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Coastal fish assemblages associated with rocky substrates: differences between a marine reserve (Pianosa island) and an unprotected area (Elba island).

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ABSTRACT

Along the coastline of Italy, there are many areas of high environmental value. Nevertheless, only for few of these the protection proposed in 1982 by low has been really enforced. Understanding if marine reserves work, requires two levels of analysis. The first requires that particular reserves be evaluated in the context of the goals inherent in their establishment. The second level of analysis should be a broad comparison across reserves in different ecosystems to understand if reserves in general have a local or regional effect. Fish assemblage inside the MPAs, usually include many species targeted by fishing, so that they are primarily expected to benefit from protection, especially those having no-take reserve. The evaluation of the benefits, in terms of increase in density and size of target fishes can be useful to assess the ecological effectiveness of reserves. Up to now, there are very few long-term series of density estimates in and around MPAs before and after protection to verify these assumptions in the context of protected populations.

Fish fauna distributions patterns can be useful for testing the effectiveness of habitat protection in marine reserves. Most target fishes are high-levels predator and their functional extinction may cause community wide changes. The impact of fishing has consequent top-down perturbations in marine communities structures and organization. Benthic communities in the shallow Mediterranean rocky sublittoral have 2 extreme alternative succession endpoints: coralline barrens and complex macroalgal beds. In the Mediterranean, Paracentrotus lividus is a key species that a high densities influences the dynamics of the phytobenthos, by eliminating erect algae and seagrasses, and inducing the formations of coralline barrens. Despite ecologists' increasing interest in fish fauna living in marine protected areas, fish assemblage of some Italian marine reserves are still completely unknown..

Pianosa represents the ideal area where to verify the hypothesis that are the subject of this PhD. The PhD thesis is aimed at: 1- determine the structure of the infalittoral fish assemblage inhabiting the protected (Pianosa Island) and adjacent (Elba Island) non protected areas of Tuscan Archipelago, in terms of species composition, diversity and relative density, assessing differences in the community parameters in different areas; 2- to estimate the fish community responses to the trophic cascades in shallow rocky shores as possible results that indicate the reserve effect.

In infralittoral area, is possible to assess that structure of rocky fish assemblage testing the effectiveness of habitat protection in MPAs?

Underwater Visual Census (UVC) is choised as the more appropriate method to conduct study on infralittoral fish community in Pianosa and Elba islands. Abundance data were collected by using a scale based on a geometric progression factor which was approximately 2. Fish density was calculated by considering the mid point of each abundant class. The size of individual fish was assigned to one of four size classes. Extensions of barren cover, was visually estimated as relative percentage by means of square of 1 m^{-2} *located on the bottom. P. lividus was chosen as indicator liable to possible shift from macroalgal cover to barren.*

During the survey the degree of exploitation on fish species at Elba island, was evaluated on the basis of direct observation of fishing landings in Marina di Campo harbour.

A one way ANOVA was performed to test for differences among the areas in the assemblage parameters. Permanova was performed to test differences in analysis fish community using locality as fixed factor. The reserve effect (response ratio, R), was calculated for trophic categories and exploitation levels, as ln of ratio between value of response variable (density) inside the protected area and in fished areas.

During this study, 46 fish species were identified. Labridae and Sparidae were the most relevant families in relation to species richness. Some large sized specimens, belonging to species of high commercial value (i.e. Seriola dumerilili, Dentex dentex, and Ephinephelus marginatus), were also observed.

Taxonomic composition of fish community in Pianosa and Elba islands agrees with other similar northern-western in Mediterranean. The abundance of target species correlates strongly with the reserve effect, and therefore shows that the reserve has been effective in providing protection for such species. Some highly spearfished species such as E. marginatus, S. umbra, or D. dentex, have been censed exclusively within the reserve. Other such as Sparus aurata are far more abundant in the reserve. Species diversity index, and Evenness, show significant differences between sites located in Pianosa. Here, Diplodus vulgaris and Diplodus sargus, showed higher abundance in protected than in unprotected area. The study of four size classes of some target species has confirmed that large individuals were significantly more abundant in the protected area.

The increase of the size of fishes is a phenomenon widely reported in other MPAs and this general pattern has primarily attributed to the lack of fishing impact. Protected area seems to have the potential to restore depleted fish stocks. The response ratio evaluated for exploitation level shows that the reserve has been effective in protection for heavily fished species. Protected area, support greater density of D. sargus and D. vulgaris as predators on sea urchin P. lividus, than unprotected area. At the same time, the cover of barrens was significantly higher in unprotected area than in Pianosa. This study has evidenced a reserve effect or refuge effect characterized by the presence of large individuals. This refuge effect, within the presence of large individuals and species vulnerable to fishing, is one of the characteristics described in the definition of reserve effect. The results of the present study suggest that the recovery of Diplodus species in Pianosa seems to reestablish predatory control upon P. lividus similar to what is observed in other temperate regions and in other Mediterranean protected areas. The results of this study can corroborate the hypothesis of a transition from macroalgal beds, which harbour hundred of species of algae and invertebrates, to barrens through sea urchins grazing.In protected area, the effect of predator density on prey and on its population parameter is density-mediated indirect interactions (DMIIs).

This study, is a further step toward a better knowledge of the rocky reef fish assemblage of this area. At the community level, the present study will provide a useful reference point for planning future investigations on inter-specific relationship such as competition. Pianosa is an protected area where is possible to demonstrate that reserves aimed at restoring whole assemblages and ecological processes should be established as permanent no take area. This studies confirms that besides direct effects on target species, fishing may cause changes in trophic relations and in ecosystem functioning. Despite Pianosa is not a typical MPA since actual permanent presence of penal institute, it represent an important site where to study isolated infralittoral fish community on rocky substrata. It is clear that future experiments testing the reserve *effect on fish biodiversity will have to take into account the necessity of being repeated on larger spatial and temporal scales.*

1. INTRODUCTION

1.1 Marine protected areas (MPAs) as a tool for maintenance and protection of biodiversity

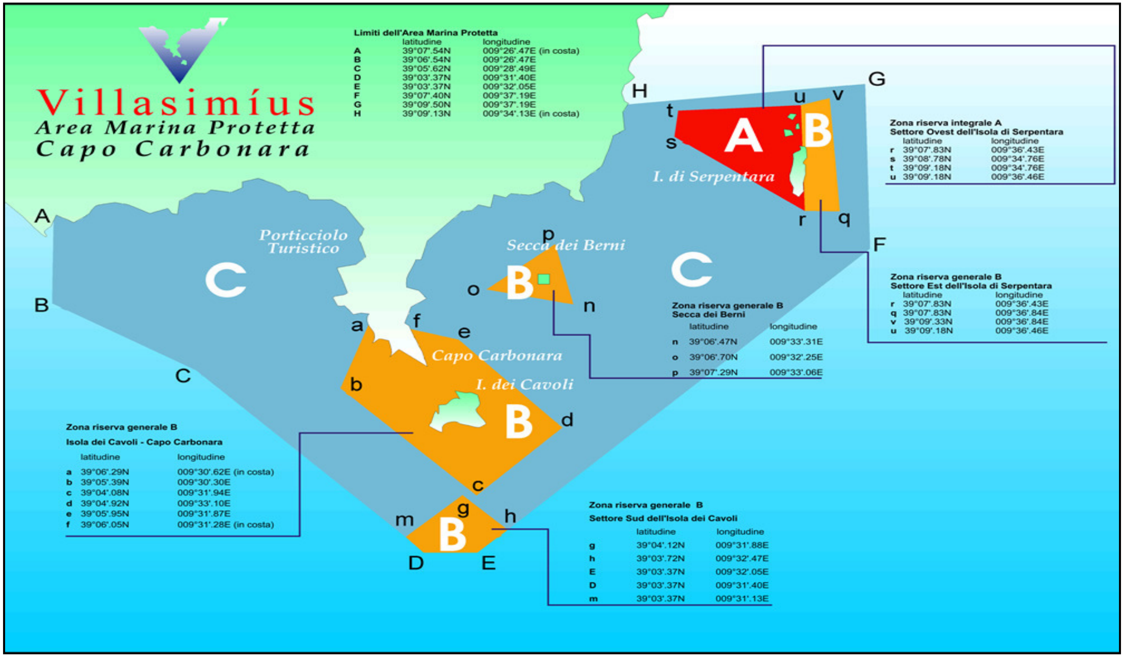
A large number of marine protected areas (MPAs) have been established around the world from second half of XX century, and while some offer protection to pristine natural communities, others attempt to halt further deterioration of sensitive habitats or serve as fisheries management tools for long-terms sustainability of fisheries (Ramos and McNeill 1984; Kelleher et al., 1995).

Generally speaking, MPAs refer to portion of the coastline where human activities, especially fishing, are restricted or banned (Agardy et al., 2003).

In the Mediterranean Sea there has been a rush in recent years to establish MPAs and reserves (Juanes, 2001). Along the coastline of Italy, there are many areas of high environmental value. Nevertheless, only for few of these the protection proposed in 1982 by low has been really enforced. In Italy, actually have been formally establishe 27 MPAs, and a total of 222.000 hectares are protected. (Fig. 1). Italian MPAs include one or more no take/ no access zones, formally defined as "A zones" according to Italian law, surrounded by buffer zones defined as "B and "C Zones" (Fig. 2), where restrictions to human uses, including fishing, become progressively more lax (Villa et al., 2002).

Figure 1 : MPAs actually established along Italian coasts.

Figure 2: typical division of a MPA in different area with different level of protection.



Limiti dell'Area Marina Protetta

	latitudine	longitudine
A	39°07'.54N	009°26'.47E (in costa)
B	39°06'.54N	009°26'.47E
C	39°05'.62N	009°28'.49E
D	39°03'.37N	009°31'.40E
E	39°03'.37N	009°32'.05E
F	39°07'.40N	009°37'.19E
G	39°09'.50N	009°37'.19E
H	39°09'.13N	009°34'.13E (in costa)

Zona riserva integrale A
Settore Ovest dell'Isola di Serpentara

	latitudine	longitudine
r	39°07'.83N	009°36'.43E
s	39°08'.78N	009°34'.76E
t	39°09'.18N	009°34'.76E
u	39°09'.18N	009°36'.46E

Zona riserva generale B
Settore Est dell'Isola di Serpentara

	latitudine	longitudine
r	39°07'.83N	009°36'.43E
q	39°07'.83N	009°36'.84E
v	39°09'.33N	009°36'.84E
u	39°09'.18N	009°36'.46E

Zona riserva generale B
Secca dei Berni

	latitudine	longitudine
n	39°06'.47N	009°33'.31E
o	39°06'.70N	009°32'.25E
p	39°07'.29N	009°33'.06E

Zona riserva generale B
Isola dei Cavoli - Capo Carbonara

	latitudine	longitudine
a	39°06'.29N	009°30'.62E (in costa)
b	39°05'.39N	009°30'.30E
c	39°04'.08N	009°31'.94E
d	39°04'.92N	009°33'.10E
e	39°05'.95N	009°31'.87E
f	39°06'.05N	009°31'.28E (in costa)

Zona riserva generale B
Settore Sud dell'Isola dei Cavoli

	latitudine	longitudine
g	39°04'.12N	009°31'.88E
h	39°03'.72N	009°32'.47E
E	39°03'.37N	009°32'.05E
D	39°03'.37N	009°31'.40E
m	39°03'.37N	009°31'.13E

From conceptual point of view, among the many different "types" of MPAs, there are three broad kinds whose underlying objectives are quite different:

a) fisheries enhancement MPAs focus on protecting local populations as a management tool to augment regional fisheries yields;

b) ecosystem diversity MPAs focus on the preservation and maintenance of broad-scale marine biological diversity;

c) special-feature MPAs focus on preservation of a particular locality because of its cultural importance or its value to a particularly vulnerable life history stage.

The broad definition of a MPA, adopted by the International Union of Concerned Scientists (UCS), states that a MPA is "any area of subtidal terrain, together with its overlying waters and associated flora, fauna, historical and cultural features, which has been preserved by legislation or other effective means to protect all or part of the enclosed environment" (Kelleher and Recchia, 1998).

Although this definition may seem too broad, the International Union for Conservation of Nature (IUCN) also established six categories of MPAs that begin to reflect the diversity of practical uses and ecological settings that MPAs include (Tab. 1) (Kelleher and Recchia, 1998).

Biodiversity reserves, those designed primarily to enhance costal ecosystem stability and multispecies interactions, are most stringently defined in categories I and II. Categories III, V, and VI also have ecosystem protection as a major goal.

Frequently a MPA might play more than one role. In particular, fisheries enhancement, the preservation of ecosystem diversity, and nursery ground protection often can go hand in hand. MPAs are generally instituted to solve, particular, local problems. Because each local problem is unique and because the design of MPAs is linked tightly to their objectives, there are so many different types of MPAs that their categorization is often complex. Sometimes this complexity is reflected in a mosaic of different use patterns, as in the Australian "Great Barrier Reef Marine Park", a large protected area (http://www.gbrmpa.gov.au/) with different types of protection in different areas. Some area in the Park are no-takes reserves, designed to enhance ecosystem diversity, and these are found scattered inside the larger park, where extractive uses are allowed (Fig.3).

Figure 3: Australian great barrier reef marine park.

Understanding if marine reserves work, requires two levels of analysis. The first requires that particular reserves be evaluated in the context of the goals inherent in their establishment. The second level of analysis should be a broad comparison across reserves in different ecosystems to understand if reserves in general have a local or regional effect. Each reserve is an unique area, often established because of its topographic, biodiversity, or cultural attributes. It is thus difficult to directly assess the effectiveness of reserve status in the strict sense, because each reserve is an un-replicated area. In addition, comparison of reserve to non-reserve area is often done without the spatial and temporal controls that have become the chief analytical tool of experimental ecology (Paine, 1984)

Table 1: Categories of Marine Protected Areas Established by the IUCN.

Category I	Strict Nature Reserve/Wilderness Area: protected area managed mainly for science or wilderness protection.
Category Ia	Strict Nature Reserve: protected area managed mainly for science-definition: Area of land and/or sea possessing some outstanding or representative ecosystems, geological or physiological features, and/or species, available primarily for scientific research and/or environmental monitoring.
Category Ib	Wilderness Area: protected area managed mainly for wilderness protection-definition: Large area of unmodified or slightly modified land, and/or sea, retaining its natural character and influence, without permanent or significant habitation, protected and managed so as to preserve its natural condition.
Category II	National Park: protected area managed for ecosystem protection and recreation-definition: Natural area of land and/or sea, designed to (a) protect the ecological integrity of one or more ecosystems for present and future generations, (b) exclude exploitation or occupation inimical to the purposes of designation of the area, and (c) provide a foundation for spiritual scientific, recreational, and visitor opportunities, all of which must be environmentally and culturally compatible.
Category III	Natural Monument: protected area managed mainly for conservation of specific natural features-definition: Area containing one, or more, specific natural or natural /cultural features which is of outstanding or unique value because of its inherent rarity, representative or aesthetic qualities or cultural significance.
Category IV	Habitat/Species Management Area: protected area managed mainly for conservation through management intervention-definition: Area of land and/or sea subject to active intervention for management purposes so as to ensure the maintenance of habitat and/or meet the requirements of specific species.

Category V	Protected Landscape/Seascape: protected area managed mainly for landscape/seascape conservation and recreation-definition: Area of land, with coast and sea appropriate, where the interaction of people and nature over time has produced an area of distinct character with significant aesthetic, ecological, and/or cultural value, and often with high biological diversity. Safeguarding the integrity of this traditional interactions is vital to the protection, maintenance, and evolution of such an area.
Category VI	Managed Resource Protected Area: protected area managed mainly for the sustainable use of natural ecosystems-definition: Area containing predominantly unmodified systems, managed to ensure long-term protection and maintenance of biological diversity, while providing at the same time a sustainable flow of natural products and services community needs.

The relationship between protection and diversity is complex. In some cases, species richness increase inside reserves. In others, the major impact of reserve status is a relevant shift in which species are present, without necessarily an increase in overall specie numbers. Species with strong recruitment, fast generations, and large growth rates are most likely to show increases in short periods of time after reserve establishment. More slowly growing species, or those with low recruitment, take much longer to show a reserve effect. Although there is ample evidence that reserve status has a marked impact on overall community composition, there is relatively little attention given to non commercial species within biodiversity reserves and little understanding of the complicated ecological cascades produced by major shifts in exploitation (Guidetti, 2006).

Changes in biomass can be obvious after only a period of several years. Increase in abundance can occur more slowly, but have been visible in numerous species that can recruit from a regional larval pool.

Fish assemblage inside the MPAs, usually include many species targeted by fishing, so that they are primarily expected to benefit from protection, especially those having no-take reserve (Dayton et al., 1995; Micheli et al., 2004; McClanahan et al., 2007). The evaluation of these benefits, in terms of increase in density and size of target fishes (Mosquera et al., 2000; Cote et al., 2001; Halpern, 2003; Micheli et al., 2004; Claudet at al., 2006; Guidetti and Sala, 2007), can be useful to assess the ecological effectiveness of reserves (Guidetti et al., 2008).

Studies of fish diversity patterns inside MPAs, indicate that protection may have an effect on overall community richness, but that the major changes are in the species previously hunted. Several studies have demonstrated the many potential benefits of no-take marine reserves or marine protected areas. Two of these are local emigration of large juveniles and adults of fishes, to adjacent fished areas and the enhancement of broad scale or regional fisheries by means of larval export as a result of conservation of spawning stock biomass (Rowley, 1994).

Many investigations of marine reserves, have been hampered by a lack of replication either in time or in space, so it is difficult to gauge whether the significant differences in density and size of target species reflect:

(1) greater fishing pressure adjacent to the reserve since closure;

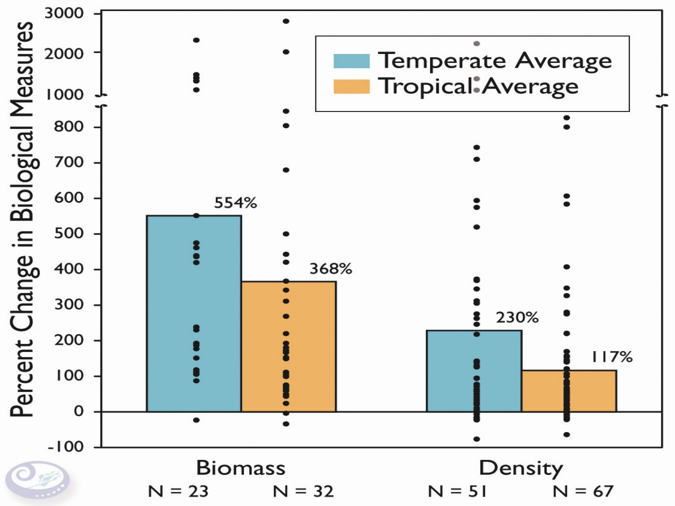
(2) preservation of the *status quo* as of the closure;

(3) an increase in density and biomass after closure.

Monitoring, can help resolve the issue to some extent, but one of the important considerations for future MPAs is to collect the baseline information before closure.

Fishing reduces population abundance removing larger and older fish, thus changing the size and age structure of exploited populations (Jennings and Polunin, 1996). The cessation or reduction of fishing, may promote an increase of abundance as well as the mean size and age of the protected populations (Fig.4).

