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Influenze antropiche dirette e indirette sulla biodiversità dei mari sardi

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Direct and indirect anthropogenic effects on biodiversity of the Sardinian seas

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INDEX

FIRST PART

SEA-BASED INFLUENCES ON THE MIDDLE-SLOPE ASSEMBLAGE

SECOND PART

LAND-BASED INFLUENCES ON THE ATLANTIC BLUEFIN TUNA

INTRODUCTION

All organisms modify their environment, and humans are no exception. As the human population has grown and the power of technology has expanded, the scope and nature of this modification has changed drastically. Until recently, the term “human-dominated ecosystems” would have elicited images of agricultural fields, pastures, or urban landscapes; now may be applied to all of Earth. Many ecosystems are dominated directly by humanity, and virtually no ecosystem on Earth’s surface is free of human influence (Vitousek et al. 1997).

Because coastal areas have been focal points of human settlement and marine resource use throughout history, neither a not properly “human habitat” such as the marine environment, is completely free of its influence. In particular humans depend on ocean ecosystems for important and valuable goods and services, but human use has also altered the oceans through direct and indirect means, modifying marine environments and its functioning (Botsford et al. 1997; Vitousek et al. 1997; Addis et al. 2008; Halpern et al. 2008; Jackson et al. 2001; Lotze et al. 2006). Land-based activities affect the runoff of pollutants and nutrients into coastal waters (Syvitski et al. 2005; Vitousek et al. 1997) and remove, alter, or destroy natural habitat. Ocean-based activities extract resources, add pollution, and change species composition (Pauly et al. 2005). These human activities vary in their intensity of impact on the ecological condition of communities (Halpern et al. 2007) and in their spatial distribution across the seascape (Halpern et al. 2008).

Fishing activity in particular is a significant human enterprise. The world fishing fleet comprised about 4 million units at the end of 2004 (FAO, 2006); moreover fisheries provide direct employment and food for millions of people. Recent estimates indicate that fishing activity increased fourfold between 1950 and 2003 (Watson and Pauly 2001). Such fast increasing of fishing effort, supported by incentives aimed to increase the fleets and to develop new technologies and the inadequate governments management policies, is considered responsible of an important decrease of the fishing resource (Bosford et al. 1997; Watson and Pauly 2001). Fishery can alter marine ecosystems through both direct and indirect effects. Major changes can lead to modification in the composition of the fish fauna (Kaiser et al. 2002), reduction of the abundance of target population, affection of the life-history parameters (growth rate and age at maturity). Fishing has also indirect long-term effects on habitat through the removal of predators that control bio-engineering organisms (Kaiser et al. 2002), and, in extreme circumstances, can results in the local extirpation of species (Rogers and Ellis 2000). Finally this may results in a reduction of habitat complexity that makes ecosystems more vulnerable to other natural and human disturbances (Jackson et al. 2001; Kaiser et al. 2002).

Considering the high degree of interaction between marine organisms and environment, a more holistic approach incorporating interspecific interactions and physical environmental influences would contribute to greater sustainability by reducing the uncertainty in predictions (Bosford et al. 1997) and promoting an ecosystem based management. A holistic, ecosystem approach to fishery management requires the integration of information from a wide range of disciplines, levels of ecological organization, and temporal and spatial scales.

In the light of the above, this thesis considers a sea-based and a land-based anthropogenic impacts and attempts to analyse the effects of both on two different ecosystems the demersal and pelagic, by means of two different “biological indicators”. Moreover, the influence of natural dynamics on shaping assemblages has also taken into account by analysing the structure of demersal communities of a peculiar environment such as a submarine canyon. Finally has been considered the impacts of traditional fishing practices on the quality of the bluefin tuna destined to human consumption.

The First Part considers the impact of fishing activity on demersal fishing resources (Chapters 1 and 2) in a increasing fishing effort area (Sabatini et al. 2006) and the influence of natural dynamics on shaping deep-sea assemblages in a submarine canyon (Chapter 3). Classical biodiversity indexes, statistical simulation and multivariate analysis ordination techniques were used in order to carry out the investigations.

