benthic inv.</i>					0.52					0.49		0.80		0.50	0.80			
17 <i>Zooplankton</i>					0.17			0.80	1.0	0.35		0.01			0.01	$6 \cdot 10^{-3}$	0.10	
18 <i>Zooplankton/pens</i>													0.90					0.10
19 <i>Phytoplankton</i>								0.20								0.06	0.90	
20 <i>Phytoplankton/pens</i>													0.10					0.90
21 <i>Discards</i>				0.15	0.19										0.01	10^{-5}		
22 <i>Detritus</i>					0.07							0.01				0.94		
23 <i>Import</i>				0.09														
24 Sum	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

Table 3.3. Fisheries landings ($t \cdot km^{-2}$) for the northeastern Ionian Sea model

	Functional groups	Purse seiners	Trawlers	Beach seiners	Small artisanals	Big artisanals
1	<i>Common d.</i>					
2	<i>Bottlenose d.</i>					
3	<i>Monk seal</i>					
4	<i>Sea birds</i>					
5	<i>Sea turtle</i>					
6	<i>Tuna</i>	$3 \cdot 10^{-3}$			$9 \cdot 10^{-5}$	$7 \cdot 10^{-4}$
7	<i>Swordfish</i>				$5 \cdot 10^{-4}$	$6 \cdot 10^{-3}$
8	<i>Sardine</i>	0.06		0.03	$4 \cdot 10^{-3}$	0.01
9	<i>Anchovy</i>	0.09		0.03	$5 \cdot 10^{-3}$	0.02
10	<i>Other pelagics</i>	0.12	0.04	0.05	0.06	0.13
11	<i>Hake</i>		0.02	$6 \cdot 10^{-4}$		
12	<i>Other demersals</i>	0.02	0.03	0.01	0.01	0.08
13	<i>Planktivores/pens</i>					
14	<i>Cephalopod</i>	0.01	0.02	$5 \cdot 10^{-3}$	$5 \cdot 10^{-3}$	0.01
15	<i>Crustacean</i>		0.01			
16	<i>Benthic inv.</i>					
17	<i>Zooplankton</i>					
18	<i>Zooplankton/pens</i>					
19	<i>Phytoplankton</i>					
20	<i>Phytoplankton/pens</i>					
21	<i>Discards</i>					
22	<i>Detritus</i>					
23	Sum	0.29	0.12	0.12	0.09	0.25

Table 3.4. Discards ($t \cdot km^{-2}$) for the northeastern Ionian Sea model

	Functional groups	Purse seiners	Trawlers	Beach seiners	Small artisanals	Big artisanals
1	<i>Common d.</i>					
2	<i>Bottlenose d.</i>					
3	<i>Monk seal</i>					
4	<i>Sea birds</i>					
5	<i>Sea turtle</i>					$3 \cdot 10^{-4}$
6	<i>Tuna</i>					
7	<i>Swordfish</i>					
8	<i>Sardine</i>		$7 \cdot 10^{-3}$		10^{-3}	$3 \cdot 10^{-3}$
9	<i>Anchovy</i>	0.01	0.02			
10	<i>Other pelagics</i>	$2 \cdot 10^{-3}$	0.01	$3 \cdot 10^{-3}$	$4 \cdot 10^{-3}$	0.01
11	<i>Hake</i>		$4 \cdot 10^{-4}$		$7 \cdot 10^{-4}$	$2 \cdot 10^{-3}$
12	<i>Other demersals</i>	0.04	$9 \cdot 10^{-3}$	$2 \cdot 10^{-3}$	$3 \cdot 10^{-3}$	0.01
13	<i>Planktivores/pens</i>					
14	<i>Cephalopod</i>		10^{-3}			
15	<i>Crustacean</i>		$9 \cdot 10^{-4}$		$3 \cdot 10^{-4}$	$8 \cdot 10^{-4}$
16	<i>Benthic inv.</i>					
17	<i>Zooplankton</i>					
18	<i>Zooplankton/pens</i>					
19	<i>Phytoplankton</i>					
20	<i>Phytoplankton/pens</i>					
21	<i>Discards</i>					
22	<i>Detritus</i>					
23	Sum	0.049	0.046	0.006	0.009	0.026

Table 3.5. Functional groups and their basic modified *Ecopath* parameters for the northeastern Ionian Sea balanced model. Modified parameters are in italics.

Group no.	Functional group	B	P/B	Q/B
1	<i>Common dolphins</i>	0.019	0.09	24.67
2	<i>Bottlenose dolphins</i>	0.013	0.08	21.07
3	<i>Monk seals</i>	0.015	0.12	12.59
4	<i>Sea birds</i>	0.0031	4.6	105.43
5	<i>Sea turtles</i>	0.0069	0.19	2.54
6	<i>Tuna</i>	0.04	0.2	3.22
7	<i>Swordfish</i>	0.06	<i>0.3</i>	<i>4.6</i>
8	<i>Sardines</i>	2.1	<i>0.7</i>	<i>8.68</i>
9	<i>Anchovies</i>	1.2	<i>0.8</i>	<i>12.3</i>
10	<i>Other pelagics</i>	1.95	<i>1.1</i>	<i>7.7</i>
11	<i>Hake</i>	0.18	<i>1.2</i>	<i>3.46</i>
12	<i>Other demersals</i>	0.9	1.35	7.43
13	<i>Planktivores with pens</i>	0.002	1.275	7.563
14	<i>Cephalopods</i>	0.4	3.3	7.0
15	<i>Crustaceans</i>	3.0	1.35	6.5
16	<i>Benthic invertebrates</i>	28.39	2.5	22.0
17	<i>Zooplankton</i>	3.7	<i>18.0</i>	40.0
18	<i>Zooplankton with pens</i>	0.0013	<i>18.0</i>	40.0
19	<i>Phytoplankton</i>	2.33	<i>285.26</i>	—
20	<i>Phytoplankton with pens</i>	0.001	<i>285.26</i>	—
21	<i>Discards</i>	0.135	—	—
22	<i>Detritus</i>	69.87	—	—

Table 3.6. Functional groups and their diet matrix for the northeastern Ionian Sea balanced model. Modified parameters are in *italics*.

Prey/Predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
1 <i>Common d.</i>																		
2 <i>Bottlenose d.</i>																		
3 <i>Monk seal</i>																		
4 <i>Sea birds</i>																		
5 <i>Sea turtle</i>																		
6 <i>Tuna</i>																		
7 <i>Swordfish</i>																		
8 <i>Sardine</i>	0.50	0.02		0.25		0.24	0.28			0.01	0.21	8·10 ⁻³		2·10 ⁻³				
9 <i>Anchovy</i>	0.30	0.07		0.09		0.13	0.14			0.01	0.15	6·10 ⁻³		6·10 ⁻³				
10 <i>Other pelagics</i>	0.10	0.05	0.20	0.05		0.40	0.36			0.02	0.13	9·10 ⁻³		3·10 ⁻³				
11 <i>Hake</i>		0.44					2·10 ⁻⁴											
12 <i>Other demersals</i>	0.05	0.38	0.45	7·10 ⁻³			0.01			4·10 ⁻³	0.07	8·10 ⁻³		4·10 ⁻³	8·10 ⁻⁴			
13 <i>Planktivores/pens</i>		0.01	0.25															
14 <i>Cephalopod</i>	0.05	0.03	0.10	0.03		0.07	0.10				0.04			4·10 ⁻³	4·10 ⁻³			
15 <i>Crustacean</i>		3·10 ⁻³		0.07	0.04	0.15	0.12				0.08	0.06		0.44	0.05			
16 <i>Benthic inv.</i>					0.52					0.52	0.32	0.36		0.56	0.58			
17 <i>Zooplankton</i>					0.17			0.80	1.0	0.44		0.54			0.36	6·10 ⁻³	0.10	
18 <i>Zooplankton/pens</i>													0.90					0.10
19 <i>Phytoplankton</i>								0.20								0.06	0.90	
20 <i>Phytoplankton/pens</i>													0.10					0.90
21 <i>Discards</i>				0.20	0.19										10 ⁻⁵	10 ⁻⁵		
22 <i>Detritus</i>					0.07							5·10 ⁻³				0.94		
23 <i>Import</i>				0.31														
24 Sum	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

3.2 System productivity (nutrient forcing function)

The changes in nutrient concentration found by *Ecosim* are shown in Fig.3.1. This trend has been compared with the trajectory of primary production (PP), from the year 1964 to the year 2006, estimated by *Ecosim* at the end of the fitting. Indeed, primary production rates for producer groups reacted differently to variation in nutrient concentration during each simulation. The generated trend of primary production showed low concentration of PP through time, except for the years 1991-1994 and, to less extend 1996-1999 and 2003-2005, where high peaks were observed. The high peak shown at the beginning of the nutrient anomaly load is probably due to the initialization of the simulation; for this reason, it will not be considered further.

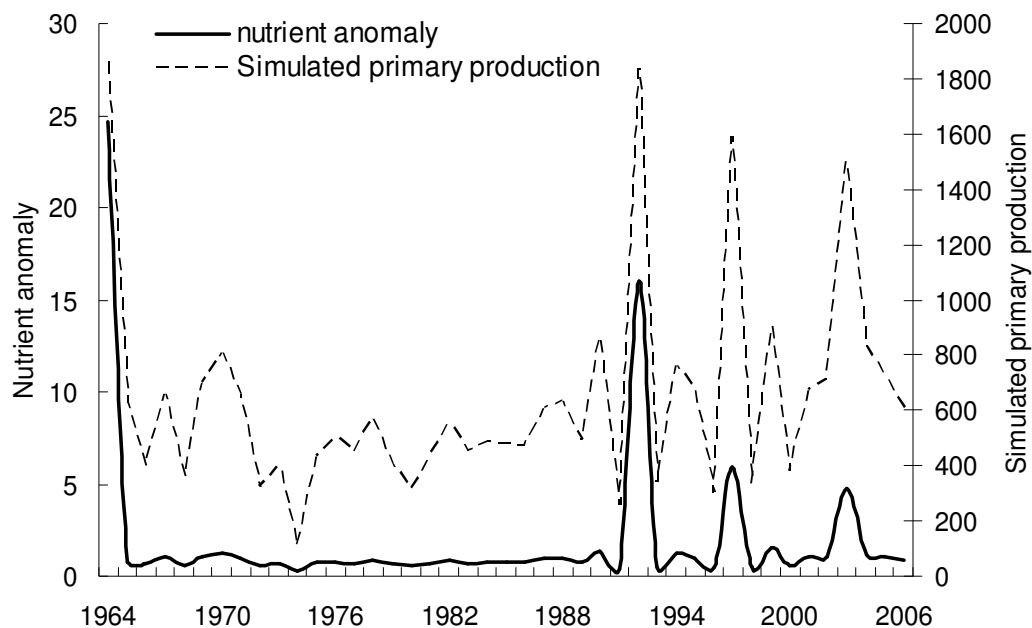


Fig.3.1. Simulated nutrient anomaly for the NE Ionian Sea, and modelled primary production estimated by *Ecosim*.

3.3 Time-series fitting

In Fig.3.2, the observed common dolphin biomass was compared with the simulated biomass. The model completely reflected the rapid decline of the population in the area suggesting also that the biomass observed in the year 1995 was far from its carrying capacity (shown at the initial year 1964). It is

interesting to notice that the main drop around 1976 is a consequence of the decline in this year of the majority of fish species.

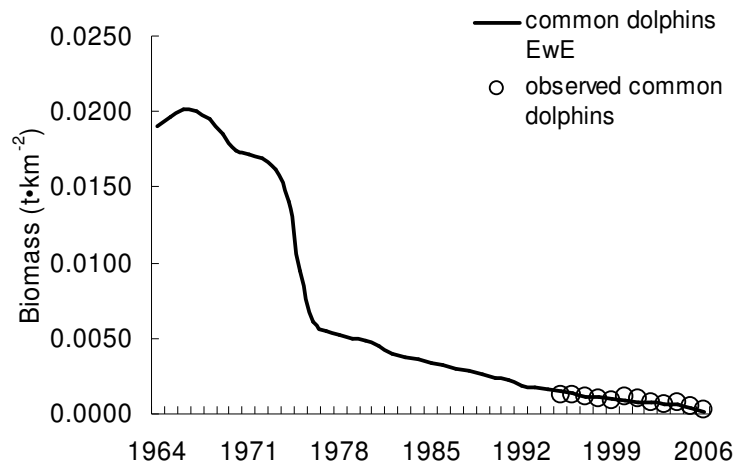


Fig.3.2. Time series fitting between predicted common dolphin biomass and observed common dolphin biomass (Tethys, unpublished data) from 1964 to 2006.

Bottlenose dolphins, on the other hand, have been observed to increase in the area for the period 1995-2006. Initially, the model was not able to predict this trend, suggesting that trophic interactions were not likely to be the reason of their increase. A mediation function was applied into the model, assuming that a third organism was affecting the feeding rate of bottlenose dolphins. In particular, it has been hypothesized that the decline of common dolphins had increased the feeding area of bottlenose dolphins. Unfortunately, no correlation was observed. Secondly, it has been hypothesized that bottlenose dolphins have increased as a consequence of the increase of fish farm cages present in the study area. The result of this analysis is shown in the Fig.3.3. The model was able to reconstruct this biomass trajectory.

