Journal of Marine Systems 102–104 (2012) 39–51

Contents lists available at SciVerse ScienceDirect

Journal of Marine Systems

journal homepage: www.elsevier.com/locate/jmarsys

Fishing management scenarios to rebuild exploited resources and ecosystems of the Northern-Central Adriatic (Mediterranean Sea)

Nadia Fouzai [a], Marta Coll [a,b,*], Isabel Palomera [a,b], Alberto Santojanni [c], Enrico Arneri [c,d], Villy Christensen [b,e]

[a] Institute of Marine Science (ICM-CSIC), Passeig Marítim de la Barceloneta, 37-49, 08002 Barcelona, Spain
[b] Ecopath International Initiative Association, Barcelona, Spain
[c] Istituto di Scienze Marine (CNR), Sede di Ancona, Largo Fiera della pesca, 2-60125 Ancona, Italy
[d] FAO AdriaMed and MedSudMed Projects, Viale delle Terme di Caracalla, 00153 Rome, Italy
[e] Fisheries Centre, University of British Columbia, 2202 Main Mall, Vancouver, British Columbia, Canada V6T 1Z4

ARTICLE INFO

Article history:
Received 19 January 2012
Received in revised form 1 May 2012
Accepted 3 May 2012
Available online 10 May 2012

Keywords:
Northern-Central Adriatic Sea
Trophic models
Ecopath with Ecosim
Ecospace
Fisheries management
Marine protected areas (MPA)

ABSTRACT

We examined various fishing management options to recover exploited marine resources and ecosystems of the Northern-Central Adriatic Sea. Dynamic simulations were based on a spatial ecological model previously calibrated with time series of data. Scenarios regarding spatial management were evaluated with the establishment of two marine protected areas, respectively, in the Pomo pit and the northern region. In addition, three temporal simulations of temporary closures and overall reduction of fishing effort of demersal and pelagic fleets (bottom, mid-water trawls and purse seines) were also considered. Simulations were run for 45 years (1975–2020), including the calibration period (1975–2002), and changes in biomass and catch of marine resources were analyzed. Our results confirm that current fishing management in the Adriatic Sea does not have clear beneficial impacts for the recovery of exploited resources, which will remain depleted in 2020 if "business as usual" continues. Simulations of alternative management suggest that both protected areas could be beneficial for fish population recovery predicting an increase in the biomass of commercial fish and predatory organisms. Simulations of temporary closures and overall reduction of fishing effort also show significant benefits for several commercial resources. We argue that both management measures may be effective tools to recover exploited ecosystems of the Northern-Central Adriatic Sea and halt the decline of marine resources.

1. Introduction

Over the past, fishing activity has been recognized to be a source of dramatic, direct and indirect impacts on both coastal and open-sea marine ecosystems (Christensen et al., 2003; Coll et al., 2008b; Jackson et al., 2001; Pauly et al., 2002). Many marine ecosystems show a growing degradation, with frequent destruction of benthic habitats, depletion of target stocks, modifications of species abundance and alterations of the food web (Lotze et al., 2006; Pauly et al., 1998, 2003). These impacts illustrate the challenges being faced while managing marine resources using traditional single species analysis tools, and the growing need to implement additional ecosystem approaches (FAO, 2003; Link, 2011; Roberts, 1997).

Fishery research is shifting towards adopting an ecosystem-based approach to strive for a sustainable and profitable exploitation, while maintaining healthy ecosystems (Browman and Stergiou, 2004; Garcia et al., 2003). The 'Ecosystem-based fisheries management' (EBFM) has been proposed as a new paradigm of fisheries management that should consider not only fisheries, but also other biotic, abiotic, and human components of ecosystems and their interactions (FAO, 2003). This approach intends to use multivariate and interdisciplinary evaluations to improve existing management frameworks (Chen et al., 2008; Garcia and Cochrane, 2005). Ecosystem analyses are useful tools to better understand and investigate the ecosystem impacts of fishing and pose strategic questions (Christensen and Walters, 2004; Martell and Walters, 2008). In the last few decades, trophic network models of aquatic ecosystems are increasingly appearing in the scientific literature, among them many based on the Ecopath with Ecosim modeling approach (EwE) (Christensen and Pauly, 1992; Christensen and Walters, 2004; Christensen et al., 2008; Polovina, 1984). EwE is a widely-used ecosystem modeling tool for the analysis of exploited aquatic ecosystems and can be a useful tool to contribute to the EBFM (Christensen et al., 2008; Coll and Libralato, 2012; Palomares et al., 2009).

The Northern-Central (NC) Adriatic Sea is one of the most productive areas of the Mediterranean Sea and one of the major fishing grounds in southern Europe. It plays an important role in the economies of European countries such as Italy, Croatia and Slovenia (Mannini and Massa, 2000). However, a dramatic expansion of marine capture fisheries has taken place since early 1970s (Coll et al., 2009, 2010b; Fortibuoni et al., 2010;

* Corresponding author. Tel.: +34 230 95 00; fax: +34 230 95 55.
E-mail address: mcoll@icm.csic.es (M. Coll).

0924-7963/$ – see front matter
doi:10.1016/j.jmarsys.2012.05.003

Lotze et al., 2011a). This expansion has been followed by fluctuations in annual landings and a general decline of marine resources. Since the late 1980s, marine capture has progressively declined, especially for European anchovy *Engraulis encrasicolus* and European sardine *Sardina pilchardus* stocks (Azzali et al., 2002; Mannini et al., 2004; Santojanni et al., 2003, 2005). Several demersal stocks such as European hake *Merluccius merluccius* have been reported as highly exploited or overexploited already in the 1980s (Arneri, 1996; Jukić-Peladić et al., 2001; Vrgoč et al., 2004) and important amounts of discards are produced (Pranovi et al., 2000, 2001; Sánchez et al., 2007). In the last forty years catches have increasingly been dominated by juveniles and small-sized species, such as small pelagic and cephalopods. Previously abundant catches of large, long-lived, high-trophic-level and high-value organisms have markedly declined (Coll et al., 2007, 2009, 2010b; Fortibuoni et al., 2010; Lotze et al., 2011a).

To facilitate the recovery of marine resources and to rebuild ecosystems, the Italian government has recently adopted a series of management measures, which included; (i) freezing the number of fishing licenses, (ii) declaring closed fishing seasons, (iii) implementing seasonal trawl spatial and temporal closures, (iv) banning harmful fishing gear (e.g., drifting gillnets fishery in the Adriatic Sea since 2003) and (v) protecting juveniles (minimum size for several target species) (AdriaMed, 2005; Anon, 2007). Additional management measures include technical regulations (mesh size and fishing gear) and stock rebuilding projects (protected areas and fishing zones) (Table 1).

Of these measures, two sets are of particular interest. The first one includes spatial and temporal closures applied to all or selected gear types. These measures cover (1) temporal bottom and mid-water trawl nets closures during summer time for 30 to 45 days (AdriaMed, 2005; Demestre et al., 2008), although in 2011 the closure was extended to 60 days, and (2) permanent spatial closures to specific trawl and seine gear types. For example, the use of trawls, seines or similar nets have been prohibited within three nautical miles of the coast or within the 50 m isobaths where this is closer to the coast (Anon, 2007) (Table 1).

A second set of measures incorporate the establishment of marine protected areas (Fig. 1). Under Italian jurisdiction there are 22 marine protected areas (MPAs) that already cover 1840 km^2 of territorial marine waters (AdriaMed, 2005), and where human activities have been limited (transport, fisheries, tourism, etc.). In addition, 11 biological conservation zones ('*Zone di tutela biologica*') have been created to experiment with models of sustainable self-management (AdriaMed, 2005). In the NC Adriatic Sea there are currently two marine protected areas (*Miramare* and *Tremiti*) and four biological conservation zones (areas of *Tenue*, *Tegnue di Porto Falconera*, *Fuori Ravenna* and *Barbare*) (Anon, 2009) (Fig. 1 and Table 1).

Table 1
Summary of the current and additional tested management measures in the Northern-Central Adriatic Sea.

Current management measures	Additional tested management measures
– Limitations through licensing system of the size of the fleet – Limitations to the fishing power of the vessel (engine power, vessel size, and gear size) – Minimum landing sizes – Daily quota/vessel	– Implementation of a marine protected area in the Pomo pit (Central Adriatic Sea) – Implementation of a marine protected area in the northern part of the Adriatic Sea – Implementation of a seasonal fishing ban, applicable to bottom and mid-water trawls, for three months every year, from November to January
– Banning harmful fishing gear (e.g., drifting gillnet fishery in the Adriatic Sea since 2003) – Mesh size regulations	– Reduction of fishing effort of mid-water trawls and purse seine fleets by 25% of the level – Reduction of fishing effort of the bottom trawls by 25% of the level
– Minimum size for several target species. – Temporal bottom and mid-water trawl nets closure during summer time for 30 to 45 days extended in 2011 to 60 days – Permanent spatial closures to specific trawl and seine gear types – Closed areas through the establishment of two marine protected areas ('Área Miramare' and 'Área Tremiti') and four biological conservation zones ('Área Tenue', 'Área Tegnue di Porto Falconera', 'Área fuori Ravenna' and 'Área Barbare')	

Given the exceptional importance of marine resources of the Adriatic Sea, and past worrisome trends, appropriate and sustainable fisheries management in this area is critical. Here, we used an ecological model representing the NC Adriatic Sea ecosystem (Coll et al., 2007), previously fitted to available time series of data from 1975 to 2002 (Coll et al., 2009, 2010b), to evaluate the effects of various alternative strategies for fisheries management. We used the dynamic temporal and spatial modeling routines of the *Ecopath with Ecosim* (EwE) software, Ecosim and Ecospace (Walters et al., 1997, 1999) to explore the benefits of existing spatial management measures on commercial and non-commercial marine resources and on main fishing activities. Moreover, we explored possible resource trends from reduction of fishing effort through establishment of larger MPAs or directly reducing fishing effort through temporal closures. In a few cases elsewhere, reduction in exploitation has helped rebuild depleted marine fish populations and ecosystems (Lotze et al., 2011b; Worm et al., 2009). This has been achieved by merging diverse management actions, including effort restrictions, gear modification, and closed areas. Reduction in fishing effort directly reduces the exploitation rate of target species. Gear modifications may be used to increase selectivity and reduce by-catch of non-target species. Closed areas, especially MPAs, can initiate recovery by protecting spawning stock and rich habitats (Agardy, 2000; Goñi et al., 2008; Roberts, 1997).

2. Materials and methods

2.1. Study area

The NC Adriatic Sea is a semi-enclosed basin located in the northernmost part of the central Mediterranean. This region of the Adriatic Sea is mostly characterized by the presence of the muddy and sandy bottoms (Brambati et al., 1983). The Adriatic Sea can be divided into three main water types: surface water, deep water and the Modified Levantine Intermediate Water (MLIW) (Artegiani et al., 1997a). In addition, the general circulation is baroclinic (Artegiani et al., 1997b). The primary

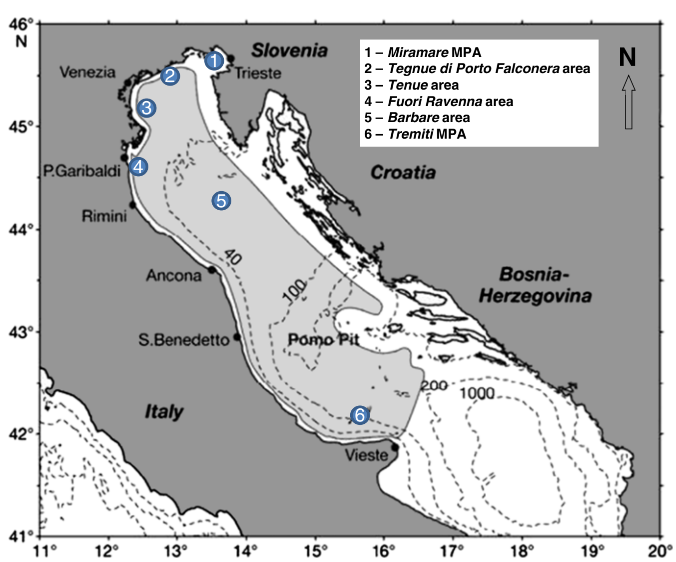

Fig. 1. The Northern-Central (NC) Adriatic Sea study area. The light-gray area represents the spatial coverage of the ecological model. Current protected areas and biological conservation zones are also indicated.

production varies from a productive (potentially eutrophic) shallow northern basin to an oligotrophic deeper central basin (Zavatarelli et al., 2000). This production is influenced by a large number of rivers discharging into the basin, particularly the Po River in the northern basin (Artegiani et al., 1997a; Zavatarelli et al., 1998). The northern and middle regions of the Adriatic Sea are characterized by a high diversity of the environmental conditions that translates to high biodiversity (Ott, 1992).