Following the international trend, in the last decade, also Sardinian fishing fleet has undergone over the last 10 years a considerable renewal as a consequence of government policy (Sabatini et al. 2006). The main change has involved the replacement of the old, low-tonnage wooden boats with large deep-sea iron trawler boats. As a result slope fisheries have progressively increased moving the fishing pressure toward deep-sea resources, which in the early 90s, were lightly exploited in the Sardinian seas (Sabatini et al. 2006). This should have produced a certain degree of anthropic disturbance that can be detrimental in particular for middle-slope assemblages, that are expected to have low resilience to increasing disturbances (CIESM 2003) because of their late maturation, extreme longevity, low fecundity and slow growth (Koslow et al., 2000). For this reason deep-sea fish are potentially much more vulnerable than their shallow-water counterparts. Moreover, bottom trawling has the potential to cause long-term impacts on sediment nutrient fluxes (Olsgard 2008) and can destroy bottom habitat (particularly biotic structure) with poorly known long-term consequences (Koslow et al., 2000). In the light of the above, is extremely important form both ecological and economic point of view, in order to preserve the range of goods and services that ocean ecosystem provides, to develop new methods aimed to increase the knowledge on both the state of the resources and ecosystem health. Biodiversity has proven to be an excellent tool to assess the health of ecosystems as it is extremely sensitive to degradation (Heip et al. 2003). In fact, monitoring biological diversity rather than simply few charismatic ecosystem components acknowledges the multi-scale nature and complexity of ecosystems and envisions a proactive rather than reactive approach to species conservation, likely a more effective and cost-efficient approach in the long run (Beever 2006). Furthermore, monitoring biodiversity seems more likely than other approaches to simultaneously track health and function of ecosystems as well as the capacity to promote human well-being (Beever 2006).

Our main aim was to propose a new method able to provide information about environmental stress due to fishery, monitoring biodiversity. First of all (Chapter 1) is provided the temporal evolution of classical biodiversity indexes (Shannon's entropy, species richness, Pielou's evenness and Simpsons' dominance), with regard to middle-slope assemblages in four specific locations of Sardinian seas (SE, NE, NW, SSW) which underwent different degrees of fishing pressure. The validity and reliability of these index as a measure of environmental stress and their potential use for the monitoring of demersal ecosystems has been evaluated. Successively (Chapter 2) a statistical model of Shannon's diversity index temporal behaviour has been calculated combining the biodiversity components and fishing effort via a two-explanatory variable multiple linear regression model. Were evaluated the dependence of biodiversity to fishing effort and discussed the usefulness of the model as tool to forecast the temporal biodiversity behaviour.

Besides the well proven increasing human influence on marine ecosystems, natural dynamics play an important role on shaping the middle-slope assemblages biodiversity and can be an important source in the variability of the data. A reliable testing ground for such hypothesis are submarine canyons. In the seas around Sardinia a number of submarine canyons with highly variable morphology divide the upper slope, mainly along the eastern coast (Western part of Central Tyrrhenian Sea). These areas are peculiar because are rich in nutrients, due to both their geomorphology and their material and hydrographic flows; they are areas with strong turbidity currents that cause morphology changes (Shepard et al., 1974). Submarine canyons are also considered as complex environments, described as unstable (Thorne-Miller & Catena, 1991) and species that live there have greater mobility than those of typical deep-sea assemblages (Rowe, 1971). Their influence on the movements of a number of species is widely known (Tudela et al., 2003; Sardà et al., 1997; Tursi et al., 1996), but information concerning day-night movements in submarine canyons is still lacking. The aim of the Chapter 3 is to describe for the first time the demersal assemblage related to the Quirra canyon (Eastern coast of Sardinia), highlighting the influence that, a singular marine environment and the natural dynamics which undergoes, may have on the diversity of the assemblage. Day-night vertical movements of the species which live in the canyon have been considered.

The second part of this thesis considers the pelagic ecosystem and use as “biological indicator” the Atlantic bluefin tuna (*Thunnus thynnus* L.).
Chapter 4 take into account the influence of a typically land-based source of anthropic perturbation (mining activity) on bluefin, via time series analysis. Chapter 5 focus the attention on the impact that the traditional fishing practices may have on the final quality of the product. Data has been collected on a traditional trap fishery, a passive capture method that relies upon the natural swimming behaviour of the tuna on their reproductive migration routes along the Mediterranean coast. Information from the traps has been the basis of many scientific studies that have resulted in a better understanding of the biology, ecology and behaviour of BFT (De Metrio, 2001; ICCAT, 2006; Sarà, 2007).
Thunnus thynnus L., undertake a feeding and spawning migration, covering distances of several thousands of miles primarily over the northern Atlantic Ocean, Gulf of Mexico, and Mediterranean Sea (Mather et al., 1967, 1973; Farber and Lee, 1980; Lutcavage et al., 1999; Block et al., 2001, 2005). These migrations are genetically controlled, but the timing and routes can be conditioned by a large variety of factors: environmental conditions, bottom topography and oceanic currents, availability of food, population