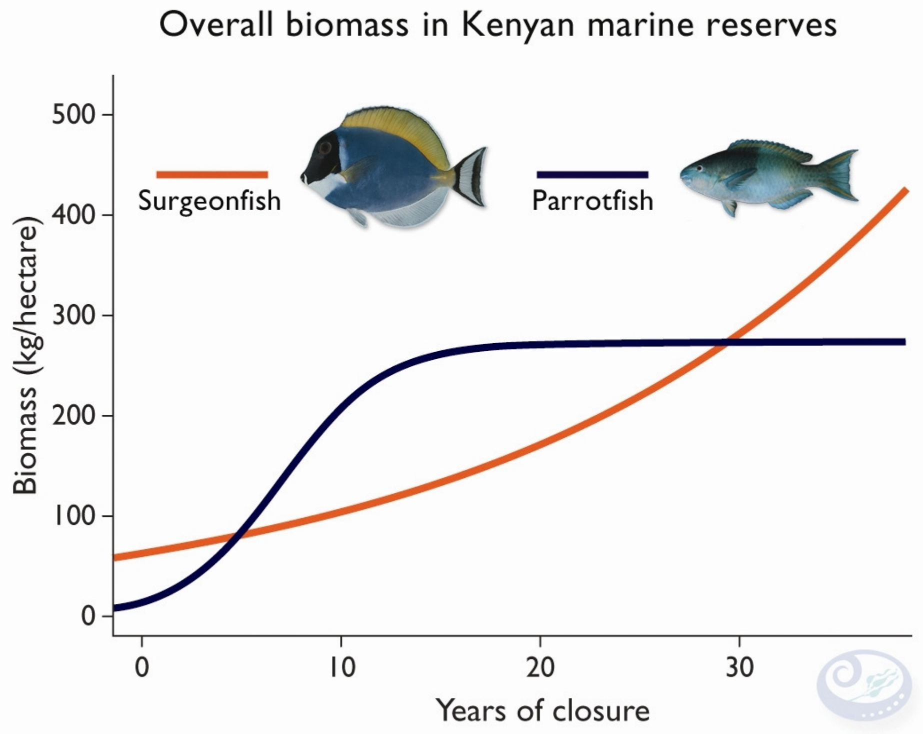
Figure 4: variation in density and biomass due to the MPAs establishment.



Average changes in fishes, invertebrates, and seaweeds within marine reserves from temperate (blue bars) and tropical (orange bars) regions around the world. Although changes varied among reserves (black dots), most reserves had positive changes in both regions. *Data: Lester et al. in revision*

The species more likely to respond to the cessation of fishing in MPAs (reserve effect) are large, long-lived predators, organisms highly vulnerable to fishing and those the populations of which are overexploited (Bohnsack 1996) (Fig. 5)

Figure 5: increase in biomass after MPA istitution



Increases in surgeonfish species (red line) and parrotfish species (blue line) in 4 Kenyan marine reserves. *Data: McClanahan et al. (2007) Ecological Applications*

Other species may not be influenced by protection or may show the opposite response (lower abundance or biomass in MPAs), presumably due to inter-specific interactions (Pinnegar *et al*., 2000).

Few data support enhancement of fishers' catch by emigration of adults from marine reserves to fished areas (Dugan and Davis, 1993). The degree to which larval production and settlement are increased by marine reserves is even more difficult to test because individual planktonic larvae cannot be tracked. Moreover, complex factors such as transport process and settlement, affects the success of larval recruitment (Lipicius and Cobb, 1993). Larval enhancement is likely if adequate spawning-stock biomass is considered within marine reserves, if larvae are transported to an area where they can recruit to a fishery, if an increase in larvae leads to an increase in number of larvae that settle, and if larvae settlement is the limiting factor for the fishery rather than habitat availability.

Marine reserves should be large enough to retain a large portion of protected individuals, and old enough for the effects of protection (on e.g., density, size, fecundity) to be realized. The appropriate size for a marine reserves is easier to determine if the movement patterns of the species chosen for protection and the fishing pressure in adjacent areas are known (Sala et al., 1998).

The shape of marine reserves can influence the movement of organisms across boundaries and it is primarily a function of the perimeter-to-area ratio of the protected area (Roberts and Polunin, 1991). The type and spatial extent of habitat bordering the reserve will influence emigration, e.g., whether the reserve is within a larger patch of similar habitat or borders very dissimilar areas. The shape of MPAs will also influence larval recruitment, dependant on the intersection of the reserve area with prevailing currents (Roberts and Polunin, 1991).

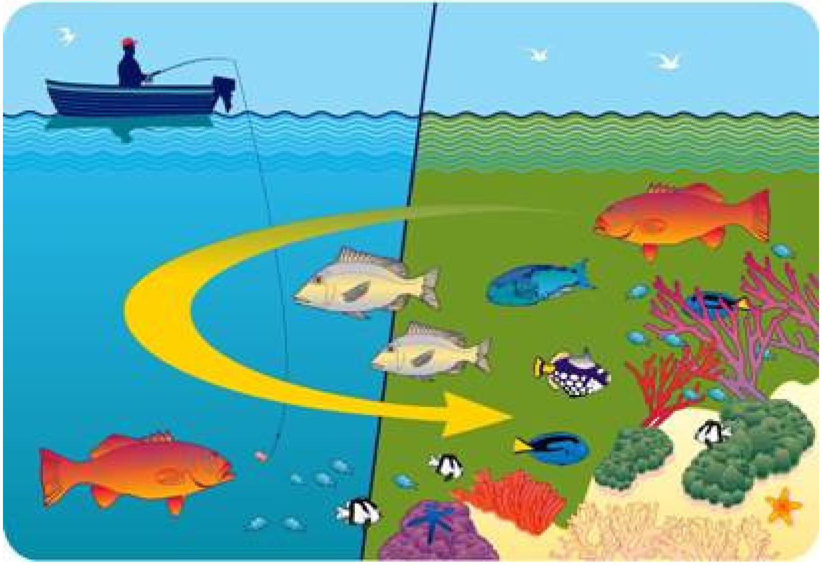
Effective marine reserves location depends critically on proximity to fishery activities, habitats encompassed, and effective enforcement capacity (Rowley, 1994). Because many marine invertebrate and fish species produce planktonic larvae, marine reserves should ideally be located such that their larval production is exported to both suitable habitats and fished areas.

Species considerations in marine reserve design are important because those species most likely to show dramatic responses to protection, are probably those limited movement and age-size distributions that have been lowered by fishing pressure. An important reserve design criterion for such species is to include spawning aggregations within reserve boundaries.

Emigration of large juveniles and adults from marine reserves to fished areas is related to the ability of fished species to reach greater densities and larger sizes in the reserve, and to the movement patterns of individuals across reserve boundaries (Rowley, 1994). Greater adult densities and sizes in MPAs than in fished areas, are the most common patterns and direct responses of fishery target species to protection from fishing, but other more complex responses can affect other species and trophic levels (Roberts and Polunin, 1991).

When the density of a population is higher inside a MPA than in adjacent unprotected areas, random movements are expected to produce a net emigration from the reserve (Ratikin and Kramer, 1996). This emigration of individuals from MPAs, has also been proposed as the explanation for the lack of difference in abundance of some species between MPAs and fished areas (Fig. 6) reported in some studies (Childress, 1997).

Figure 6: spillover of individuals from MPAs.



Potential emigration could be important for non-pelagic fishes and some invertebrates such as lobster and shrimps. These species may spend enough time inside MPAs to have a significant reduction in fishing mortality while having the ability to move outside the MPAs. Highly mobile species like pelagic fish, could move easily in and out of MPAs, but additional production of biomass, if any, attributable to protection will be low, and will be related to time spent in the MPAs (Bohnsack 1996). Sessile or highly sedentary species, as many invertebrates, could not emigrate significantly during their adult life.

Despite the potential of biomass export from MPAs as a management tool for fisheries and species replenishment, there is remarkably little evidence of this effect so far. Some studies have demonstrated that fish tagged and released inside may be caught outside them (Bohnsack 1998), however this researches usually do not demonstrate that emigration is higher than immigration.

It has been noted that fisheries that exploit species with some natural refuge from current fishing techniques, appear to be more robust and dependable over time than fisheries on species that have no natural refuges (Beverton and Holt, 1957; Dugan and Davies, 1993). Similarly, MPAs, as natural refuges, may increase fish stocks in adjacent areas through emigration of adult, juvenile and larval stages (Alcala and Russ, 1990). When compared with other fishing grounds used by fishers, the CPUE (Catch Per Unit Effort), of the artisanal fishery around a MPA in the western Mediterranean was higher (Mas and Barcala, 1997). The total yield of the small-scale fishery around this MPA did not change significantly after the MPA was created, but the catch of some highly sought-after species such as *Dentex dentex*, *Ephinepelus marginatus*, and *Pagrus pagrus* increased (Mas and Barcala, 1997).

Population inside the MPAs cannot increase endlessly and resource availability, recruitment, inter-specific interactions, as well as the rates of immigration and emigration, will determine in each case the maximum size that a population can attain inside a MPA. Reduction of fishing mortality should lead to increase survival and to a shift towards bigger and older individuals. This in turn will increase the reproductive potential of the protected populations. The resulting higher abundance and biomass in the MPA may result in increased competition that may in turn trigger compensatory mechanisms which will tend to dampen the rate of population growth.

Up to now, there are very few long-term series of density estimates in and around MPAs before and after protection to verify this assumption in the context of protected populations.

Fish that are not vulnerable to fishing (not catchable) due to a small size or to behavioural traits, such as avoiding or escaping from the fishing gear, should not be directly affected by the reserve. Not catchable fish should show an even distribution across the reserve boundary. As catchability increases, population sizes outside the reserve should be depressed to a greater extent (Ratikin and Kramer, 1996).

In some marine species, emigration from MPAs could be related to changes in habitat requirements during their life history, as many marine species undergo seasonal and ontogenetic bathymetric or habitat changes (MacDiarmid and Breen, 1993). Because diversity of habitats is often dependent on area, this condition is closely related to the sizes of MPAs. The effective sizes of MPAs depend on the home ranges and habitat requirements of the species to be protected (Bohnsack 1996). Even small MPAs have been shown to protect some species with restricted movements, but large areas may be needed to protect more mobile species (Childress, 1997). Also, high perimeter/area ratios will favour biomass export from MPAs and several small MPAs also influencing this ratio. The permeability of MPAs edges may be increased by placing MPAs within large patches of similar habitat. Alternatively, spillover may be limited from MPAs that protect separate patches of similar habitat separated by expanses of unfavourable habitat or deep water (Chapman and Kramer, 1999).

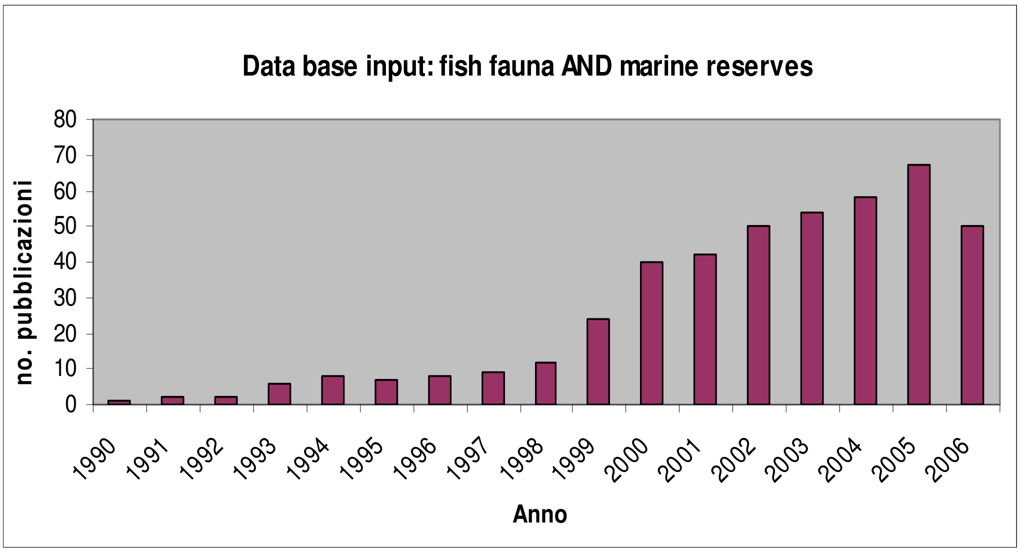
In Mediterranean Sea, MPAs have been established with the aims of providing refuge to threatened species, protecting ecosystems or habitats with particularly high biodiversity, facilitating recovery of already damaged areas, protecting breeding stocks, improving recruitment to neighbouring areas, or restocking marine species of commercial interest (Harmelin *et. al.*, 1995).

In Mediterranean, the size of MPAs is generally too small to produce significant increase in the fisheries at regional scales, but higher catch rates around the borders could compensate for the loss fishing grounds for local fishers and may increase local support for MPAs. In Mediterranean sea, it is still unclear whether the export of adults reflect an increase in population size inside the protected area and concomitant density-dependent emigration from the reserve, or simply a re-arrangement of the spatial distribution of the populations.

There is an increase in international publications about MPAs and its effectiveness as management tool to increase fish biodiversity (Fig. 7).

Several studies of coastal fish assemblages have been made in marine reserves of the Mediterranean using visual censuses. In French national park of Port Cros, investigations have focused either on the whole fish community (Harmelin-Vivien and Harmelin, 1975; Harmelin, 1984; 1987) or species of high interest, such as the dusky and *Sciaena umbra* (Harmelin and Marinopoulos, 1993). Other studies conducted in the marine reserves of Banyuls-Cèrbere, of Las Islas Medas (Spain) and of Scandola (Corsica) indicated the effects of protection on fish assemblage (Bell, 1983; Garcia-Rubies and Zabala, 1990; Francour, 1991, Dufour et al., 1995).

Figure 7: number of international publications on MPAs and fish communities.



Along the Italian coast, visual assessment of the coastal fish assemblage has been performed in some areas of high naturalistic interest, such as Gorgona Island (Vacchi et al., 1997a) Torre Guaceto (Marconato et al., 1996), Portofino (Tunesi and Vacchi, 1993; Vacchi and tunesi, 1993), and the Aeolian Islands (Vacchi et al, 1997b) which are under environmental protection, Asinara Islands (Pais et al., 2004). MPAs can offer valuable opportunity to evaluate specific hypothesis concerning effects of fishing on community structure (Jennings and Polunin, 1996; Russ and Alcala, 1998a).

1.2 The biology and ecology of infralittoral fish communities in Mediterranean Sea.

The reserve effect on fish communities have been described and analysed in the Mediterranean Sea (Harmelin-Vivien, 1984; Francour, 1991; Dufour et al., 1995; La Mesa and Vacchi; 1999; Macpherson et al. 2002; Lipej et al. 2003). Fish communities represent one of the most appropriate means of evaluating the reserve effects within more or less protected areas (Vacchi et al., 1997a). The general benefits to the fish communities inhabiting a marine reserve is the restoration of an equilibrated community structure (Harmelin et al, 1995) whit normal proportions of large adults (spawners) and a high-level predators.

The actual distribution of the species within the Mediterranean Sea reflect their biogeographic origins and the different geomorphological, edaphic and hydroclimatic condition found within the Mediterranean.

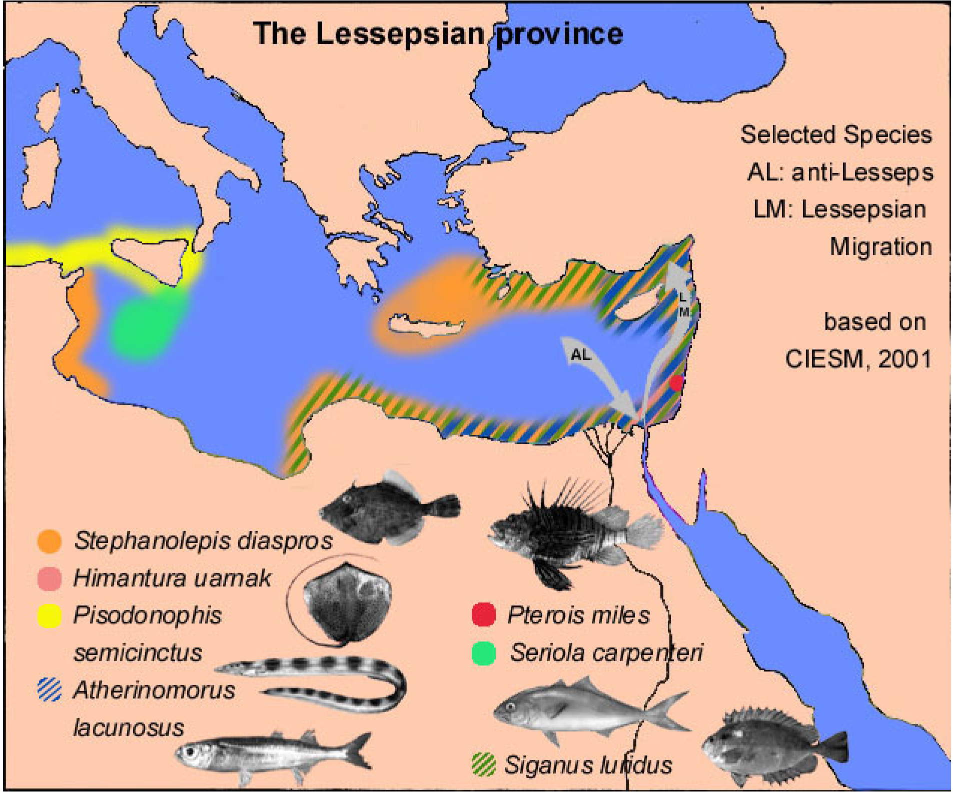
Species coming from the north Atlantic are strictly or preferentially found in the northern areas of the Mediterranean. Saharian and subtropical species are mostly found in the eastern sector, while the ponto-caspic species are far north in the Aegean Sea and Adriatic Sea.

Some characteristics of the Mediterranean fish community are considered: biodiversity decreases from west to east and depth as well due to the “canal effect”, the increasing poverty toward the east and the permanent homothermia.

The FNAM writers (1984-86), estimated the fish biodiversity of the geographical area, located between latitude 30° and 80° north, longitude 30° west and longitude 60° east, including Mediterranean and Black Sea, at 1256 species, including 125 Chondrichthyes. Fish biodiversity in the Mediterranean can be evaluated at 575 Osteichthyes, 86 Chondrichthyes and 3 Cyclostomes; making a total of 664 (Quignard and Tomasini, 2000). The opening of the Suez Canal resulted in a noticeable enrichment of the Mediterranean fish fauna. The lessepsian immigration (Fig 8), is enduring and even intensifying. The majority of these species are found in levantine waters.

The reported divergence in faunal composition between the western and eastern Mediterranean basins appears to be increasing. This may be explained by the reported increase in the migratory flux of the lessepsian fishes (Ben Tuvia and Golani, 1995), and the limit of this migration to the east of the Siculo-Tunisian Strait. Some faunal mixing also has occurred as a result of change in hydroclimatic conditions and of recent habitat modifications (Quignard and Tomasini, 2000). The extension of the distribution area of thermophile fishes and the decreasing number of fish of "boreal type" specie tend to homogenize the Mediterranean fish community, especially around the western area and suggest a "meridionalization" of its northern coast (Riera et al., 1995).

Figure 8: Lessepsian fish species (www.sbg.ac.at/ipk/avstudio/pierofun/)



The role of the Strait of Gibraltar and of its west to east current should not be underestimated. The discovery of Atlantic species in some areas of Mediterranean, may be related to a possible greater incoming flux of Atlantic waters.

Relative to endemism, in Mediterranean itself, endemism was evaluated at 6,9% in the Alboran Sea (Reina-Hervas, 1987), 15 % in the Adriatic, 11% on the Greek sea (Papacostantinou, 1988, 1990 b) and 8% in the Levant Sea (Golani, 1996). 46% of the Mediterranean endemic species belong to the Gobioid family which is numerically dominant in the sea. The other family have a maximum of four endemics each. Considering species eco-biology, endemic are small and the majority of them inhabit the coastal zone. They are benthic or nektonic fishes, and their spawn is frequently fixed on substrate. About 68% of the endemic species are benthic, 29% are nektonic and 3% are pelagic.

The major factors that influence the distributions patterns of fishes, are separated into biotic (mainly composition of the benthos, competition between organisms, predation and recrtuitment), abiotic (mainly environmental considerations such as depth, salinity, temperature and exposure to trade winds), and historical factors (Letourner et al., 2003).

The terms “subtidal fishes” are defined as species which relies utilization of the subtidal zone for completion of all or an essential part of its life history. Such as a definition takes into account the much greater mobility of fishes when compared with other subtidal animals and does not exclude those species which inhabit the subtidal zone only a certain states of the tide or season of the year.