Our study area was located in the NC Adriatic Sea, geographical subarea (GSA) 17 of the General Fisheries Commission for the Mediterranean (GFCM, FAO). This area was chosen because of its ecological and fisheries characteristics (Bombace, 1992) and the availability of ecological models (Coll et al., 2007, 2009). Its total area is approximately 55,500 km^2, an average depth of 75 m, with maximum depths (in the Pomo pit) at about 273 m (Fig. 1). This area includes Italian territorial waters and the international waters from the 12 miles off the coast of Italy to 12 miles from Croatia and Slovenia. The area within 3 nautical miles from the Italian coast (or less than 10 m depth), where the artisanal fleets mainly operate and trawling is banned, was excluded from the western part of the study area. The area included in Slovenian and Croatian territorial waters from the eastern coast was excluded from the eastern part of the area.

2.2. The ecological model of the NC Adriatic Sea

We developed temporal and spatial simulations using a trophic model built with the Ecopath with Ecosim approach, version 6 (Christensen and Walters, 2004; Christensen et al., 2008). The trophic mass-balance model (Ecopath) had been previously developed to characterize the food-web structure and functioning of the NC Adriatic Sea and to quantify the ecosystem impacts of fishing during the 1990s (Coll et al., 2007). This model was composed of 40 functional groups (defined as single species, several trophically similar species, or just a specific life stage of an individual species), including the main trophic components of the ecosystem, from primary producers to top predators, natural detritus and discards from fishing activities. The most common fishing activities included in the analysis were bottom and beam trawls (here called bottom trawling), mid-water trawls, purse seines and tuna fishing fleets.

The Ecopath model (Christensen and Walters, 2004; Christensen et al., 2008) is built on a system of linear equations to describe the average flows of mass and energy between the species groups during a period of time, normally a year. The flow to and from each functional group is described by the following main equation:

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

where B_i is the biomass of i, P/B_i is the production/biomass ratio, Y_i is the total fishery catch rate, E_i is the net migration rate (emigration − immigration), BA_i is the biomass accumulation rate, EE_i is the "ecotrophic efficiency", the proportion of the production that is utilized in the system, B_j is the biomass of consumers or predators j, $(Q/B)_j$ is the consumption per unit of biomass of j, and DC_{ji} is the fraction of i in the diet of j. The second equation of Ecopath describes consumption as a function of production, respiration and unassimilated food (Christensen and Walters, 2004).

The ecological model describing the NC Adriatic Sea had been fitted to time series of data (Coll et al., 2009, 2010b) using the temporal dynamic module Ecosim (Walters et al., 1997). This calibration enabled the authors to characterize changes in marine resources in the Northern-Central (NC) Adriatic Sea ecosystem from 1975 to 2003, and explore the extent to which these changes were driven by trophic interactions, environment and or fishing.

Ecosim is used to simulate ecosystem effects of fishing mortality changes and environmental forcing over time (Christensen and Walters, 2004; Walters et al., 1997). The process uses a system of time-dependent differential equations from the baseline mass-balance model, where the biomass growth rate is expressed as:

$$dB_i/dt = g_i \cdot \sum Q_{ji} - \sum Q_{ij} + I_i - (M_i + F_i + e_i) \cdot B_i \quad (2)$$

where dB_i/dt represents the growth rate of group i during the time interval dt in terms of its biomass B_i, g_i is the net growth efficiency (production/consumption ratio), M_i is the non-predation $((P/B)_i B_i (1 - EE_i))$ natural mortality rate, F_i is the fishing mortality rate, e_i is the emigration rate, and I_i is the immigration rate ($e_i B_i - I_i$ is the net migration rate). The two summations estimate consumption rates, the first expressing the total consumption by group i, and the second the predation by all predators on the same group i.

The consumption rates, Q_{ji}, are calculated based on the 'foraging arena' concept, where B_i's are divided into vulnerable and invulnerable components (Ahrens et al., 2011; Walters et al., 1997). Ecopath simulations are especially sensitive to the 'vulnerability' settings, which incorporates density-dependency and expresses how far a group is from its carrying capacity (Christensen and Walters, 2004; Christensen et al., 2008). The consumption rate is expressed as:

$$Q_{ij} = \frac{a_{ij} \cdot v_{ij} \cdot B_i \cdot B_j \cdot T_i \cdot T_j \cdot S_{ij} \cdot M_{ij} / D_j}{v_{ij} + v_{ij} \cdot T_i \cdot M_{ij} + a_{ij} \cdot M_{ij} \cdot B_j \cdot S_{ij} \cdot T_j / D_j} \quad (3)$$

where, a_{ij} is the effective search rate for predator j feeding on a prey i, v_{ij} is the base vulnerability expressing the rate with which prey move between being vulnerable and not vulnerable, B_i is the prey biomass, B_j is the predator biomass, T_i represents prey relative feeding time, T_j is the predator relative feeding time, S_{ij} is the user-defined seasonal or long term forcing effects, M_{ij} is the mediation forcing effects, and D_j represents effects of handling time as a limit to consumption rate.

Further details on parameterization and calibration of the NC Adriatic models can be found in Coll et al. (2007, 2009, 2010b).

2.3. Spatial dynamic modeling and parameterization

2.3.1. Spatial model

We used the mass-balance model of the NC Adriatic Sea representing 1975 to develop a new spatial dynamic model using the spatial module Ecospace v6 (Christensen et al., 2008; Walters et al., 1999). The 1975 mass-balanced model was used as the starting point to develop the spatial dynamic model, which was structured on biomass pools of 40 functional groups, linked by trophic flows that move in a spatial grid. Ecospace is a spatially-explicit time dynamic model. It is based on the same set of differential equations as used in Ecosim (Eq. (2)) and represents the biomass (B) and consumption (C) dynamics over a two-dimensional space as well as time that are, varying within spatial coordinates x, y and time t. The space, time and state are considered discrete variables by using the Eulerian approach, which treats movements as 'flows' of organisms among fixed spatial reference cells.

After rectangular grids of spatial cells are defined, cells are assigned to land or water and to a given habitat type, areas of enhanced primary production, and restricted areas to fishing. For trophic interaction, fishing, and movement calculations, biomass is considered as homogeneous within each cell and movement of biomass and flows is allowed across the faces of adjacent cells. For each cell, the immigration rate I_i of Eq. (2) is assumed to consist of up to four emigration flows from the surrounding cells. The emigration flows (Eq. (2)) are in turn represented as instantaneous movement rates m_i times' biomass density B_i in the cell:

$$Bi(x,y) = mi(x,y+1) \cdot Bi(x,y) \quad (4)$$

where (x, y) represents cell row and column.

The instantaneous emigration rates m_i from a given cell in Ecospace are assumed to vary based on the pool type, preferred habitat, and response of organisms to depredation risk and feeding condition. The probability of movement of organisms towards favorable habitats can be calculated by a 'habitat gradient function' for each mapped habitat type and species or group *i*.

2.3.2. Basemap and additional parameterization

We developed a basemap depicting the study area with geographical coordinates 45° 80′ N, 41° 60′ N and 12° 00′ E, 17° 00′ E (Fig. 1). The basemap was represented as a grid consisting of 42 rows and 50 columns, where each grid cell was 5 km in length or 25 km^2 in area. The basemap was associated with relative primary production and average depth of the study area for each cell (Christensen et al., 2008).

Based on the depth strata and the bottom type in the study area (Brambati et al., 1983), six habitat types were defined in the basemap (Fig. 2). Species preferences were assigned to these habitat types (Table 2) based on information available about the biology and ecology of the species included in each functional group, their depth distributions and their preferred sediment type (sources of information: FishBase (www.fishbase.org); SealifeBase (www.sealifebase.org); ICTIMED database (www.cmima.csic.es/ictimed); Riedl, 1986; Fischer et al., 1987a, 1987b).

In addition, (i) the dispersion rate of each functional group in terms of habitat, (ii) the relative dispersal rate in 'non-preferred' habitats, and (iii) the relative feeding rate in non-preferred habitat by functional group were established using information available in the literature (Chen et al., 2009; Christensen et al., 2003, Christensen et al., 2008; Martell et al., 2005; Ortiz and Wolff, 2002; Zeller and Reinert, 2004) or default values from Ecospace when data was not available (Table 2):

i) Regarding dispersion rate, which are not rates of directed migration, but rather basic relative population dispersal rates as a result of random movements, we combined the information provided in the literature and considered the ecological knowledge available for each functional group. Values were assumed to be of three relative magnitudes (3, 30 and 300 km/year) representing essentially non-dispersing, and demersal and pelagic groups, respectively;
ii) For the relative dispersal rate in 'non-preferred' habitats we assumed values from 1 to 5 times the basic movement rate from sessile and planktonic organisms to large pelagic fish;
iii) The increased vulnerability to predation (or grazing) of various organisms outside their 'preferred' habitat can be changed using a multiplier. We assumed that groups were twice more vulnerable to predation in non-preferred habitats than in preferred habitats;
iv) The relative feeding rate in non-preferred habitat was based on the trophic level of each functional group and we assumed it was equal to 0.95 for plankton and trophic levels TL = 1 and 2; equal to 0.01 for intermediate trophic levels (TL = 2 to 3.5); equal to 0.3 for medium–high trophic levels (TL = 3.5 to 4); equal to 0.6 to higher trophic levels (TL > 4) and equal to 1 for infauna (Table 2).

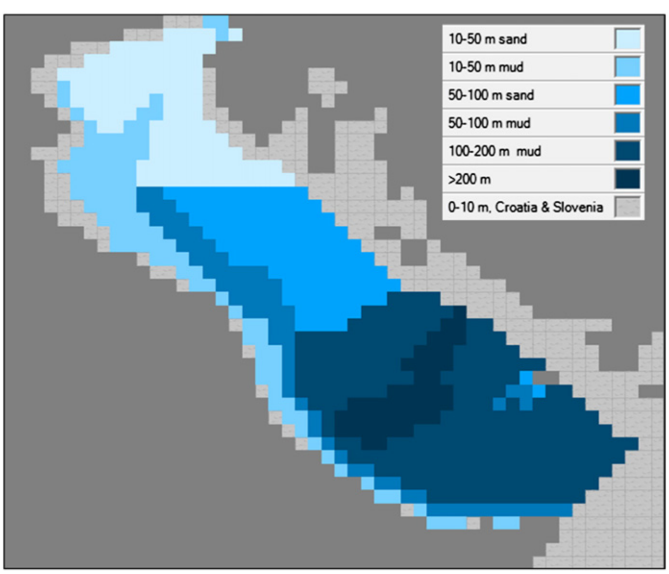

Fig. 2. Spatial basemap with the identification of habitats in the Northern-Central Adriatic Sea: (1) 10–50 m sand, (2) 10–50 m mud, (3) 50–100 m sand, (4) 50–100 m mud, (5) 100–200 m mud, and (6) > 200 m mud. Dark gray cells represent land; light gray cells represent areas excluded from the model: 0–10 m, Croatia and Slovenia and the southern Adriatic Sea.