density, and anthropogenic perturbations (Cort, 1990; Sarà et al., 2007). Bluefin tuna show steady 'homing behaviour' prior to spawning when schools move in the same general direction. Traditional traps (tonnare) have documented the 'natal homing' of BFT from the 12th century to today (Sarà, 1998; Cury et al., 1998; Doumenge, 1998). Because there is variability in trap landings data it is important to identify the sources. The variability has been evaluated on large geographical scales with some evidence of synchronous fluctuations over periods of about 100-120 years which are considered directly related to global environmental variables (Ravier and Fromentin, 2004). On a smaller geographical scale, historical documentation based on anecdotal data, evidence such as environmental conditions may account for trap landing variability (Parona, 1919; Neuparth, 1923; Sella, 1929; Lozano Cabo, 1958; Rodriguez-Roda, 1966). According to Lemos and Gomes (2004), local weather and coastal ocean variability are the probable sources of short-term oscillations in trap catches. Another possible explanation of variability might be so called boundary effects, or the well-recognized fact that tunas tend to concentrate on boundary fronts in the ocean (Murphy, 1959; Blackburn, 1965; Sund et al., 1981; Laurs et al., 1984; Fielder and Bernard, 1987). According to Hubbard et al. (2004), environmental boundaries can affect pelagic migration patterns. Their definition of boundaries included specific isotherms, confines between warm and cool water masses, specific depth contour lines and bodies of water with different physical and chemical characteristics.

The south-western region of Sardinia has been the location of one of the most important Italian trap fisheries since the late 15th century and fishing continues to this day with three active traps (Addis et al., 2006). For at least 2000 years this region was also historically important for lead, silver and copper mining (Manconi, 1986). South-western Sardinia's mining areas produced about 15 Mton of metal, which makes the region comparable to Ireland in total metal production (Salminen et al., 2005). The cumulative impacts of environmental hazards in south-western Sardinia resulting from mining and industrial activities is well documented: on land (Leita et al., 1989; Caboi et al., 1993; Boni et al., 1999), in coastal areas (Caredda et al., 1999; Pagano et al., 2002), and in human health risk factors (Cardia et al., 1989; Sanna et al., 2002; Cocco et al., 1996). However, there is little information available on the impact of such factors on pelagic marine fisheries. The migration of the Atlantic bluefin tuna in coastal waters is genetically controlled. As the bluefin make their spawning migration past Sardinia, it is reasonable to consider local perturbations, generated by social and economic events and environmental changes, as disruptive to the pathways of bluefin schools and thus account for variability in the Mediterranean trap captures. Therefore, in Chapter 4 is intended to investigate patterns of catch variability in three historical traps of south-western of Sardinia (Porto Paglia, Portoscuso and Isola Piana), to test for mining perturbation effects on impact and control trap sites using a BACI experimental design (Green, 1979; Underwood, 1994), and to analyze the interactions between trap landings and multiple environmental variables (Mining, Run-off and SST) using a Dynamic Factor Analysis (DFA) (Zuur et al., 2007).