In general the more stable the shore and the more cover, the grater the fish population. On a sandy or muddy shore, the fish population fluctuates greatly, with fish moving in and out with the tides, although some may remain buried in the sand at low water. In common with other organisms, fish are usually scarce or absent on shingle beaches. It is on rocky shores, which provide plenty of cover in the form of crevices, spaces beneath boulders, and rock pools, that fish populations attain their greatest numbers. Fishes which show the greatest adaptations to the changing conditions in the littoral zone are rarely found below low water, and are Blenniidae and the many of the Gobiidae.

Fish show adaptations to shallow water life, which allow them to remain in favourable places in the littoral zone, but whose distribution also extends below low water. The large water movements associated with waves and tides, means that large body size and consequent lack of manoeuvrability would be disadvantageous; hence, shore fish rarely exceed 20-30 cm in length. Their small sizes also enables them to inhabit holes and crevices, giving further protection against turbulence. Consequent upon this mode of life, scales which would be rubbed off in such habitats are often absent (Blenniidae) or very firmly attached (Gobiidae). The copious mucus which many of them secrete may act as a lubricant when moving in confined spaces, although it also been suggested that it may play some role in osmoregulation. The lateral line is also usually reduced, possibly because its usefulness is small in turbulent waters.

Turbulence can also be avoid by keeping close to the bottom where water movement is less. The bottom-living mode of life of most shore fish, is reflected in their body shape which is usually either compressed and somewhat elongate or depressed. The absence or reduction of the swim-bladder also aids this bottom life. Because of the absence or reduction of the swim-bladder, most shore fish keep close to the bottom and perform only short hops and darts from one place to another. They rarely swim for any distance in open water, thereby reducing they chances of being damaged by waves or swept out sea. To enable them to perform this type of locomotion their pectoral fins are large and used as sculls.

The head is normally large with a terminal mouth and often bears tentacles or other appendages. The function of these appendages is for the most part unknown. They eyes are generally large, set forward, and high on the head, giving an extensive visual field anteriorly and dorsally as well laterally.

Fish which are left out of water when the tide recedes face the problem of exposure to air and subsequent desiccation. Exposure to air also means that the availability of oxygen to the fish is changed for the normal respiratory movements passing water over the gills are no longer of use and oxygen must be obtained directly from the air. For all fish, survival seems to depend on keeping the body and particularly the respiratory surfaces, moist so that oxygen exchange can continue.

In the littoral zone of Mediterranean, the fish show one of two basic reaction to waves: they move to places where water movement is smaller, either deeper or by staying in their holes, or they attach the bottom by means of suckers.

The enormous wealth and variety of food available in the littoral zone is paralleled by the large number of feeding types found among the fish that live there, from pure herbivores through omnivores to carnivores. Demersal eggs are laid on the shore attached to stones and boulders, where they are guarded by the male parent. Development takes one to six weeks depending upon the species and the temperature, after which planktonic larvae hatch. The planktonic phase last about two months, during which time the larvae gradually metamorphose into the adult form and finally become benthic. Although the pelagic phase of the life history is shortened by the laying of attached eggs, the attached larvae still have to spend many days in the plankton where survival is generally low.

Growth proceeds rapidly for the first 1-2 years, but gradually slows down with age. Sexual maturity is reached in the first or second year and spawning may occur once or several times each season, usually in the spring and summer. The eggs are carefully deposited on a hard surface in a single layer and adhere to the substratum by means of glutinous adhesive threads. In the Blenniidae an accessory gland is present in the testis which secrets a lipoidal substance whose function has been variously interpreted as causing an increase in the adhesive properties of the egg.

The higher survival rate of eggs may be linked with the relatively low fecundity of shore fish. The reproductive behaviour of shore fish has received much attention. The comparison of several species from different families reveals great similarities in the general patterns of their behaviour and may be regarded as convergent evolution in response to the same selection pressures.

The main cyclic changes which occur in conditions of the littoral zone have a considerable effect upon the fish that live there. The main cycles are, in order of increasing frequency, seasonal, lunar, diurnal, and tidal. The majority of fish which have been investigated shown to exhibit behaviour which is related to a least one of these cycles in the environment (Gibson, 1982).

The most important seasonal cycle seems to be that of temperature, under whose influence several species migrate, presumably to avoid the low temperatures found on the shore in winter.

Such restriction of movement, which occurs in mobile specie, implies that fishes possess a general exploratory tendency, are able to perceive, learn, and remember specific features of the local topography and distinguish them from those in “strange” areas. Once familiarity whit an area has been acquired, it is possible for the fish to move about within it and return to any part with consistency, a process generally named homing. It is possible that fishes have an intimate knowledge of the centre of its area but only recognize a few landmarks at its outer limits. Homing, that is the ability of fish to return to the same small area whether displaced artificially or by they own efforts, has been shown to occur in several species.

Another behaviour pattern which may cause restriction of movement is territoriality. A territory is defined as any defended area and is generally considered to be for feeding or reproductive porpoises. Territory holding in the non reproductive state seems to be relatively rare.

In common with most littoral invertebrates and algae, shore fish are found in limited horizontal zones. This zonation is not usually as well defined as it is in sessile animals and plants, possibly because of the much greater mobility of fish. In each type of habitat there are a vertical limits to the distribution of a species. These vertical limits are usually sharpest on rocky shores where each level possess its own physical and biological characteristics, whereas on sandy shores the whole subtidal habitat is much more uniform.

On rocky shores each species inhabits a fairly well defined level and although the levels of two species may broadly overlap, the portion of the habitat occupied is rarely the same.

Behaviour seems to be the most important factor in determining the habitat occupied, and differences in this behaviour lead to the observable separation of one species typical habitat from that of another. Two other factors may also be of importance. The availability of the right type of food. The fact that most of the necessary food occurs in zones may lead to the limitation of the fish to the level where its prey is most abundant. Secondly, as physical conditions become more variable at higher levels on the shore, the extent to which species can tolerate changes in factors such as temperature, salinity, and turbulence, may govern the upper limit at which they can survive or maintain themselves.

In general littoral fish have well developed colour change mechanisms which enable them to match varying coloured background. This adaptability is necessary for the fish to camouflage itself amongst the patchwork of shades and colours on the shore, and typical tide-pool species have a much greater ability to match their background than species which only enter the pool occasionally. In many species the sexes differ in colouration and during the breeding season this difference is heightened when the males of many species assume specific colour patterns associated with their reproductive behaviour.

Fishes inhabiting the littoral rocky shores are not only valuable elements of coastal biodiversity, but they also exert an important ecological role in the functioning of littoral ecosystems (La Mesa *et al.,*2004). Fish fauna distributions patterns can be useful for testing the effectiveness of habitat protection in marine reserves (Harmelin et al. 1995; Agardy, 2000; Guidetti, 2002). At small spatial scale, the structure of rocky fish assemblages, is know to be ecologically linked to certain features of their habitat such as substrate complexity, availability of shelters and resources, algal and macrobenthic rocky cover, and other substrate variables (Letourner et al, 1003).

The relationship between fish assemblages and habitat has been found both in tropical coral reef (Sale, 1991; Chabanet et al., 1997; Adjeroud et al., 1998) and temperate waters (Falcon et al., 1996; Gillanders and Kingsford, 1998).

1.3 Trophic cascades in MPAs

Fishing may have both direct and indirect effects on benthic community structure. The fishing activities in an unfished area leads to reduction in target stocks and changes the growth, production and recruitment of the target species (Pinnegar et al., 2000). Subtidal rocky reef in temperate waters harbour a variety of fish species which derive food and /or shelter from the reef substratum. Indirectly effect of fishing can be even more consequential for marine ecosystems structure and dynamics than removal of the target species (Botsfrod et al., 1997), because many fishing gears lead to direct degradation of benthic community by removing sessile species that provide a critical structural habitat important in recruitment and prey protection.

Relations between predator-prey governing structure of natural communities (Duffy, 2002). However predators may indirectly influence the entire community through trophic cascades (Paine, 1980; Witman and Dayton, 2001).

Food webs in shallow rocky habitat in the Mediterranean are diverse and complex as coral reefs. Although total species richness in Mediterranean is probably lower than of tropical seas, food-web topology and structure can be as complex as coral reefs. This trophic complexity is associated with the high diversity of Mediterranean benthic communities.

Mediterranean shallow food webs have been exploited for thousands of year (Hughes, 1994), with many coastal fisheries now abandoned or declining. In addition, introduced species have dramatically altered food webs (Bouduresque and Verlaque, 2002).

Most target fishes are high-levels predator and their functional extinction may cause community wide changes (Sala et al., 1998; Jackson et al., 2001). The impact of fishing has consequent top-down perturbations in marine communities structures and organization (Sala et al., 1998). Benthic communities in the shallow Mediterranean rocky sublittoral have 2 extreme alternative successional endpoints: coralline barrens (bare rocky substrata) and complex macroalgal beds (Sala et al., 1998).

In the Mediterranean, *P. lividus* is a key species (Sala et al, 1998) that a high densities influences the dynamics of the phytobenthos, by eliminating erect algae and seagrasses, and inducing the formations of coralline barrens.

In temperate subtidal rocky reef we can identify three trophic levels (Sala et al, 1998; Pinnegar et al., 2000):

1- predators (e.g. fish, lobster, otters);

2- grazers (e.g. sea urchins, some fishes);

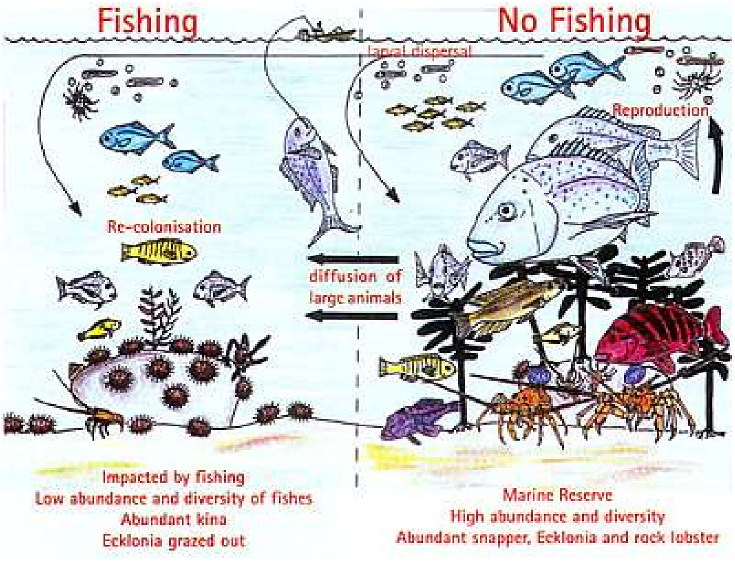
3- macroalgal assemblage (e.g. *Cystoseira* beds).

When the density of sea urchin *Paracentrotus lividus* exceed a threshold of 7 to 9 urchins m^{-2}, a shift between macroalgal assemblages and coralline barrens has been observed to occur (Verlaque, 1987; Hereu, 2004).

Many studies in the Mediterranean rocky littoral have demonstrated that large piscivorous and invertebrate feeding, are more abundant within MPAs compared to sites outside (Francour, 1994; Harmelin et al., 1995), and this is often particularly so for the sparid fishes *D. sargus* and *D. vulgaris* (Bell 1983; Harmelin et al., 1995). These species have been identified as the most effective predators of sea urchins, with *D. vulgaris* being the most important grazer in rocky reefs (Sala et al., 2008). Large sized of the wrasses *Coris julis* and *Thalassoma pavo* (preying upon sea urchins < 1 cm in diameter) are also predators. Sea breams are targeted by many types of fishery, while wrasses have a low commercial value and are fished only as by catch in area where angling and traps were intensively used.

Even though the food web of subtidal rocky reef include complex and multiple trophic interactions, the link between predatory fish, sea urchins and macroalgae, has been found to be the path through which fishing may affect the overall community (Sala et al., 2008) (Fig.9).

Figure 9: trophic cascades in MPAs and unprotected areas.



It has been suggested that the transition from macroalgal beds, that harbour hundred of species of algae and invertebrates, to barrens, dominated by sea urchins and a few species of encrusting algae, can enhanced by the removal of large predatory fish that feed on sea urchins, and that marine reserves may helps the recovery of predatory fish and thus potentially re-establish their predatory control on sea urchin population (Sala et al., 2008; Guidetti, 2006).

When released from predation control, sea urchins may increase in density and overfeed on macroalgae, which in turn may cause the transition from macroalgal beds to barrens (Sala et al., 1998). Since the recovery of sea breams *D.vulgaris* and *D. sargus* (and other predator fish) was observed within reserves, along with lower urchin density and less extended barrens (Guidetti and Sala, 2007).

If marine reserves cause a recovery of predatory fish densities, sea urchins density are reduced and we would expect reserves to harbour macroalgal beds. On the contrary, when population size of predatory is no enough to control population size of sea urchins, we would expect these areas to harbour coralline barrens.

The effect of predator density on prey and on its population parameter is defined as density mediated indirect interactions (DMIIs)

That once a high population of sea urchins has developed, recovery of those fish that eat algae might be slowed or impossible because their food resources drop below the minimum threshold necessary for in situ population development (McClahanan and Sala, 1997).

1.4 **Aims of the Ph.D. Thesis**

The number of studies on the ecology of temperate reef fish communities has increased exponentially in the last twenty years. Although the body of information does not yet match that for their tropical counterparts, a consideration of the status of temperate reef fish ecology has been necessary.

The difficulties in measuring reserve effectiveness should be overcome by appropriate sampling designs, distinguishing between the influence of management and the intrinsic natural variability of ecological systems due to factors other than protection. Comparing protected with unprotected areas, should provide a better understanding of the effects of human activities in coastal marine environments, above and beyond the natural variation that is likely to exist among apparently similar habitats.

In spite of the increasing number of MPAs, no general evaluations have been done to assess the ecological responses to protection from fishing. The effects of protecting marine areas, have been studied extensively and the studies have shown the benefits and limits of MPAs. Despite ecologists' increasing interest in fish fauna living in marine protected areas, fish assemblage of some Italian marine reserves are still completely unknown.

Pianosa Island benefits from a condition of total protection, due to which any unauthorized activity is prohibited within one nautical mile from the coast. Such protection, enforced in 1858 following the establishment of the penitentiary, continues still today with the inclusion of the island in the Arcipelago Toscano National Park, and has actually prevented any significant type of human activity for a long period. Therefore, oppositely to the case of most MPAs, for which protective measures follow periods of intense exploitation, the coastal waters of Pianosa represents a pristine habitat that has not experienced significant modifications in time. Although not regarded as a MPA, the island can be virtually considered as a total protection zone of a MPA.

This aspect results in being particularly important in the evaluation of the parameters concerning the fish population. Therefore Pianosa represents the ideal area where to verify the hypothesis that is the subject of this Ph.D.

The comparison with a nearby area (Elba Island), where no protection measures are enforced and where fishing activities take place, is extremely important in evaluating the answers to the working hypothesis.

In Mediterranean Sea, assessment of community-wide changes following protection results in contradictory analysis. In some case, no cascading effects on prey were significantly found in differences in predatory fish abundance and size (Guidetti, 2006). Other study showed recovery of fish within reserve, but effect on the benthic assemblages were evident only in some cases. Sala et al., (1998) stated the importance of further research aimed at understanding the mechanisms underlying the realization of trophic cascades in temperate rocky reef.

Protection from fishing can directly restore population of target fishes and indirectly drive whole communities towards an unfished state (Sala et al., 1998; Guidetti, 2006).

Stronger and more widespread indirect effects on protection may become apparent in reserves protected for sufficient durations of time to allow build-up to top predator abundances and biomass. Build of top predators may results in major changes in community structure through cascading trophic interations (Sala et al., 1998) in older reserves or sites protected for decades.

No take area as Pianosa Island, represent an opportunity to test for cascading effect of predator removal at spatial and temporal scales (Sala et al., 2008; Shears and Babcock, 2002).

Most of the studies concerning the reserve effect of MPAs on fish communities are based on the study of population parameters. While maintaining these parameters as the base for evaluating the reserve effect, the working hypothesis for this Ph.D. is to verify if the establishment of the MPA has determined a reestablishment of predatory interactions, that might be altered in the control area outside the MPA.

In such sense, any change in the parameters of fish populations in the marine protected area compared to the control area could be ascribed to the reestablishment of predatory interactions.

Moreover, this study assessing the possible differences between protected and unprotected rocky reef analyzing distribution patterns of predatory fish, sea urchins, barrens and macroalgal beds.

I examined the community wide responses to protection in the form of trophic cascades in the case of fish, sea urchins, and barrens. Even though there is empirical evidence that abundant predatory fish population can regulate sea urchin populations little is know about the relationship between fish and sea urchin abundance in area as Pianosa were fishing is absent for a long time.

Critical questions are: pattern distribution of invertebrate feeders, sea urchins and barrens, supporting trophic cascade model in rocky littoral ecosystems? Does the community responds to protection in the form of trophic cascades?

In conclusion, the Ph.D. thesis is aimed at:

1) determine the structure of the infalittoral fish assemblage inhabiting the protected (Pianosa Island) and adjacent (Elba Island) non protected areas of Tuscan Archipelago, in terms of species composition, diversity and relative density, assessing differences in the community parameters in different areas;

2) to estimate the fish community responses to the trophic cascades in shallow rocky shores as possible results that indicate the reserve effect.

2. MATERIALS AND METHODS

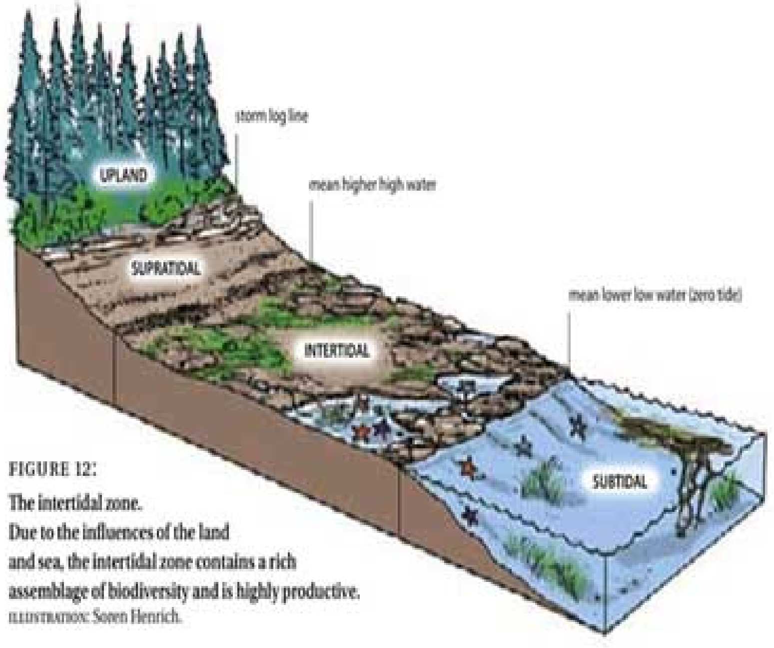
2.1 The shallow rocky habitats of Mediterranean Sea

The marine coastal zone is important to many littoral fish assemblages consisting of several components:

- resident species (Hilldén, 1984)
- juvenile fish, which utilize shallows as a nursery (Zijlstra, 1972);
- pelagic and demersal species which temporarily migrate to, and forage in coastal areas (Pihl, 1982).

Infralittoral zone is defined as the region of shallow water closest to the shore. The lowest zone is the part of the shore exposed only at the lowest tides. It is constituted from the Infralittoral Fringe and Infralittoral - (Sublittoral) - zones. Because of the longer submersion period, organisms here do not have to be as sturdy and adaptive. This is why more organisms make this zone their home. A variety of algae grow here like surf grass and sea palm and provide shelter for many small animals who find plentiful food in this zone. This area (Fig. 2.1), includes the sub tide zone and is normally underwater. The organisms here can only tolerate slight exposure from the air or sun.

Figure 2.1: infralittoral zone in Mediterranean Sea.