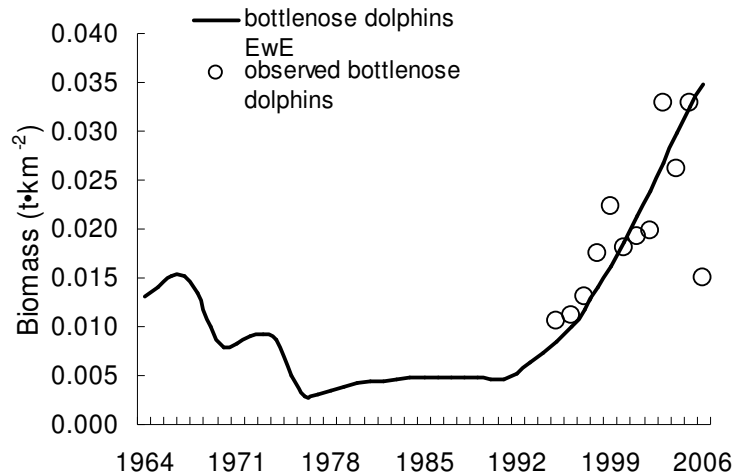


Fig.3.3. Time series fitting between predicted bottlenose dolphin biomass and observed bottlenose dolphin biomass (Tethys, unpublished data) from 1964 to 2006. The predicted bottlenose dolphin biomass trend was generated using time series number of fish farm cages as a driver.

Monk seal biomass was available for only three years 1977, 1986 and 1987 (Goedicke 1981; Panou et al., 1993); the model reflected the decline observed in those years and, as for common dolphins, suggesting a rapid decrease from 1976.

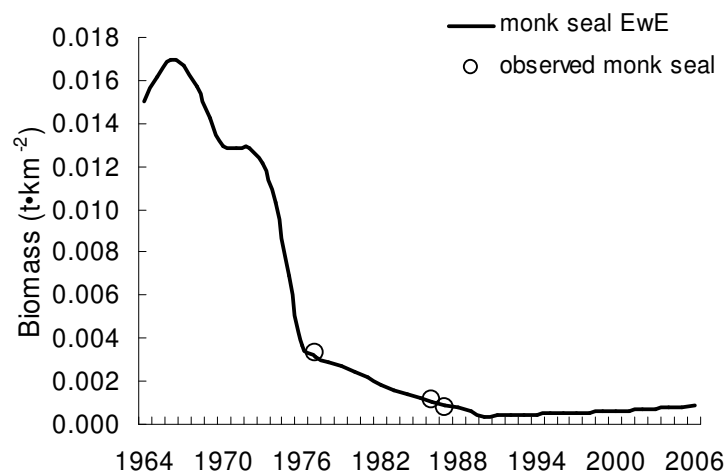


Fig.3.4. Time series fitting between predicted monk seal biomass and observed monk seal biomass (Goedicke, 1981; Panou et al., 1993,) from 1964 to 2006.

Time series data on sardine CPUE from 1964 to 2003 were compared with the simulated trajectory. The model, initially, had some difficulties to explain the decline of sardine since the 1964. No evidence was observed in fishing effort or catches that could clarify these changes (Fig.3.5). Different vulnerabilities and density dependent values have been changed several times, but the best fit was found when a nutrient loading forcing function was added to the model. This function was used to simulate changes in nutrient concentration within the ecosystem. The result is shown in the Fig.3.6.

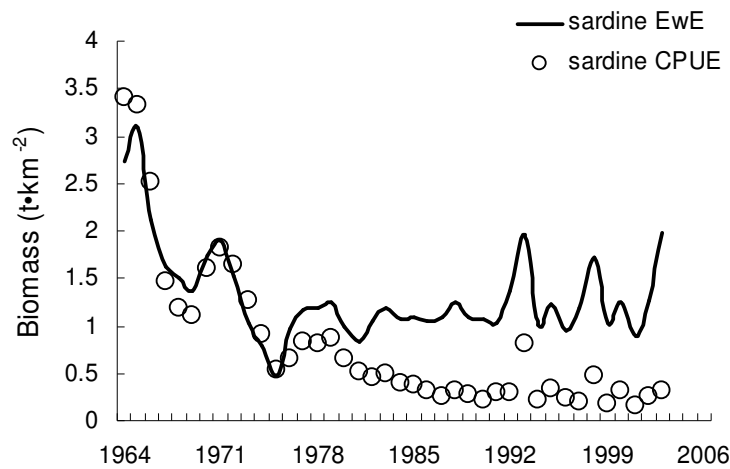


Fig.3.5. Time series fitting between predicted sardine biomass and estimated sardine CPUE, from 1964 to 2003, without the nutrient loading forcing function.

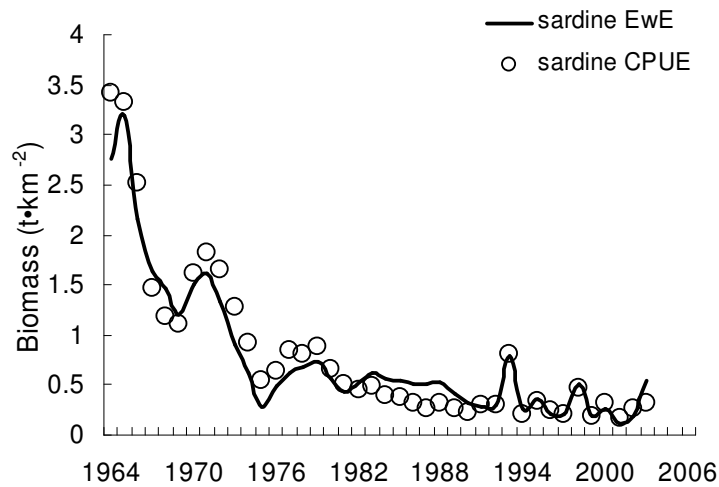


Fig.3.6. Time series fitting between predicted sardine biomass and estimated sardine CPUE, from 1964 to 2003, with the nutrient loading forcing function.

The methodology applied to anchovy biomass was the same one used for sardines. The predicted CPUE matched quite well the observed CPUE showing, as for sardine, a marked decline in CPUE since 1964 (Fig. 3.8).

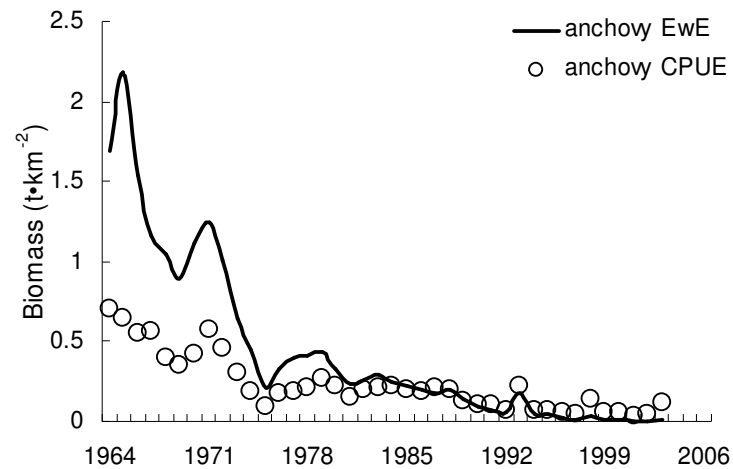


Fig.3.7. Time series fitting between predicted anchovy biomass and estimated anchovy CPUE, from 1964 to 2003, without the nutrient loading forcing function.

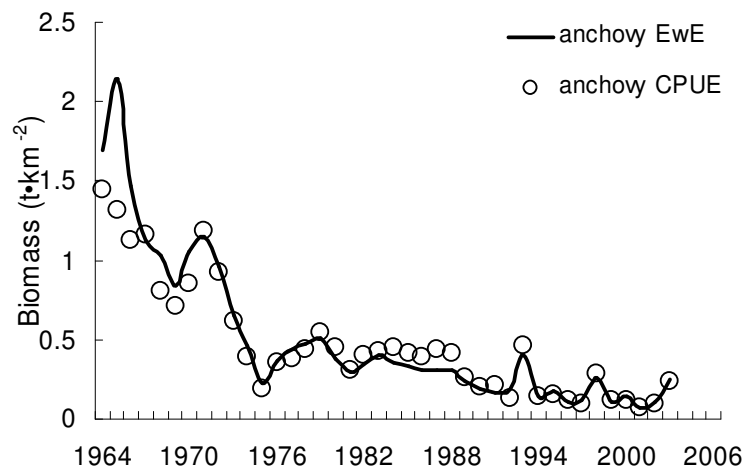


Fig.3.8. Time series fitting between predicted anchovy biomass and estimated anchovy CPUE, from 1964 to 2003, with the nutrient loading forcing function.

As it is clear from Fig.3.9, *Ecosim* failed to reproduce the tuna CPUE trend. Several vulnerabilities, emigration/migration values were tested in the model, but none of them were able to represent the fluctuations shown by the tuna in the

area. As discussed in Section 4, this is likely to be because the tuna dynamics are influenced by events at a much larger scale than represented here.

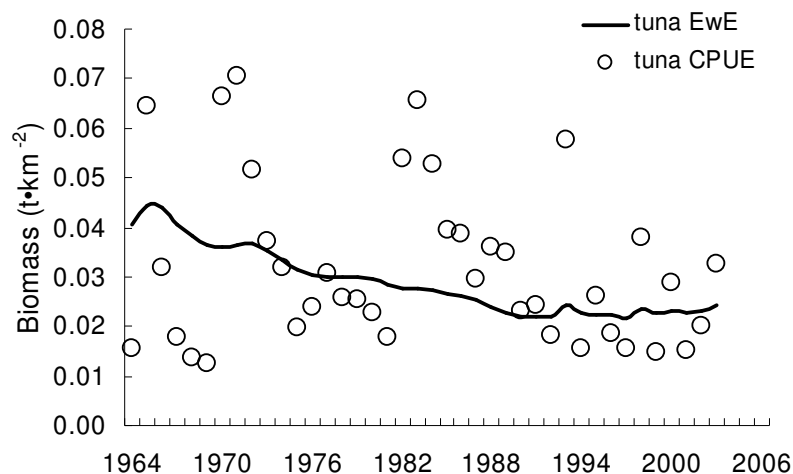


Fig.3.9. Time series fitting between predicted tuna biomass and estimated tuna CPUE, from 1964 to 2003.

Similar scenario has been observed in the swordfish CPUE trend; the model, indeed, was not able to reflect the annual fluctuations of the species in the area. However, a more defined trajectory may suggest that this stock has decreased since the 1970s (Fig.3.10).

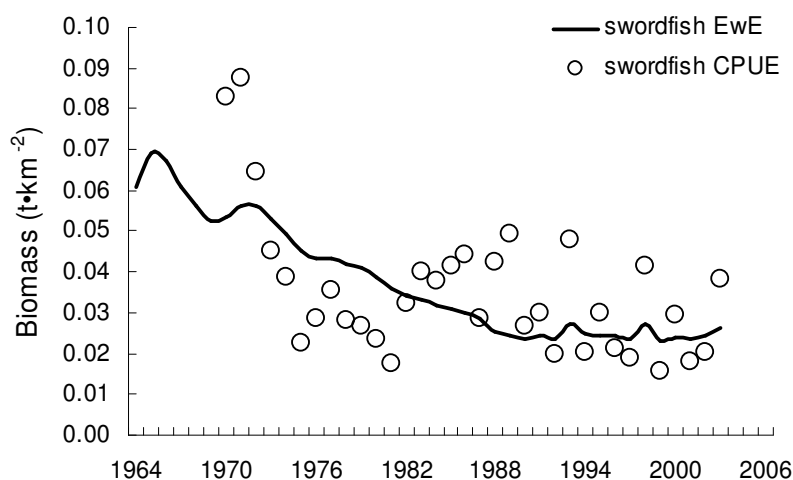


Fig.3.10. Time series fitting between predicted swordfish biomass and estimated swordfish CPUE, from 1964 to 2003.

Hake CPUE trend was compared with the predicted biomass; the model reconstructed quite well the changes in CPUE especially for the period 1968-1982 while it was not capable of following the trajectories shown in 1964-1981 and the 1983-2003 periods (Fig.3.11).

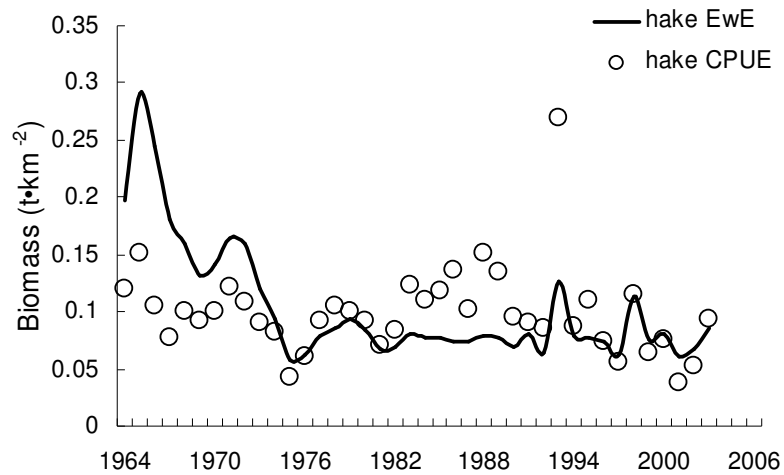


Fig.3.11. Time series fitting between predicted hake biomass and estimated hake CPUE, from 1964 to 2003.