2.3.3. Advection, migrations and fishing grounds

In our study area, surface water circulation remains fairly constant throughout the year and production is basically determined by Po River discharges and nutrients (Artegiani et al., 1997a, 1997b; Pinardi et al., 2006; Zavatarelli et al., 1998). Therefore, we did not use the advection module available in Ecospace (Christensen et al., 2008). However, to simulate changes in the primary production of the study area due to changes in the Po River discharges, we used annual mean time series of the Po River discharges (from 1975 to 2005, hm^3 y^{-1} obtained from Magistrato del Po (Parma, Italy)) to force the primary production of the immediate Po River area, which then affected the annual primary production of the entire area.

In addition, a seasonal migration field was incorporated to capture the seasonal movements of three important species: (i) European sardine, (ii) European anchovy and (iii) European hake. We defined a monthly sequence of 'preferred' cell positions in the basemap (as defined by Christensen and Walters (2004) and Christensen et al. (2008)):

i) For the European sardine, we included two migrations towards the coast, the first in search of food and the second during the sexual maturation season in spring (March, April and May) and late summer to early autumn (August, September and October), respectively. In late autumn (November), the sardine migrates offshore towards the deeper, colder waters of the outer Dalmatian islands to spawn (e.g. Morello and Arneri, 2009; Mužinić, 1973);
ii) For the European anchovy, we signaled the migration from the open sea towards the coast (to the gulfs of Trieste and Venice) to spawn during the period from April to October and the migration to the open sea in winter (from November to March) (Morello and Arneri, 2009);
iii) For the European hake, we included the migration of adults to the waters of the Pomo pit in the winter (from November to March) and to the shallow water of the continental platform in the spring and autumn (from April to October) (Arneri and Morales-Nin, 2000; Županović and Jardas, 1986).

To establish main fishing grounds by fishing fleet, individual fishing gear types were allocated to depth strata (Table 3) and excluded from gear-specific closed areas (Fig. 3).

2.4. Spatial simulations and analysis

Taking into account the trajectory of the NC Adriatic ecosystem (Coll et al., 2009, 2010b; Fortibuoni et al., 2010; Lotze et al., 2011a), which illustrates a decline in marine fisheries resources and an increase in fishing effort, we simulated spatial closures and alternative management policies to reduce fishing effort from 2002 to 2020, after the calibration period of the model from 1972 to 2002 (Coll et al., 2009). Available time series of fishing effort by fleet from the area showed a general increase with time from mid-1970s to mid-2000s (Coll et al., 2009, 2010b). Vulnerability parameters resulting from the calibration were used during the simulations. The annual mean time series of the Po River discharges from 1975 to 2005 was

Table 2
Input parameters by functional group for the Ecospace model of the Northern-Central Adriatic Sea. (+) indicates application of habitat preferences for a specific functional group.

Functional group		All habitats	10–50 m sand	10–50 m mud	50–100 m sand	50–100 m mud	100–200 m mud	>200 m mud	Base dispersal rate (km/year)	Relative dispersion in non-preferred habitat	Vulnerability to predation in non-preferred habitat	Relative feeding rate in non-preferred habitat
1	Phytoplankton	+							3	1	2	–
2	Micro- and mesozooplankton	+							3	1	2	0.95
3	Macrozooplankton	+							3	1	2	0.95
4	Jellyfish	+							3	1	2	0.95
5	Suprabenthos	+							3	1	2	0.01
6	Polychaetes	+							3	1	2	0.95
7	Comm. scallops and gasterop.		+	+	+	+			3	1	2	0.95
8	Benthic invertebrates	+							3	1	2	0.95
9	Shrimps	+							30	2	2	0.01
10	Norway lobster					+	+	+	30	2	2	0.3
11	Mantis shrimp			+					30	2	2	0.01
12	Crabs	+							30	2	2	0.01
13	Octopus	+							30	2	2	0.01
14	Squids		+	+	+	+	+	+	30	2	2	0.6
15	Vul. hake				+	+	+	+	30	2	2	0.3
16	Non vul. hake				+	+	+	+	30	3	2	0.6
17	Gadids		+	+	+	+	+		30	3	2	0.01
18	Mullets	+							30	2	2	0.01
19	Conger eel		+		+				30	3	2	0.6
20	Anglerfish		+	+	+	+	+	+	30	3	2	0.6
21	Flatfish		+	+	+	+	+		30	2	2	0.3
22	Turbot and brill		+	+	+	+			30	3	2	0.6
23	Demersal sharks	+							30	3	2	0.3
24	Demersal skates		+	+	+	+	+		30	5	2	0.6
25	Demersal fish (1)	+							30	3	2	0.01
26	Demersal fish (2)		+	+	+	+	+	+	30	3	2	0.3
27	Benthopelagic fish	+							300	3	2	0.3
28	Anchovy		+	+	+	+	+		300	4	2	0.01
29	Sardine		+	+	+	+	+		300	4	2	0.01
30	Other small pelagic fish	+							300	4	2	0.01
31	Horse mackerel	+							300	5	2	0.01
32	Mackerel	+							300	5	2	0.01
33	Atlantic bonito		+	+	+	+	+	+	300	5	2	0.6
34	Large pelagic fish			+	+	+	+	+	300	5	2	0.6
35	Dolphins		+	+	+	+	+	+	300	5	2	0.6
36	Marine turtles		+	+	+	+	+		300	5	2	0.3
37	Sea birds		+	+	+	+			300	5	2	0.6
38	Discards	+							–	–	–	–
39	By-catch	+							–	–	–	–
40	Detritus	+							–	–	–	–

Comm. scallops and gasterop.: commercial scallops and gastropods; Vul. hake: vulnerable hake; Non vul. hake: non vulnerable hake (Coll et al., 2007).

kept constant from 2005 to 2020. Two types of management regimes were considered:

a) The establishment of new marine protected areas as a method to selectively reduce fishing effort in different areas of the NC Adriatic Sea using spatial closures, which included:

Scenario 1. The implementation of a marine protected area in the Pomo pit (Central Adriatic Sea) to cover an area of 4000 km^2, or 7.5% of the modeled area (Fig. 3, area A). The Pomo pit is a sensitive and critical zone for spawning and nursery for important Adriatic demersal resources especially for European hake (Arneri and Morales-Nin, 2000; Županović and Jardas, 1986). Therefore, the Pomo pit MPA was tested as a targeted fishery management closure to protect and rebuild European hake and Norway lobster (*Nephrops norvegicus*) stocks. In this simulation, all fleets are excluded from the MPA.

Scenario 2. The implementation of a marine protected area in the northern part of the Adriatic Sea that covered an area of 4000 km^2, 7.5% of the modeled area (Fig. 3, area B). This region is a highly productive area that is critical for spawning and nursery of important commercial demersal resources, but has been intensely exploited. The MPA was designed to protect and rebuild the biomass of commercial species, with emphasis on small pelagic fish. In this simulation, bottom trawls, mid-water trawls and purse seine fleets were excluded from the MPA.

b) A general reduction of fishing effort in the study area, without spatial closures but the application of a reduction of effort by fleet.

Table 3
Distribution of fishing fleets by habitat type in the Northern-Central Adriatic model. (+) indicates depths were each fleet currently fish.

Fleet/habitat use		10–50 m	10–50 m mud	50–100 m sand	50–100 m mud	100–200 m mud	>200 m
1	Bottom trawls	+	+	+	+	+	+
2	Mid-water trawls	+	+	+	+	+	
3	Purse seiner	+	+	+	+	+	
4	Tuna fleet			+	+	+	+

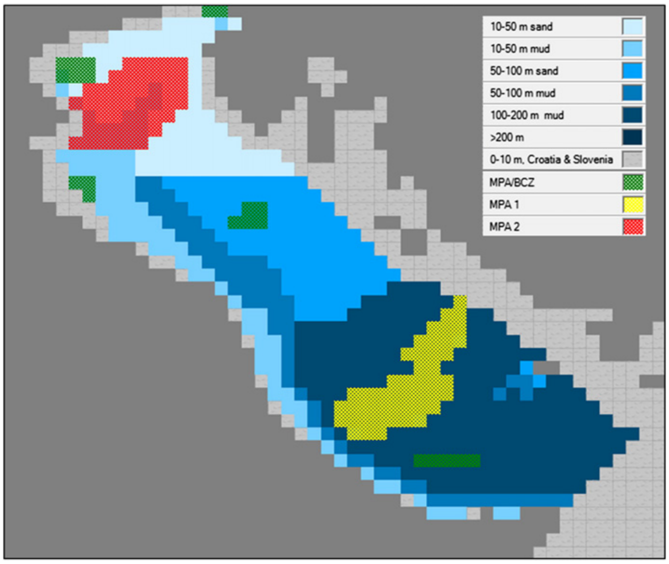

Fig. 3. Protected areas and biological conservation zones in the Northern-Central Adriatic Sea. The location of current protected areas and biological conservation zones are indicated with pointed-green cells. The location of the Pomo pit MPA (A) and the Northern MPA (B) simulated in this study are indicated in pointed-yellow and pointed-red cells, respectively. Dark gray cells represent land; light gray cells represent areas excluded from the model: 0–10 m, Croatia and Slovenia and the southern Adriatic Sea.

In the Mediterranean Sea, seasonal closures are generally imposed with the aim of protecting demersal resources at the most vulnerable point of their life cycles, the recruitment period. The duration and seasonality of these closures varies among countries and harbors (e.g. Demestre et al., 2008). In the Adriatic Sea, temporary closures of fishing trawling grounds consist of the complete cessation of fleet activity for 30 to 45 days (generally from August to September); although in 2011 the closure was extended to 60 days. However, these closures are too short to enable a recovery of benthic communities (Demestre et al., 2008). Here we simulated larger reductions of fishing effort as following:

Scenario 3. The implementation of a seasonal fishing ban, applicable to bottom and mid-water trawls, for three months every year, from November to January (thus during winter time when the number of suitable fishing days at sea are reduced). The seasonal temporal closure covered the entire study area.

Scenario 4. The reduction of fishing effort of mid-water trawls and purse seine fleets by 25% of the level in 2002. This reduction is equivalent to the cessation of fleet activity for one day per week in the entire study area.

Scenario 5. The reduction of fishing effort of the bottom trawls by 25% of the level in 2002. This reduction is equivalent to the cessation of fleet activity for one day per week in the entire study area.

Although the ecological model also included fishing activities on large pelagic fish, we did not include this fleet in our scenarios due to the fact that data for this fishing fleet is less certain and total volume of catch is lower than that from trawls or purse seines. In addition, large pelagic fish dynamics (such as that from blue fin tuna) in the Adriatic Sea are linked with a much larger stock covering the Mediterranean and east Atlantic, at least. We compared the results from the management alternatives listed above to a baseline scenario: the actual management system in the Adriatic Sea as examined in Coll et al. (2009) from 1975 to 2002, while values of fishing effort from 2002 to 2020 were kept constant. In all simulations, including the baseline, we included the biological conservation zones and MPAs currently implemented in the area (2 MPAs and 4 biological conservation zones, Fig. 3) and the temporal closures currently already applied in the area.

Changes in species biomass and catches at the end of the simulations (year 2020) were evaluated and compared to values from 1975 by management strategy. In addition, the following ecological indicators were calculated: total biomass ($t \cdot km^{-2}$), commercial species biomass ($t \cdot km^{-2}$), predator species biomass (including organisms with trophic level ≥4) ($t \cdot km^{-2}$), fish biomass ($t \cdot km^{-2}$), invertebrate biomass (excluding planktonic organisms) ($t \cdot km^{-2}$), total catch ($t \cdot km^{-2} \cdot y^{-1}$), demersal catch ($t \cdot km^{-2} \cdot y^{-1}$) and pelagic catch ($t \cdot km^{-2} \cdot y^{-1}$).

3. Results

3.1. Baseline simulation

Results from the baseline simulation highlighted a decrease in biomass of many important ecological and commercial groups by 2020

Table 4
Biomass and catch changes by functional group at the end of 45-year simulation (final biomass and catch, by 2020) in comparison with the beginning of the simulation (initial biomass and catch, 1975). Biomass values are expressed in $t \cdot km^2$ and catch in $t \cdot km^2 \cdot y^{-1}$.