Bluefin tuna has the capacity to conserve metabolic heat and can maintain a steady-state body temperature that is greater than ambient water temperature (Carey and Teal, 1969; Carey and Gibson, 1983). Is hypothesized that this characteristic enhance tuna performance (Brill, 1996), independent of environment temperature, and enables bluefin tuna to expand their vertical and latitudinal thermal range (Graham and Dickson, 2001).
Carey first shed light on the capacity of giant bluefin to thermoregulate through significant ranges in ambient temperature. Later, he also made the first observations of significant visceral warming following a meal, and he suggested that the elevated temperature would serve to speed digestion and allow tuna to feed frequently (Carey and Teal 1966; 1969, Carey et al. 1971; 1984, Carey and Lawson 1973) Since then, other studies on free-swimming wild fish, using electronic tags, have provided more comprehensive information on the eco-physiology and behaviour of bluefin tuna, including data on thermoregulatory biology over a large geographical scale (e.,g., Lutcavage et al., 2000; Marcinek et al., 2001; Brill et al., 2002) and evidence that bluefin tuna maintain muscle temperature at about 24 °C, significantly higher than the ambient temperatures of their high latitude feeding grounds in summer months. Large individuals with archival tags placed in the peritoneal cavity experienced a wide range of environmental temperatures (2.8° to 30.6°C) but maintained relatively constant internal peritoneal temperature (25°C) and a thermal excess up to 21°C above ambient temperature Block et al. (2001).
The bluefin tuna behaviour, when is entrapped and fished in a traditional Mediterranean trap, suggested the possible correlation between body temperature and low/high stress conditions of the fish. Due to the big dimensions and the chamber structure of the

Mediterranean trap once the bluefin enter into the chamber, they tend to adjust quickly to captivity, and display low levels of activity, and are thus considered to be in a low stress condition. In contrast, during fishing operations, or the "mattanza", the tuna are withdrawn from the water by raising the bottom of the net mesh floor of the death chamber and are then hoisted up onto the fishing vessel via hooks. The act of confining the fish in a small space to be killed presumably produces a high degree of stress, and a cascade of metabolic processes that deplete the oxygen reserves in the muscles. Moreover, the increase in swimming speed, a consequence of a high tail beat frequency observed during fishing operations, results in increased metabolic heat within the slow-twitch muscle (red) and contributes to body temperature elevation (Blank et al., 2007).

The main aim of the Chapter 5 is to verify if body temperature can be used as an indicator of stress, ranging from the presumed "minimal stress" condition of free swimming bluefin tuna in the chambers (i.e "pre-fishing phase" experimental condition) to the high degree of stress associated with fishing operations in a traditional trap in Sardinia (i.e. "fishing phase" experimental condition). Additional data have been provided on white (Tw) and red muscle (Tr) temperature and the excess of mean red muscle temperature compared to mean ambient water temperature (Ta) (Tx=Tr-Ta) for individual bluefin tuna.

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FIRST PART

Sea-based influences on the middle-slope assemblage

Chapter 1

Biodiversity time trends of red shrimp-related assemblages in Sardinian seas (Central-Western Mediterranean).

Biodiversity time trends of red shrimp-related assemblages in Sardinian seas (Central-Western Mediterranean).

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Abstract – Species diversity is considered as important factor for ecosystem stability. While is well proved that the commercial fishing affects overall biomass, size and species structure of demersal fish communities, just few studies considered effects of fishery on species diversity. The aim of this study is to examine the temporal evolution of classical biodiversity indexes for red shrimp-related assemblages in specific locations of Sardinian seas, that have been subject to increasing fishing pressure over the past eleven years. Data come from 11 MEDITS trawl surveys carried out from 1994 to 2004, concomitant to Sardinian fleet renewal. Were estimated the temporal trends of fishing effort and biodiversity indexes (Shannon's entropy, species richness, Pielou's evenness and Simpson's dominance). Analysis was conducted on four separate areas (NW, NE, SE, SSW) into which the Sardinian seas were sub-divided, due to their different geomorphology and fishing practices. Statistically significant variations in both fishing pressure and biodiversity indexes were found for southern sub-zones. Precisely, SE sub-zone showed a fishing effort increase of 120% together with decreases in Shannon's diversity, species richness and an increase in Simpson's dominance. For SSW sub-zone that displayed a 22.6% increase of fishing effort, occurred only a decrease of species richness. According to our results, fishing pressure appeared to affect diversity of demersal species related to red shrimp, showing different biodiversity index reactions on different levels of fishing effort. The validity and reliability of this index as a measure of environmental stress and its potential use for the monitoring of demersal ecosystem is finally discussed.

Key words: Species diversity / Biodiversity index / Red shrimp / Demersal resources / Sardinian seas

1. Introduction

The exploitation of marine resources is increasing all over the world. Recent estimates indicate that fishing activity increased fourfold between 1950 and 2003 (Watson and Pauly 2001). The situation in the Mediterranean Sea is not far off this mark: indeed, "*most of its resources are overexploited. […] the situation has got worse over the last ten years due to increased efficiency in fishing devices and methods*" (Anon. 2002). Sardinian seas have seen a considerable increase in the pressure on fishing stocks as well, mostly on demersal species (Sabatini et al. 2006), due to the renewal of fishing fleets over the last 10 years.