The simulated changes in biomass for other pelagics and other demersals fitted the observed CPUE trends (Fig.3.12 and Fig.3.14). This was possible by searching for vulnerabilities and nutrients anomalies that would minimize the summed squared residuals between observed and predicted estimates. The model reflected the decline of these groups of fish since the 1964. The trends, before and after the use of the nutrient loading forcing function, are shown respectively in Fig.3.12, 3.13 and Fig. 3.14, 3.15.

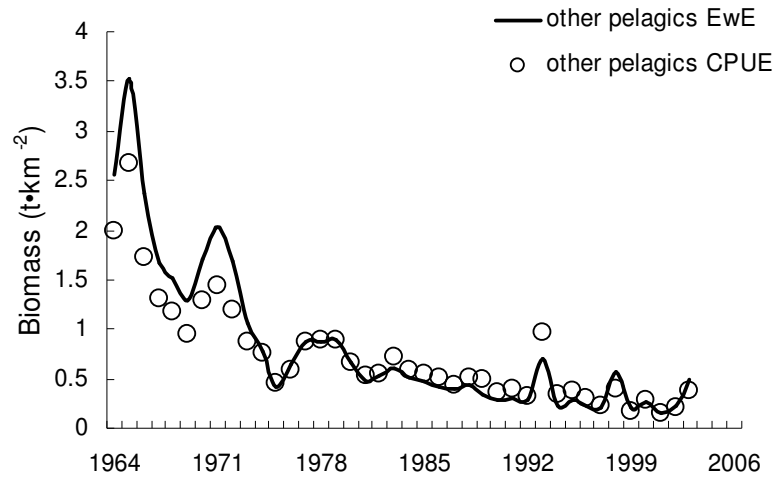


Fig.3.12. Time series fitting between predicted other pelagics biomass and estimated other pelagics CPUE, from 1964 to 2003, without the nutrient loading forcing function.

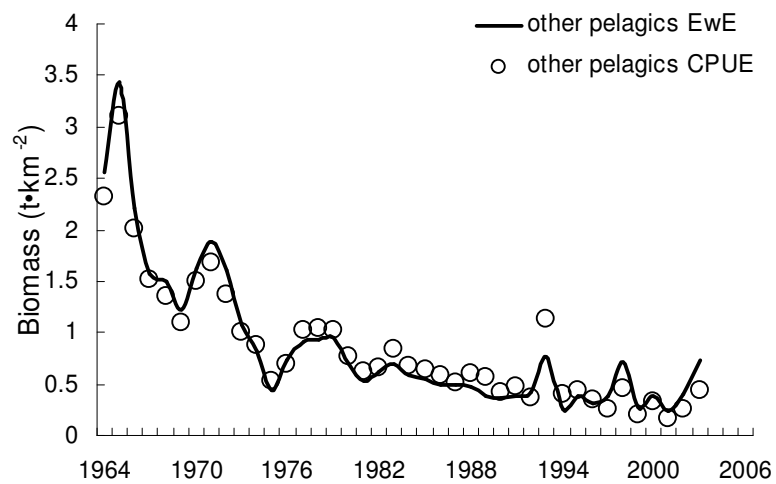


Fig.3.13. Time series fitting between predicted other pelagics biomass and estimated other pelagics CPUE, from 1964 to 2003, with the nutrient loading forcing function.

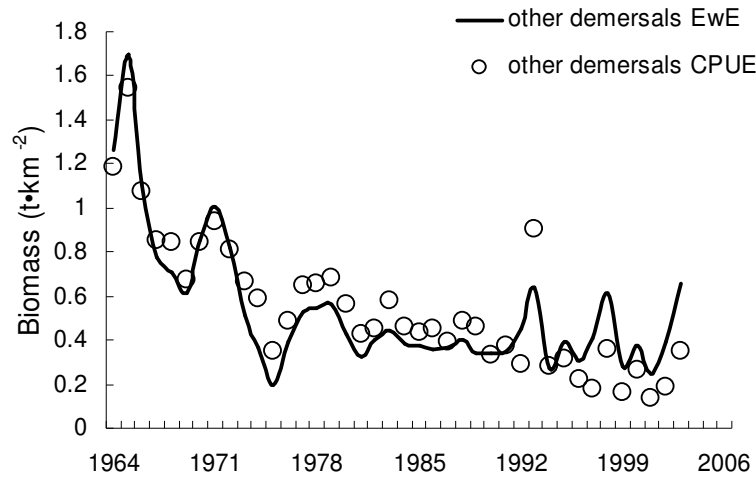


Fig.3.14. Time series fitting between predicted other demersals B and estimated other demersals CPUE, from 1964 to 2003, without the nutrient forcing function.

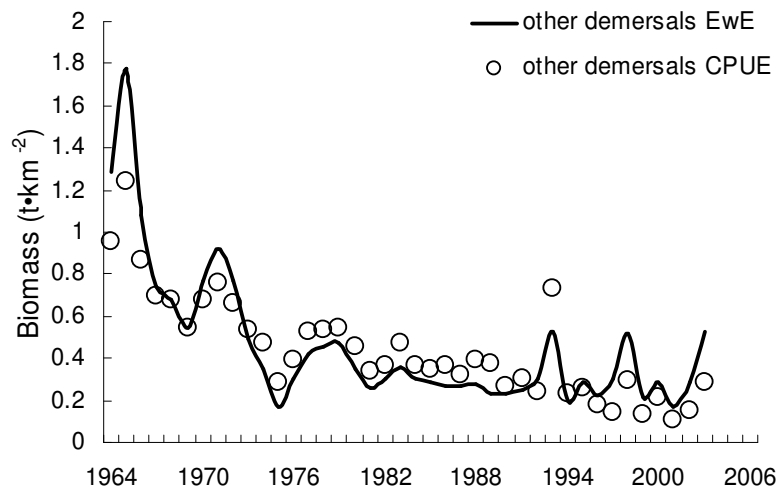


Fig.3.15. Time series fitting between predicted other demersals B and estimated other demersals CPUE, from 1964 to 2003, with the nutrient forcing function.

Cephalopods and crustaceans modelled biomass trends had similarities with the observed CPUE changes, particularly between 1964 and 1982; however, *Ecosim* was not able to find a good fit for the following years (Fig.3.16. and Fig.3.17). In the crustaceans trend, even without a good match, the model seemed to capture the increase in CPUE from the year 1983 onwards.

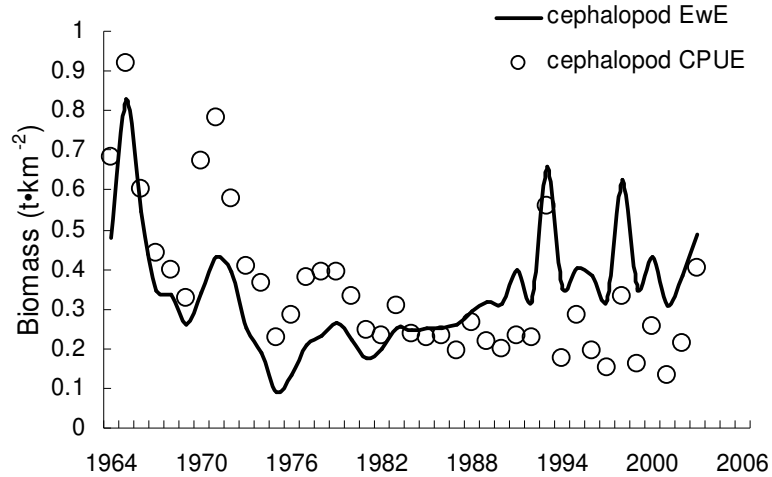


Fig.3.16. Time series fitting between predicted cephalopod biomass and estimated cephalopods CPUE, from 1964 to 2003.

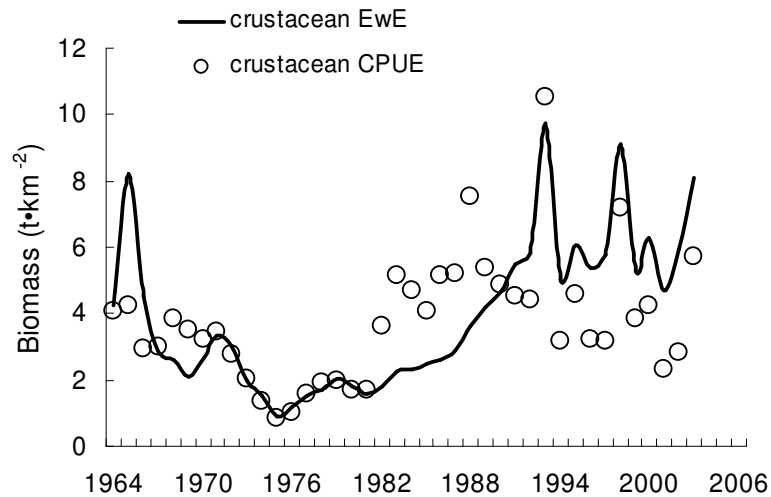


Fig.3.17. Time series fitting between predicted crustacean biomass and estimated crustaceans CPUE, from 1964 to 2003.

Catches time series fitting are presented below. In particular, the predicted sardine landings and, to a less extend also, the anchovy and cephalopod landings showed higher trends compared to the observed one (Fig. 3.18 and Fig.3.19). This may suggest a possible underestimation of the reported catches.

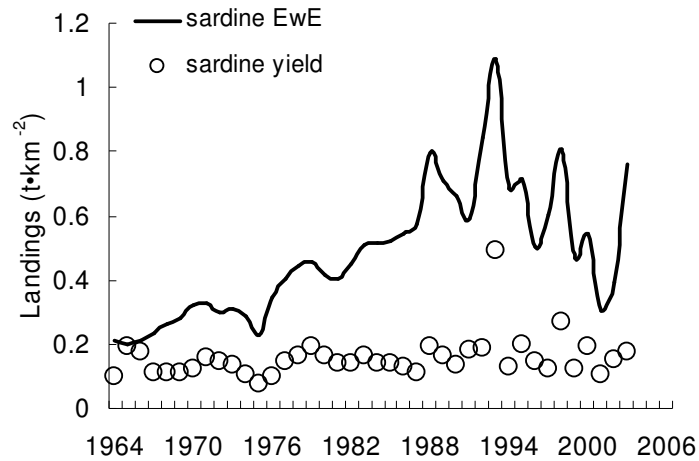


Fig.3.18. Time series fitting between predicted and estimated sardine yield, from 1964 to 2003.

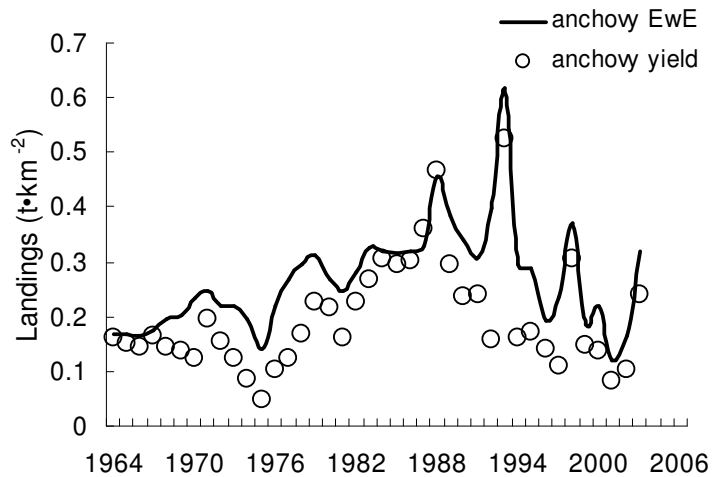


Fig.3.19. Time series fitting between predicted and estimated anchovy yield, from 1964 to 2003.

The model was not able to fit tuna, hake and swordfish catches probably related to the bad fits observed in the CPUE trajectories; in particular tuna and hake catches are underestimated by *Ecosim* predictions. Regarding the other functional groups, other pelagics, other demersals and crustaceans, the simulations seemed to fit quite well the observed trends.

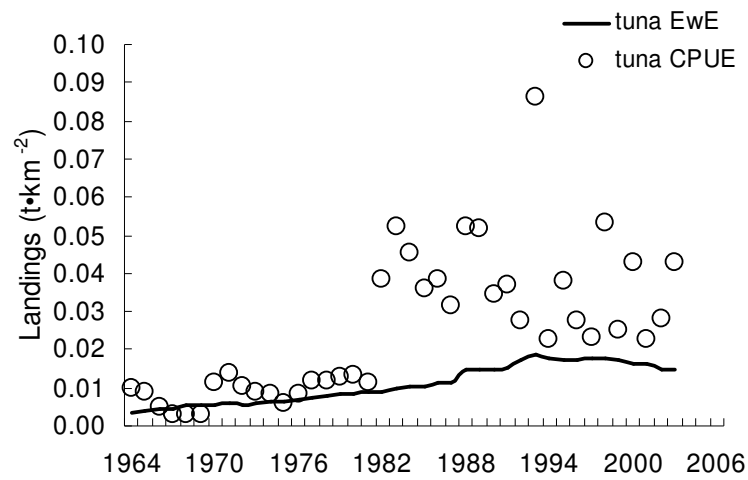


Fig.3.20. Time series fitting between predicted and estimated tuna yield, from 1964 to 2003.