	Functional group	Initial biomass (Bi)	Final biomass (Bf)	Bf/Bi	Initial catch (Ci)	Final catch (Cf)	Cf/Ci
1	Phytoplankton	15.83	17.17	1.08	–	–	–
2	Micro and mesozooplankton	8.85	7.65	0.86	–	–	–
3	Macrozooplankton	0.54	0.55	1.03	–	–	–
4	Jellyfish	2.11	1.75	0.83	–	–	–
5	Suprabenthos	0.98	1.27	1.30	–	–	–
6	Polychaetes	9.82	10.04	1.02	–	–	–
7	Comm. scallops and gasterop.	0.95	1.35	1.43	0.01	0.05	4.49
8	Benthic invertebrates	78.88	80.30	1.02	0.01	0.03	3.04
9	Shrimps	0.32	0.42	1.32	0.03	0.13	4.07
10	Norway lobster	0.05	0.05	0.96	0.04	0.11	2.72
11	Mantis shrimp	0.05	0.00	0.08	0.02	0.01	0.35
12	Crabs	0.51	0.68	1.33	0.01	0.04	4.00
13	Octopus	0.15	0.09	0.57	0.10	0.16	1.58
14	Squids	0.06	0.02	0.28	0.08	0.05	0.59
15	Vul. hake	0.21	0.03	0.14	0.22	0.21	0.97
16	Non vul. hake	0.23	0.01	0.03	–	–	–
17	Gadids	0.24	0.02	0.08	0.10	0.03	0.27
18	Mullets	0.06	0.03	0.54	0.06	0.09	1.51
19	Conger eel	0.02	0.01	0.78	0.02	0.04	2.08
20	Anglerfish	0.02	0.00	0.09	0.02	0.00	0.22
21	Flatfish	0.07	0.01	0.13	0.09	0.03	0.32
22	Turbot and Brill	0.04	0.01	0.28	0.01	0.01	0.67
23	Demersal sharks	0.05	0.02	0.40	0.03	0.04	1.06
24	Demersal skates	0.07	0.05	0.71	0.03	0.07	2.44
25	Demersal fish (1)	0.28	0.12	0.42	0.16	0.17	1.06
26	Demersal fish (2)	0.12	0.16	1.29	0.03	0.12	3.67
27	Benthopelagic fish	0.64	0.40	0.62	0.04	0.07	1.90
28	Anchovy	2.50	2.09	0.84	0.73	0.74	1.02
29	Sardine	4.52	5.59	1.24	0.78	0.97	1.25
30	Small pelagic fishes	1.02	0.77	0.75	0.04	0.06	1.43
31	Horse mackerel	1.33	1.28	0.96	0.06	0.17	2.78
32	Mackerel	0.63	1.04	1.65	0.01	0.04	4.57
33	Atlantic bonito	0.31	0.44	1.45	0.00	0.01	3.39
34	Large pelagic fishes	0.12	0.11	0.94	0.00	0.01	2.25
35	Dolphins	0.01	0.01	0.95	0.00	0.00	0.80
36	Marine turtles	0.03	0.00	0.08	0.00	0.00	0.31
37	Sea birds	0.00	0.00	0.81	0.00	0.00	2.25
38	Discards	0.73	1.09	1.49	–	–	–
40	Detritus	193.61	199	1.03	–	–	–
	Total	325.97	333.66	1.02	2.74	3.46	1.26

(Table 4 and Fig. 4). The biomass of most large predator groups showed evident changes with large declines for large hake, angler fish (*Lophius* sp.), turbot (*Psetta maxima*), brill (*Scophthalmus rhombus*), demersal sharks, seabirds, and dolphins. Intermediate consumers also showed important declines, with the exception of small tunid Atlantic bonito (*Sarda sarda*) that increased marginally. The spatial distribution of the biomass of functional groups of the ecological model showed depletions of numerous ecological groups of the Adriatic Sea ecosystem by 2020 (Fig. 4). In contrast, the biomass of low-trophic-level species such as sardine and mackerel (*Scomber* sp.) increased due to declining predation (Table 4). The biomass of commercial bivalves, gastropods, and various benthic crustaceans, which included shrimps and crabs, showed an increase of 43%, 32% and 33%, respectively (Table 4).

Under this baseline scenario, catches increased for several groups, including commercial bivalves and gastropods, Norway lobster, other benthic crustaceans, octopus, red mullets (*Mullus* sp.), conger eel, demersal sharks and skates, other demersal fish, and various pelagic fish, such as anchovy, sardine, small horse mackerel, mackerel, Atlantic bonito, and large pelagic fishes. The increases varied from 4% to 358%. Some groups such as mantis shrimp (*Squilla mantis*), squids, hake, anglerfish, flatfish, and turbot and brill showed catch declines from 18% to 79% (Table 4).

3.2. MPA implementations

3.2.1. Scenario 1: MPA in the Pomo pit area

Under this scenario our results showed that the predator species' biomass increased slightly within the Pomo pit MPA (Fig. 5a). In addition, several target species benefit from the establishment of this MPA (Table 5). For example, the biomass of hake and Norway lobster increased by 33% and 3%, respectively, and the biomass of anglerfish group increased over 100% in comparison with the baseline simulation (Table 5). Other groups of predators also slightly benefited from the MPA such as demersal skates, demersal sharks, conger eel and Atlantic bonito (Table 5). The remaining groups, which included low and medium trophic-level species, showed little response to the establishment of the MPA.

The establishment of the Pomo pit MPA led to a 1% decline in the total catch compared to the baseline simulation (Fig. 5a), although catch of some demersal groups showed a substantial increase, such as for shrimps, Norway lobster, hake and anglerfish. Catch by different fleets showed little or no change as a result of MPA establishment (Fig. 6a).

3.2.2. Scenario 2: implementation of a marine protected area in the North Adriatic Sea

As a response to the establishment of the MPA in the North Adriatic Sea, the biomass of several commercial species and fish groups showed important increases, such as mantis shrimp, anglerfish and flatfish (Table 5). The predator groups also showed benefits from the establishment of the MPA (Fig. 5b), and most groups showed increases that varied between 4% and 390% (Table 5). On the contrary, some low and medium trophic-level species, such as the Norway lobster, anchovy, and sardine, or commercial bivalves and gastropods, did not display a significant change in biomass as a result of this MPA establishment.

The establishment of the northern protected area led to a 2% reduction in the total catch (Fig. 5b). Under this scenario, and in comparison with the baseline 2020 simulation, most groups showed a decline in catch, although never greater than 27%. On the contrary, mantis shrimp and flatfish showed a significant increase in the catch (Table 6). When examined by fleets, the only one that showed a slight overall variation in catch following the establishment of the MPA was the bottom trawl, which decreased by 3% (Fig. 6b).

3.3. Scenarios of fishing effort reduction

3.3.1. Scenario 3: implementation of a seasonal fishing ban

Under this scenario, commercial species showed small biomass changes (approximately 3% in respect of the baseline 2020 simulation) and only the biomass of the predators increased by 14% (Fig. 5c). In

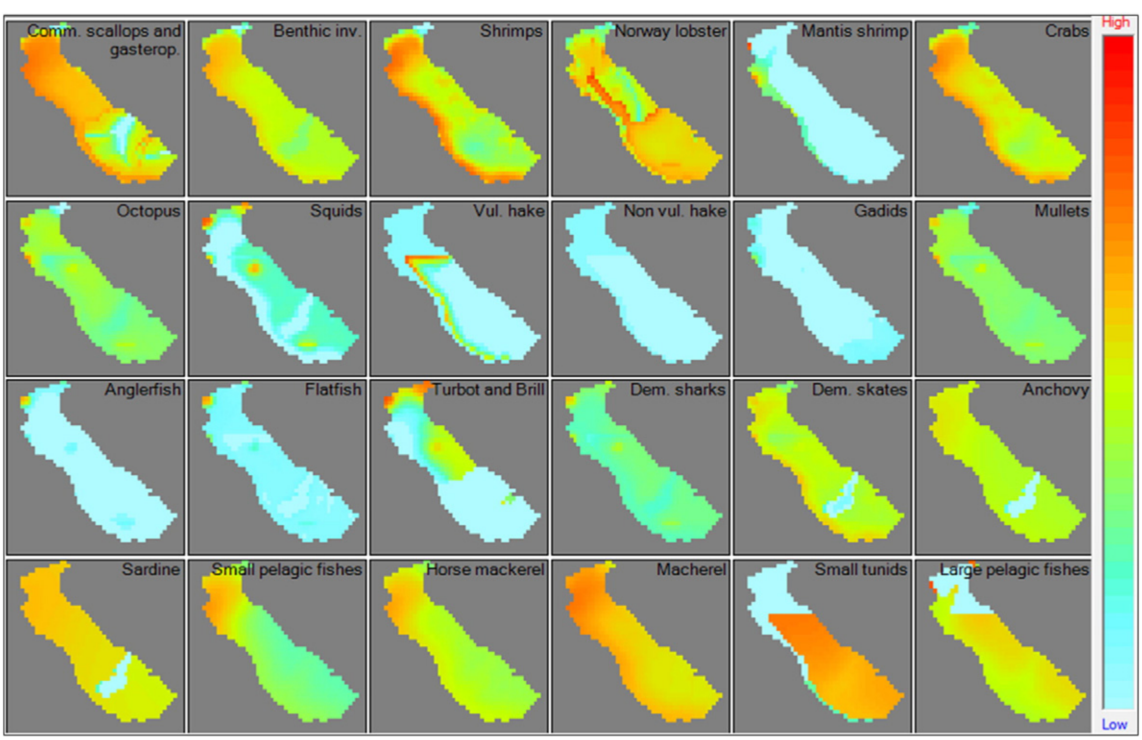

Fig. 4. Spatial distribution of the biomass of several functional groups after 45 years of simulation (by year 2020) under the baseline scenario. The color scale (in the right) indicates an increase (red) or decrease (blue) from the baseline in 1975.

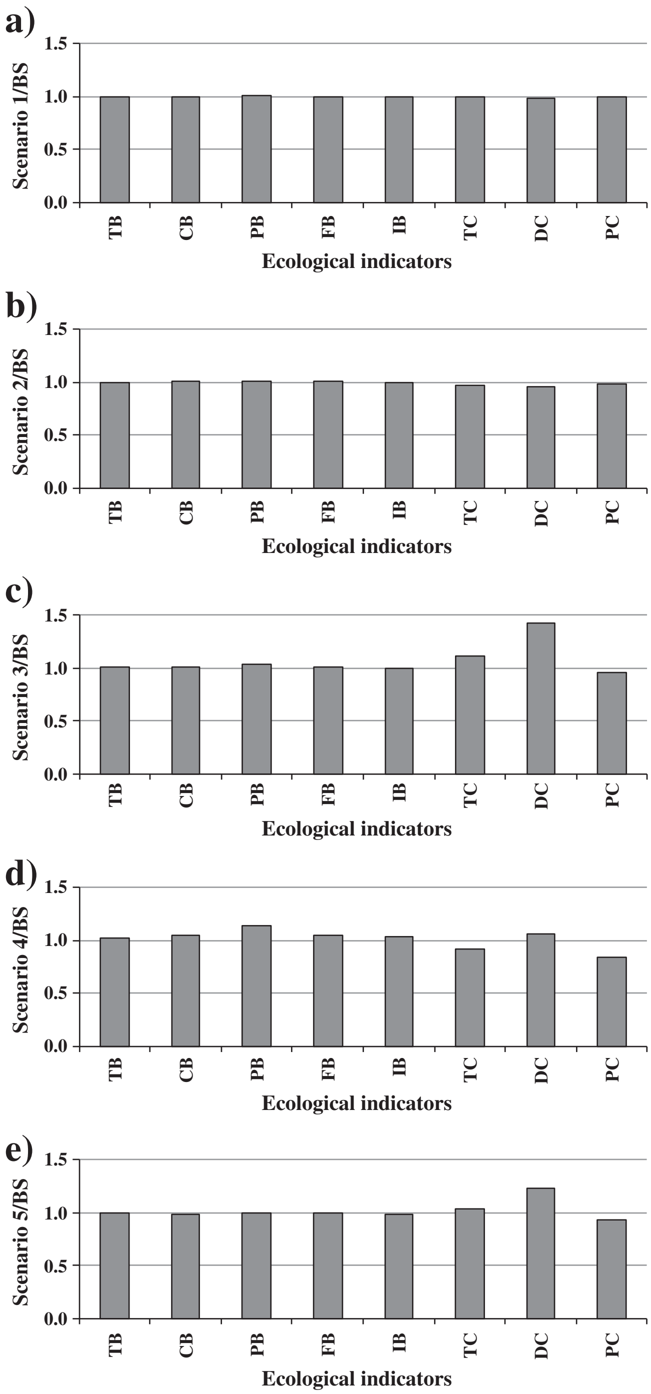

Fig. 5. Comparative changes of ecological indicators between baseline simulation (BS) and management scenarios 1 to 5. Ratios less than 1 indicate a decline in biomass or catch over the 45-year simulation (from 1975 to 2020). TB = total biomass, CB = commercial species biomass, PB = predator species biomass, FB = fish biomass, IB = invertebrates (except plankton) biomass, TC = total catch, DC = demersal catch, PC = pelagic catch.