The biocenosis of infralittoral algae is found on more or less well lit rocky substrates. The fauna is rich and includes various crustaceans and various molluscs. The Corallinales form, with the basal part of the calcareous thalluses and with the aid of the sessile gastropod, *Vermetus triquetus*, and the sedentary polychaete *Spirobranchus polytrema,* a very important basal concretion. Calcareous concretions are also due to the gastropod *Dendropoma petraeum* and to the serpulid polychetes *Serpula vermicularis*. The Mediterranean mean depth of infralittoral zone is 35 m. Fish assemblages in Mediterranean infrallittoral rocky habitats are typically composed of several tens of species, and the fish species pool is almost identical throughout the western Mediterranean basin and Adriatic Sea (Fischer et al., 1987).

The shallowest zone (0-6m) of the Mediterranean rocky littoral displays high biotope heterogeneity with alternation of patches boulders, pebbles, rocky flats, grave, and sand., with a biotic cover dominated by macrophytes. More important algae in this zone are *Cystoseira* spp.

Many species of the littoral fish communities settle this habitat with precise specie-specific microhabitat requirements (Garcia Rubes and Macpherson, 1995; Zazetta, 1996).

In unprotected areas, frequent disturbances caused directly or indirectly by man (e.g. pollution, littoral encroachment, aggressive fishing, excessive grazing, introduction of invasive species) affect this narrow belt, leading to habitat destruction and severe depletion of the capacity of sites to house settlers. The disappearance of the three-dimensional algal cover, such as that of *Cystoseira* spp. and the associated invertebrate fauna, has negative consequences for settlement success through depletion of both shelter and food resources.

The shallow benthic (infralittoral zone) Mediterranean includes three main biocenoses:

- sand flats;
- seagrass beds
- rocks.

The latter can be further divided into rocky bottoms, with variously sized blocks and into rocky slopes, where the concretion of algae and other organisms with calcareous skeletons create a high structural complexity.

The study of littoral ichthyofauna on rocky bottoms has increased markedly due to the development of direct observation techniques. In particular, the investigations carried out rocky bottom, have provided much information about the influence of some ecological factors.

The composition and structure of the fish communities are as much mediated by their relationships with the physical environment as by biotic factors such as, habitat preference, competition, predation, recruitment or post-settlement mortality.

Habitat structure affect both adult and juvenile fishes through its effect on processes such as proportionating refuge against predation, fishing pressure, hydrodinamics, light. Increase in structural complexity, causes an increase in diversification of resources and finally and increase in species richness (Fig. 2.2).

Figure 2.2: infralitoral substratum complexity.

Among physical factors, nature and complexity of substrate, habitat complexity, shelter and refuge availability, nutrient supply, hydrodynamism, and depth have an important influence on fish assemblages (Milicich, 1994; Letoruner, 1996).

Hydrological factors, such as temperature, can also be significant, not only for their effect on the distribution of the species, but also for their influence on life history, population dynamics and synecological relations (Sarà, 1985).

The physical heterogeneity, implies a strongly patchy distribution of the benthic assemblages, and fish richness and diversity are directly related to this environmental "rugosity" (Luckhurst and Luckhurst, 1978).

Shallow benthic communities are high primary production areas with substantial secondary production, so they are able to maintain the richest littoral fish assemblages (Garcia-Rubies and Zabala, 1990). The exposed sub-emergent rocky reef were covered by a photophilic algae community overgrazed by sea urchins *Arbacia lixula* and *Paracentrotus lividus.*

In the shallowest areas, strong differences of direct human predation pressure above and beyond 10 m depth should be expected. Historical factor, can also play a significant role in structuring fish communities in infralittoral zone.

Here is possible assess that structure of rocky fish assemblage testing the effectiveness of habitat protection in MPAs.

2.2 Study area

Study on fish communities of Tuscan Archipelago National Park, are limited at some islands and locations.

A preliminary study of the fish fauna of the Gorgona Island, the northernmost of the Tuscan Archipelago, was carried out in 1989 by Vacchi et al. (1997a). Some measures of protection were in force along the coast of the Island, due to the presence of a penitentiary. Diurnal and nocturnal strip-transect were performed at different sites characterized by rocky bottom and patches *P. oceanica.* As species richness was concerned, Labridae and Sparidae were the dominant families. Some species heavily fished were represented by large and very large size specimen.

A survey on the fish assemblage of Meloria bank, located at 7 km off Livorno coast, was conducted in 1996 (Biagi et al., 1998). The investigated fish assemblage covers 31 fish species, a lot of which belonging to Labridae and Sparidae.

The coastal fish assemblage of Capraia Island, was investigated in June 1994 (Santangelo et al., 1996). During this study, 46 fish species were identified. Labridae and Sparidae were the most relevant families in relation to species richness. Some large sized specimens, belonging to species of high commercial value (*i.e. Seriola dumerilili*, *Dentex dentex*, and *E. marginatus*), were also observed.

In 1995 a preliminary study to describe fish assemblage was carried out at the locality “Le Ghiaie” (Biagi et al., 1997), the only marine area of Elba Island where some protection measures on fish fauna are in force (fishing prohibition). During the survey 46 fish species, belonging to 15 families were recorded. In terms of species number, Labridae and Sparidae were the most important families.

The literature on the fish community of Pianosa Island (Central Thyrrenian Sea) is scarce, despite its inclusion inside the National Park of Archipelago Toscano.

The present study was conducted in Pianosa island and Elba island, both located inside the National Park of Tuscan Archipelago, established in 1996 (Fig. 2.3 and 2.4). For administrative purposes, Pianosa forms part of the Elba island municipality. The attention of this studies was directed towards fish communities of the infralittoral rocky zone. Each sampling site within and outside the protected area, was comparable in habitat and rugosity. The bottoms of these sites was formed by largely rocky plates with relatively few shelters.

Figure 2.3: the national Park of Archipelago Toscano.

Figure 2.4: Pianosa island

Pianosa is formally protected since 152 years, first penal settlement, and high security prison up today. At present, its population is only made of convicts and guards. All kinds of fishing have been prohibited within the boundaries since institution of penitentiary in 1858. Pianosa is the island nearest to Elba (Tab 2.1).

Table 2.1:geographic localization of Pianosa.

Poisition	***Lat Nord 42°34' - Long Est 10° 05'***
Area	***Kmq 10,25***
Perimeter	***26 km***
Height	***29 mt.***
Distance from Elba	***7 mg***
Distance from Montecristo	***17,5 mg***
Distance from continent	***21,8 mg***
Distance from Corse	***21,5 mg***

Pianosa Island is a small portion of a ridge submarine emerged (a few miles W of Monte Cristo) stretches with NS direction and closing a few miles of the island of Capraia. This structure, which is named Pianosa Ridge, divides the Tyrrhenian basin here in two parts: one between the dorsal and Corsica, which reaches a depth of 800m and that between the ridge and the continent with a maximum depth 400m.

Pianosa consists of two formations: the Formation of Marina del Marchese (Miocene) and Pianosa Formation (middle Pliocene). The first one, warped and bent, outcrops along the cliff erosion in the NW coast and consists mainly of marls with interbedded sandstone and rarely as conglomerate (maximum size of pebbles 1-5 cm), the emerged portion is of 463m. Based on studies of the Foraminifers training has been referred to the Burdigalian and in part to the lower portion Lang, it would be unsettled in an environment of open sea and fairly deep. Formation of Pianosa (thickness of 25-30m), discordant on the Marina del Marchese, the island is almost completely. It was deposited in shallow water and is ready to form am anticline with the direction of EW shortening. The main rock consists of organogenic calcarenites and fossil sands (Echinoids, bryozoans, molluscs, brachiopods and fish).

The depths of Pianosa are shallow and deepen gently, the bathymetry of 50 m is reached on average at about 1500 m from the coast.

Pianosa is the only Tuscan island composed of sedimentary rocks. Thanks to its calcareous nature and to its level ground it has been cultivated since the ancient times. The neighbouring Elba island is more heterogeneous and geologically complex.

Shoreline contour, bottom topography and exposure to dominant wave directions are more or less the same in each site. The eastern the southern and the western coast of Pianosa are exposed to hydrodynamism. On these coasts, a steep rocky bottom with large boulders are predominantly at 0-40 m depth range and are replaced by sand in deeper water. The northern coast is less exposed do the closeness of some rocks that reaching some meters of height. The benthic community of the areas are composed mainly of rocky boulders of diverse sizes interspersed with patches of sand and *P. oceanica.*

A preliminary survey was made in April 2007 in both islands, to choose zones with similar physical substrate complexity, in order to minimize variation of this factor, which have large influence on the structure of fish communities. For the proposal of the study were choose 5 areas with the same spatial and morphological features. Three of these areas were located in Pianosa and two in Elba. The experimental design adopted is illustrated as follows:

- 5 Areas (3 in protected area of Pianosa, and 2 in unprotected area of Elba);
- at each area were choose 3 sites;
- at each sites were conducted one strip transect.

Three areas were selected in Pianosa island to lead information of fish community. The first was located near the little harbour of the island and is called “Cala Giovanna” (Fig. 2.5). This area, located to the east side of the island, is constituted by hard bottom, sand and *P. oceanica*.

Figure 2.5: sampling site of Cala Giovanna.

The second site located in Pianosa is called "Cala Ruta" (Fig. 2.6) and is placed to the southwest side of the island. The bottom has the same characteristics of the other chosen sites.

Figure 2.6: sampling site of "Cala Ruta".

Third site, is located to the extreme north and it is named “Porto Romano” (Fig. 2.7). As the other two stations, this one is formed by hard rocky bottom and with some patches of sand and *P. oceanica.*

Figure 2.7: the northern site of “Porto Romano”.

Sampling area used as reference point to evaluate differences in fish assessment, were chosen at Elba island in respect to some considerations. As the island is the closest to Pianosa, is best suited to the comparison of the characteristics of coastal fish populations.

Furthermore, the shape of the coastline in some places is very similar to what is observed in Pianosa.

This allowed to select two areas, showing the same characteristics as those for the study of fish fauna in Pianosa, or rocky bottoms with rocks of medium to large size areas where they may be alternating between sea grass and sand.

The depth has similarities with the study areas also, and it gradually increase from the coast.

The effects of protection on fish populations should be assessed only on equivalent bottom carefully selected for similar biota, rugosity and depth (Garcia-Rubies and Zabala, 1990).

Sites located in Elba Island are two. The first called “Procchio” (Fig. 2.8) is placed on south side of the island. The second is sited on to the opposite, north side, and is called “Galenzana” (Fig. 2.9).

Figure 2.8: site of Procchio.

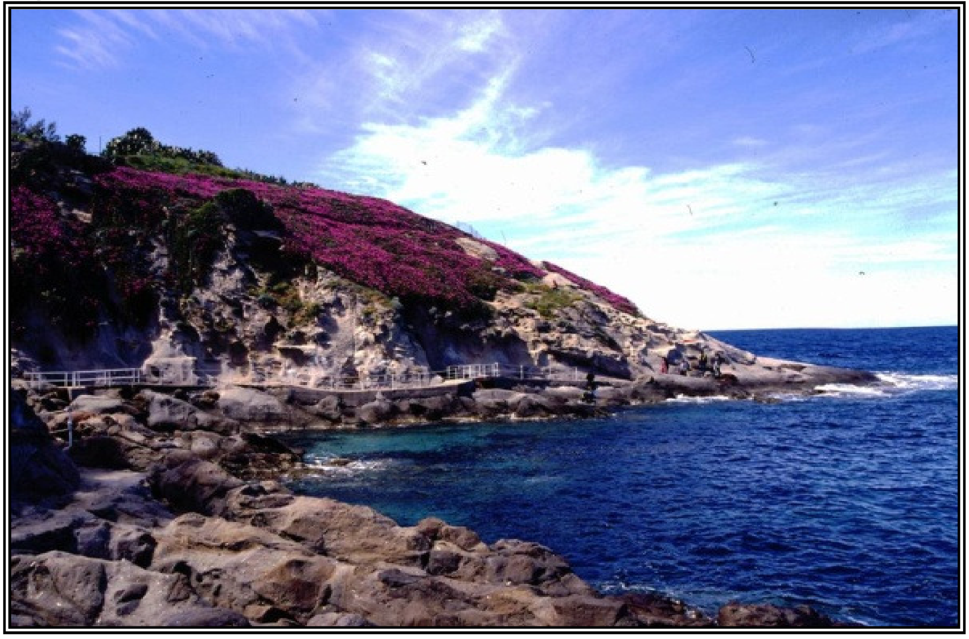
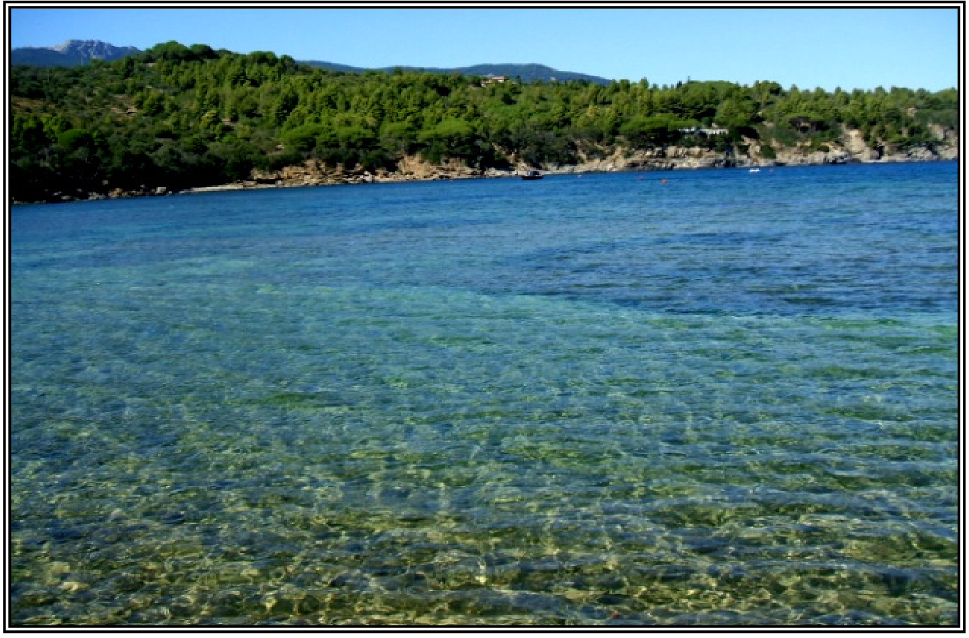


Figure 2.9: location of “Galenzana”.



2.3 Underwater visual census and sampling methods

Study on the fishes assemblages along Italian coasts have been performed by means of visual census in some marine protected areas, as well as in area not yet protected, but where the protection or the recovery of the environmental integrity needed.

UVC method to assess fish have been widely employed in Mediterranean Sea to investigate the association between fish community and habitats (Harmelin, 1987; Guidetti, 2000), the social organization and reproductive patterns (De Girolamo et al., 1999; Verginella et al., 1999), human impacts (Guidetti et al., 2003), distribution paterns of juvenile fishes (Vigliola er al., 1998), study of fish community in MPAs (Vacchi et al., 1998; Garcia-Charton et al, 2004). Only in a few cases stationary point have been employed (Vacchi and Tunesi, 1993; Francour, 1994), where the observer remains stationary in the centre of a circular area of preestablished radius while recording species, number and size of fishes.

Ecologists have used underwater visual census (UVC) techniques for recording fish densities on reef since 1950s (Brock, 1954; Barans and Bortone, 1983; Harmelin-Vivien et al, 1985).

The accuracy of UVC has frequently been questioned (Andrew and Mapstone, 1987). Several source of bias have been identified, such as: the failure of an observer to notice individuals, the presence of observer, observer experience, observer speed, and fish detectability (Lincoln Smith, 1988). Moreover, UVC can significantly underestimate cryptic fish density (Brock, 1982; Kulbicki, 1990; Ackerman and Bellwood, 2000). Census techniques routinely used to detect highly mobile or easily observable species, will likely miss or significantly underestimate morphological or behavioural cryptic species that are usually hidden among intricate substrata. Both readily observable and cryptic fishes have complex ecological and behavioural interrelationships, and human and natural events that alter coral reefs impact the entire fish community. By supplementing data obtained from studies of visible fauna with collections of cryptic organisms, a more complete assessment can be achieved.

As a relative measure of fish abundance a biased visual survey is not a problem if the bias remain constant. Since the method has several advantages, notably being non-destructive and relatively quick to execute.

The strip transect is one of the most frequently used visual survey methods. This technique involve a diver swimming a measured distance along the bottom while counting fish within a fixed width. The density of fish is the number counted within the defined area of the transect, expressed per unit area.

The study of fish assemblages in MPAs must based on non-distruptive sampling methods which do not alter the environmental integrity ensured by the protection regime.

Visual census method permit a community assessment without removal of organisms, conversely by other traditional techniques (*i.e.*, sampling by trawls and dredges) (Bortone and Kimmel, 1991), and it can be used both on heterogeneous rocky bottom, such as coral and artificial reef, that sand bottom.

Strip transect (Harmelin-Vivien et al., 1985) were used to collect qualitative and quantitative data in the Mediterranean Sea (Bell, 1983; Fasola et al., 1997; Garcia Charton & Pèrez Rufaza, 1998; Garcia-Rubies & Zabala, 1990; Harmelin, 1987; Harmelin et al., 1995; Marconato et al., 1996) and it has been previously used to study fish communities in similar Mediterranean sites (Mazzoldi and De Girolamo, 1998).

For the mentioned arguments, UVC is choised as the more appropriate method to conduct study on infralittoral fish community in Pianosa and Elba islands.

Fish sampling design by means of UVC is illustrated in table 2.2.

Table 2.2: fish sample design adopted.

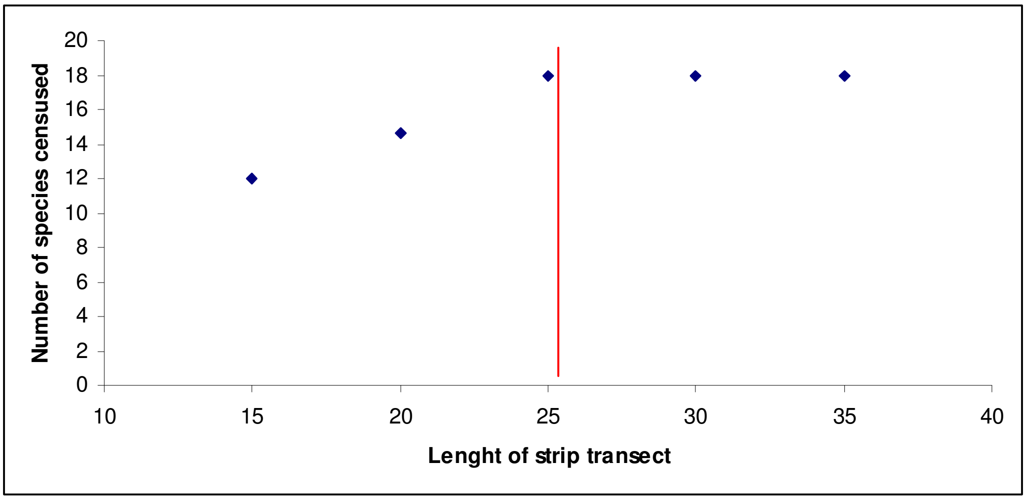
FISHING SAMPLE DESIGN		
Time	***2007 2009 2010***	
Area	***ELBA ISLAND***	***PIANOSA ISLAND***
Site	***Galenzana (GA) Procchio (PH)***	***C. Giovanna (CG) P. Romano (PR) C. Ruta (CR)***
Depth	***0-4 m***	
Transect each site	***18***	
Total Transect	***90***	

Sampling were conducted from June 2007 to September 2010. A forced stop to the activities was induced during 2008. In each site were conducted three strip transect randomly chosen, for a total of 90 visual counts.

Sampling, in a experimental area chosen during training, were made during the cold season, and in order to minimize environmental parameters such as water transparency and surface weather, sampling were carried in the morning and with good sea-whether conditions.