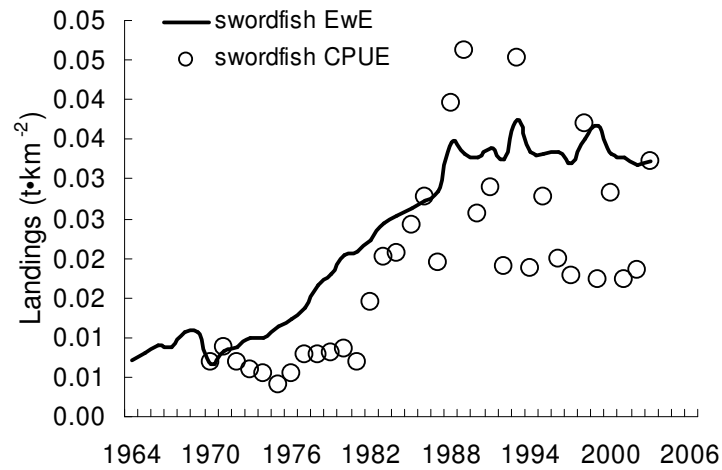


Fig.3.21. Time series fitting between predicted and estimated swordfish yield, from 1964 to 2003.

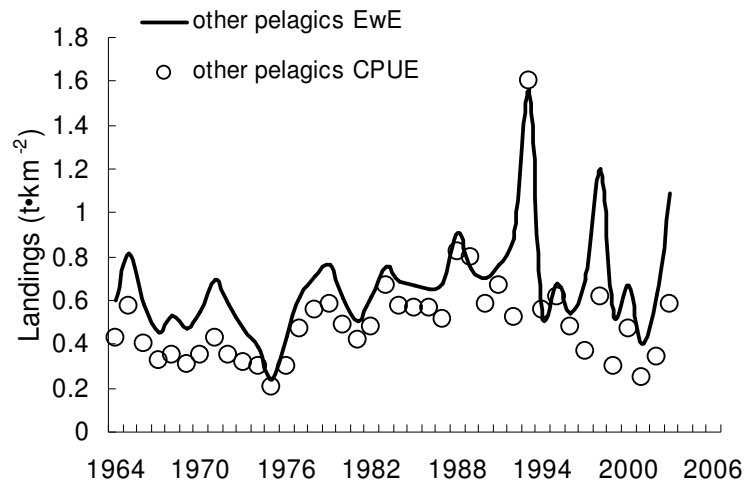


Fig.3.22. Time series fitting between predicted and estimated other pelagics yield, from 1964 to 2003.

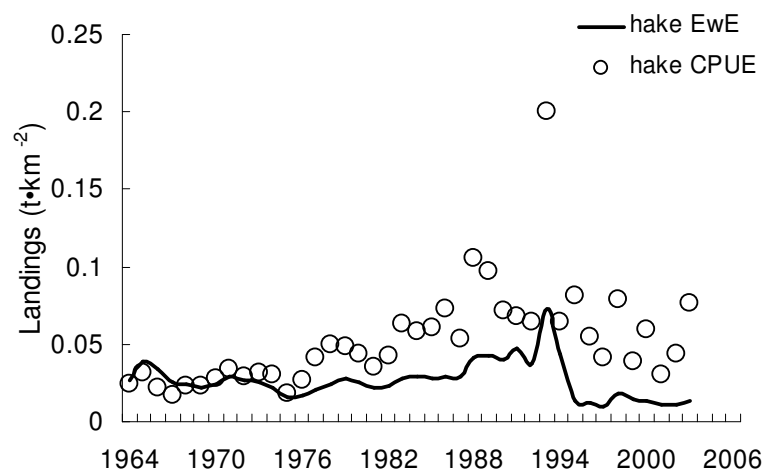


Fig.3.23. Time series fitting between predicted and estimated hake yield, from 1964 to 2003.

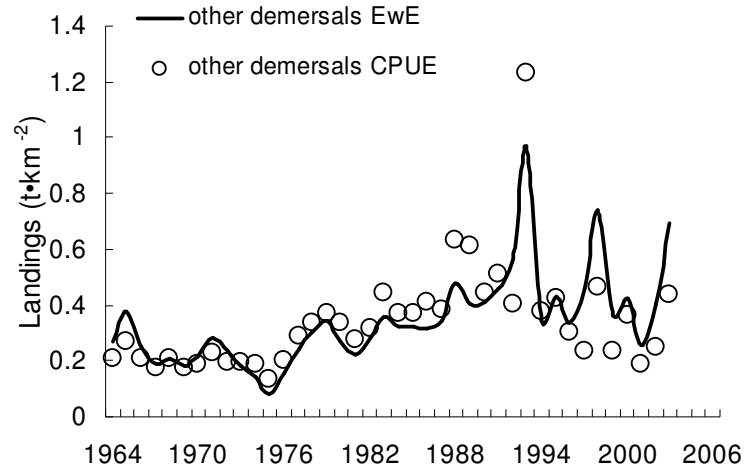


Fig.3.24. Time series fitting between predicted and estimated other demersals yield, from 1964 to 2003.

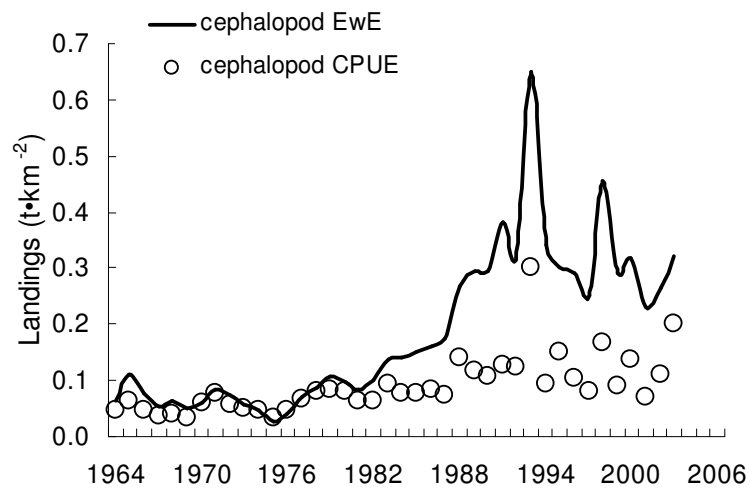


Fig.3.25. Time series fitting between predicted and estimated cephalopod yield, from 1964 to 2003.

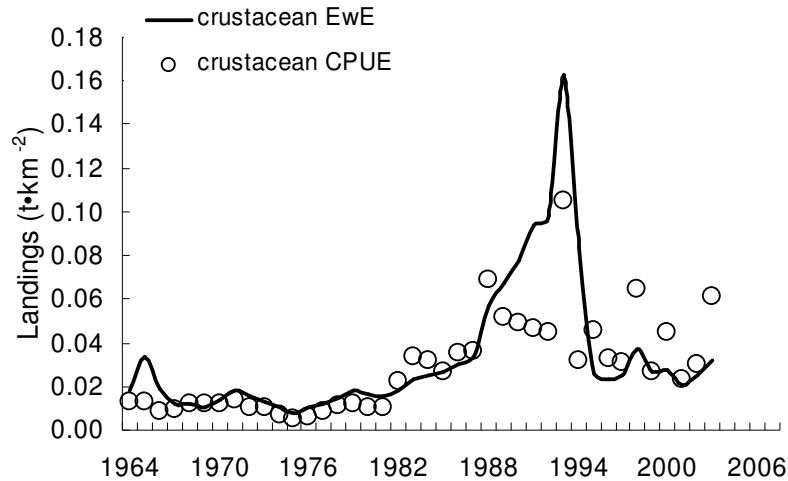


Fig.3.26. Time series fitting between predicted and estimated crustaceans yield, from 1964 to 2003.

3.4 Fishing closures scenarios

In order to evaluate the impact of fishing pressure through time on the marine ecosystem, and especially on the common dolphin population, I simulated four different scenarios for fisheries regulations from the present up to 2030. In the first scenario, I decided to keep constant the fishing effort of each fishery present in the study area from the year 2003 to the year 2030. In the second scenario, I assumed a closure for the purse seine fishery from the year 2009 up to 2030. In the third scenario, the main industrial fisheries (purse seiners, trawlers and beach seiners) were closed from 2009 to 2030; while in the fourth one, I closed the area to all fisheries from 2009 to 2030. The simulations are presented only for the main key species of the study area. In the first scenario simulated by the model, if the fisheries are allowed to harvest as in the present days, common dolphins and anchovies would slowly disappear (Fig.3.27 and Fig.3.29) while sardines would gradually increase (Fig.3.28).

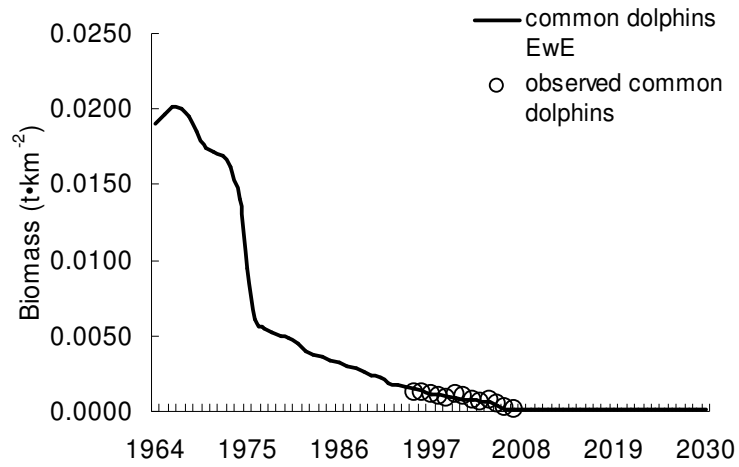


Fig.3.27. Simulated common dolphin biomass until 2030.

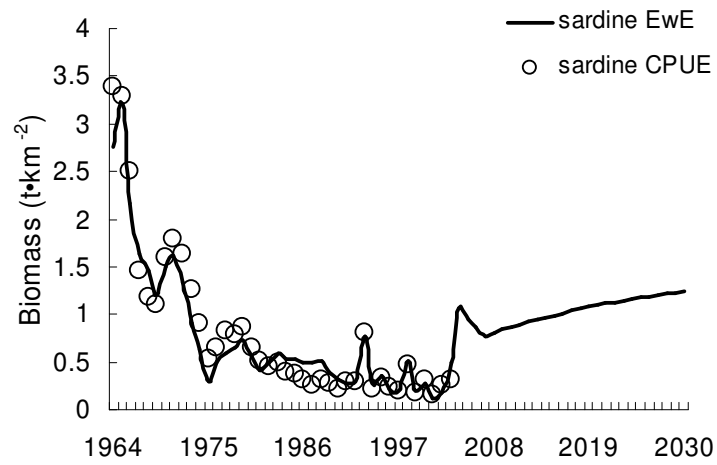


Fig.3.28. Simulated sardine biomass until 2030.

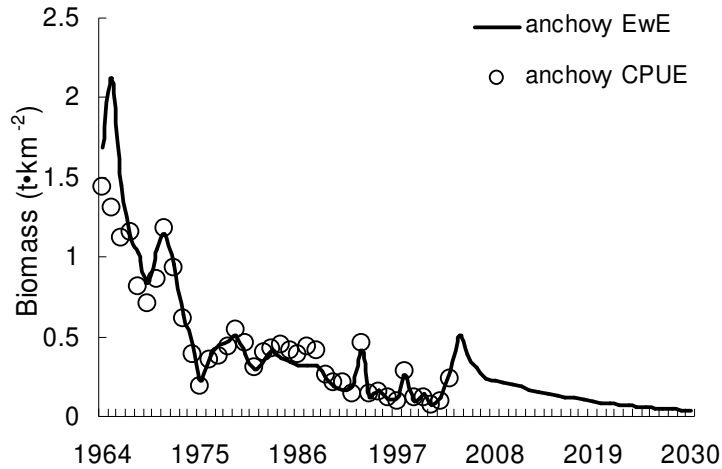


Fig.3.29. Simulated anchovy biomass until 2030.

In the following scenarios, the trends of common dolphins are represented starting from the year 1995 in order to better evaluate the changes in biomass of this functional group, related to the closure of fisheries (Fig.3.30). The second scenario, that shows the closure of purse seiners in the area, would have a positive effect on all the considered species (Fig.3.30, 3.31, 3.32). However, as it is possible to notice in Fig.3.30, common dolphin biomass is predicted to only increase slightly.