addition, biomass results by functional groups highlighted that many groups were positively affected by this management measure (Table 5), especially most demersal and large predators. On the contrary, the pelagic groups (large pelagic fishes, horse mackerel, mackerel, Atlantic bonito, small pelagic fishes, sardine, and anchovy) and some low and medium trophic-level species, (such as benthic invertebrates, commercial bivalves and gastropods, crabs and shrimps), showed small response to the implementation of the temporary closure. Their change in biomass did not exceed 15% (Table 5).

Overall, demersal catch increased by 39% and pelagic catch declined by 16% in comparison with the baseline 2020 simulation (Fig. 5c). Changes in catches were also variable depending on which group was investigated (Table 6). Mantis shrimp showed the largest increase in catch in comparison with the baseline 2020 simulation. Other species, including shrimps and crabs, commercial bivalves and gastropods, flatfish, and pelagic fishes showed a decrease in their catch of 3% to 23% (Table 6). Most of the fleets showed non-significant changes in catch as a result of the temporary closure, although bottom trawls showed increase in catch of 39% (Fig. 6c).

3.3.2. Scenario 4: 25% reduction of mid-water trawls and purse seine fishing effort

Under this simulation, commercial species, predator species, and fish biomass increased by 5%, 15% and 5%, respectively. The biomass of individual groups showed moderate change (Table 5) with the exception of mantis shrimp, hake, turbot and brill, demersal skates, and seabirds that showed an increase of 20%, 44%, 20%, 25%, and 27%, respectively, in comparison with the baseline 2020 scenario (Table 5).

This simulation yield an overall 8% decline in total catch in comparison with the baseline 2020 simulation (Fig. 5d). Catch by individual groups showed a decline for anchovy, sardine, small pelagic fishes, and horse mackerel by 8%, 24%, 18%, and 15%, respectively. The remaining groups showed a slight increase in catch that varied between 2 and 23% (Table 6). The effect of this simulation was different between fleets. Bottom trawls and tuna fleets showed an increase on the catch by 4% and 11%, respectively, while mid-water trawls and purse seiners showed a decrease by 20% and 21%, respectively (Fig. 6d).

3.3.3. Scenario 5: 25% reduction of bottom trawling fishing effort

Under the last simulation, the biomass of commercial and predator species decreased by 3%. Individually, the biomass of all demersal resources, and especially the medium and high trophic level species, increased (Table 5). For example, hake increased by 216% and Norway lobster by 12% in comparison with the baseline 2020 simulation. On the contrary, pelagic organisms did not show large changes.

Overall, under this scenario, total catch increased by 4% (Fig. 5e). Demersal catch showed an increase of 27%, while pelagic catch decreased by 8% in comparison with the baseline 2020 scenario. Various individual demersal groups also showed important catch increase (Table 6), while catch of pelagic species moderately declined (Table 6). This scenario had limited effect on catch by fleet. Catch of bottom trawl increased slightly by 12%, while catch of the remaining fleets showed slight declines not exceeding 6% (Fig. 6e).

4. Discussion

Global declines in quantity and quality of fishing stocks have created a need for complementary methods to manage fisheries (Botsford et al., 1997; Christensen and Maclean, 2011; Link, 2011; Walters and Martell, 2004). Multispecies and ecosystem analysis are valuable tools, not only for fishery scientists but also for fisheries management that need to take decisions about exploitation, conservation and restoration of marine fisheries resources.

The spatial modeling tool Ecospace is a simulation model that provides an initial screening capability for protected area policy and generates strategic predictions of how taxon specific biomass will change under different marine management scenarios (Salomon et al., 2002; Walters et al., 1999). It is a useful tool to help limit the pathology inherent in current marine resource management and help design an adaptive management approach (Salomon et al., 2002).

The aim of this study was to evaluate existing spatial management regimes and potential MPAs for the NC Adriatic Sea marine ecosystem using this modeling tool. We also attempted to predict alternative management options to globally reduce the impact of fishing on the marine ecosystem. However, it is important to take into account

Table 5
Biomass changes for several functional groups by 2020 in comparison with 1975 under different management scenarios, and comparative ratios with the baseline simulation.

	Functional group	Sc.1 (Bf/Bi)	Sc.2 (Bf/Bi)	Sc.3 (Bf/Bi)	Sc.4 (Bf/Bi)	Sc.5 (Bf/Bi)	Sc.1/BS	Sc.2/BS	Sc.3/BS	Sc.4/BS	Sc.5/BS
1	Phytoplankton	1.09	1.09	1.10	1.11	1.06	1.00	1.00	1.01	1.03	0.98
2	Micro and mesozooplankton	0.86	0.86	0.84	0.83	0.89	1.00	1.00	0.98	0.96	1.03
3	Macrozooplankton	1.03	1.05	1.02	1.01	1.05	1.00	1.02	0.99	0.98	1.01
4	Jellyfish	0.83	0.83	0.82	0.80	0.85	1.00	1.00	0.99	0.97	1.02
5	Suprabenthos	1.30	1.28	1.24	1.32	1.21	1.00	0.99	0.96	1.02	0.94
6	Polychaetes	1.02	1.02	1.04	1.06	1.00	1.00	1.00	1.02	1.03	0.98
7	Comm. scallops and gasterop.	1.44	1.43	1.49	1.48	1.42	1.00	1.00	1.04	1.04	0.99
8	Benthic invertebrates	1.02	1.02	1.04	1.05	0.99	1.00	1.00	1.02	1.03	0.98
9	Shrimps	1.33	1.29	1.29	1.38	1.23	1.01	0.98	0.98	1.05	0.93
10	Norway lobster	0.99	0.99	1.10	1.00	1.07	1.03	1.03	1.15	1.04	1.12
11	Mantis shrimp	0.05	0.54	0.72	0.10	0.60	0.61	6.65	8.86	1.20	7.43
12	Crabs	1.35	1.32	1.20	1.37	1.16	1.01	0.99	0.90	1.03	0.87
13	Octopus	0.52	0.60	0.92	0.62	0.85	0.91	1.05	1.61	1.09	1.49
14	Squids	0.28	0.36	0.47	0.31	0.46	0.99	1.26	1.68	1.10	1.63
15	Vul. hake	0.19	0.14	0.56	0.20	0.44	1.33	1.00	4.04	1.44	3.16
16	Non vul. hake	0.03	0.03	0.05	0.03	0.03	1.00	1.00	1.54	1.00	1.04
17	Gadids	0.04	0.16	0.62	0.09	0.50	0.57	2.03	8.07	1.16	6.48
18	Mullets	0.54	0.76	0.65	0.56	0.64	0.99	1.39	1.20	1.03	1.17
19	Conger eel	0.78	0.91	0.89	0.81	0.89	1.01	1.18	1.15	1.04	1.14
20	Anglerfish	0.20	0.45	0.15	0.09	0.16	2.24	4.90	1.67	0.99	1.72
21	Flatfish	0.14	0.74	0.14	0.13	0.14	1.04	5.64	1.04	1.02	1.06
22	Turbot and Brill	0.22	0.39	0.69	0.33	0.59	0.79	1.41	2.49	1.20	2.14
23	Demersal sharks	0.44	0.49	0.56	0.43	0.54	1.10	1.23	1.40	1.07	1.35
24	Demersal skates	0.84	0.85	1.00	0.89	0.96	1.19	1.20	1.40	1.25	1.36
25	Demersal fish (1)	0.39	0.43	0.94	0.45	0.89	0.94	1.04	2.25	1.07	2.13
26	Demersal fish (2)	1.29	1.24	1.19	1.33	1.16	1.00	0.97	0.93	1.03	0.90
27	Benthopelagic fish	0.62	0.53	0.66	0.57	0.71	1.00	0.85	1.06	0.92	1.14
28	Anchovy	0.83	0.86	0.78	0.85	0.80	0.99	1.03	0.94	1.01	0.96
29	Sardine	1.24	1.24	1.32	1.35	1.18	1.00	1.00	1.06	1.09	0.95
30	Small pelagic fishes	0.74	0.78	0.64	0.67	0.74	0.99	1.04	0.86	0.90	0.98
31	Horse mackerel	0.97	0.98	0.89	0.96	0.91	1.01	1.02	0.92	0.99	0.95
32	Mackerel	1.65	1.46	1.41	1.67	1.38	1.00	0.89	0.85	1.01	0.84
33	Atlantic bonito	1.46	1.43	1.59	1.71	1.28	1.01	0.99	1.10	1.18	0.89
34	Large pelagic fishes	0.94	0.99	0.91	1.01	0.90	1.00	1.05	0.97	1.07	0.95
35	Dolphins	0.95	0.93	1.02	1.00	0.94	1.00	0.97	1.08	1.06	0.98
36	Marine turtles	0.06	0.09	0.25	0.08	0.15	0.77	1.14	3.30	1.06	2.01
37	Sea birds	0.79	0.85	1.13	1.03	0.88	0.97	1.04	1.39	1.27	1.09
38	Discards	1.43	1.39	1.84	1.53	1.69	0.96	0.94	1.24	1.03	1.14
40	Detritus	1.03	1.03	1.04	1.05	1.02	1.00	1.00	1.01	1.02	0.99
Total		1.02	1.03	1.04	1.05	1.01	1.00	1.00	1.01	1.02	0.99

Sc.1: scenario 1; Sc.2: scenario 2; Sc.3: scenario 3; Sc.4: scenario 4; Sc.5: scenario 5; BS: baseline simulation; Bf/Bi: final biomass in 2020/initial biomass in 1975.

that this modeling application was parameterized to mainly represent the ecological dynamics of Italian territorial waters and international waters (which include the majority of the surface of the NC Adriatic Sea, Fig. 1). The parameterization excluded as well the areas above 10 m from the western coast. Therefore, our results represent suitable conditions for the Italian territorial waters and international waters only (North and Central areas) and are not representative of ecological dynamics neither in the Eastern NC Adriatic nor in the immediate coastal areas near Italy. Future developments of ecological modeling in the NC Adriatic Sea should aim to include Slovenian and Croatian territorial waters and coastal dynamics to expand this study and incorporate further realism in spatial and temporal simulations.