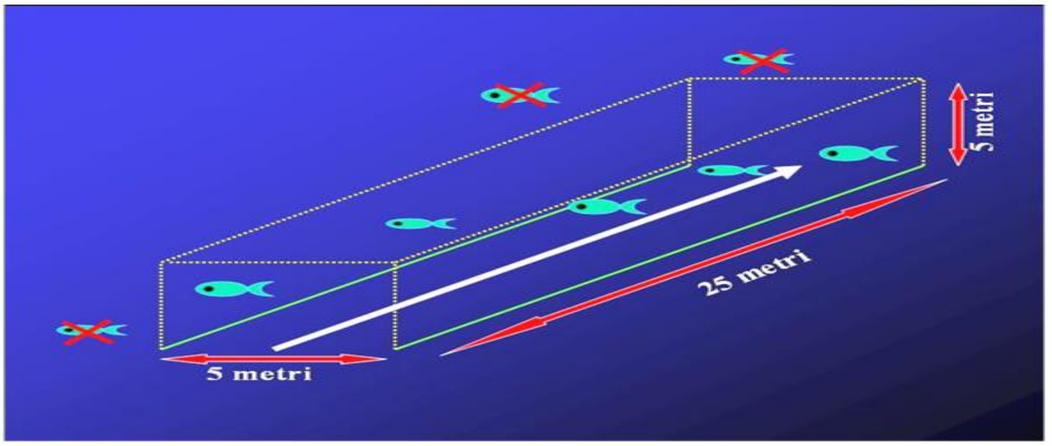
The appropriate length of transects was estimated by means of cumulative number of species (Fig. 2.10) obtained counting number of species at progressive transept length. Number of surveyed species is roughly steady at length of 25 meters. Transects lengths were chosen in order to include homogeneous habitat in all sites considered.

Figure2.10: cumulative number of species at measured length of strip transect.



that were performed in snorkelling, were 25 m long and 5 m wide (total surface area 125 m^{-2}) (Fig 2.11). The depth of transects ranged between 0 and 4 meters.

Figure 2.11: strip transect pattern used in fish visual census.



To allow the organism to re-establish their normal activities, surveys was conducted after waiting at least 15 minutes after the transect had been set. An interval at least 15 minutes was also used, during successive surveys along the same transect during experimental trials. Survey were conducted when the visibility was greater than 10 m. In situ data were recorded by pencil on PVC slates. To take censuses of the fishes, the diver swimming slowly along the transect (3-4 m min^{-1}) depending on substratum complexity. Sampling time long the transect were ranged between 30 and 60 minutes. Transect were at minimum distance of 50 m form each other to avoid spatial dependence in the density estimate of fish.

As a descriptor of substratum complexity was used rugosity, which was estimated as the ratio *L/l*, where *L* is the actual distance between two points measured whit a rope contoured over the bottom and *l* is the linear distance between such points (Luckurst and Luckurst, 1978; La Mesa et al., 2004).

Abundance data were collected by using pre-established discrete scale based on a geometric progression factor which was approximately 2:

1, 2-5/, 6-10/ 11-30/, 31-50/, 51-100/, 101-200/, 201-500, >500. Fish density was calculated by considering the mid point of each abundant class (Harmelin-Vivien et al., 1985).

The size (total length) of individual fish was assigned to one of four size classes (small, medium, large, very large) each corresponding to one quarter of the recorded minimum total length of the species (Fischer et al., 1987). In addition, small individuals of some species were recorded as juveniles on the basis of morphological features (*e.g.* the colour pattern).

Experimental tests were made previously in training activity, in order to avoid errors of size assessment using a shape of fish with predefined total length.

Algal stature was evaluated as (1) encrusting; (2) medium, up to 10-cm height; or (3) tall, > 10 cm.

Extensions of barren cover, was visually estimated as relative percentage by means of square of 1 m^{-2} located on the bottom. At each transect, the square was randomly located at two point of investigated area. The percentage extension of barren valuated at each square, was fitted to 2 m^{-2} and was considered as the value of % barren extension of the transect.

About its importance as key species in Mediterranean, *P. lividus* was chosen as indicator liable to possible shift from macroalgal cover to barren. During each of 90 strip transect were recorded the density of *P. lividus* and this was successively evaluated in relation to density of fishes such as *Diplodus* spp.

Each fish species was assigned to 1 of 7 trophic groups using the available information about diet in the FishBase database (www.fishbase.org) and in Mediterranean studies (Sala, 2004). To evaluate the working hypothesis on the existence of trophic cascades due to the protection in protected area, fish were classified as.

large piscivores (LP); Small piscivores (SP), Invertebrate feeders Type 1 (major predators of sea urchins) (INF1); invertebrate feeders Type 2 (whose diets seldom include sea urchins) (INF2); detritivores (DET); planktivores (PLA); and herbivores (HER).

I split the invertebrate feeders into two groups because the major role a few fish species can have in regulating sea urchins populations and hence in affecting the entire community (Sala et al., 1998; Hereu et al.,, 2004, Guidetti, 2006, Guidetti and Sala, 2007). In order to avoid that total densities of *P. lividus* was influenced by juvenile stages of sea urchins that are not palatable for invertebrate feeders 1, sea urchins less than 1 cm of diameter were excluded from evaluation of response ratio.

During the survey the degree of exploitation on fish species at Elba island, was evaluated on the basis of direct observation of fishing landings in Marina di Campo harbour. Landings catch of artisanal fishery using trammel and longline net, was evaluated by means of direct observations in spring and summer 2007, 2009, 2010: a total of 32 censuses were made.

On the basis of qualitative assessment of landing composition, I have defined three exploitation level: non target species (NT), lightly fished (LF), heavily fished (HF).

2.4 Data analysis

Composition and structure of assemblage structure were specified by calculating number of species (*S*), number of individual per area (*N*), Shannon-Wiener (*H*') diversity index as: $H' = -\sum p_i \log_2 p_i$; where p_i is the proportion of each species, and Evenness index (*J'*): $J' = H/H_{max}$.

The descriptions of habitat structure was carried out using the mean, and standard deviation.

A one way ANOVA was performed to test for differences among the areas in the assemblage parameters (*S, H'*, and *J*), total density and relative density of the dominant species. Homogeneity of variances was tested by Cochran's test; whenever necessary data were transformed and re-tested (Underwood, 1997). In a few cases, transformation did not produce homogeneous variance, despite this ANOVA was used after setting $\alpha = 0.01$. This was done to compensate for the increased likelihood of type 1 error (Underwood, 1997).

Post hoc multiple comparison after ANOVA were made by post hoc Snk test whit STATISTICA 6 ® software package (Statsoft, 2007). Permanova (Permutational Multivariate Analysis of Variance) was performed to test differences in analysis fish community using locality as fixed factor (Anderson and Willis, 2003).

I explored possible general responses of fish assemblages to reserve in the year in the five sites of both islands. This was made using a constrained ordination named canonical analysis of principal coordinates (CAP Anderson and Willis, 2003). CAP can uncover patterns that are masked in unconstrained ordination and fix axes that maximise the degree of correlation between a set of predictor and response variables. Here the relative importance of sites in influencing fish assemblages was assessed by virtue of the strength of their individual correlations with the canonical axes. Analyses were conducted at species level. The canonical correlation were tested using 4999 random permutations.

Density of some commercial species and invertebrate feeders, sea urchins and barrens extension, was calculated in the areas, and differences tested using χ^2 test and ANOVA respectively. In calculations of barrens cover data were Arcsin transformed.

The reserve effect (response ratio, R), was calculated for trophic categories and exploitation levels, as ln of ratio between value of response variable (density) inside the protected area and in fished areas (Olkin, 1985, Micheli et al., 2004). Positive response ratio indicate greater density within reserves than in references conditions, whereas negative values are indicative of greater abundances in reference conditions compared to reserves.

3 RESULTS

3.1 Fish assemblage

During the three year of observations a total of 46 six species belonging to 15 family were observed (table 3.1).

Table 3.1: fish species censused within the five sites: GA (Galenzana), PH (Procchio), PR (Porto Romano), CG (Cala Giovanna), CR (Cala Ruta).

Family	Species	Zone
Apogonidae	*Apogon imberbis* (Linnaeus, 1758)	all sites
Atherinidae	*Atherina* spp. Linnaeus 1758	all sites
Blennidae	*Parablennius gattorugine* (Brünnich, 1768)	GA,PR,PH
	Parablennius pilicornis (Cuvier, 1829)	GA,PH
	Parablennius rouxi (Cocco, 1833)	CG,PR
	Parablennius sanguinolentus (Pallas, 1811)	PH
	Parablennius tentacularis (Brünnich, 1768)	GA,PH
	Parablennius zvonimiri (Kolombatovic, 1892)	GA,PH
Gobidae	*Gobius bucchichi* (Steindachner, 1870)	GA,CG,PR,PH
	Gobius cobitis (Pallas, 1870)	GA,CG,PH,CR
	Gobius couchi (Miller & El-Tawil, 1974)	GA
	Gobius geniporus (Valenciennes, 1837)	GA,PH
	Gobius paganellus (Linnaeus, 1758)	GA,PH
Labridae	*Coris julis* (Linnaeus, 1758)	all sites
	Labrus merula (Linnaeus, 1758)	all sites
	Labrus viridis (Linnaeus, 1758)	all sites
	Symphodus cinereus (Bonaterre, 1788)	GA,CG,PH,CR
	Symphodus mediterraneus (Linnaeus, 1758)	GA,PR,PH,CR
	Symphodus melanocercus (Risso, 1810)	GA,PR,PH,CR
	Symphodus melops (Linnaeus, 1758)	GA, PH
	Symphodus ocellatus (Linnaeus, 1758)	all sites
	Symphodus roissali (Risso, 1810)	all sites
	Symphodus rostratus (Bloch, 1797)	all sites
	Symphodus tinca (Linnaeus, 1758)	all sites
	Thalassoma pavo (Linnaeus, 1758)	all sites
	Xyrichthys novacula (Linnaeus, 1758)	CG,PR
Moronidae	*Dicentrarchus labrax* (Linnaeus, 1758)	PH
Mugilidae		all sites
Mullidae	*Mullus barbatus* (Linnaeus, 1758)	all sites
Muraenidae	*Murena helena* (Linnaeus, 1758)	GA,PR,PH
Pomacentridae	*Chromis chromis* (Linnaeus, 1758)	all sites
Sciaenidae	*Sciaena umbra* (Linnaeus, 1758)	CG,PR,CR
Scorpaenidae	*Scorpaena porcus* (Linnaeus, 1758)	all sites
Serranidae	*Epinephelus marginatus* (Lowe, 1834)	CG,PR,PH,CR
	Serranus scriba (Linnaeus, 1758)	all sites
Sparidae	*Dentex dentex* (Linnaeus, 1758)	CG,PR,CR
	Diplodus annularis (Linnaeus, 1758)	all sites
	Diplodus puntazzo (Cetti, 1777)	CG,PR,PH,CR
	Diplodus sargus (Linnaeus, 1758)	all sites
	Diplodus vulgaris (Geoffroy Saint-Hilaire, 1817)	all sites

	Lithognathus mormyrus (Linnaeus, 1758)	CG,PR,PH
	Oblada melanura (Linnaeus, 1758)	all sites
	Sarpa salpa (Linnaeus, 1758)	all sites
	Sparus aurata (Linnaeus, 1758)	CG,PR,PH,CR
	Spondyliosoma cantharus (Linnaeus, 1758)	GA,CG,PH
Tripterygiidae	*Tripterygion delaisi* (Cadenat & Blache, 1971)	GA,CG,PR,PH
	Tripterygion tripteronotus (risso, 1826)	CG,PH,CR

The families with a higher number of species were wrasses (Labridae, with 12 species, which represent 26% of species richness) and sea breams (Sparidae, with 10 species). Other families with several species were Gobidae and Blennidae.

The mean and standard deviation were calculated at each area for number of species (*S*), number of individual per area (*N*), Shannon-Wiener index diversity (*S*), evenness index (*J*), are illustrated in table 3.2

Table 3.2: mean values and standard deviation of parameters of fish community measured in the five considered area.

	S	***N***	***H'***	***J'***
PH	**14,3 ± 3,2**	**190,5 ±167,3**	**2,3 ± 0,8**	**0,6 ± 0,2**
GA	**13,9 ± 2,2**	**117,5 ± 112,9**	**2,7 ± 0,7**	**0,7 ± 0,2**
CG	**14,9 ± 2,7**	**93,5 ± 31,9**	**3,13 ± 0,3**	**0,81 ± 0,1**
PR	**14 ± 2,4**	**95,7 ± 27,1**	**2,99 ± 0,4**	**0,8 ± 0,1**
CR	**13,5 ± 2,5**	**105,3 ± 57,3**	**2,9 ± 0,5**	**0,78 ± 0,1**

Abundance was 120,5 (s.d. 100,3) individual per transect; this parameter varied between 602 and 32 individuals per transect. Only one species could be considered abundant, with a mean abundance greater than 20 individual per transect; this was the case of *Atherina* spp. that form large shoals living throughout the water column. Some species are less common with mean abundance between 3 and 7 individual per transect. This is the case of *Mugilidae* spp., white sea bream *D. sargus*, common two banded sea bream *D. vulgaris*, salema *Sarpa salpa*, saddled sea bream *Oblada melanura*, East Atlantic peacock wrasse *Shymphodus tinca*, ornate wrasse *Thalassoma pavo*, Mediterranean rainbow wrasse *Coris Julis*, damselfish *Chromis chromis*. Most species are rare, appearing only in some transect, such as pearly razorfish *Xyrichtys novacula*, Mediterranean moray *Muraena helena*, European seabass *Dicentrarchus labrax*.

All the rest of species showed a narrower distribution being recorded only in some transect, especially cryptobenthic fish as Blennidae and Gobidae, are surveyed only in a few number of transect.

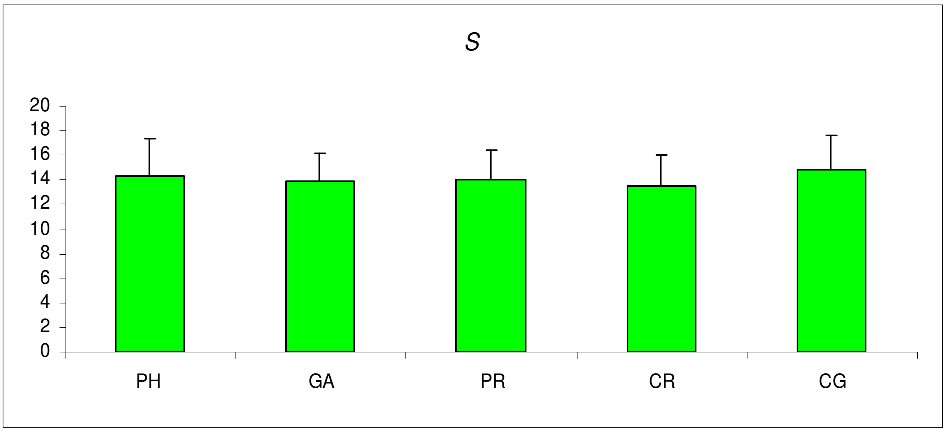
Relevant, from its importance in terms of conservation of biodiversity is the presence of an individual of *Gobius couchi*. Known distribution of Couch's goby, (Miller and El-Tawil, 1974) is formerly limited to the British Isles, and appears to be extended to the Mediterranean. A specimen was collected at the island of Ischia (Stefanni and Mazzoldi, 1999).

Total number of species is certainly affected from “occasional” species as *Lythognathus mormyrus* as their presence on rocky substrate accidental.

Differences in community parameter between the five areas investigate were tested using analyses of Variance (ANOVA), and post hoc SNK test.

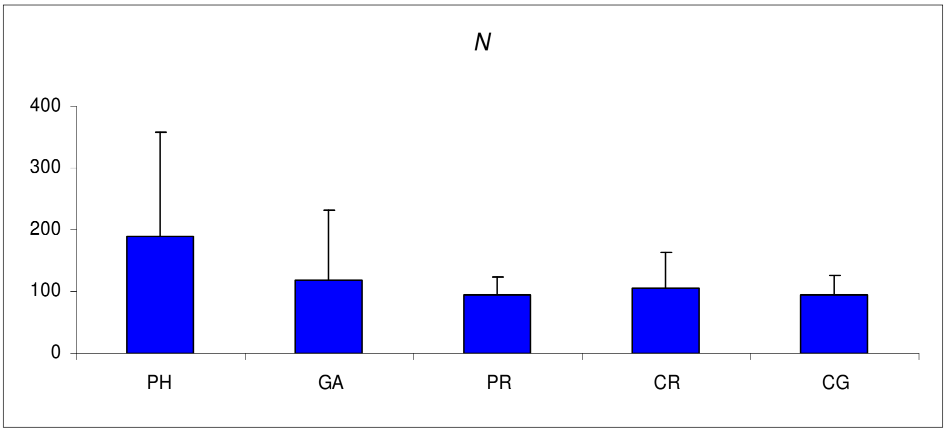
Number of species (*S*), don’t show any significant difference between sites (*P*>0.05) (Fig.3.1).

Figure 3.1: ANOVA testing differences in number of species between sites



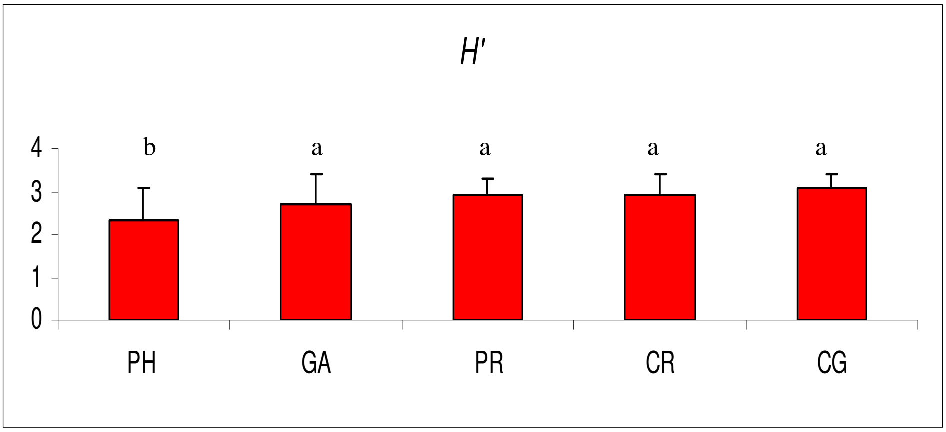
The number of individuals per area don’t differs from the five sites. Here is needed to consider that Cochran’s test on homogeneity of variance is significant. For this reasons differences highlighted if figure 3.2 are considered not significant to avoid making an type 1 error.

Figure 3.2: ANOVA testing differences in number of individuals between sites.



The Shannon-Wiener index diversity (*H'*), differs significantly between sites. Procchio site (PH) located in Elba island has mean value of *H'* lower than other four sites ($P< 0.05$) (Fig. 3.3).

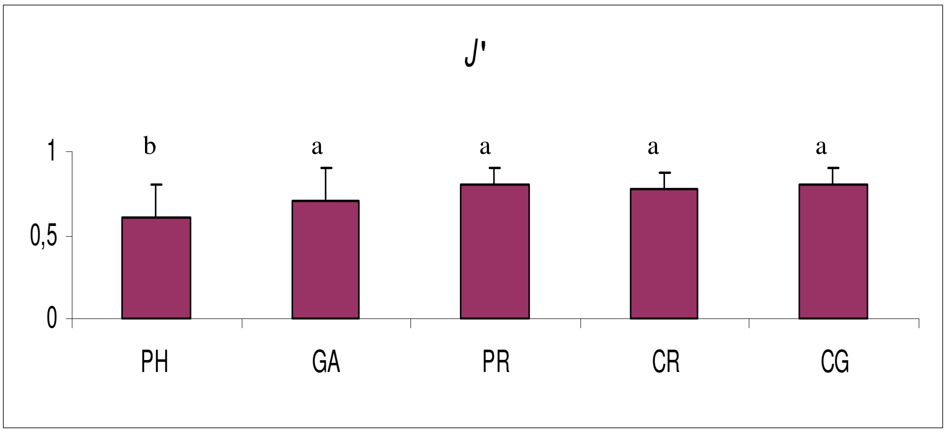
Figure 3.3: ANOVA testing differences in Shannon-Wiener index between sites.