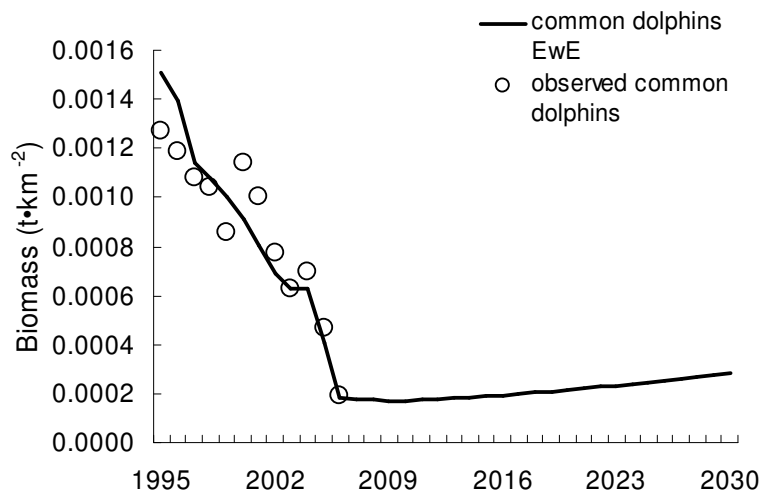


Fig.3.30. Simulated common dolphin biomass, assuming a closure to the purse seine fishery from 2009 until 2030.

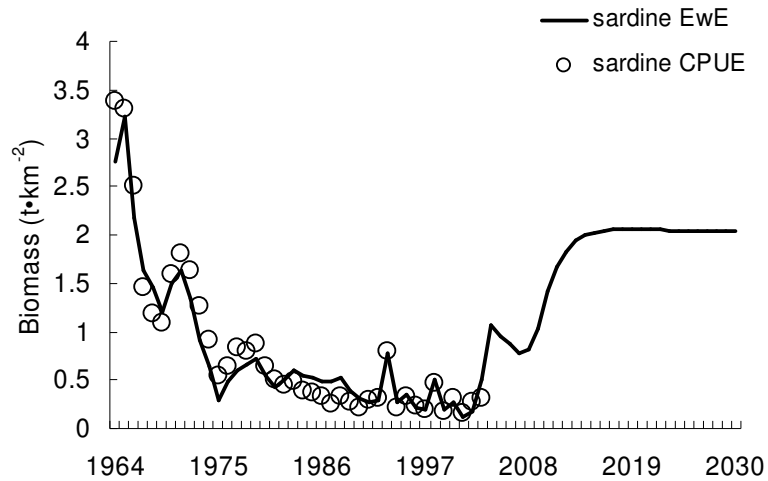


Fig.3.31. Simulated sardine biomass, assuming a closure to the purse seine fishery from 2009 until 2030.

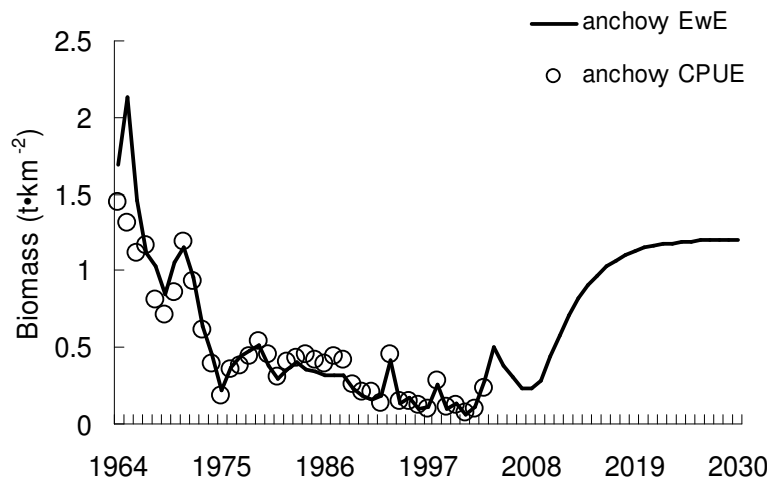


Fig.3.32. Simulated anchovy biomass, assuming a closure to the purse seine fishery from 2009 until 2030.

The closure of the main commercial fisheries is predicted to restore the sardine and the anchovy stock, reaching a plateau similar to the 1960's abundance (Fig.3.34 and Fig.3.35); common dolphins would benefit more using this strategy, showing a more pronounced increase through time (Fig.3.33).

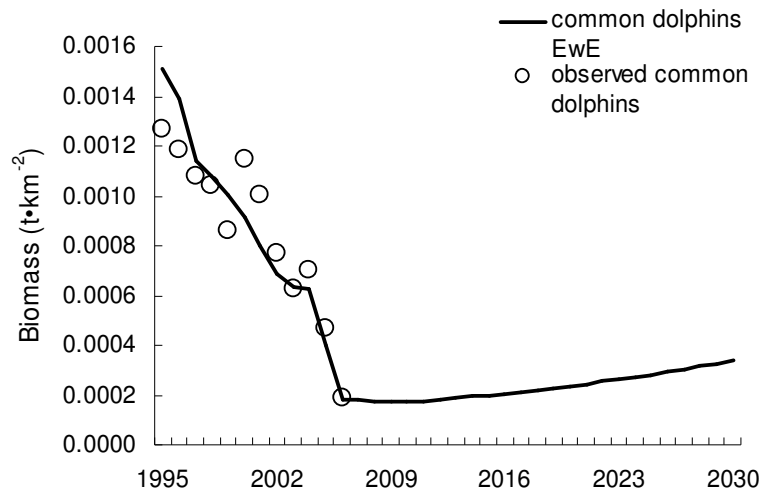


Fig.3.33. Simulated common dolphin biomass, assuming a closure to the purse seine, beach seine and trawl fisheries from 2009 until 2030.

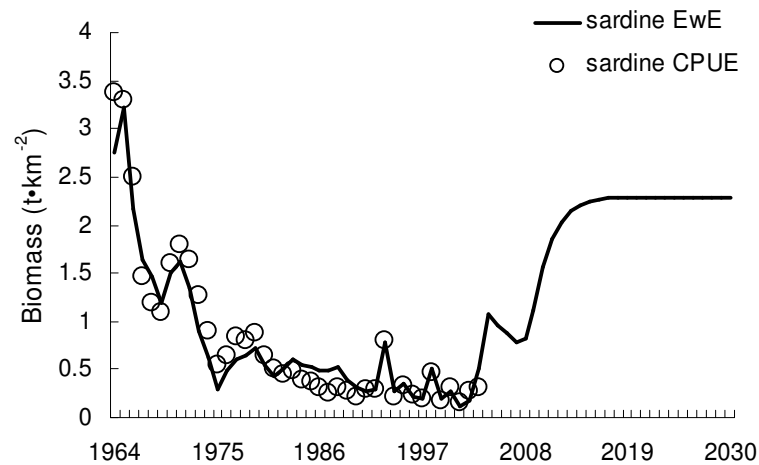


Fig.3.34. Simulated sardine biomass, assuming a closure to the purse seine, beach seine and trawl fisheries from 2009 until 2030.

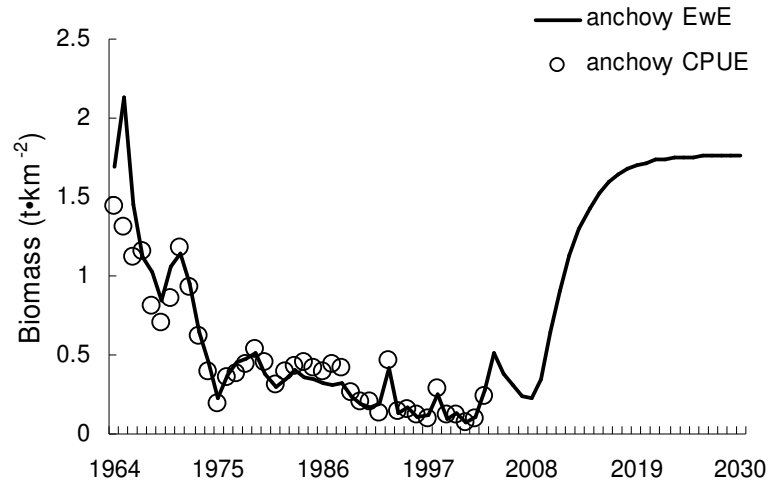


Fig.3.35. Simulated anchovy biomass, assuming a closure to the purse seine, beach seine and trawl fisheries from 2009 until 2030.

Closing all the area to fishing activities, would bring common dolphins to the same biomass level observed in 2005, while sardines and anchovies would keep the same trajectories delineated in the previous scenario (Fig.3.36, 3.37, 3.38).

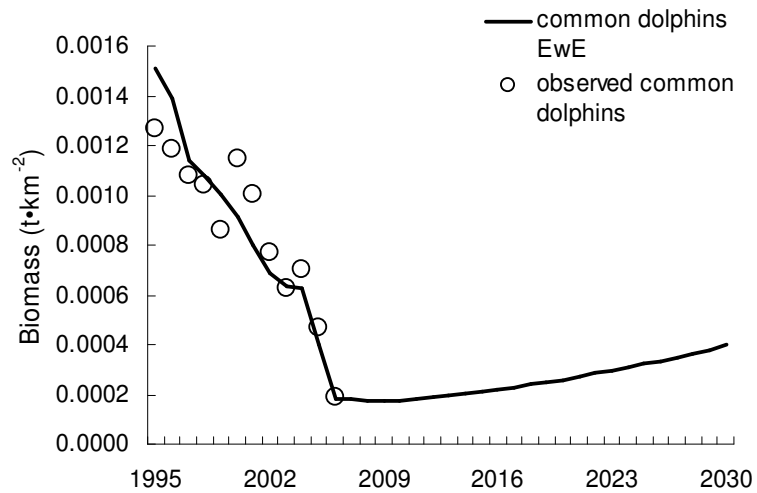


Fig.3.36. Simulated common dolphin biomass, assuming a closure to all the area to fishing activities from 2009 until 2030.

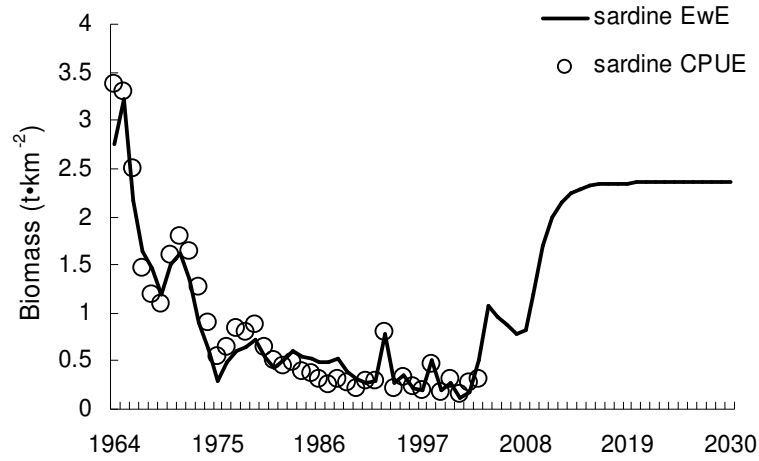


Fig.3.37. Simulated sardine CPUE, assuming a closure to all the area to fishing activities from 2009 until 2030.

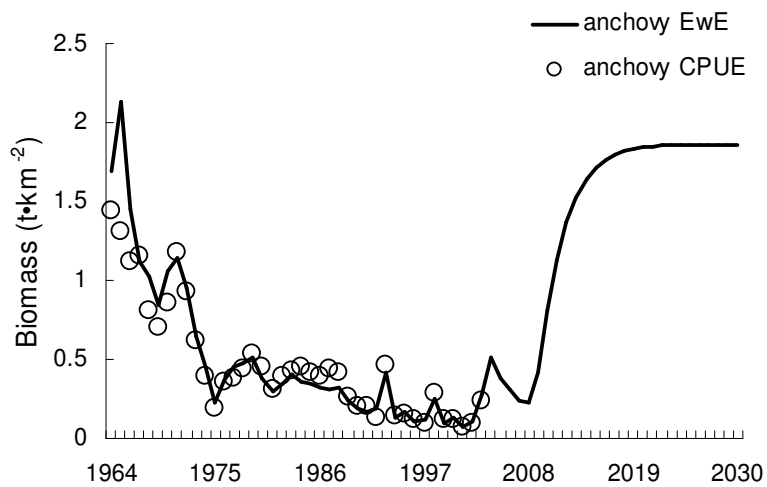


Fig.3.38. Simulated anchovy CPUE, assuming a close to all the area to fishing activities from 2009 until 2030.

3.5 Ecospace

A graphic representation of the state of the ecosystem in 2006 has been made using *Ecospace*. Each map considered a single functional group within the ecosystem. Colours are expressing relative biomass with green that is equal to average biomass for the given group in 1964, red that is two times or more high biomass and blue that corresponds to lower biomass (Fig.3.39). The red rectangles shown in pen-associated zooplankton and pen-associated phytoplankton maps, represented fish farms presence in the area. As it can be seen, high concentration of bottlenose dolphins and pen-associated planktivores are surrounding the marine fish culture areas. Almost all the functional groups of this ecosystem show clearly low levels of biomass, except for cephalopods, crustaceans, benthic invertebrates, zooplankton and phytoplankton.