Despite above limitations, to our knowledge this study represents the first attempt to evaluate different fishing management alternatives for the NC Adriatic Sea in an ecosystem context using a spatial–temporal food web model and can contribute to advancement towards an ecosystem-approach to fishing in the Mediterranean Sea (Coll and Libralato, 2012). Our results highlight that under the baseline 2020 scenario, and developing management simulations using a model fitted to data, several marine resources of the Northern-Central Adriatic Sea will remain depleted or decline, in line with previous modeling exercises (Coll et al., 2009, 2010b) and independent analysis of ecological indicators (Coll et al., 2010a, 2010b; Fortibuoni et al., 2010; Lotze et al., 2011a). Results highlight how fish populations and larger predators are at low abundance in the study area, and therefore some of their preys are abundant, such as invertebrates and small pelagic fish. This is also consistent with other studies on marine resources in the Adriatic basin. For example, independent results highlight that European hake is overexploited since the 1960s (Vrgoč et al., 2004). Demersal sharks and skates also showed declines in biomass through time (Jukić-Peladić et al., 2001). On the contrary, European anchovy's catch strongly declined until 1987 but a partial recovery in stock biomass and catch has occurred since the 1990s (Santojanni et al., 2003). This partial recovery may be a result of the decline in the larger predators' biomass and the reduction of total fishing effort by implementing different fishing strategies since 1988 (Cingolani et al., 1996).

Existing regulations in the NC Adriatic Sea include spatial and temporal closures for trawling, marine protected areas, and biological conservation zones (Table 1) (AdriaMed, 2005; Anon, 2007, 2009). However, present observations and model forecasts point to the fact that these regulations may not be effective to reverse decline trends and recover marine resources. Fishing temporal closures prohibit trawls at certain times to protect juveniles and the spawning grounds. However, after the fishing closure is re-opened, the beneficial effects are short-lived and the achievement of a temporal fishing closure is subsequently diminished with the resumption of trawling activity in the area (Demestre et al., 2008). Additionally, the enforcement of management measures in the Mediterranean is low (Mora et al., 2009).

MPAs are increasingly advocated as an effective tool to safeguard the declining coastal fishery resources and restore overexploited stocks and degraded areas (Agardy, 2000; Chen et al., 2009; Goñi et al., 2008;

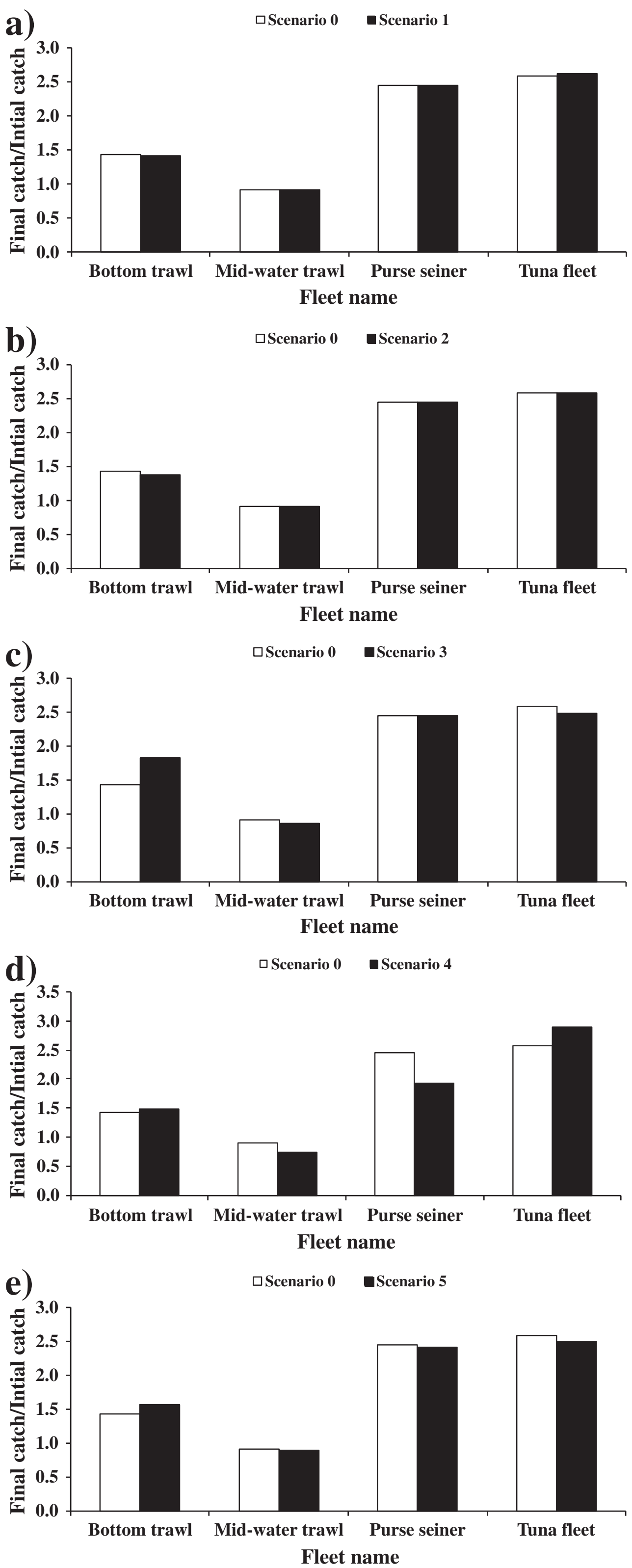

Fig. 6. Catch changes by fleet for the baseline simulation (scenario 0) and management scenarios 1 to 5 at the end of the 45-year simulation (year 2020). Ratios lower than 1 indicate a decline catch over the 45-year simulation (from 1975 to 2020).

Roberts, 2000; Worm et al., 2009). MPAs are recognized for their contribution to the protection of spawning stock and biologically rich habitats, as well as for their valuable contribution to marine fishery management (Roberts, 1997). Our simulations of two MPAs in different areas of the NC Adriatic Sea predict that they could indeed be beneficial for fish stock recovery and ecosystem rebuilding in the Adriatic. We found they could have a positive impact increasing the biomass of several fish and predator groups. These results are consistent with studies from other marine ecosystem (Chen et al., 2009; Zeller and Reinert, 2004).

Considering our overall results, the northern MPA appeared to be more effective than the Pomo pit MPA, especially for demersal fish and top predator species. In addition, the northern region of the Adriatic Sea is an important juvenile nursery ground for many commercial species, and a significant role is played by the considerable water run-off from the mainland brought by numerous rivers especially the Po River that enter the NC Adriatic Sea ecosystem (Zavatarelli et al., 1998, 2000). Therefore, our results suggest that the implementation of an MPA in this area could be ecologically optimal. The implementation of the Pomo pit MPA, already discussed by the AdriaMed Demersal Resources Working Group (AdriaMed, 2009), may also be beneficial for several marine resources especially the target species European hake and Norway lobster.

Simulations of seasonal restrictions of fleet activity during November to January showed that significant reductions in fishing effort would be required to change the trend in current declining biomass projections. Results showed significant benefits both in terms of biomass and catch for most large predator groups and important commercial species, and especially for demersal resources. Therefore, the extension of temporal closures to fishing could be effective for rebuilding of several demersal stocks. These results are in line with previous recommendations that a longer temporal closure in the area would be beneficial and allow the maintenance of more structured systems (Agardy, 2000; Demestre et al., 2008).

An effort reduction in the pelagic fleet by 25% could be also effective to advance in the recovery of marine resources of the NC Adriatic Sea. The effort reduction for the demersal fleet by 25% showed benefits for most of the demersal resources. This result is in line with results for other exploited ecosystems. For instance, an effort reduction in the vicinity of 50% for the gear taking the majority of the blue whiting catch, and a reduction by approximately 20% for each of the two major gear types targeting Greenland halibut would be required to reverse the projected biomass decline for both species in the Faroe Islands, Northeastern Atlantic Ocean (Zeller and Reinert, 2004). In the Mediterranean, similar trends were described by Coll et al. (2008a) in the South Catalan Sea, where simulations of a reduction of total fishing effort by 20% (one day of fishing for all fleets), and additional 20% reduction of fishing effort for the demersal fleets, showed higher recoveries of commercial species and vulnerable species such as top predators. Our results are also in line with calls for the reduction of global fishing fleet overcapacity and exploitation levels (Worm et al., 2009).

In all simulations, the establishment of an MPA or the reduction in fishing effort led to variation in the qualitative response of catch by the different fleets and organisms of the ecosystem. In general, results of all scenarios showed decline or no change in the projected overall catch. However, when examined by fishery group or fleet, some sectors or groups of the fishery remain unaffected, or showed some benefits. These results indicate a conflict between socioeconomic and ecological goals, and demonstrate somehow a negative correlation between economic and social performance and ecological stability of the ecosystem. This phenomenon was also observed in the Hong Kong marine and Thailand Gulf ecosystems (Christensen, 1998; Pitcher et al., 2000). However, clear benefits were observed for bottom trawl fleets with the implementation of temporal restriction, the reduction in the effort of pelagic fleet, and the reduction in the effort of demersal fleet. A detailed socioeconomic study, which should include accurate data for prices per species, fleet operational costs, and employment indicators for each fishery should complement this study to inform on best management options not only from an ecological point of view, but also taking into account socio-economic and market factors.

Table 6
Changes in catch for each fishery group after 45-year simulation in all simulations and comparative ratios with the baseline simulation.

	Functional group	Sc.1 (Cf/Ci)	Sc.2 (Cf/Ci)	Sc.3 (Cf/Ci)	Sc.4 (Cf/Ci)	Sc.5 (Cf/Ci)	Sc.1/BS	Sc.2/BS	Sc.3/BS	Sc.4/BS	Sc.5/BS
7	Comm. scallops and gasterop.	4.40	4.50	4.27	4.65	3.44	0.98	1.00	0.95	1.03	0.76
8	Benthic invertebrates	3.12	2.95	2.81	3.14	2.27	1.02	0.97	0.92	1.03	0.75
9	Shrimps	4.33	3.95	3.67	4.28	2.96	1.05	0.96	0.89	1.04	0.72
10	Norway lobster	2.87	2.68	2.89	2.88	2.32	1.03	0.96	1.04	1.03	0.83
11	Mantis shrimp	0.14	0.93	2.46	0.30	1.69	0.56	3.67	9.66	1.19	6.64
12.	12 Crabs	4.21	3.93	3.32	4.15	2.70	1.04	0.97	0.82	1.03	0.67
13	Octopus	1.43	1.37	2.52	1.76	1.91	0.89	0.85	1.56	1.09	1.18
14	Squids	0.59	0.66	1.02	0.68	0.78	0.96	1.08	1.66	1.11	1.27
15.	15 Vul. hake	0.76	0.63	1.58	0.78	1.37	1.05	0.88	2.19	1.08	1.90
17	Gadids	0.12	0.29	1.70	0.25	1.11	0.55	1.37	8.00	1.03	5.22
18	Mullets	1.52	1.52	1.70	1.57	1.35	0.99	1.00	1.11	1.16	0.89
19.	19 Conger eel	2.11	2.16	2.28	2.23	1.84	0.98	1.01	1.06	1.03	0.86
20	Anglerfish	0.30	0.38	0.34	0.22	0.28	1.35	1.68	1.52	1.04	1.26
21	Flatfish	0.33	0.63	0.31	0.34	0.26	0.99	1.90	0.95	0.99	0.79
22	Turbot and Brill	0.51	0.79	1.80	0.85	1.23	0.75	1.15	2.62	1.02	1.80
23	Demersal sharks	1.10	1.09	1.40	1.16	1.09	1.01	1.00	1.29	1.23	1.00
24	Demersal skates	2.42	2.38	2.68	2.58	2.15	0.97	0.95	1.08	1.07	0.86
25	Demersal fish (1)	0.92	0.99	2.52	1.19	1.95	0.83	0.90	2.28	1.03	1.76
26	Demersal fish (2)	3.86	3.55	3.20	3.86	2.58	1.03	0.95	0.85	1.07	0.69
27	Benthopelagic fish	1.88	1.55	1.77	1.72	1.61	1.01	0.83	0.95	1.03	0.87
28	Anchovy	1.03	1.04	0.97	0.79	1.00	0.99	1.00	0.94	0.92	0.97
29	Sardine	1.27	1.25	1.29	1.04	1.21	1.00	0.99	1.02	0.76	0.96
30	Small pelagic fishes	1.45	1.39	1.14	1.22	1.19	1.01	0.96	0.79	0.82	0.82
31	Horse mackerel	2.80	2.72	2.29	2.72	2.00	1.02	0.99	0.83	0.85	0.73
32	Mackerel	4.66	4.05	3.53	4.60	2.96	1.02	0.88	0.77	0.99	0.65
33	Atlantic bonito	3.59	3.49	3.91	4.17	3.13	1.02	0.99	1.11	1.01	0.89
34	Large pelagic fishes	2.30	2.38	2.21	2.44	2.17	1.00	1.04	0.97	1.19	0.95
35	Dolphins	0.82	0.80	0.80	0.65	0.81	1.00	0.97	0.97	1.07	0.99
36	Marine turtles	0.16	0.25	0.65	0.23	0.33	0.76	1.14	3.00	0.79	1.51
37	Sea birds	2.26	2.58	3.17	3.07	2.02	0.95	1.08	1.33	1.07	0.85
Total		1.24	1.24	1.49	1.15	1.31	0.99	0.98	1.18	0.92	1.04

Sc.1: scenario 1; Sc.2: scenario 2; Sc.3: scenario 3; Sc.4: scenario 4; Sc.5: scenario 5; BS: baseline simulation; Cf/Ci: final catch in 2020/initial catch in 1975.