Commercial fishing exerts a considerable effect on demersal fish communities, in terms of overall biomass, the modification of size and species structure (Pauly 1979; Sharp and Csirke 1984; Pauly 1988; Suvapepun, 1991; Greenstreet and Hall 1996; Haedrich and Barnes 1997, Sainsbury, et al. 1997; Fogarty, and Murawski 1998; Bianchi et al 2000), as well as interfering with ecosystems (ICES, 1995; Greenstreet and Hall 1996; Hunt and Furness, 1996, Heip 2003). In order to gain an overall vision of such effects, some more markedly ecosystem-addressed management approaches have recently been developed, (May 1976; Koslow et al. 1988; Jennings and Polunin, 1996; Stergiou 2003; Rochet et al. 2005). Among these, biodiversity is surely one of the best methods to evaluate the effects of overfishing on ecosystems.

The great variety in the geomorphology and natural environments of Sardinian sea beds (Cau 1994), together with the intensity of exploitation, which has been constantly monitored since 1994 by trawl surveys (MEDITS) (Bertrand et al. 2002), make Sardinian seas a reliable testing ground for evaluating the varying consistency in biodiversity indexes in different areas, and we believe may provide efficient indicators for assessing the state of health of ecosystems.

Other similar studies carried out in recent years have compared different bio-geographical areas (Atlantic sea and Mediterranean sea) (Blanchard 2001) employing only certain kinds of indexes, and using a low sampling effort (Ungaro et al. 1998; D'Onghia et al. 2003). Other studies have concentrated on some particular areas with high biodiversity gradients, such as tropical marine ecosystems (Lobry et al. 2003).

The aim of this study is to examine the temporal evolution of classical biodiversity indexes (Shannon's entropy, species richness, Pielou's evenness and Simpsons' dominance), with regard to capture depths of red shrimp in specific locations of

Sardinian seas that have been subject to increasing fishing pressure over the past eleven years. We have evaluated the validity and reliability of this index as a measure of environmental stress and its potential use for the monitoring of demersal ecosystems.

2. Materials and methods

2.1. The study area

Sardinia is one of the most interesting areas of Mediterranean sea Central western basin both for geographical position and the extent of fishing bottoms (Fig. 1) (Cau et al. 1994).

Geomorphology of Sardinian sea bottoms is diverse. The South Western and Western seafloor is characterized by a wide continental shelf with a gradual decline that ends between 150-200 m depth and a gradual continental slope. The South Eastern and Eastern bottoms are characterized by a narrow continental shelf that terminates at about 50-100 m of depth and a really steep continental shelf (Fig. 1).

Sea bottoms we considered for our investigation regard a depth interval from between 400-800 m and of about 4220 km^2 (Cau et al. 1994) (Fig. 1). At the considered depths, Western and South Western sea floors are characterized by sandy muds (until 450 m) and deep muds (>500 m depth), whereas the South Eastern and Eastern bottoms are rocky, irregular and spaced out by active canyons, that allow transportation of sediments and nutrients to deeper regions. Due to differences on geomorphology of sea bottom and fishing habits, seas around Sardinia were subdivided in 4 sub-zones (North Western 595 km^2, North eastern 855 km^2, South eastern 502 km^2 and South-South Western 2268 km^2) (Fig. 1). The Northern seas were excluded due to the lack of deep strata.