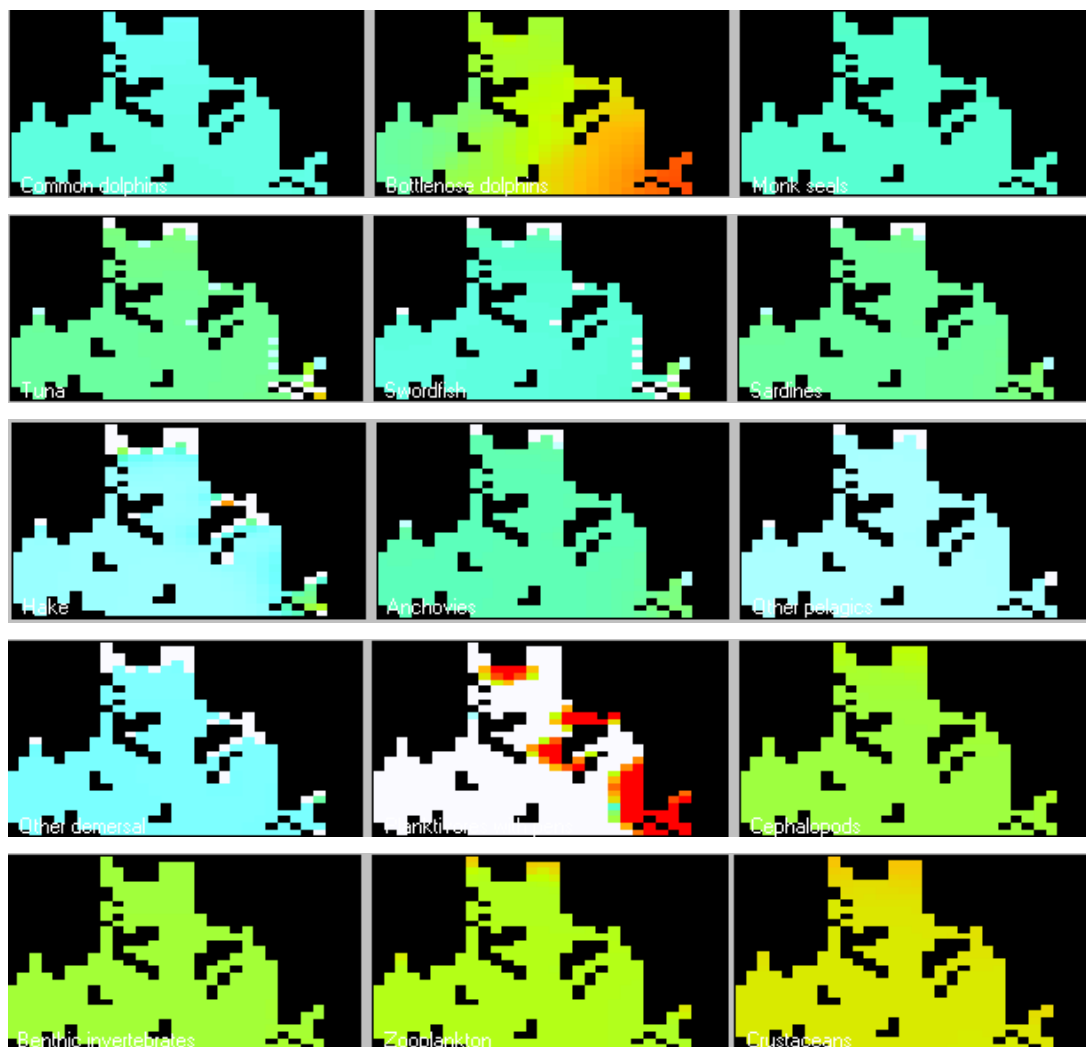




Fig.3.39. Ecospace representation of the 2006 model for each functional group. Colours are expressing relative biomass with green that is equal to the average biomass for the given group in 1964, red that is two times or more high biomass and blue that corresponds to lower biomass.

Furthermore, two different scenarios using MPAs were created, as in *Ecosim*, starting from the year 2009. In the first one, the entire area was closed to all fisheries (MPA1), while in the second scenario, the MPA2 was prohibited just for the three industrial fisheries. The Fig.3.40 and 3.41 show the graphic representations of the implementation of MPAs up to the year 2030, for these two scenarios. In the Fig. 3.40, the positive effect of MPA closure is clearly visible for the entire marine ecosystem. All the functional group would benefit from this management action. In the Fig. 3.41, by allowing small and big artisanal fishery to operate, the restoration of sardines and anchovies is achieved, while common dolphins, monk seals, swordfish, other pelagics and other demersals don't show any significant variation. However, even if, in the last figure (Fig.3.41), no changes in common dolphin biomass were observed, as it possible to see from the Table 3.7, common dolphin biomass would increase slightly with the implementation of the MPA2. This is the case also for monk seals, swordfish, other pelagics and other demersals. These spatial results agree with the similar scenarios simulated using *Ecosim* (Fig.3.36, Fig. 3.37, Fig. 3.38).

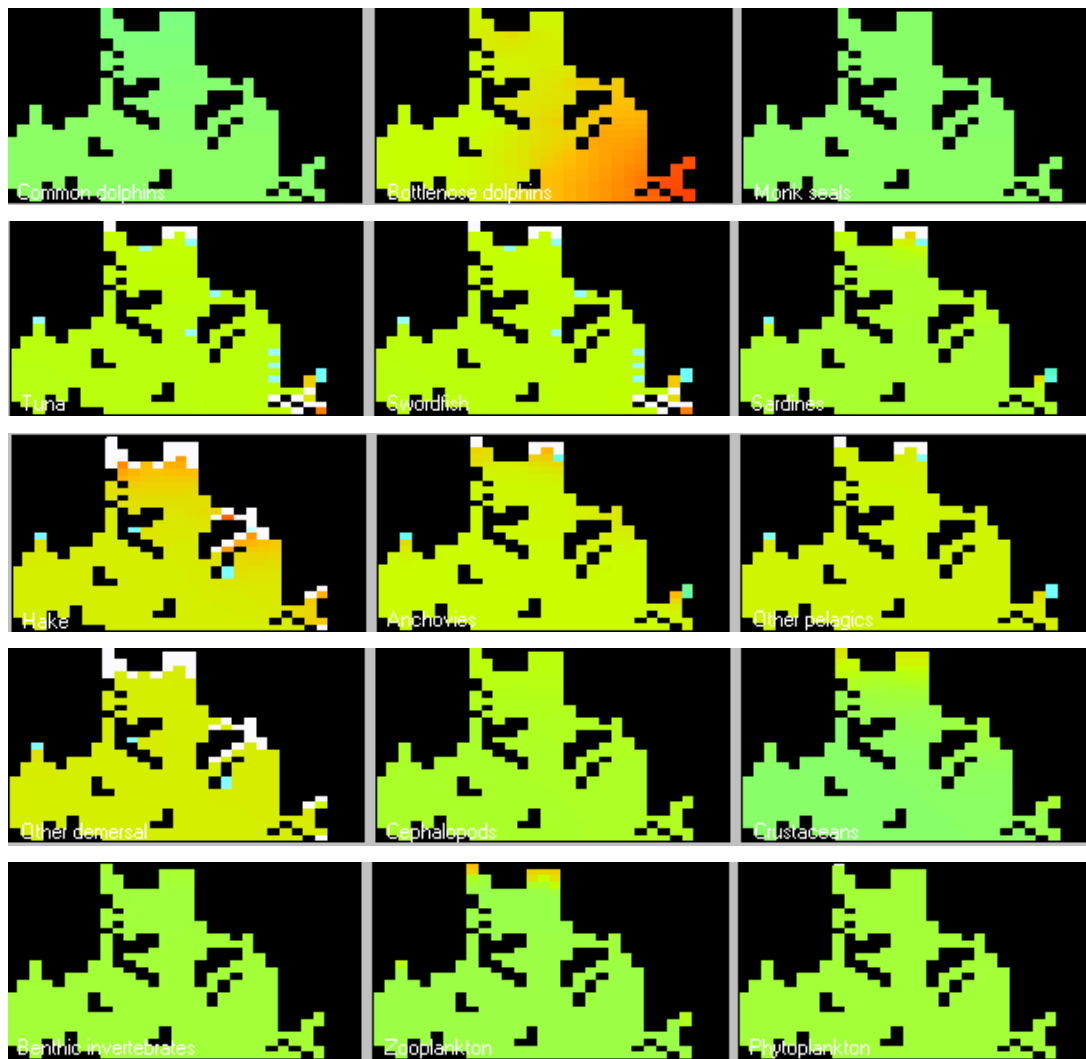


Fig.3.40. Ecospace representation of the 2030 model state, for each functional group, after the creation of a MPA1 for the entire area, closing the whole fishing industry.

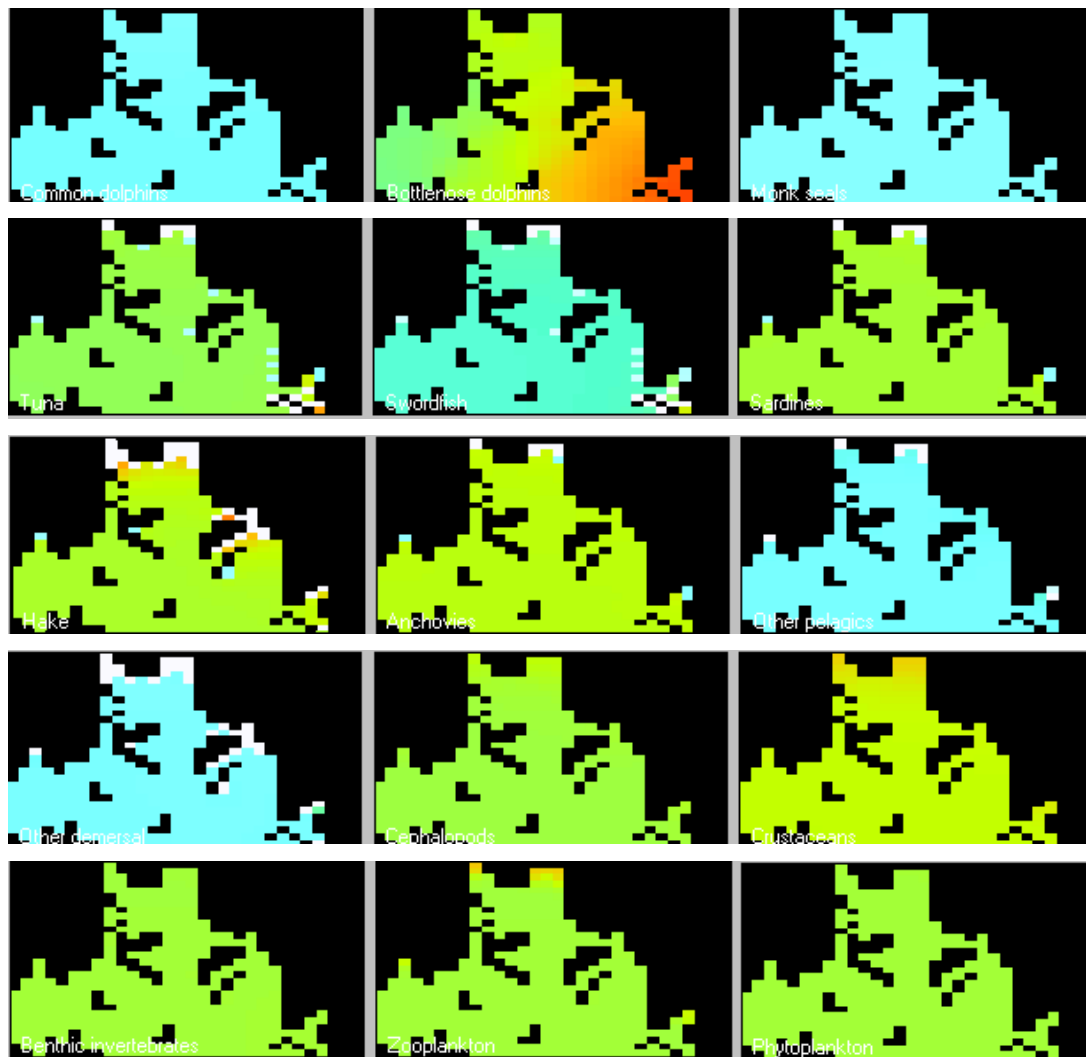


Fig.3.41. Ecospace representation of the 2030 model, for each functional group, after the creation of a MPA2 for the entire area, closing the three industrial fisheries.

Table 3.7. Ecospace output for three different scenarios. The biomass ($t \cdot km^{-2}$) of the main functional groups is shown for the '2006' model (without MPA), for the '2030' model with the closure of all fisheries (MPA1) and for the '2030' model with the closure of the three industrial fisheries (MPA2).