Overall simulations showed limited benefits for European anchovy, sardine and small pelagic fish, even in simulation of reduction of fishing effort for the pelagic fleets. These trends are related to the intra-specific competition and higher predation mortality on these species by recovering predators. On the contrary, important positive results were showed for the demersal species, especially European hake, anglerfish, and flatfish. These results are directly related with lower fishing mortality and higher availability of prey. Results also highlighted increasing trends in NC Adriatic sea birds and turtle populations. These recoveries were an evident result of the exploitation reduction and the habitat protection through MPA, especially with the MPA in the Northern Adriatic Sea area. This is interesting since the region hosts most of the coastal habitats that are currently protected (Lotze et al., 2006). Thus, different simulated strategies of management can also help to protect and conserve endangered and predatory species since they benefit from recoveries of their prey populations.

It is important to take into account that the spatial model Ecospace does not represent the full variety of physical transport and migratory processes that may be critical in the spatial organization of ecosystems. Furthermore, the spatial results generated by Ecospace are average responses and do not intend to mimic actual transitional dynamics (Walters, 2000). However, our results are drawn on the assumption that the spatial model used in this study offer reasonable qualitative predictions of the responses of the ecosystems to MPA designation and fishing effort reduction and it can provide some insights about the likely efficacy of alternative management policies in an ecosystem within a trophic context (Walters et al., 1999). In fact, there is potential for a large number of estimation errors in the parameters of the EwE model used, though this error will mainly affect the quantitative outputs from the model, rather than the qualitative results (Walters et al., 1999). Therefore, our study presents important results that can be used by policymakers in a strategic way and that can certainly be improved if more data becomes available. Future simulations should consider habitat-specific effects of fishing gear and fishing fleet movement, and the effects of environmental changes. An additional socioeconomic study including accurate data for prices per species, fleet operational costs, and employment indicators per fleet should complement this study to inform on best management options to stop depletion of marine resources and foster recovery of ecosystems in the NC Adriatic Sea.

5. Conclusions

This study illustrates that the current management regime in the NC Adriatic Sea is not effective enough in stopping the decline of commercial stocks and rebuilding depleted ecosystems. However, our results highlight that recovery is possible if substantial management changes are undertaken. In this context, both the establishment of MPAs and seasonal reductions in fishing effort seem to be suitable options that may yield significant positive ecological results in a near future if strongly implemented. In both cases, we observed a reorganization of the food web with larger populations of predatory fish and other predators and a reduction of smaller prey organisms.

MPAs are becoming a popular management tool to promote the conservation of marine resources and ecosystems. Our spatial simulations suggest that Northern Adriatic Sea MPAs could be beneficial for commercial stock recovery and rebuilding of biomass of several fish populations and top predators. Also, the implementation of the Pomo pit MPA may also be beneficial for several marine resources especially the target species European hake and Norway lobster. In addition, seasonal closure for 3 months and reduction of effort from both pelagic and demersal fleets could also yield significant benefits for most resources, especially for the demersal ones. Comparing the effectiveness of different scenarios, we argue that both management measures could lead to a better situation than the current and future ones under "business as usual" context.

Acknowledgments

During the work, N.F. was supported financially by a fellowship from the International Centre for Advanced Mediterranean Agronomic studies (CIHEAM). M.C. was supported financially by postdoctoral fellowships from the Spanish Ministry of Science and Technology and by the European Community Marie-Curie OIF Post-doctoral Fellowship to the ECOFUN project. This work was developed within the context of the SESAME project—EC Contract No. GOCE-036949, funded by the Sixth Framework Programme. The authors wish to thank Chiara Piroddi from the EU Joint Research Centre, Ispra, and E.B. Morello from the Istituto di Scienze Marine (ISMAR) for useful advice for the development of this work. Particular thanks go to S. Martell from the Fisheries Centre (University of British Columbia) for help and advice during the construction of the Ecospace model.

References

AdriaMed, 2005. General outline of marine capture fisheries legislation and regulations in the Adriatic Sea countries. FAO-MiPAF Scientific Cooperation to Support Responsible Fisheries in the Adriatic Sea. GCP/RER/010/ITA/TD14 (rev. 1): AdriaMed Technical Documents, 14 (rev. 1): 68 pp.
AdriaMed, 2009. Report of the Tenth Meeting of the AdriaMed Coordination Committee. FAO-MiPAAF Scientific Cooperation to Support Responsible Fisheries in the Adriatic Sea.GCP/RER/010/ITA/TD26: AdriaMed Technical Documents, 26. 35 pp.
Agardy, T., 2000. Effects of fisheries on marine ecosystems: a conservationist's perspective. ICES J. Mar. Sci. 57 (3), 761–765.
Ahrens, R.N.M., Walters, C.J., Christensen, V., 2011. Foraging arena theory. Fish Fish., http://dx.doi.org/10.1111/j.1467-2979.2011.00432.x
Anon, 2007. Council Regulation (EC) No 1967/2006 of 21 December 2006 concerning management measures for the sustainable exploitation of fishery resources in the Mediterranean Sea, amending Regulation (EEC) No 2847/93 and repealing Regulation (EC) No 1626/94. Off. J. Eur. Union L36/6.
Anon, 2009. Zone di tutela biologica: nuove determinazioni. DECRETO 22 gennaio 2009. Ministero delle Politiche Agricole Alimentari e Forestali. (GU n. 37 del 14-2-2009).
Arneri, E., 1996. Fisheries resources assessment and management in Adriatic and Ionian Seas. FAO Fish. Rep. 533, 7–20.
Arneri, E., Morales-Nin, B., 2000. Aspects of the early life history of European hake from the central Adriatic. J. Fish Biol. 56, 1368–1380.
Artegiani, A., Bregant, D., Paschini, E., Pinardi, N., Raicich, F., Russo, A., 1997a. The Adriatic Sea general circulation. Part I: air sea interactions and water mass structure. J. Phys. Oceanogr. 27, 1492–1514.
Artegiani, A., Bregant, D., Paschini, E., Pinardi, N., Raicich, F., Russo, A., 1997b. The Adriatic Sea general circulation. Part II: baroclinic circulation structure. J. Phys. Oceanogr. 27, 1515–1532.
Azzali, M., De Felice, A., Luna, M., Cosimi, G., Parmiggiani, F., 2002. The state of the Adriatic Sea centered on the small pelagic fish populations. P.S.Z.N. Mar. Ecol. 23 (Suppl. 1), 78–91.
Bombace, G., 1992. Fisheries of the Adriatic Sea. Marine Eutrophication and Population Dynamics. 25th European Marine Biology Symposium. Olsen and Olsen, Fredensborg, pp. 379–389.
Botsford, L.W., Castilla, J.C., Peterson, C.H., 1997. The management of fisheries and marine ecosystems. Science 277, 509–515.
Brambati, A., Ciabatti, M., Fanzutti, G.P., Marabini, F., Marocco, R., 1983. A new sedimentological textural map of the northern and central Adriatic Sea. Boll. Oceanol. Teorica Appl. I 4, 1–7.
Browman, H.I., Stergiou, K.I., 2004. Perspectives on ecosystem-based approaches to the management of marine resources. Mar. Ecol. Prog. Ser. 274, 269–303.
Chen, Z., Qiu, Y., Jia, X., Xu, S., 2008. Simulating fisheries management options for the Beibu Gulf by means of an ecological modelling optimization routine. Fish. Res. 89, 257–265.
Chen, Z., Xu, S., Qiu, Y., Lin, Z., Jia, X., 2009. Modeling the effects of fishery management and marine protected areas on the Beibu Gulf using spatial ecosystem simulation. Fish. Res. 100, 222–229.
Christensen, V., 1998. Fishery-induced changes in a marine ecosystem: insight from models of the Gulf of Thailand. J. Fish Biol. 53 (Suppl. A), 128–142.
Christensen, V., Maclean, J., 2011. Ecosystem Approaches to Fisheries: A Global Perspective. Cambridge University Press, Cambridge. 325 pp.
Christensen, V., Pauly, D., 1992. ECOPATH II — a software for balancing steady-state ecosystem models and calculating network characteristics. Ecol. Model. 61, 169–185.
Christensen, V., Walters, C.J., 2004. Ecopath with Ecosim: methods, capabilities and limitations. Ecol. Model. 172, 109–139.
Christensen, V., Guénette, S., Heymans, J.J., Walters, C.J., Watson, R., Zeller, D., Pauly, D., 2003. Hundred-year decline of North Atlantic predatory fishes. Fish Fish. 4, 1–24.
Christensen, V., Walters, C., Pauly, D., Forrest, R., 2008. Ecopath with Ecosim version 6. User Guide — November 2008. Lenfest Ocean Futures Project 2008. 235 pp.
Cingolani, N., Giannetti, G., Arneri, E., 1996. Anchovy fisheries in the Adriatic Sea. Sci. Mar. 60 (Suppl. 2), 269–277.
Coll, M., Libralato, S., 2012. Contributions of food web modelling to the ecosystem approach to marine resource management in the Mediterranean Sea. Fish Fish. 13, 60–88.
Coll, M., Santojanni, A., Palomera, I., Tudela, S., Arneri, E., 2007. An ecological model of the northern and central Adriatic Sea: analysis of ecosystem structure and fishing impacts. J. Mar. Syst. 67, 119–154.
Coll, M., Bahamon, N., Sardà, F., Palomera, I., Tudela, S., Suuronen, P., 2008a. Improved trawl selectivity: effects on the ecosystem in the South Catalan Sea (NW Mediterranean). Marine Ecology Progress Series 355, 131–147.
Coll, M., Lotze, H.K., Romanuk, T., 2008b. Structural degradation in Mediterranean Sea food webs: testing ecological hypotheses using stochastic and mass-balance modelling. Ecosystems 11, 939–960.
Coll, M., Santojanni, A., Palomera, I., Arneri, E., 2009. Food-web changes in the Adriatic Sea over the last three decades. Mar. Ecol. Prog. Ser. 381, 17–37.
Coll, M., Shannon, L.J., Yemane, D., Link, J.S., Ojaveer, H., Neira, S., Jouffre, D., Labrosse, P., Heymans, J.J., Fulton, E.A., Shin, Y.-J., 2010a. Ranking the ecological relative status of exploited marine ecosystems. ICES J. Mar. Sci. 67, 769–786.
Coll, M., Santojanni, A., Palomera, I., Arneri, E., 2010b. Ecosystem assessment of the North-Central Adriatic (Mediterranean Sea): towards a multivariate reference framework. Mar. Ecol. Prog. Ser. 417, 193–210.
Demestre, M., De Juan, S., Sartor, P., Ligas, A., 2008. Seasonal closures as a measure of trawling effort control in two Mediterranean trawling grounds: effects on epibenthic communities. Mar. Pollut. Bull. 56, 1765–1773.
FAO, 2003. The ecosystem approach to fisheries. FAO Technical Guidelines for Responsible Fisheries, 4, Suppl. 2. FAO, Rome. 112 pp.
Fischer, W., Bauchot, M.L., Schneider, M., 1987a. Fiches FAO d'identification des espèces pour les besoins de la pêche. Méditerranée et mer Noire. Zone de pêche 37. Végétaux et Invertébrés, vol. I. FAO, Rome. 760 pp.
Fischer, W., Bauchot, M.L., Schneider, M., 1987b. Fiches FAO d'identification des espèces pour les besoins de la pêche. Méditerranée et mer Noire. Zone de pêche 37. Vertébrés, vol. II. FAO, Rome. 761–1530 pp.
Fortibuoni, T., Libralato, S., Raicevich, S., Giovanardi, O., Solidoro, C., 2010. Coding early naturalists' accounts into long-term fish community changes in the Adriatic Sea (1800–2000). PLoS One 5 (11), e15502.
Garcia, S.M., Cochrane, K.L., 2005. Ecosystem approach to fisheries: a review of implementation guidelines. ICES J. Mar. Sci. 62, 311–318.
Garcia, S.M., Zerbi, A., Aliaume, C., Do Chi, T., Lasserre, G., 2003. The ecosystem approach to fisheries. Issues, terminology, principles, institutional foundations, implementation and outlook. FAO Fisheries Technical Paper, No. 443. FAO, Rome. 71 pp.
Goñi, R., Adlerstein, S., Alvarez-Berastegui, D., Forcada, A., Reñones, O., Criquet, G., Polti, S., Cadiou, G., Valle, C., Lenfant, P., Bonhomme, P., Pérez-Ruzafa, A., Sánchez-Lizaso, J.L., García-Charton, J.A., Bernard, G., Stelzenmüller, V., Planes, S., 2008. Spillover from six western Mediterranean marine protected areas: evidence from artisanal fisheries. Mar. Ecol. Prog. Ser. 366, 159–174.
Jackson, J.B.C., Kirby, M.X., Berger, W.H., Bjorndal, K.A., Botsford, L.W., Bourque, B.J., Bradbury, R.H., Cooke, R., Erlandson, J., Estes, J.A., Hughes, T.P., Kidwell, S., Lange, C.B., Lenihan, H.S., Pandolfi, J.M., Peterson, C.H., Steneck, R.S., Tegner, M.J., Warner, R.R., 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293, 629–637.
Jukić-Peladić, S., Vrgoć, N., Krstulović-Šifner, S., Piccinetti, C., Piccinetti-Manfrin, G., Marano, G., Ungaro, N., 2001. Long-term changes in demersal resources of the Adriatic Sea: comparison between trawl surveys carried out in 1948 and 1998. Fish. Res. 53 (1), 95–104.
Link, J.S., 2011. Ecosystem-based Fisheries Management: Confronting Tradeoffs. Cambridge University Press, Cambridge. 224 pp.
Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Matthew, C., Kay, M.C., Kidwell, S.M., Kirby, M.X., Peterson, C.H., Jeremy, B.C., Jackson, J.B.C., 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312, 1806–1809.
Lotze, H.K., Coll, M., Dunne, J.A., 2011a. Historical changes in marine resources, food-web structure and ecosystem functioning in the Adriatic Sea, Mediterranean. Ecosystems 14 (2), 198–222.
Lotze, H.K., Coll, M., Magera, M.A., Ward-Paige, C., Airoldi, L., 2011b. Recovery of marine animal populations and ecosystems. Trends in Ecology and Evolution 26 (11), 595–605.
Mannini, P., Massa, F., 2000. Brief overview of Adriatic fisheries landings trends (1972–97). In: Massa, F., Mannini, P. (Eds.), Report of the First Meeting of AdriaMed Coordination Committee. FAO-MiPAF Scientific Cooperation to Support Responsible Fisheries in the Adriatic Sea. GCP/RER/010/ITA/TD-01, pp. 31–49.
Mannini, P., Massa, F., Milone, N., 2004. Adriatic Sea fisheries: outline of some main facts. AdriaMed Seminar on Fishing Capacity: Definition, Measurements and Assessment. Report of the First Meeting of the AdriaMed Working Groups on Small Pelagic Resources, Termoli, Italia Maggio 2004. GCP/RER/010/ITA, pp. 13–33.
Martell, S.J.D., Walters, C.J., 2008. Experimental policies for rebuilding depleted stocks. Can. J. Fish. Aquat. Sci. 65 (8), 1601–1609.
Martell, S.J.D., Essington, T.E., Lessard, B., Kitchell, J.F., Walters, C.J., Boggs, C.H., 2005. Interactions of productivity, predation risk, and fishing effort in the efficacy of marine protected areas for the central Pacific. Can. J. Fish. Aquat. Sci. 62, 1320–1336.
Mora, C., Myers, R.A., Coll, M., Libralato, S., Pitcher, T.J., Sumaila, R.U., Zeller, D., Watson, R., Kevin, J., Gaston, K.J., Worm, B., 2009. Management effectiveness of the world's marine fisheries. PLoS Biol. 7 (6), e1000131.
Morello, E.B., Arneri, E., 2009. Anchovy and sardine in the Adriatic Sea — an ecological review. Oceanogr. Mar. Biol. Annu. Rev. 47 (209), 256.
Mužinić, R., 1973. Migrations of adult sardines in the central Adriatic. Neth. J. Sea Res. 7, 19–30.
Ortiz, M., Wolff, M., 2002. Spatially explicit trophic modelling of a harvested benthic ecosystem in Tongoy Bay (central northern Chile). Aquat. Conserv. Mar. Freshwater Ecosyst. 12, 601–618.
Ott, J., 1992. The Adriatic benthos: problems and perspectives. In: Colombo, G., Ferrari, I., Ceccherelli, V.U., Rossi, R. (Eds.), Marine Eutrophication and Population