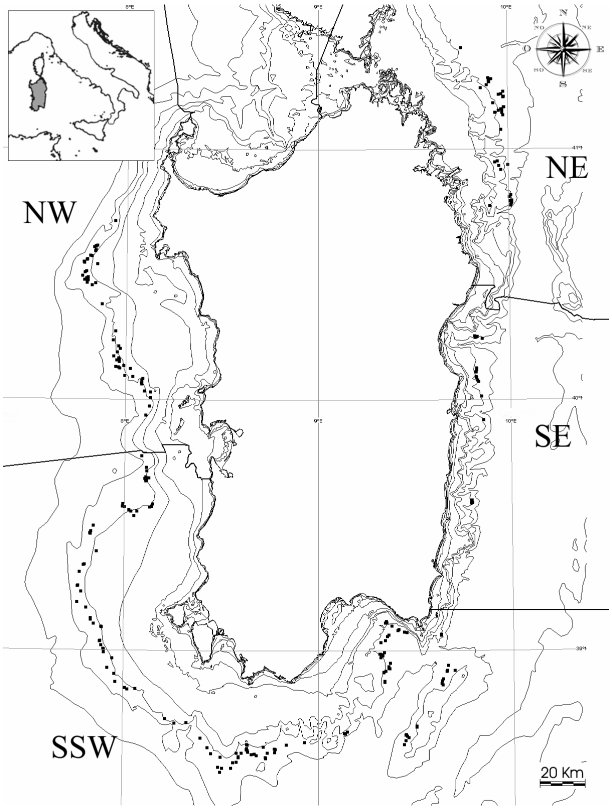

Fig. 1. – Seas around Sardinia. Location of the trawl surveys during the period 1994-2004. Depth from between 400-800 m were considered.

2.2. Data collection

Data were collected over a eleven year period between 1994-2004, during eleven trawl surveys carried out in the seas around Sardinia during summer months, within the framework of the international MEDITS research programme. MEDITS is a international trawl survey started in 1994 and carried out in the Mediterranean, intended to produce information on benthic and demersal species (Bertrand et al. 2002). Samples were collected using a stratified random strategy, by means of a trawler of 168 GT of the local fishery. Fishing gear is a GOC 73, a bottom trawl (Fiorentini et al. 1999) designed for experimental fishing with scientific purpose. Its codend meshsize is 20 mm (stretched mesh) (Bertrand et al. 2002). Teleosts, selachians, crustaceans and cephalopods captured in each haul were sorted into species, then counted and weighed.

Were considered a total of 294 hauls of the duration of one hour each, at depths of between 400 and 800 m (Fig. 1). In order to evaluate species strictly associated to red shrimps, were selected exclusively those hauls where *Aristeus antennatus* and *Aristaeomorpha foliacea* were present. Abundance data (N) were successively standardized per 1 h (N h^{-1}), in order to homogenize data.

In order to verify if annual samplings are comparable between them and the number of individuals caught is suitable for a biodiversity investigation, rarefaction method have been carried out. Yearly rarefaction curves calculated for every sub-zone, where rarefied to the minimum number of individuals for the considered sub-zone. Species richness estimates where then compared with observed species richness.

2.3. Biodiversity indices

For biodiversity calculation, four indexes have been used.

Species diversity has been calculated with the Shannon index *H'* (Shannon, 1948):

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$

The quantity p_i is the proportion of individuals found in the *i*th species (Pielou 1969).

$$p_i = \frac{n_i}{N}$$

Where n_i is the number of specimen of the *i*th species and *N* is the number of individuals of the sample. The value of the Shannon index obtained from empirical data usually falls between 1.5 and 3.5 and rarely surpasses 4 (Magurran 2004).

For species richness estimation were used the maximum Shannon's entropy (Ganis 1991):

$$H_{\max} = \ln S$$

Maximum entropy is a logarithmic function of the only numerousness of species and its measurement is used to evaluate the "species richness" component of specific diversity (Ganis 1991).

Evenness computation has been carried on by Pielou's *J'* index (Pielou 1969), presented as the ratio of observed diversity (*H'*) and maximum entropy ($H_{\max}$):

$$J' = \frac{H'}{H_{\max}} = \frac{H'}{\ln S}$$

Evenness is a measure of how similar species are in their abundances (Magurran 2004). *J'* value, is comprised between 0 (when only one species is present) and 1 (when all species have the same abundance). Evenness regards the potential maximum diversity of a ecosystem ($H_{\max}$); that is the capability of the ecological system to accept species in equivalent proportions. Therefore *J'* should be considered as a "synthesis" of previous indexes (Lobry et al. 2003).

In order to have another point of view on species structure of investigated communities, the dominance index have been also evaluated by the Simpson index:

$$D=\sum_{i=1}^{S} p_i^2=\sum_{i=1}^{S}\left(\frac{n_i\left[n_i-1\right]}{N\left[N-1\right]}\right)$$

Is part of another group of diversity indices that are weighted by abundances of the commonest species, while information statistics indices described above tend to emphasize the species richness component of biodiversity (Magurran 2004).

k-dominance curves, showing the cumulative abundance of ranked species (Lambshead et al. 1983), were finally calculated for abundance data within years and used in the interpretation of assemblage structure as follows: species assemblage x is more diverse than y if the curve for y is everywhere below or touching that of x.