The other four sites, located both in Pianosa and in Elba, showed roughly value.

The same assessment is made testing variation of Evenness index between the considered sites. This index is different ($P<0.05$) between PH and the other sites investigated (Fig 3.4).

Figure 3.4: ANOVA testing differences in evenness index between sites.



Eveness index in Procchio is lower than the other four sites which are similar and statistical comparison did non show significant differences between these.

Permutational Multivariate Analysis of Variance allowed comparisons between fish communities among different sampling sites by means of similarity matrix based on the species recorded and on abundance values. Abundance data were transformed to log + 1 so that each species contributed fairly evenly to the analysis (Tab. 3.3).

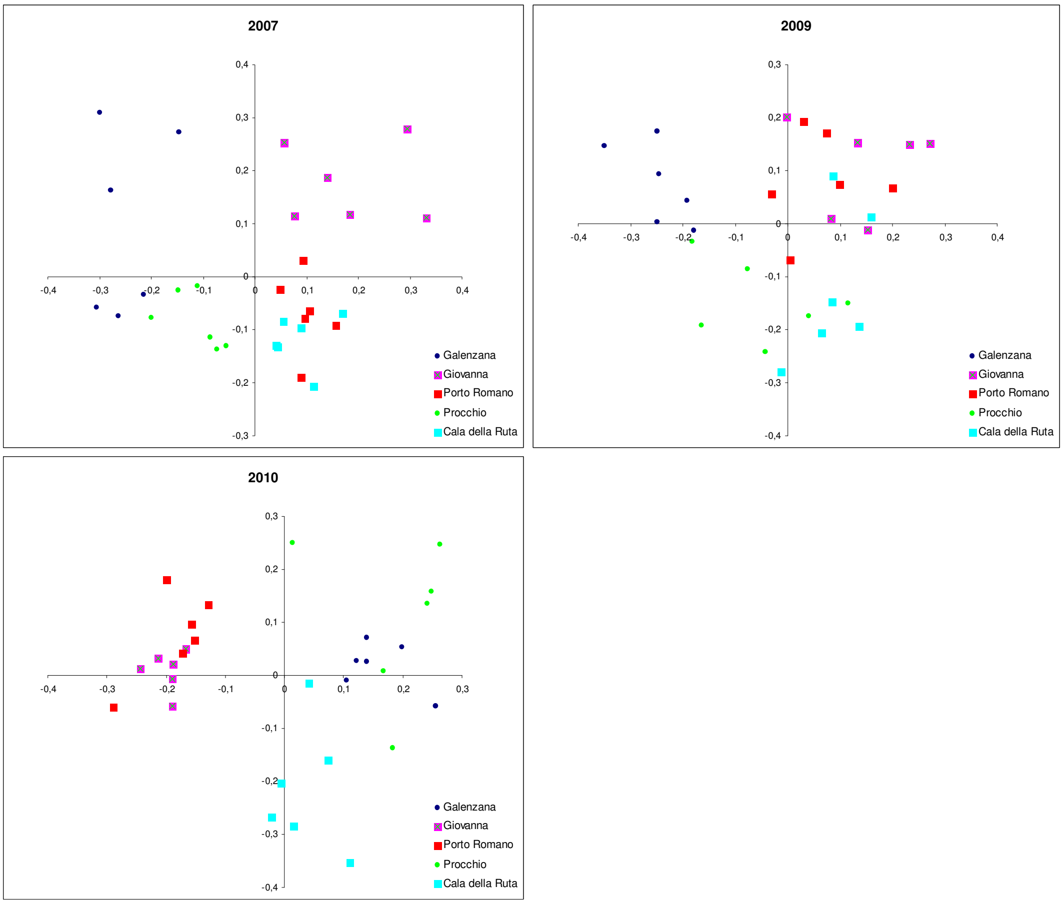
Table 3.3: Permanova analysis

Pair wise Tests		
Locality		
Groups	**t**	***P* (perm)**
GA-CG	3,1624	0,0002
GA-PR	2,6762	0,0778*
GA-CR	2,4411	0,0002
GA-PR	1,4663	0,0292
CG-PH	2,8462	0,0002
CG-CR	2,3588	0,0002
PR-PH	2,3569	0,0002
PR-CR	1,5982	0,0116
PH-CR	1,9492	0,0004

Galenzana and Procchio, both from Elba Island, were the only sites which did not show significant differences, whereas all other sites within the study area showed significant differences.

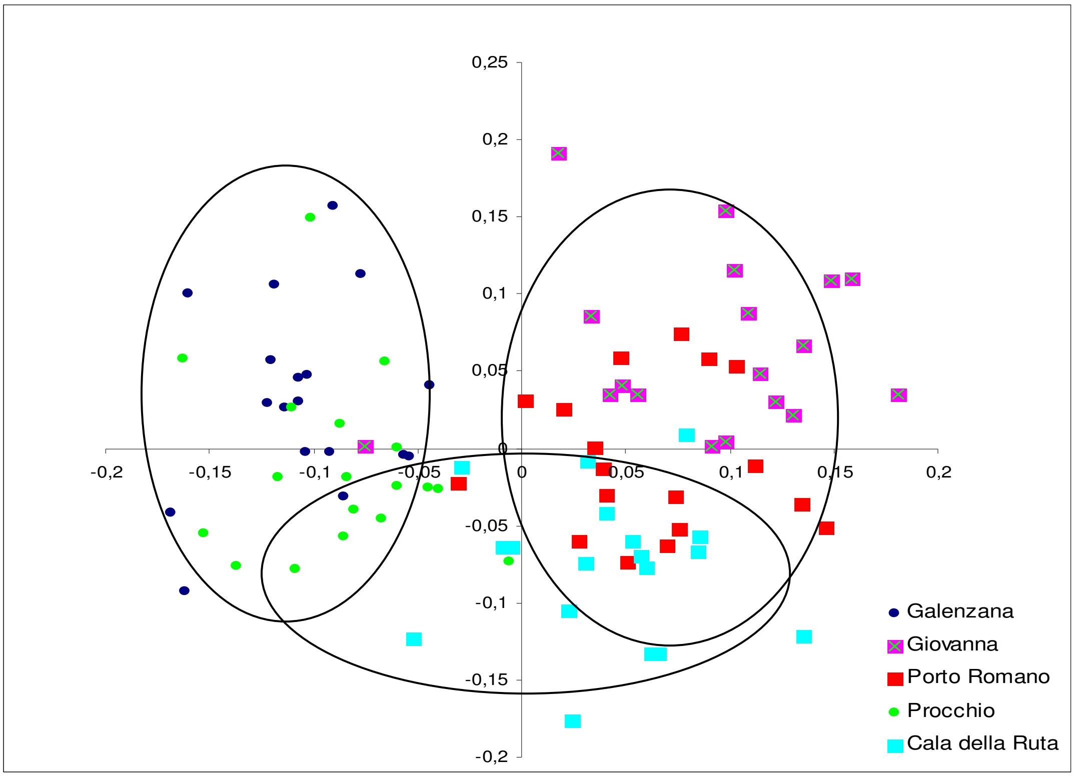
The canonical correlation analysis using CAP was made in the three years of study. (Fig 3.5). Significant differences among sites were recorded in the three year sampling period (square correlation δ^2= 0,86, *P*= 0,0002; δ^2= 0,75, *P*= 0,02; δ^2=0,89, *P*= 0,0002).

Figure 3.5: Canonical correlation ordination (CAP) of fish assemblage data in the three years. Each point refers to a single site.



The same analysis was carried out taking into consideration the whole sampling period. Also in this case, the plot of canonical axes displayed a spatial separation, which indicate significant differences among areas (Fig 3.6). (square correlation δ^2= 0,75, *P*= 0,0002).

Figure 3.6: Canonical correlation ordination (CAP) of fish assemblage data of total year.

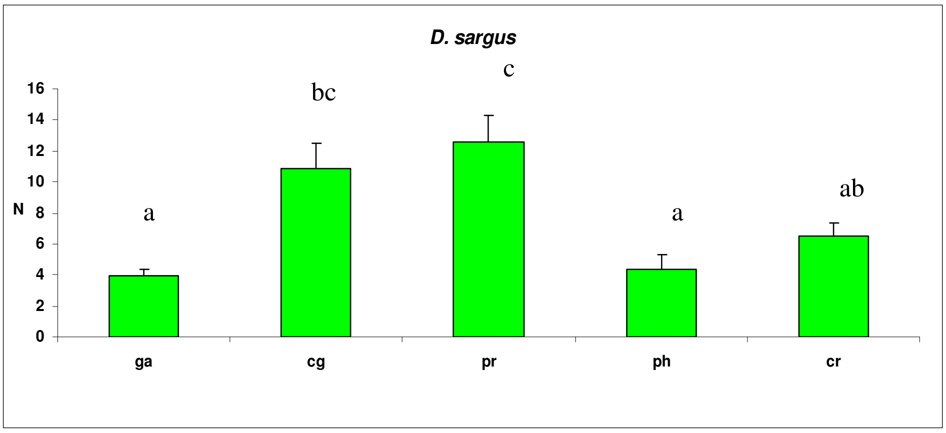


Results of CAP analysis show a clear pattern of variation, especially when the whole sampling period is taken into consideration. In particular, sites GA and PH, from Elba Island, result partially overlapping and localized in the left part of the graph in correspondence of the negative values of axis 1. Two sites from Pianosa, CG and PR, show a similar pattern being both localized on the opposite side of the graph. Although it is grouped with the other two sites in the protected area, the third site from Pianosa, CR, overlaps slightly also with PH (Elba).

In order to evaluate the responses to protection from unprotected (Pianosa) with respect to protected (Elba) areas, the frequency of distribution of the most represented species was analyzed. Differences between mean values and their correspondent standard deviations, were tested by means of ANOVA.

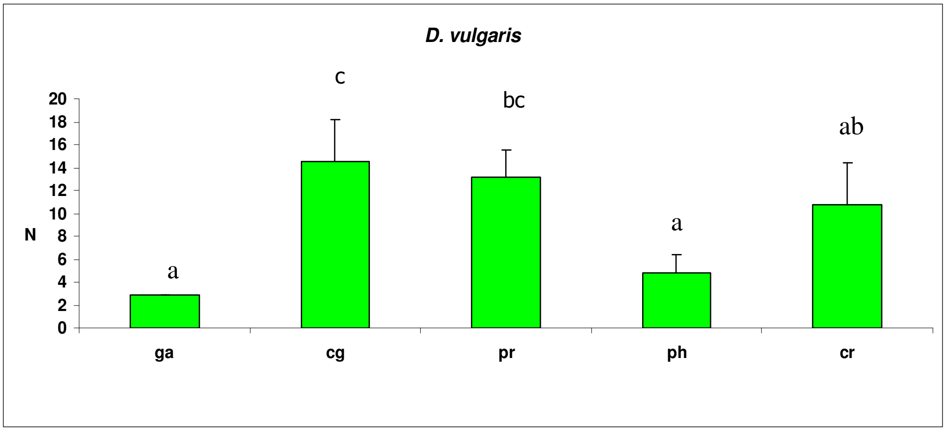
With regards to its distribution, *D. sargus* (Fig. 3.7) shows significant differences ($P>$ 0.05). Results of post hoc (SNK test) reveals similarities between the sites GA and PH (Elba Island), partial overlapping between CR and GI (Pianosa), and also of the latter site with PH (Elba Island).

Figure 3.7: *D. sargus* frequency of distribution between sites.



As with *D. sargus*, also *D. vulgaris* is considered a species of great commercial interest. Significant differences (*P*<0.05) were recorded for this last species with regards to its frequency of distribution among sites (Fig. 3.8). *D. sargus* and *D. vulgaris* show a similar pattern for sites located at Pianosa, however, partial overlapping between PH (Elba) and CR (Pianosa) was observed for the latter species.

Figure 3.8 : *D. vulgaris* frequency of distribution between sites.



The frequency of distribution of either *C. julis* or *T. pavo* (Fig 3.9) and (Fig. 3.10) among sites, do not show significant differences (*P*>0,05).

Figure 3.9: *C. julis* frequency of distribution between sites.

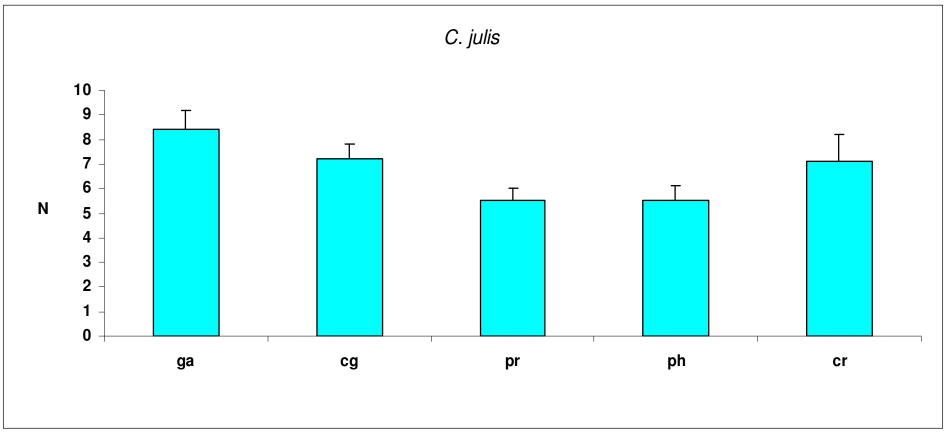
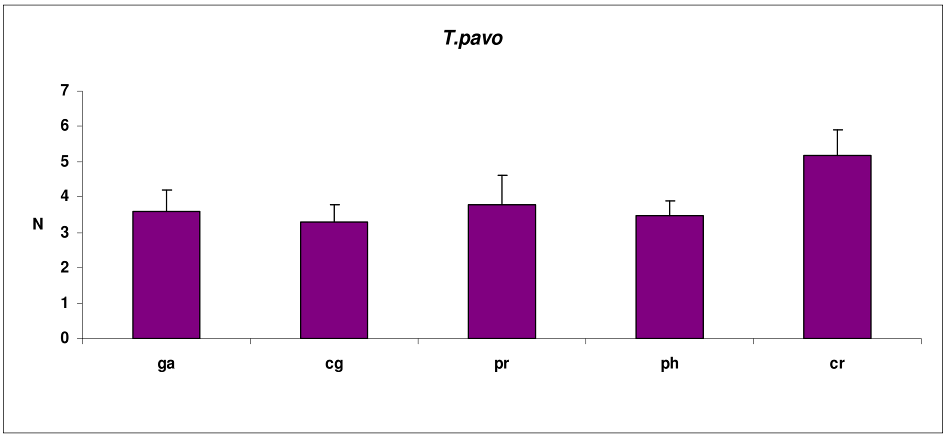


Figure 3.10: *T. pavo* frequency of distribution between sites.



Either one of these species are considered of commercial interest.

S. scriba and *S. tinca* are two benthic species of little commercial value in the study area. They do not show significant differences (*P*>0,05) in their distribution between sites from Elba and Pianosa (Fig. 3.11 and 3.12).

Although it cannot be properly considered as a target species, *O. melanura* represents in the study area an accessory capture of certain commercial value. Differences in its frequency of distribution among sites shows significant differences (*P*<0.05), exactly as demonstrated for commercial species such as *D. sargus* and *D. vulgaris* (Fig.3.13).

The main differences in terms of mean abundance were recorded between sites located in the unfished area and in the ones from Elba (GI=PR=CR>PH=GA).

Figure 3.11: *S. scriba* frequency of distribution between sites.

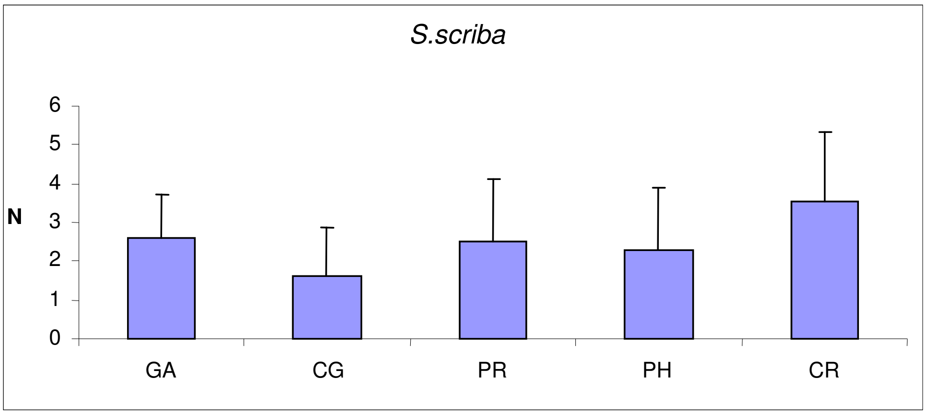


Figure 3.12: *S. tinca* frequency of distribution between sites.

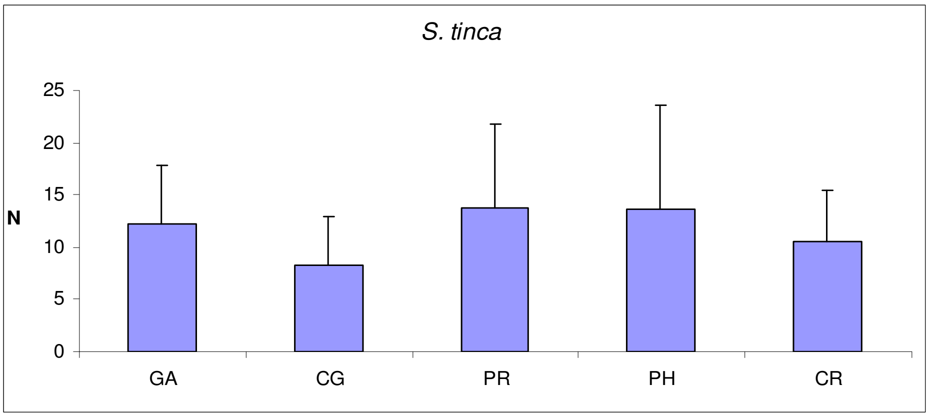
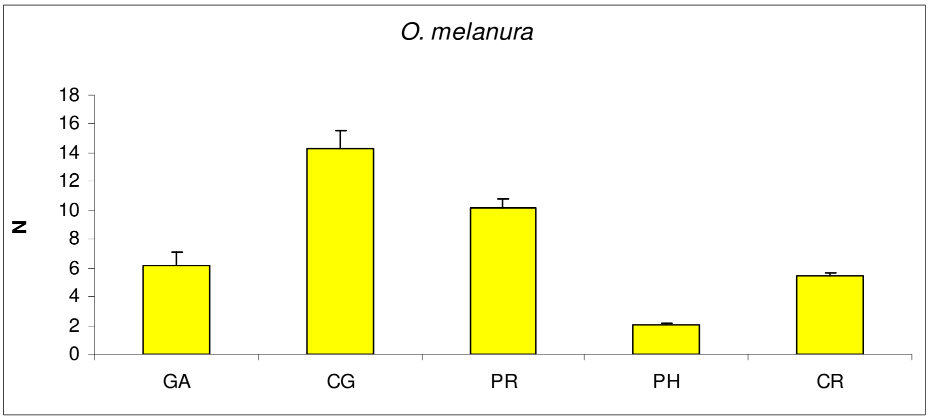


Figure 3.13: *O. melanura* frequency of distribution between sites.