Dynamics. 25th European Marine Biology Symposium. Olsen and Olsen, Fredensborg, pp. 367–378.
Palomares, M.L.D., Morissette, L., Cisneros-Montemayor, A., Varkey, S., Coll, M., Piroddi, C., 2009. Ecopath 25 years conference: extended abstracts. Fish. Cent. Res. Rep. 17, 171.
Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., Torres, F.J., 1998. Fishing down marine food webs. Science 279, 860–863.
Pauly, D., Christensen, V., Guénette, S., Pitcher, T.J., Sumaila, U.R., Walters, C.J., Watson, R., Zeller, D., 2002. Towards sustainability in world fisheries. Nature 418, 689–695.
Pauly, D., Alder, J., Bennett, E., Christensen, V., Tyedmers, P., Watson, R., 2003. The future for fisheries. Science 302, 1359–1361.
Pinardi, N., Arneri, E., Crise, A., Ravaioli, M., Zavatarelli, M., 2006. The physical, sedimentary and ecological structure and variability of shelf areas in the Mediterranean Sea (27). In: Robinson, A.R., Brink, K.A. (Eds.), The Sea, vol. 14. Harvard University Press, pp. 1245–1331.
Pitcher, T.J., Watson, R., Haggan, N., Guénette, S., Kennish, R., Sumaila, R., Cook, D., Wilson, K., Leung, A., 2000. Marine reserves and the restoration of fisheries and marine ecosystems in the South China Sea. Bull. Mar. Sci. 66 (3), 530–566.
Polovina, J.J., 1984. Model of a coral reef ecosystem. I. The Ecopath model and its application to French Frigate Shoals. Coral Reefs 3, 1–11.
Pranovi, F., Raicevich, S., Franceschini, G., Farrace, M.G., Giovanardi, O., 2000. Rapido trawling in the northern Adriatic Sea: effects on benthic communities in an experimental area. ICES J. Mar. Sci. 57, 517–524.
Pranovi, F., Raicevich, S., Franceschini, G., Torricelli, P., Giovanardi, O., 2001. Discard analysis and damage to non-target species in the rapido trawl fishery. Mar. Biol. 139, 863–875.
Riedl, R., 1986. Fauna y Flora del Mar Mediterráneo. Ed. Omega, Barcelona, 858 pp.
Roberts, C.M., 1997. Connectivity and management of Caribbean coral reefs. Science 278, 1454–1457.
Roberts, C.M., 2000. Selecting marine reserve locations: optimality versus opportunism. Bull. Mar. Sci. 66 (3), 581–592.
Salomon, A.K., Waller, N.P., McIlhagga, C., Yung, R.L., Walters, C., 2002. Modeling the trophic effects of marine protected area zoning policies: a case study. Aquat. Ecol. 36, 85–95.
Sánchez, P., Sartor, P., Recasens, L., Ligas, A., Martin, J., De Ranieri, S., Demestre, M., 2007. Trawl catch composition during different fishing intensity periods in two Mediterranean demersal fishing grounds. Sci. Mar. 71 (4), 765–773.
Santojanni, A., Arneri, E., Barry, C., Belardinelli, A., Cingolani, N., Giannetti, G., Kirkwood, G., 2003. Trends of anchovy (*Engraulis encrasicolus*, L.) biomass in the northern and central Adriatic Sea. Sci. Mar. 67 (3), 327–340.
Santojanni, A., Cingolani, N., Arneri, E., Kirkwood, G., Belardinelli, A., Giannetti, G., Colella, S., Donato, F., Barry, C., 2005. Stock assessment of sardine (*Sardina pilchardus*, Walb.) in the Adriatic Sea, with an estimate of discards. Sci. Mar. 69 (4), 603–617.
Vrgoč, N., Arneri, E., Jukič-Peladič, S., Krstulovič Šifner, S., Mannini, P., Marčeta, B., Osmani, K., Piccinetti, C., Ungaro, N., 2004. Review of current knowledge on shared demersal stocks of the Adriatic Sea. FAO-MiPAF Scientific Cooperation to Support Responsible Fisheries in the Adriatic Sea. GCP/RER/010/ITA/TD-12: AdriaMed Technical Documents, 12. 91 pp.
Walters, C.J., 2000. Impacts of dispersal, ecological interactions, and fishing effort dynamics on efficiency of marine protected areas: how large should protected areas be? Bull. Mar. Sci. 66, 745–757.
Walters, C., Martell, S., 2004. Fisheries Ecology and Management. Princeton University Press, Princeton, NJ (USA).
Walters, C.J., Christensen, V., Pauly, D., 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Rev. Fish Biol. Fish. 7, 139–172.
Walters, C.J., Pauly, D., Christensen, V., 1999. Ecospace: predictions of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. Ecosystems 2, 539–554.
Worm, B., Hilborn, R., Baum, J.K., Branch, T.A., Collie, J.S., Costello, C., Fogarty, M.J., Fulton, E.A., Hutchings, J.A., Jennings, S., Jensen, O.P., Lotze, H.K., Mace, P.M., McClanahan, T.R., Minto, C., Palumbi, S.R., Parma, A.M., Ricard, D., Rosenberg, A.A., Watson, R., Zeller, D., 2009. Rebuilding global fisheries. Science 325, 578–585.
Zavatarelli, M., Raicich, F., Bregant, D., Russo, A., Artegiani, A., 1998. Climatological biogeochemical characteristics of the Adriatic Sea. J. Mar. Syst. 18, 227–263.
Zavatarelli, M., Baretta, J.W., Baretta-Bekker, J.G., Pinardi, N., 2000. The dynamics of the Adriatic Sea ecosystem. An idealized model study. Deep Sea Res. I (47), 937–970.
Zeller, D., Reinert, J., 2004. Modelling spatial closures and fishing effort restrictions in the Faroe Islands marine ecosystem. Ecol. Model. 172, 403–420.
Županović, S., Jardas, I., 1986. A contribution to the study of biology and population dynamics of the Adriatic hake, *Merluccius merluccius* (L.). Acta Adriat. 27, 97–146.