Group name	Biomass 2006	Biomass 2030	Biomass 2030
	No MPA	MPA1	MPA2
Common d.	0.0061	0.084	0.0069
Bottlenose d.	0.021	0.026	0.022
Monk seal	0.004	0.012	0.008
Tuna	0.026	0.047	0.034
Swordfish	0.027	0.071	0.029
Sardines	1.39	2.18	2.07
Anchovies	0.82	1.66	1.40
Other pelagics	0.45	2.73	0.70
Hake	0.11	0.26	0.19
Other demersals	0.24	1.24	0.29
Cephalopods	0.37	0.44	0.38
Crustaceans	4.56	2.63	3.99

CHAPTER 4: DISCUSSION

4.1 Time-series fitting

Ecopath with *Ecosim* was used for the NE Ionian Sea to better understand the trophic interactions between two different populations of dolphins and the other components of the marine ecosystem, and, in particular, to explain why common dolphins have decreased rapidly in the study area. Results shown in Chapter 3 support the hypothesis that the observed decline of these dolphins is due to overfishing (Bearzi 2005, 2006). In particular, the model suggested that this decline gradually increased since the end of the 1970s as a consequence of the decline of sardines and anchovies stocks, the main prey items of common dolphins. It is likely that bycatch of dolphins has added to the decline, but there are only a few records of bycatch reported in the area (Bearzi, 2006). Moreover, surveys conducted outside the study area have recorded no migration or a change in spatial distribution of these photoidentified dolphins, suggesting that the population has not moved elsewhere (Bearzi et al., 2005). Therefore, the lack of prey availability appeared to be the most plausible reason for the rapid decrease of common dolphin numbers in the area. Moreover, as an effect of this decline, low reproductive rates have been observed in the area with a P/B that was set to a $0.05 \cdot \text{year}^{-1}$ to improve the fit with the observed data.

According to *Ecosim* predictions, the catches of sardines and anchovies have increased through time reaching a peak in 1993 with a decline afterwards. An interesting aspect of these scenarios (Fig.3.13 and Fig.3.14) is related to the fact that predicted sardine landings and in part also anchovy landings, have shown higher trajectory compared to the observed one, suggesting an underestimation of the catch data. This agrees with other studies that have pointed out how Greek landing data are unreliable (Stergiou et al., 1997, 1998; Briand, 2000) and in particular in the study area where fishers have often deliberately misreported their catches to avoid more restricted regulations or taxations (Bearzi et al., 2006). Unfortunately, as shown in Chapter 3, no significant results were obtained for other large marine predators, such as tuna and swordfish, feeding on epipelagic prey (sardines and anchovies), even if they have also been observed declining in the area (Bearzi et al., 2006). This is must

probably due to the fact that these two pelagic fish are highly migratory and influenced by events at a scale much larger than studied here.

Bottlenose dolphins, on the other hand, have progressively increased in the study area. The model suggested that this increase was due not to the decline of common dolphin, but instead, to the rapid development of marine fish culture along the coasts of the study area. The opportunistic feeding behaviour of bottlenose dolphins was previously observed and discussed by Bearzi et al. (2004b), and it has been confirmed in this study. The association of bottlenose dolphins with fish farms is a phenomenon that is not uncommon in the Mediterranean Sea; different studies have demonstrated how the distribution of bottlenose dolphin is directly linked to the presence of fish farms (Pulcini et al., 2004; Díaz López, 2006). The concentration of bottlenose dolphins around fish farm areas is mainly due to a high abundance of prey species near the fish pens. Several works have, indeed, shown the positive effect of fish farms on the increase of wild fish distribution especially in the surrounding areas. Here, wild fish are generally attracted by pellets lost through the cages and particulate organic matter (Dempster et al., 2002; Machias et al., 2004, 2005; Giannoulaki et al., 2005).

Moreover, Belias et al. (2003) have discussed the effect of coastal aquaculture on the studied marine ecosystem; the study has demonstrated how fish farm operations have affected the environmental parameters especially in the vicinity of the fish pens. In particular, while salinity, pH, dissolved oxygen and temperature registered normal values, the concentrations of ammonia, phosphates, silicates and dissolved organic carbon have increased compared to the reference sites (areas without fish farms). These new environmental conditions have created a new trophic resource for wild fish, favouring, therefore, bottlenose dolphin foraging activities.

The simulated biomass trajectory of monk seal reproduced very well the observed decline of this marine mammal in the area. Even in this case, the interaction with fisheries is considered the main cause of the decrease. However, in contrast to common dolphins, prey availability does not seem to be the only reason of the collapse of monk seal. Indeed, as suggested by the model, direct and incidental killings caused by fishers played an important role for the decline

of monk seal biomass. In Greece, for the past forty years, fishers have harassed seals as competitors for fish resources. Because of its opportunistic feeding behaviour, and because of the decline of the main demersal and pelagic stocks, monk seal have been attracted to feeding close to fishing nets and recently fish farm cages (Panou et al., 1993; Androukaki et al. 1999). As results, the increase of fishing gear damage, and fish loss have intensified fishers' anger toward these animals. In particular, in the Ionian Sea, deliberate killing and deaths in fishing gears seemed to be the main cause of the population's decline (Cebrian 1992; Panou et al., 1993; Androukaki et al., 1999).

For most of the commercial fish stocks, the model was able to follow the observed trends; in particular, sardines, anchovies, pelagic and demersal CPUE have shown a remarkable decline since the late 1960s. The collapse is mainly due to an intensive fishing effort that kept on increasing until the end of the 1990s, as discussed also by Stergiou et al. (2007) and Kapadagakis et al. (2001). The simulated trajectories shown by the model concord with other studies for the Eastern Ionian Sea that demonstrated how trawl and purse seine catches have considerably declined between 1996 and 2000, suggesting a decline in both pelagic and demersal stocks (EC, 2004). Another interesting aspect of these scenarios is that, while the most important pelagic and demersal stocks have clearly decreased through time, crustaceans CPUE has gradually increased. This could be related to the so called 'Fishing Down Marine Food Web' described by Pauly et al. (1998). This phenomenon occurs when higher trophic level predators are overfished and removed from the ecosystem, and gradually replaced by lower trophic level organisms.

Furthermore, as mentioned in Chapter 3, the predicted trends of sardines, anchovies, pelagic and demersal CPUE, fitted the observed trajectories when changes of nutrient concentration were added to the model. Changes through time in nutrient loading have consequently generated changes in primary production (PP), as shown in the Chapter 3 (Fig.3.1). The simulated trend of primary production has evidenced a general low concentration of PP through time with the exception of three high peaks observed for the periods 1991-1994, 1996-1999 and 2003-2005. Unfortunately, it was not possible to compare this predicted

PP trajectory with observed data because of a lack of such data from the study area.

A possible explanation to the modelled trend could be related to observed climate changes that have influenced the thermohaline circulation of the Eastern Ionian Sea. Several studies, indeed, have evidenced changes in water masses features of the South Aegean Sea between 1986-1997 that have affected the Eastern Ionian Sea circulation (Theocharis et al., 1992; Stergiou et al., 1997; Theocharis et al., 1999; Manca et al., 2002). According to these researches, since 1990, waters coming from the Cretan Sea were found for the first time in the deep and bottom waters of the Ionian Sea. The mechanisms that have induced these hydrological changes were mainly due to two different forcing factors in two distinct periods inside the Cretan Sea. The first one was related to an increase of salinity, between 1987-1992, due to a continuous period of reduced rainfall, while the second one, corresponded to a decrease of temperature between 1992-1994 (Theocharis et al., 1992, 1999). The combinations of these two factors created a mass of dense water and a strong continuous outflow towards the deep parts of the Ionian Sea. This phenomenon has changed the hydrological structure of the Eastern Mediterranean bringing warm, saline, and very dense waters rich in oxygen into the Ionian Sea (Stergiou et al., 1997; Theocharis et al., 1999).

Furthermore, Manca et al. (2002) have demonstrated that these climatic shifts have modified the dynamics of the upper, intermediate and bottom layers of the Ionian Sea also in the period 1997-1999. In particular, the upper layer was mainly influenced by a warmer and less saline water mass, the intermediate layer had higher values of salinity and waters richer in oxygen, and the bottom layer was influenced by denser waters of Aegean origin. It is worth noticing that the main changes observed in the Ionian Sea water mass match the peaks in system productivity estimated by the model.

4.2 Fishing closure scenarios

Fishing closures were proposed in this study as a management strategy for the conservation of common dolphins and for the preservation of the main key species (sardines and anchovies) that inhabit the NE Ionian Sea. As mentioned in

the previous Chapter, four different scenarios were evaluated. In the first one, the fishing effort of each fishery was maintained at the 2003-level up to 2030. This was done in order to assess the impact of fisheries on common dolphins and its prey, keeping constant the current situation. It is clear that this predicted scenario call into question the survival of common dolphin's survival in the area; the pressure of fisheries on sardines and anchovies CPUE doesn't allow the restoration of dolphin population. The same result is obtained for anchovies that keep on declining through time until they collapse, while sardines will gradually increase.

It is worth mentioning that sardine and anchovy trends are influenced by the density-dependent catchability parameter estimate. Indeed, the decrease of anchovies CPUE is related to the low abundance of the stock observed in the last year (2003). This allows the fisheries to deplete them all. The opposite scenario is shown for the sardines: a higher abundance in the last year, will let them spread more and increase through time. The other three scenarios have been proposed to evaluate the effects of different fishing closures for the entire study area. By closing the area to certain fisheries or to the entire fishing fleets, common dolphins would gradually increase, and sardine and anchovy stocks would reach a plateau.

However, the social structure of common dolphins could already have reached a point of social collapse that may not allow the population to rebound. As discussed by Bearzi et al. (2005), the common dolphins of the NE Ionian Sea have been observed to progressively distribute themselves in smaller and dispersed groups, and exhibit low reproductive rates. Generally, this situation occurs, as a behaviour strategy, when the prey availability is scarce (Norris and Dohl, 1980; Chapman and Chapman, 2000). Moreover, Chapman and Reiss (1999) have pointed out how the lack of food resources plays an important role in the reproductive rates of a population becoming one of the main factors to control animal population sizes. This could be particularly significant in common dolphins where sperm competition is the characteristic of the reproductive strategy (Murphy et al., 2005). Both males and females mate several times with different partners and, therefore, living in large groups becomes essential for their reproductive success.

Another essential limitation of these scenarios is related to fisheries regulations. As mention in the first Chapter, measures to regulate fisheries issues have been provided since the early 1960s. However, despite the existing legislations, demersal and pelagic fisheries resources have been overfished. This could be attributed not only to the inadequacy of the current management system (Stergiou et al., 1997, 2007) but also to illegal activities and non-respected regulations surveyed in the study area (Tethys Research Institute, 2007). Therefore, in order to adopt these fishing closures and make them effective, it is necessary to implement a severe control of the fishing activities especially regarding the fulfilment of the existing legislations.

4.3 Ecospace

The creation of MPAs was evaluated using the spatial model *Ecospace*. This was done by exploring two alternative scenarios: the implementation of a MPA1, in which the entire study area was closed to all fisheries, and the implementation of a MPA2, in which the three main industrial fisheries were prohibited. The results shown in the first scenario clearly showed the importance of closing all the area in order to allow the rebuilding of common dolphin population and the most important commercial stocks. In the second scenario, which allows the artisanal fisheries to operate, the restoration of sardines and anchovies is achieved, while common dolphins, monk seals, other pelagics and other demersals only showed slight increase.

In Greece, particularly in the study area, the adoption of MPAs has been considered one of the most effective measures for the restoration of common dolphins and for the protection of epipelagic spawning habitats (Stergiou et al., 1997; Bearzi et al., 2006; Stergiou et al., 2007). These refuges could provide more abundant recruits and larger adult sizes, not only inside the MPA, but also in the surrounding fished areas. This in turn may become beneficial for marine predators and for the local fisheries also outside the MPAs. The creation of a Marine Protected Area could be particularly successful in coastal areas like the NE Ionian Sea where the extreme multi-species and multi-gear nature of the inshore fisheries make their management difficult (Stergiou et al., 1997).

The model appears to be an effective management tool for the implementation of MPAs in the study area. However, in order to create a successful marine harvest refuge, a deeper assessment on the spatial distribution and the dispersal rate of the target species living in the area is required. Indeed, it has been demonstrated that a small MPA can be valuable for the restoration of target species with low dispersal rates, while it is not effective for species with higher dispersal rates (Beattie et al., 2002).

Moreover, results obtained using *Ecospace*, have demonstrated that larger MPAs should be preferred over smaller ones due to spatial cascade effects, where predators are well protected and abundant, and prey densities consequently low. This situation favours movements of predators towards available food (outside and MPA), and also, as a consequence, a higher concentration of fishing effort at the edge of the MPAs (Walters et al., 1999). In the study area, an effective MPA, that aims to protect small pelagics and predators (e.g., common dolphins), should be, therefore, big enough to include spatial movements of anchovies and sardines due to spawning and spatial movements of common dolphins towards food resources.

CHAPTER 5: CONCLUSIONS

An *Ecopath* with *Ecosim* (EwE) model was built for the NE Ionian Sea to better understand the dynamics of two different populations of dolphins and their trophic interactions with the ecosystem. In particular, this model was chosen to assess the impact of local fisheries on common dolphin community and its prey. The *Ecopath* model was constructed considering 22 functional groups for the year 1964; data on biomass, P/B, Q/B and diet for each group were estimated or extrapolated from the literature. Fisheries landings and discards were also included.

Simulated temporal trajectories were made using *Ecosim*. In particular the model was fitted with time series data for the most important groups from 1964 to 2006. The results of these simulations make evident a clear decline of common dolphins due to the decline of their prey. Almost all the fisheries target species faced a sharp decrease since the late 1970s; the overexploitation of the marine fish resources was mainly caused by an intensive fishing pressure that occurred in the area until the end of 1990s. Moreover, it seemed also that the level of primary production played an important role in the trajectories of the main functional groups. In particular, simulated changes through time in PP concentration impacted the abundance trends of all the commercial species, showing a cascade-up effect throughout the ecosystem.