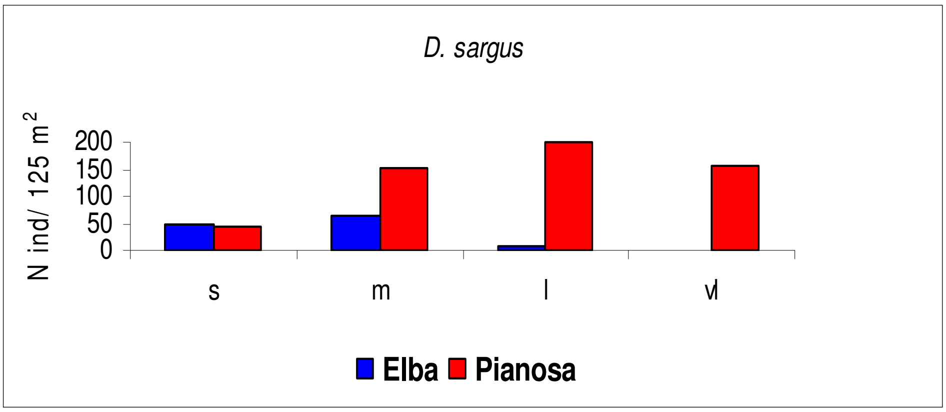


The analysis of frequency of distribution is helpful for understanding how the recorded species are distributed in the sampling sites.

Size structure (i.e. small, medium, large, very large) of the most represented species from unfished (Pianosa) and fished (Elba) areas was studied in order to evaluate ontogenetic changes in habitat use. The analysis was carried out either for species of commercial interest or for species without any commercial value. Comparisons were made between species densities within the two areas in relation to their size, the hypothesis was tested by means of χ^2.

Differences for *D. sargus* (Fig.3.14) with regards to size structure among sampling sites were significantly high (*P*<0,001).

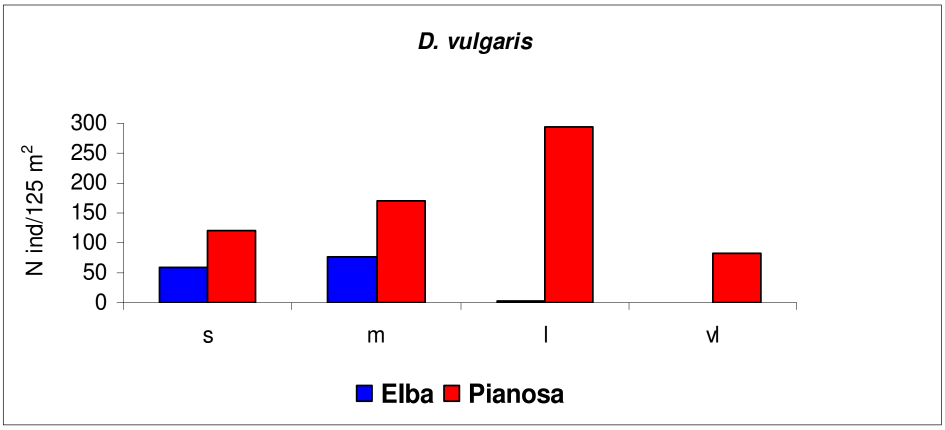
Figure 3.14: sizes distribution between areas of *D. sargus*.



Small size individuals were equally distributed between the two areas, whereas medium size individuals showed an unequal distribution since most individuals were recorded in the unprotected area (Pianosa). Large and very large individuals were recorded almost exclusively from the sites located in Pianosa. Two sites, GA and PH (Elba) show a higher number of medium size individuals with respect to those expected, from chi square value.

Differences in size structure between the two areas were recorded for *D. sargus* as well as for the other commercial species *D. vulgaris* (*P*< 0.01) (Fig.3.15).

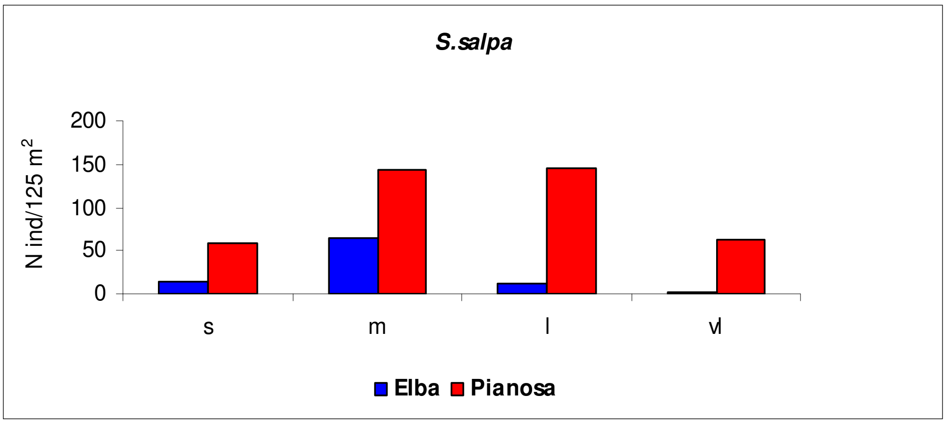
Figure 3.15: sizes distribution between areas of *D. vulgaris.*



This last species shows a higher number of young and medium size individuals in the protected area; large and very large individuals were found exclusively in the protected area. The comparison between expected and recorded values helps to understand differences between the two areas, as in PH were found a higher number of young individuals with respect to those expected and in PR were found a lower number of young individuals with respect to the expected ones. At the same time, in PH were recorded a lower number of large individuals with respect to those expected.

Sarpa salpa is a herbivorous schooling species of little or any commercial value showing a distribution similar to the two preceding species (Fig. 3.16). This species presents all size classes in the unfished area (Pianosa), whereas in the fished area (Elba) medium classes are predominant followed by small and large ones. Comparison between expected and recorded values, shows that Pianosa houses a lesser number of juveniles than expected, which explains the amount of significant difference between the two areas (P<0,01).

Figure 3.16: sizes distribution between areas of *S. salpa.*



Analysis of size structure for some species of little or none commercial interest such as *C. julis* (Fig.3.17) and *T. pavo* (Fig.3.18), does not show significant differences between the two areas.

Figure 3.17: sizes distribution between area of *C. Julis.*

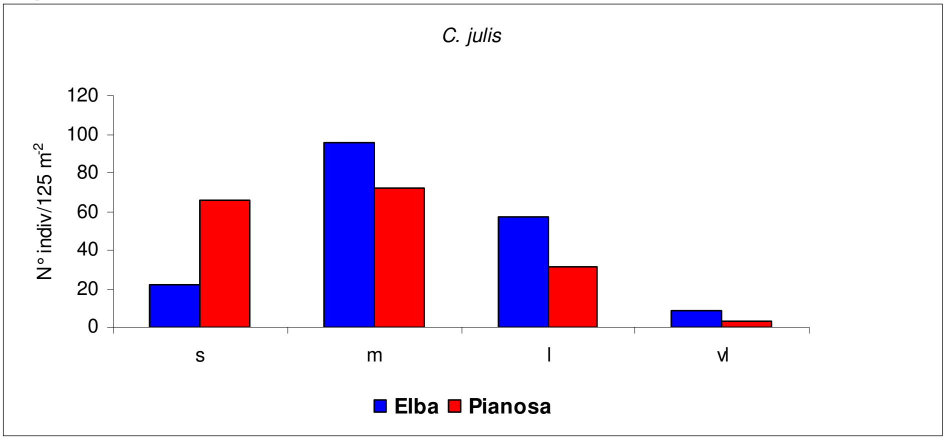
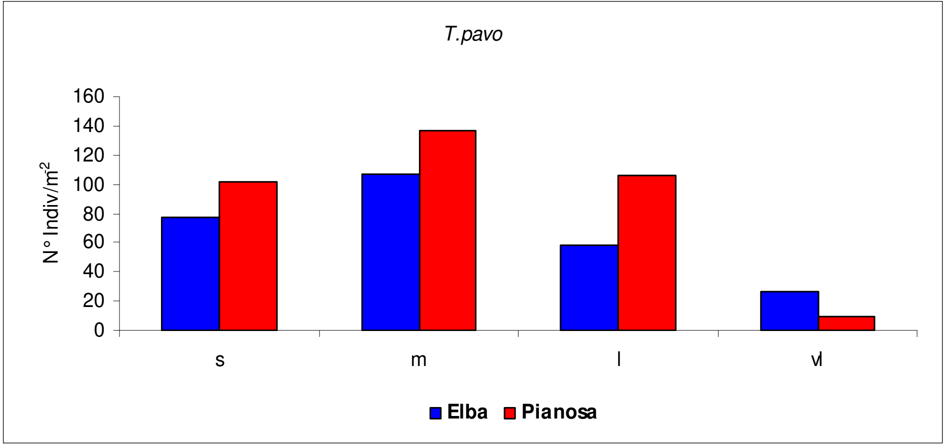
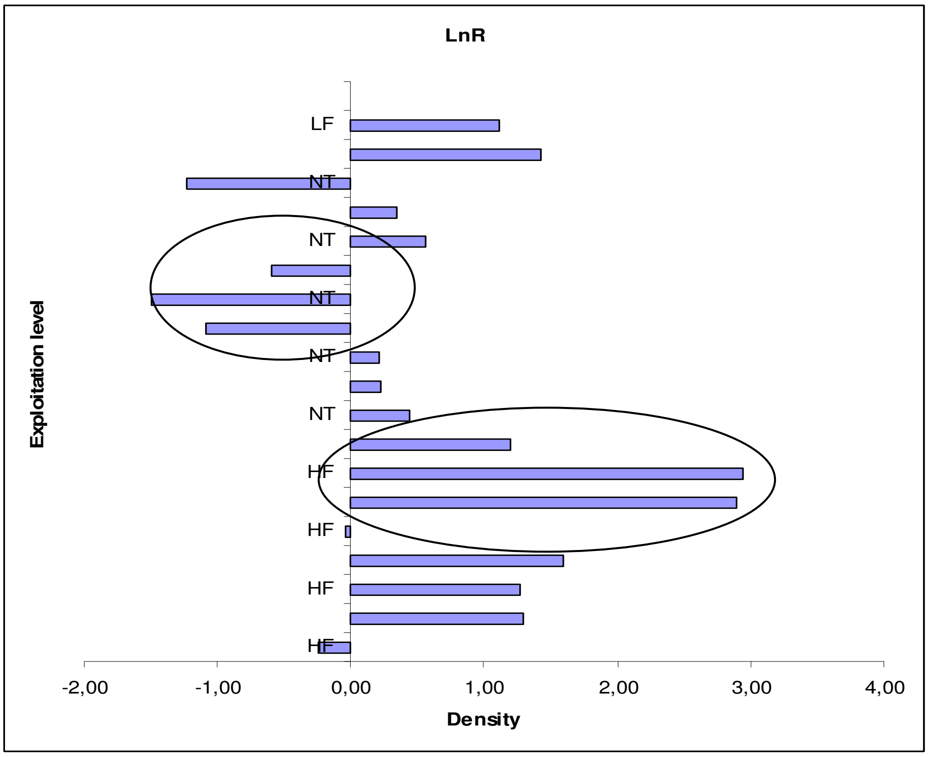


Figure 3.18: sizes distributions between area of *T. pavo*.



The response ratio based on densities of fish and their exploitation level (Fig. 3.19), showed that heavily fished species, benefit of protection more than no target species. This is particularly true for some species as *D. vulgaris, D. sargus, S. umbra, E. marginatus, D. dentex*. As observed in others MPAs Mediterranean rocky bottom, target species can find refuges from fishing activities inside the reserve and increment their population size.

Figure 3.19: response ratio for each of exploitation level.

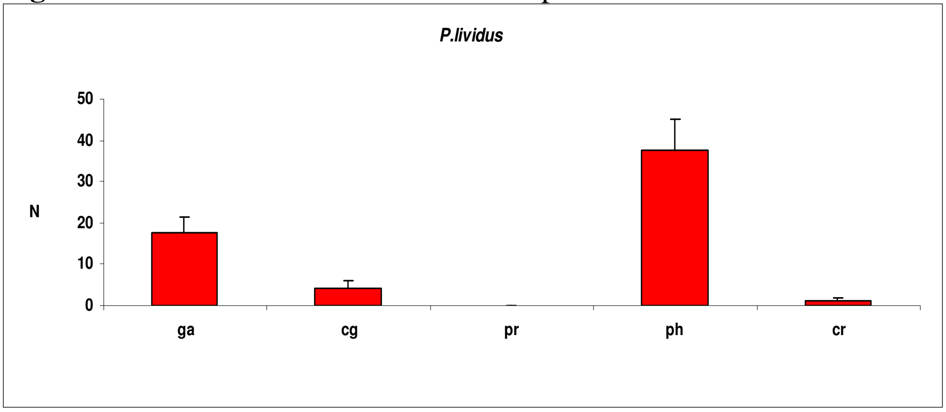


3.2 Evaluation of trophic cascades

Mechanisms causing trophic cascades either inside or outside protected areas may be evaluated by comparing mean densities of invertebrate feeders 1, sea urchins and barrens. In order to evaluate benefits of protection in relation to the recovery of trophic cascades within unfished areas, herbivores and invertebrate feeders 2 must be also taken into consideration.

As previously shown, the densities of predatory species (INF1) such as *D. sargus* and *D. vulgaris* considerably changes between sites located inside the protected area with respect to Elba Island, whereas the densities of species with a lower rate of predation on sea urchins such as *T. pavo* e *S. tinca* (INF2), did not show significant differences among sites and between areas. Therefore, the sea urchin *P. lividus* (Fig.3.20) must be considered the key species for the evaluation of trophic cascades.

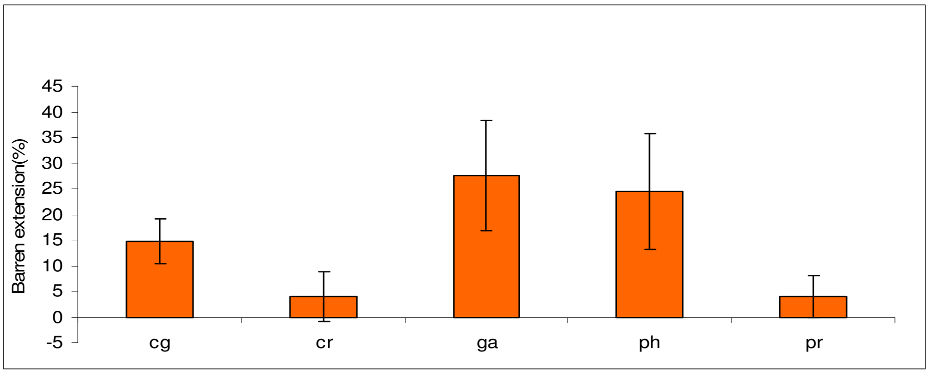
Figure 3.20: densities of *P. lividus* in sampled areas.



Differences in mean densities of *P. lividus* resulted highly significant ($P< 0{,}001$) among sampling sites. The number of sea urchins recorded in GA and PH (Elba) was much higher with respect to the ones found in the three stations from Pianosa.

Since grazing activity by sea urchins may transform macroalgal beds into "barrens", their extension in the two areas was evaluated (Fig.3.21)

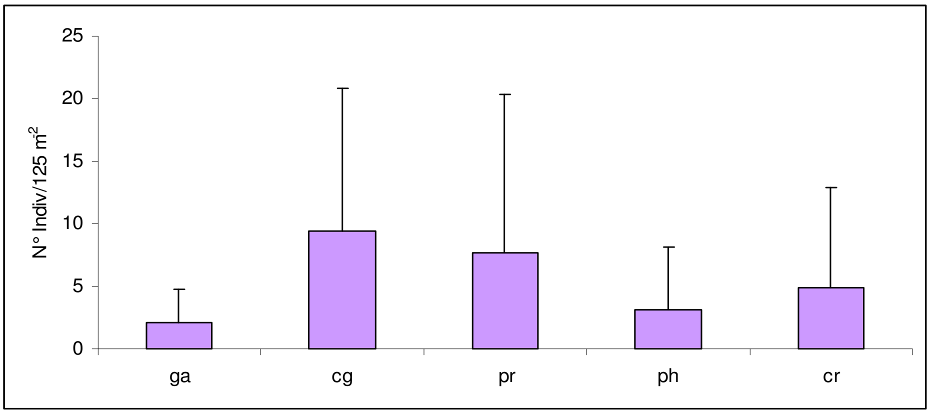
Figure 3.21: extension (mean and standard deviation) of barren in each investigated site.



Differences in the percentage of barren extension resulted highly significant ($P<0{,}0001$). Post-hoc test shows that GA and PH (Elba) have similar values which are considerably higher than CG as well as CR, PR, which in turn show similar values.

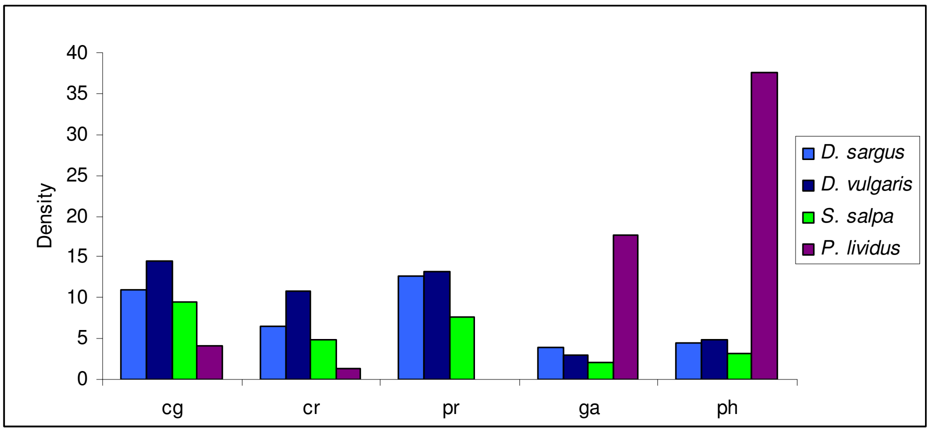
In order to correctly evaluate cascading effects deriving from protective measures, the distribution of intermediate elements should be taken into consideration. Herbivores play a key role due to their tight connections with macroalgal assemblages. *S. salpa,* which is the only herbivorous species (Fig.3.22) recorded in the study area, shows significant differences (*P*< 0.01) with regards to its frequency of distribution, especially between sites CG (Pianosa) and PR (Elba). As a whole, total density recorded in Pianosa appears to be higher with respect to that obtained from Elba. Two sites, GA and PH do not show significant differences.

Figure 3.22: *S. salpa* frequency distribution between sites.



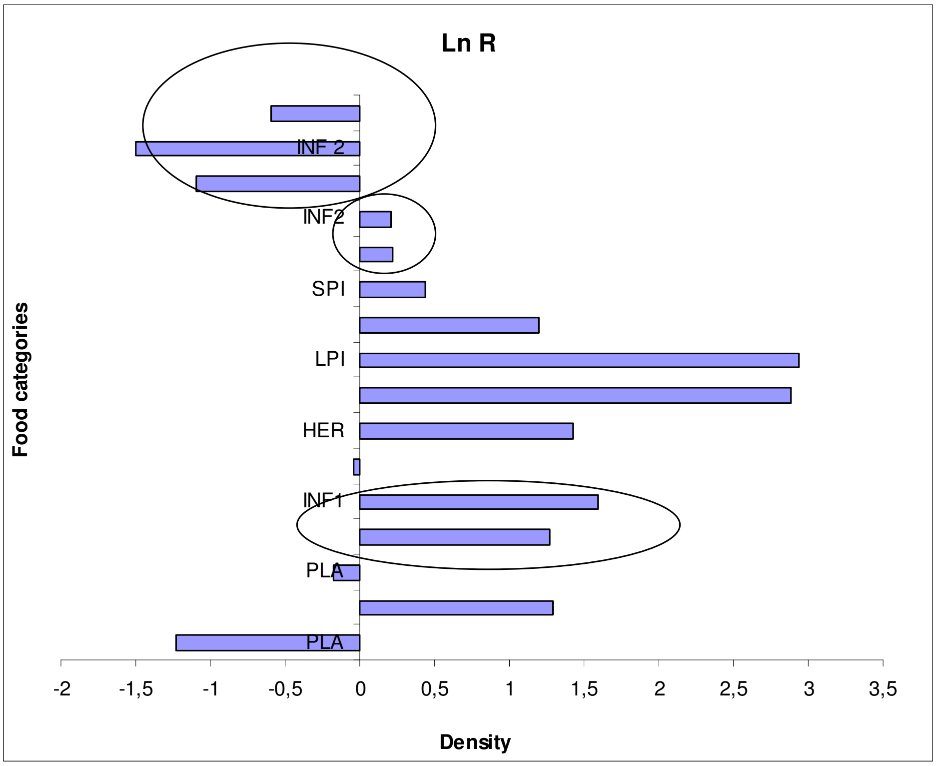
In order to evaluate their contribution, densities of the single species were compared and shown in Figure 3.23. Sparids (invertebrate feeders 1), which represent the principal predators of sea urchins, show higher densities within the protected area. Even if not as consistently, *S. salpa* (herbivores), shows the same pattern. Densities of *P. lividus* shows an opposite pattern, with higher densities recorded in sites from Elba (PH and GA).