On the other hand, bottlenose dolphins have increased through time; a positive correlation was observed between this species of dolphin and the development of the fish farm industry in the study area. The model suggested that the opportunistic feeding behaviour of bottlenose dolphins close to fish pens have favoured their increase. *Ecosim* was then used to simulate three fishing policy scenarios within the study area. Three different fishing closures were investigated:

1. The closure of purse seiners;
2. The closure of purse seiners, trawlers and beach seiners; and
3. The closure of the entire fishing fleets.

The impact of these fishing closures on common dolphins, sardines and anchovies were evaluated. According to *Ecosim* predictions, sardines and anchovies would benefit from these management strategies, starting to increase from the first scenario. Common dolphins, on the other hand, would show a more evident increase only when the entire study area would be closed to fishing industry. Same conclusions were obtained using the spatial model *Ecospace*. Two different MPAs have been evaluated using two different scenarios: in the MPA1, the entire area was closed to fisheries, while in the MPA2, purse seiners, trawlers and beach seiners were not allowed to operate.

In the first scenario, the model suggested that an MPA for the entire area would be the best solution to benefit both sardine, anchovies stocks and common dolphin population. In the second scenario, the creation of an MPA with small and big artisanal fishery free to operate, would be beneficial for the restoration of sardines, anchovies stocks and to less extend for the conservation of common dolphin population. *Ecospace* was a good starting point for the implementation of MPAs as management strategy for the study area. As mentioned in Chapter 4, more information on spatial distribution of target species is required, but overall, effective outcomes have been observed.

The application of *Ecopath* with *Ecosim* and *Ecospace* appeared to be a useful tool for the understanding of this marine ecosystem. The model has underlined that management actions are essential for the restoration and protection of common dolphin, bottlenose dolphin populations and other target species. In particular, measures to reduce overfishing, illegal fishing activities and to respect existing legislations are in need. The adoption of a marine protected area seemed to be an effective management measure to enhance prey survival and to sustain marine predators like common dolphins. These measures need to be adopted immediately if we want to stop the rapid disappearance of one of the last populations of common dolphin in the Mediterranean Sea.

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APPENDIX A

Table 3.8. Relative biomass (n. of individuals), Catch ($t \cdot km^{-2}$), CPUE ($t/HP \cdot year^{-1}$) $\cdot 1000$, and bycatch ($t \cdot km^{-2}$) time series data for the main functional groups of the model.

	1. Common d.		2. Bottlenose d.		3. Monk seal		6. Tuna		7. Swordfish		8. Sardine		9. Anchovy		10. Pelagics	
	B	By-catch	B	By-catch	B	By-catch	CPUE	Catch	CPUE	Catch	CPUE	Catch	CPUE	Catch	CPUE	Catch
1964						0.0003	0.134	0.002			10.581	0.193	7.325	0.134	14.138	0.022
1965						0.0003	0.553	0.009			10.314	0.190	6.659	0.123	18.949	0.027
1966						0.0003	0.274	0.005			7.816	0.164	5.695	0.119	12.281	0.019
1967						0.0003	0.152	0.003			4.572	0.106	5.883	0.136	9.226	0.015
1968						0.0003	0.120	0.003			3.697	0.108	4.105	0.120	8.282	0.020
1969						0.0003	0.110	0.003			3.426	0.107	3.595	0.112	6.728	0.020
1970						0.0003	0.572	0.011	0.510	0.007	4.989	0.118	4.367	0.103	9.171	0.024
1971						0.0003	0.610	0.014	0.538	0.009	5.634	0.150	6.009	0.160	10.225	0.030
1972						0.0003	0.445	0.010	0.396	0.007	5.114	0.138	4.714	0.127	8.405	0.026
1973						0.0003	0.320	0.009	0.278	0.006	3.951	0.129	3.127	0.102	6.130	0.028
1974						0.0003	0.275	0.008	0.238	0.005	2.833	0.099	2.006	0.070	5.378	0.027
1975						0.0003	0.170	0.006	0.140	0.004	1.685	0.070	0.946	0.040	3.195	0.016
1976						0.0003	0.210	0.008	0.176	0.006	2.008	0.094	1.785	0.084	4.225	0.023
1977			35			0.0003	0.263	0.012	0.219	0.008	2.610	0.138	1.942	0.103	6.200	0.036
1978						0.0003	0.222	0.012	0.174	0.008	2.489	0.156	2.229	0.140	6.348	0.044
1979						0.0003	0.220	0.013	0.165	0.008	2.717	0.184	2.750	0.186	6.295	0.043
1980						0.0003	0.195	0.013	0.145	0.008	2.011	0.154	2.310	0.177	4.740	0.039
1981						0.0003	0.153	0.011	0.108	0.007	1.575	0.132	1.582	0.133	3.790	0.031
1982						0.0003	0.462	0.038	0.198	0.014	1.425	0.131	2.023	0.186	3.960	0.038
1983						0.0003	0.563	0.052	0.247	0.020	1.541	0.157	2.151	0.219	5.117	0.056
1984						0.0003	0.453	0.045	0.233	0.021	1.227	0.134	2.302	0.251	4.134	0.052
1985						0.0003	0.340	0.036	0.255	0.024	1.163	0.134	2.105	0.243	3.857	0.054
1986			12			0.0003	0.333	0.038	0.272	0.028	1.003	0.124	1.994	0.247	3.609	0.064

Table 3.9. continued

	1. Common d.		2. Bottlenose d.		3. Monk seal		6. Tuna		7. Swordfish		8. Sardine		9. Anchovy		10. Pelagics	
	B	By-catch	B	By-catch	B	By-catch	CPUE	Catch	CPUE	Catch	CPUE	Catch	CPUE	Catch	CPUE	Catch
1987					8	0.0003	0.255	0.032	0.175	0.019	0.805	0.107	2.228	0.296	3.126	0.047
1988							0.310	0.052	0.262	0.040	1.002	0.182	2.113	0.383	3.645	0.094
1989						0.001	0.301	0.052	0.303	0.046	0.839	0.154	1.327	0.244	3.461	0.086
1990							0.198	0.035	0.165	0.026	0.694	0.129	1.054	0.196	2.516	0.063
1991							0.210	0.037	0.185	0.029	0.913	0.171	1.055	0.198	2.848	0.060
1992							0.160	0.028	0.122	0.019	0.941	0.176	0.691	0.129	2.233	0.056
1993							0.496	0.086	0.295	0.045	2.510	0.466	2.333	0.433	6.884	0.177
1994							0.132	0.023	0.125	0.019	0.667	0.121	0.736	0.134	2.422	0.057
1995	120		17				0.223	0.038	0.184	0.028	1.031	0.188	0.769	0.140	2.713	0.072
1996	112		18				0.161	0.028	0.130	0.020	0.740	0.136	0.626	0.115	2.092	0.048
1997	102		21				0.133	0.023	0.117	0.018	0.630	0.116	0.491	0.091	1.589	0.036
1998	98		28				0.325	0.053	0.256	0.037	1.457	0.256	1.428	0.251	2.833	0.070
1999	81		36				0.129	0.025	0.097	0.017	0.559	0.116	0.577	0.120	1.218	0.035
2000	108		29				0.249	0.043	0.182	0.028	0.993	0.183	0.622	0.114	2.007	0.052
2001	95		31				0.130	0.023	0.111	0.017	0.515	0.097	0.361	0.068	1.036	0.027
2002	73		32				0.174	0.028	0.126	0.018	0.824	0.145	0.488	0.086	1.544	0.038
2003	59		53				0.280	0.043	0.235	0.032	0.988	0.164	1.184	0.197	2.708	0.068
2004	66		42													
2005	44	0.0005	53	0.0003												
2006	18		24													

Table 3.10. continued

	11. Hake		12. Demersals		14. Cephalopods		15. Crustaceans		20. Fish farms
	CPUE	Catch	CPUE	Catch	CPUE	Catch	CPUE	Catch	Forced B
1964	1.823	0.399	5.281	0.149	1.654	0.047	1.098	0.011	0.001
1965	2.298	0.532	6.869	0.193	2.223	0.062	1.136	0.011	0.001
1966	1.601	0.376	4.800	0.147	1.454	0.044	0.782	0.008	0.001
1967	1.172	0.306	3.822	0.127	1.066	0.035	0.809	0.008	0.001
1968	1.524	0.326	3.747	0.148	0.961	0.038	1.032	0.011	0.001
1969	1.399	0.285	3.002	0.127	0.794	0.034	0.939	0.011	0.001
1970	1.523	0.328	3.766	0.135	1.632	0.058	0.859	0.010	0.001
1971	1.841	0.400	4.195	0.164	1.893	0.074	0.923	0.011	0.001
1972	1.653	0.325	3.627	0.140	1.400	0.054	0.740	0.009	0.001
1973	1.371	0.294	2.948	0.142	0.987	0.047	0.551	0.008	0.001
1974	1.240	0.277	2.623	0.135	0.887	0.046	0.365	0.006	0.001
1975	0.646	0.193	1.567	0.095	0.547	0.033	0.222	0.004	0.001
1976	0.913	0.278	2.163	0.142	0.684	0.045	0.277	0.005	0.001
1977	1.402	0.438	2.897	0.205	0.920	0.065	0.421	0.007	0.001
1978	1.587	0.516	2.936	0.239	0.952	0.077	0.515	0.010	0.001
1979	1.535	0.546	3.030	0.263	0.948	0.082	0.533	0.010	0.001
1980	1.388	0.455	2.496	0.240	0.799	0.077	0.458	0.009	0.001
1981	1.078	0.394	1.892	0.197	0.602	0.063	0.448	0.009	0.001
1982	1.282	0.444	2.025	0.227	0.562	0.063	0.966	0.019	0.001
1983	1.868	0.626	2.605	0.319	0.746	0.091	1.378	0.028	0.001
1984	1.685	0.539	2.036	0.266	0.571	0.074	1.254	0.027	0.001
1985	1.792	0.526	1.931	0.263	0.546	0.074	1.084	0.023	0.001
1986	2.080	0.526	2.020	0.295	0.565	0.082	1.376	0.030	0.001

Table 3.11. continued


	11. Hake		12. Demersals		14. Cephalopods		15. Crustaceans		20. Fish farms
	CPUE	Catch	CPUE	Catch	CPUE	Catch	CPUE	Catch	Forced B
1987	1.559	0.483	1.765	0.273	0.470	0.073	1.392	0.030	0.001
1988	2.291	0.767	2.156	0.454	0.648	0.136	2.015	0.058	0.001
1989	2.044	0.740	2.048	0.438	0.533	0.114	1.446	0.043	0.001
1990	1.457	0.548	1.466	0.319	0.484	0.105	1.303	0.041	0.047
1991	1.368	0.627	1.661	0.365	0.568	0.125	1.215	0.039	0.082
1992	1.297	0.488	1.310	0.286	0.553	0.121	1.187	0.038	0.123
1993	4.086	1.495	4.043	0.878	1.353	0.294	2.826	0.089	0.145
1994	1.332	0.517	1.267	0.270	0.420	0.090	0.856	0.027	0.220
1995	1.678	0.580	1.420	0.303	0.690	0.147	1.228	0.038	0.271
1996	1.121	0.452	0.996	0.215	0.467	0.101	0.868	0.027	0.345
1997	0.856	0.343	0.790	0.170	0.369	0.080	0.842	0.026	0.359
1998	1.737	0.578	1.613	0.329	0.801	0.163	1.918	0.055	0.414
1999	0.983	0.280	0.727	0.167	0.388	0.089	1.031	0.023	0.478
2000	1.147	0.436	1.196	0.260	0.620	0.135	1.137	0.038	0.520
2001	0.579	0.229	0.606	0.134	0.316	0.070	0.619	0.020	0.551
2002	0.807	0.323	0.842	0.176	0.514	0.108	0.754	0.026	0.579
2003	1.428	0.541	1.566	0.313	0.977	0.195	1.525	0.052	0.579
2004									0.579
2005									0.579
2006									0.579

APPENDIX B



The University of British Columbia
Office of Research Services and Administration
Behavioural Research Ethics Board

Certificate of Approval

PRINCIPAL INVESTIGATOR Christensen, V.		DEPARTMENT Fisheries	NUMBER B06-0367
INSTITUTION(S) WHERE RESEARCH WILL BE CARRIED OUT			
CO-INVESTIGATORS Piroddi, Chiara, Zoology			
SPONSORING AGENCIES Unfunded Research			
TITLE An Ecosystem Based Approach to Study Dolphin Population Dynamics in the Eastern Ionian Sea			
APPROVAL DATE AUG 21 2006	TERM (YEARS) 1	DOCUMENTS INCLUDED IN THIS APPROVAL: May 6, 2006, Questionnaires	
<p>CERTIFICATION:</p> <p>The application for ethical review of the above-named project has been reviewed and the procedures were found to be acceptable on ethical grounds for research involving human subjects.</p> <div style="text-align: center;">  <hr/> <p><i>Approved on behalf of the Behavioural Research Ethics Board</i> by one of the following: Dr. Peter Suedfeld, Chair, Dr. Jim Rupert, Associate Chair Dr. Arminee Kazanjian, Associate Chair</p> </div> <p>This Certificate of Approval is valid for the above term provided there is no change in the experimental procedures</p>			