Figure 3.23: compared densities of *D.sargus, D.vulgaris*, *S.salpa, P.lividus* between sites.



The reserve effect was evaluated in terms of trophic cascades by comparing the Ln of response ratio calculated densities in each trophic groups. Figure 3.24 shows that positive values of R are represented by invertebrate feeders 1, which benefit from lack of fishing pressure and have a diet based primarily on sea urchins. Instead, invertebrate feeders 2 which predate only occasionally on sea urchins (< 1 cm in test diameter), show negative values of R and thus don't seem to benefit from reserve effects. With regards to herbivores (*S. salpa*), the positive value of R, seems to demonstrate that the reserve has a positive effect on its density.

Figure 3.24: response ratio for each of trophic groups.



4 DISCUSSION

Regarding biodiversity of the fish community studied, this work has discerned the effect of protection on fish species composition and relative abundances, despite the small range of depth considered. Taxonomic composition of fish community in Pianosa and Elba islands agrees with other similar northern-western in Mediterranean (Bell, 1983; Harmelin, 1987; Garcia-Rubies and Zabala, 1990). These results are concordant with the literature concerning the factors explaining the spatial variation of local fish assemblage in tropical and temperate waters (Luckhurst and Luckhurst, 1978; Bell and Galzin, 1984; Patton, 1985; Grigg, 1994; McCormik, 1994).

Species belonging to Blennidae and Gobidae family, were censused mainly in sites located in Elba island. The less number of small cryptobentonic fishes in protected area than in non protected ones, may be a consequence of large number of piscivorous in protected reserve. The biomass of fish is limited by food, space or other limiting environmental sources; is colourable that biomass become saturated inside the reserve due to the adult fish populations. Strong competition or predation pressures could either limit recruitment or displace the smallest individual outside the reserve. Recent studies in different Mediterranean marine reserve, have shown that mortality pattern of some juvenile fishes could be higher in the reserves than outside (Dufour et al., 1995).

The species of the family Labridae are present in almost all sites where transects were conducted. This data shows that in reality there is not a reserve effect for this family. The wrasse species are characterized by strong sexual dimorphism, for which, in many cases, it was possible to assess qualitatively the presence of both sexes in the study areas. *S. tinca* proved to be particularly abundant in both sites located in the protected area than in the island of Elba. Other species such as *S. melanocercus*, *S. melops*, *S. mediterraneus*, were recorded in only sporadically.

The reserve effect is the first factor affecting both the qualitative and quantitative structure of fish assemblages (Garcia-Rubies and Zabala, 1990). The abundance of target species is correlates strongly with the reserve effect, and therefore shows that the reserve has been effective in providing protection for such species. Some highly spearfished species such as *E. marginatus*, *S. umbra*, or *D. dentex*, have been censused exclusively within the reserve. Other such as *S. aurata* are far more abundant in the reserve.

There are not significant differences between the number of species recorded in the five sites under consideration and in the two areas, as indeed shown in other studies in the Mediterranean Sea (Vacchi et al., 1998).

The effect of reserves does not discriminate on the total number of species, but on the type of species, depending on their vulnerability to be captured by commercial fishing activities and to the trophic level occupied.

The observed variability in number of individuals at Procchio (Elba), is probably due to the influence effect introduced by the erratic presence of large schools of mid-water species as *C.chromis, Atherina* spp*,* and *O. melanura.*

Full protection implemented at Pianosa seems not to affect the total number of individuals and thus the general abundance (no reserve effect). The assessment of the total abundance of species in protected area and in the reference area it should be assessed by studying the frequency distribution of the sizes of most abundant species. This abundance should be the result of greater presence of young individuals that in many species tend to form aggregates rather numerous, or the result of a large number of sub-adults and adults individuals.

Although the values of species richness is of the same order to those obtained in other areas of Mediterranean (Bell, 1983; Harmelin, 1987, 1990; Garcia-Rubies and Zabala, 1990; Dufour et al., 1995; Renones et al., 1995), some differences exist between sites in this study. Species diversity index (*H'*), and Evenness index (*J'*), show significant differences between sites located in Pianosa.

Increased levels of anthropogenic stress generally cause a decrease in both H' and J'.

The lowest values of H' and J' measured at the site of Procchio (Elba) could be the result of increased exposure of the area to the presence of tourists. In fact, although survey activities were all conducted under the same environmental conditions, they do not take into account the presence of tourists and the impact this has on the fish community. Hence the importance to study the behavior of different species against the presence of swimmers, so to assess whether it can be a cause of aggregation of species. In particular, there are species, such as *O. melanura*, which are attracted by the presence of swimmers. The potential impact on sampling area during exposure to other forms of human disturbance was not evaluated here.

Despite the uniformity of bottom type and the chosen type of sampling sites, Pianosa and Elba differ significantly considering the various parameters of the community. PERMANOVA, in particular, shows how the two sites of the Elbe (PH and GA) are the only ones not to show statistically significant differences. In this sense, therefore, there is a distinct grouping between the protected sites and the control area (CR).

The Canonical Analysis of Correlation (CAP), conducted for each year of the survey, confirms the analysis made on PERMANOVA. Both the details for each single three-year frame and those of the whole period show a clear division of sites in the two areas, with regard to the population having fixed factor protection. The sites chosen in the reference area (Elba) always show partially overlapping negative values along the axis of abscissas. The same type of overlapping exists for positive values for PR and CG sites in the target area. The third reference sites, CR, although characterized by a population similar to the one of the other two sites in Pianosa, show an overlap with both groups. The reserve effect translates into a substantial difference in the composition of the populations of the two islands. This fundamental fact has enabled to work on the structural analysis of the fish community in the two areas in order to assess which species can determine the observed differences, and to what extent.

The study of the community parameters however is centered mainly on those species of commercial interest, involved in the assessment of the trophic cascade due to the institution of protected area. The assessment of "effort" to capture small-scale fisheries conducted at one site (Marina di Campo), with a minimum number of vessels fishing mainly with trammel nets and longlines, has allowed to define locally relevant species into categories: hightly fished (HF), lightly fished (LF) and no target (NT).

Several studies from the western Mediterranean sea, report that abundance of some target species is greater within MPAs than in fishes areas (Harmelin et al., 1995, La Mesa and Vacchi, 1999; Garcia- Charton et al., 2004; Guidetti et al., 2005). Marine reserves, may be effective in restoring fish assemblages (Halpern, 2003). Sargo beams show a positive response to protection in other MPAs in Mediterranean sea (Guidetti, 2005). In Pianosa, both sargo beams species, *D. vulgaris* and *D. sargus* (higly fished species), showed higher abundance in protected than in unprotected area. It is possible that the reserve effect due to the protection from fishing activities can support observed differences in abundance between the two areas. The richness of the two considered sargo bream species (*D. sargus* and *D. vulgaris*) in protected area, can also be related to the high hydrodynamically sheltered areas suitable pro recruitment along the Pianosa coast.

The same considerations can be made for those species that do not have high commercial value but may still be subject to the fisheries, like the Saddled seabream (*O. melanura*). The species of little or no commercial value do not show significant differences in abundance at the studied sites.

The non target species could be negatively affected by the reserve effect through their ecological relationship with other fishes, such as competition or predation (Dufour et al., 1995). However frequency distribution of some no-target species in Pianosa and Elba, (*C. julis*, *T. pavo, S. scriba* and *S. tinca*), don't show differences between the investigated sites.

The demographic structure of fish community can be considered the main difference between protected and unprotected areas. This doesn't mean that marine reserves are overcrowded refuges for large individuals, but zones where natural fish populations are maintained. The analysis of size classes showed the reserve effect more clearly.

The study of four size classes of some target species has confirmed that large individuals were significantly more abundant in the protected area. In Pianosa, large and very large specimens of *D. sargus* and *D. vulgaris*, are more than small and medium.

The increase of the size of fishes is a phenomenon widely reported in other MPAs (Roberts and Polunin, 1991), and this general pattern has primarily attributed to the lack of fishing impact (Garcia-Rubies and Zabala, 1990; Harmelin et al., 1995; Mosquera et al., 2000). However processes regulating biomass partitioning related to size in fish populations are very complex and may vary not only in relation to the local fishing impact, but also to the habitat type, the local productivity, and the indirect impact caused by changes in trophic interactions among species triggered by the removal of larger-bodies predatory fish (Macpherson et al, 2002; Dulvy et al, 2004; Guidetti et al., 2005).

The investigated highly fished species that significantly respond to protection at Pianosa (e.g. *E. marginatus, S. umbra, D. dentex, S. aurata, D. sargus, D. vulgaris, D. puntazzo*), are targeted by many kind of fisheries (especially trammel net). Protected area seems to have the potential to restore depleted fish stocks particularly for species as

D. sargus and *D. vulgaris*.

As they are widely distributed, the sargo breams are the most valuable target species for testing the efficiency of fish regulations in Mediterranean areas (Garcia-Rubies and Zabala, 1990). The proportion of large individuals in local populations is a particularly interesting index (Harmelin et al., 1995). Pianosa allows large individuals of target species; the permanent concentrations of large spawners with undisturbed reproductive behaviour may enhance the recruitment potential of a whole region through dispersal of egg and larvae.

Frequency distribution of non target species such as *C. julis*, and *T. pavo* can be interpretated as out come of their social behaviour. For example, males of *C. julis* (larger than females) have as priority activity to catch available food, including bait. In zones with heavy angling pressure, the density of males tend to decrease rapidly, and the result deficiency must be compensated for by sex change (Harmelin et al.,m 1995).

Garcia- Rubies and Zabala (1990) interpreted the lower number of large and very large individuals as an effect of habitat monopolization by the large territorial individuals of others species.

It would be useful to study also the biomass of fish at each site in the two areas, in order to know if there is a higher biomass of vulnerable fishes inside the reserve than outside.

Some important parameters have not been assessed during this study and this limits the strengths of conclusion. For example, the fishing effort, gives a potential bias on the data (Saldana, 1991) because no data were available to test the hypothesis that fish community in Elba were affected by fishing. The presents results are the consequence of qualitative assessments of fishing impact on fish community.

The response ratio evaluated for exploitation level shows that the reserve has been effective in protection for heavily fished species. Protected area is a zone where natural, non harvested adult fish populations are maintained (Garcia-Rubies and Zabala, 1990). A large proportion of species negatively affected by protection are not targeted by fishing, resulting in an overall lack of response to protection for non target species. Some of these species are characterized by low mobility as families Blenniidae and Gobiidae, and their negative response ratio is evidence indirect of protections.

The distributions of abundances or biomass among different species is affected by protection and the effects on resulting community structure vary depending on the duration of protection (Micheli et al., 2004).

In addition, variation in species responses to protection was correlated to trophic level.

Fish assemblage in Pianosa was protected from fishing for many years, and has greater relative proportions of predatory abundance than fished assemblage in Elba.

Protected area, support greater density of *D. sargus* and *D. vulgaris* as predators on sea urchin *P. lividus*, than unprotected area. Conversely invertebrate feeders Type 2 abundances, were similar in the five sites probably due to the infrequent use of fishing techniques that have a high impact on wrasses (Guidetti, 2006), at the same time, the cover of barrens was significantly higher in unprotected area than in Pianosa. The presence of a greater number of high levels predators in the reserve than outside has important consequences on the environment dynamic (McClanahan, 1997), for example, the absence or low number of high level predator outside the reserve are synonymous with increasing number of preys.

This study has evidenced a reserve effect or refuge effect characterized by the presence of large individuals. This refuge effect, within the presence of large individuals and species vulnerable to fishing, is one of the characteristics described in the definition of reserve effect (Russ and Alcala, 1989).

D. sargus, is a key predatory species in the North-western Mediterranean rocky infralittoral (Sala, 1997). The results of the present study suggest that the recovery of *Diplodus* species in Pianosa seems to reestablish predatory control upon *P. lividus* similar to what is observed in other temperate regions (Shears and Babcock, 2002) and in other Mediterranean protected areas (Guidetti, 2006). Abundance of *S. salpa* (herbivorous fish), is significantly more abundant in two sites of Pianosa; the larger presence of this species in the protected areas is probably related to the higher cover of macroalgal in protected area than in Elba sites.

The available studies dealing with community wide changes in marine reserves suggest that trajectories of change are various (Sala et al., 1998; Pinnegar et al., 2000; Micheli er al., 2004; Guidetti et al., 2005). Only marine reserves that have the proper characteristics for supporting dense population of predators should be expected to undergo the back transition from barrens to macroalgal beds (Guidetti, 2006). The lower density of sea urchins inside the reserve is probably related to higher levels of fish predation (McClanahan, 1997), even though there are many factors potentially capable of affecting population density of sea urchins. The pattern of predation upon *P. lividus* may have implications for the trajectories of community change after establishment of no take reserve.

In unprotected area, where fishing activities are capable to remove large invertebrate feeders, sea urchins may increase in density and overfeed on macroalgae, which in turn may cause the transition from macroalgal beds to barrens (Sala et al., 1998). The results of this study can corroborate the hypothesis of a transition in Elba investigated sites, from macroalgal beds, which harbour hundred of species of algae and invertebrates, to barrens, through sea urchins grazing.

Diplodus density can be assumed to be an index of the potential of reserves to recover from barrens to algal beds or to maintain flourishing algal beds. Population size of invertebrate feeders type 1 in Elba sites, is not enough to control population size of sea urchins. In protected area, the effect of predator density on prey and on its population parameter is density-mediated indirect interactions (DMIIs).

Other factors than large size and high abundance of predatory fishes can affect sea urchins abundance, which may be large even in marine reserves (Sala and Zabala, 1996; Sala et al., 1998; Guidetti, 2006). This processes are pollution, disease, large scale oceanographic events and availability of refuges from predation. The relationship between predatory fishes and urchins, and the existence of trophic cascades after reserve creation, are not easily predictable.

The differences in trophic group abundances indicate a shift in the trophic structure of fish assemblages protected from fishing. A combination of increased abundances and increased individual size of piscivorous and invertebrate feeders fishes, likely explains their greater proportional contribution to fish biomass in no take reserve compared to reference conditions (Micheli et al., 2004).

All invertebrate feeders type 1, showed a positive response ratio, while invertebrate feeders type 2 were not influenced by the establishment of no take areas.

The observed variation in abundance of the studied fish assemblages attributable to the reserve effect, can confound the results of studies attempting to detect and explain the effect of other anthropogenic factors. At the same time, the knowledge about changes in habitat structure can help to separate the influence of this factor from effect of protection (Garcia-Charton, 1998). The casual relationship between fish species and their habitat have to be elucidated in order to distinguish them from other factors.

The temporal variability of fish assemblages should be considered when assessing the information obtained in this study, as this is a factor which is known to greatly influence the structure of fish communities in Mediterranean coastal habitats. One of the main difficulties in investigating the processes which may act in a MPA, is the spatial and temporal heterogeneity of the ecosystems (Kolasa and Pickett, 1991; Garcia-Charton and Peréz-Ruzafa, 1999).

Spatial and temporal pattern in rocky reef fish community are likely to be caused by different sets of processes, which differ in their importance and scales (Jones, 1988).

Significant increases in total abundances in MPAs typically occurred in few years (1-3 yr) after the reserve establishment (Halpern and Warner, 2003), and recovery of trophic structure requires decades (Micheli, 2004).

The present study confirm that focusing the sampling effort on target species provide substantial evidence of protection effects.

5. CONCLUSIONS

Micheli et al (2004) questioned if the observed changes in abundances of top trophic level and in their proportional contribution to biomass could be compared to what might be observed in pristine ecosystems. Hawaiian islands has been indicated as possibly to closest to pristine (Jackson et al, 2001): data from this islands has been indicated that approximately 50% of fish biomass is accounted by top predators, specifically large piscivorous snappers, groupers, carangids, and sharks (Friedlander and De Martini, 2002).

Despite Pianosa is not a typical MPA since actual permanent presence of penal institute, it represent an important site where to study isolated infralittoral fish biodiversity on rocky substrata. The absence of anthropic disturbance previous the institution of reserve is an important factor in this work, because most of the study conducted on littoral fish community in Mediterranean MPAs were affected by disturbance caused by fishing and other anthropic influences, such as pollution and tourism previous institution of marine reserves.

Although the absence on data on total biomass in this work, the proportion of invertebrate feeders and piscivorous fish observed in the protected area may be below what the system could potentially support.

This studies confirms that besides direct effects on target species, fishing may cause changes in trophic relations and in ecosystem functioning. Fish community parameters and proportion of invertebrate feeders, sea urchins and barren in protected area related to non protected area, support hypothesis that community changes produced by fishing interdiction, can create trophic cascades in MPAs. Depletion and size reductions of invertebrate feeders caused by fishing modify the patterns of predation on sea urchins. Fishing prohibition in protected areas may restore ecological and predatory interactions among species, with potential community wide effects. Moreover, some aspects as recruitment rate, migrations, and other factors structuring fish biodiversity should be seriously considered. For instance, relative importance of fishing predation on sea urchins depends on the absolute importance of other processes as recruitment rate, fish predation pressure and shelters availability.

The protected area causes restoring of infralittoral fish biodiversity. The reserve effect is a real consequence of total protection. This effect occurs on size, abundance and population structure of fishes species that are targeted by fishing nearby. Infralittoral fish community, being the most susceptible of changes due to biotic and abiotic modifications induced by anthropic activities, gives positive responses to protection.

However protected area do not represent an equilibrium point. It is in fact susceptible to ecologic and oceanographic events; certainly the absence of human impact within reserve may helps communities to recover after disturbances. It is clear that future experiments testing the reserve effect on fish biodiversity will have to take into account the necessity of being repeated on larger spatial and temporal scales. In particular, researches are urgently required to unequivocally establish the consequences of total protection on a larger spatial scale, in order to provide useful information for innovative and better management of the MPAs.

The current Italian legislation for the MPAs for the conservation of marine biodiversity provides full protection within the integral reserve areas (Area A) only. However, these areas do not allow the total recovery of fish communities, in terms of abundance and species diversity. In particular, although the partial ban of the fishing activities improves the abundance of the commercial fish species in the MPAs, their presence is limited within the borders of the protected areas. Considering MPAs as part of integral marine ecosystem management schemes, this situation generates low level of profitability, both in fisheries and touristic activities (i.e. diving activities).

This work evidences that the total inhibition of fisheries has a positive impact on the fish population in a large spatial scale. Among the benefits from this kind of protection, it is possible to observe a sharing of medium and large size top-predators in a larger space, avoiding the typical concentration of them in small scale areas. Moreover an increase of small size fishes is observed as consequence of the decrease of predation pressure.

Thus a larger extension of Area A in the Italian MPAs provides better conservation conditions, improving the role of the others protected zones (B and C) as “source” of fish biodiversity, while the non-protected areas close to MPAs as “since” and generating a positive economic feed-back also.

The resilience of fish populations allows that coastal fish communities can restore their natural diversity, abundance and recruitment in a few years but a correct management is urgently needed. In this framework, this study suggests that it would be necessary to consider a different zonation of MPAs, *e. g.* providing a total ban on fishing within a mile away from the coast, for the protected sites with similar characteristics to the the areas considered in this study.

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