2.4. Test for trend

Trend significance was tested using the non-parametric Mann–Kendall test (*MK* test). The Mann–Kendall test can be stated most generally as a test for whether Y values tend to increase or decrease with time (T) (monotonic change) (Helsel and Hirsch, 2002).

H_0: Prob $[Y_j > Y_i] = 0{,}5$ where time $T_j > T_i$

H_1: Prob $[Y_j > Y_i] \neq 0{,}5$ (2-sided test)

where i = 1,…, (n - 1) and j = (i + 1),…, n

The value of τ was estimated using the L. A. Goodman and W. H. Kruskal’s method (Zar, 1999).

$$\tau=\frac{N_C-N_D}{N_C+N_D}$$

Where N_C is the number of concordant values (i.e. the number of times that Y grows when T grows) and N_D is the number of discordant values (i.e. the number of times that Y decreases when T grows) that can be found in the computation process of τ.

The τ value estimated has been then compared with tabular τ_α values for the significance level chosen. Then we have:

if $\left|\tau\right| < \tau_\alpha \Rightarrow$ H_0 is accepted, no statistical significant trend

if $\left|\tau\right| \geq \tau_\alpha \Rightarrow$ H_0 is rejected, there is a statistical significant trend. This can be:

increasing if $\tau > 0$

decreasing if $\tau < 0$

The correlation between fishing effort and biodiversity indexes was estimated by means of a linear regression and then tested using Fisher’s F test.

3. Results

3.1. The fishing fleet

Fishing pressure on deep sea resources increased by 52% during the period 1991 – 2004 as a consequence of the policy designed to renew and improve productivity in Sardinian fisheries (Fig. 2, Table 1). Until the last year considered in this study (2004) there were 67 boats of TSL > 30 that operated between 400 – 800 m of depth, on a trawling area of about 4220 km^2.

The increase in fishing effort from 1994 to 2004 is statistically significant for the whole area but showed a different temporal trend in the four studied sub-zones. The highest increases were recorded in South – Eastern (SE) (+120%) and South South – Western (SSW) (+22.6%) zones; *MK* testing showed statistical differences. On the other hand, the North – Eastern (NE) area registered no fishing effort increase, whereas the North – Western (NW) area showed a non-statistically significant 12.5% increase (Fig. 2) (Table 1).

Table 1. – Fishing effort ($\text{N°}_{\text{boats}}\ \text{km}^{-2}$) and its temporal percentage increase (1994 benchmark year)

	South-East		**North-East**		**North-West**		**S.South-West**		**Total**	
Years	**F. E.**	**% Incr.**	**F. E.**	**% Incr.**	**F. E.**	**% Incr.**	**F. E.**	**% Incr.**	**F. E.**	**% Incr.**
1991	0.012	20.0	0.006	-44.4	0.012	-12.5	0.011	-16.1	0.010	-17.0
1993	0.010	0	0.011	0	0.007	-50.0	0.013	-3.2	0.011	-9.4
1994	0.010	0	0.011	0	0.013	0	0.014	0	0.013	0
1995	0.010	0	0.011	0	0.013	0	0.014	0	0.013	0
1996	0.010	0	0.011	0	0.013	0	0.014	0	0.013	0
1997	0.014	40.0	0.012	11.1	0.018	37.5	0.015	9.7	0.015	17.0
1998	0.014	40.0	0.012	11.1	0.018	37.5	0.015	9.7	0.015	17.0
1999	0.012	20.0	0.012	11.1	0.018	37.5	0.015	9.7	0.014	15.1
2000	0.016	60.0	0.011	0	0.018	37.5	0.015	12.9	0.015	18.9
2001	0.016	60.0	0.011	0	0.018	37.5	0.015	12.9	0.015	18.9
2002	0.016	60.0	0.007	-33.3	0.007	-50.0	0.016	16.1	0.013	1.9
2003	0.016	60.0	0.007	-33.3	0.007	-50.0	0.016	16.1	0.013	1.9
2004	0.022	120.0	0.011	0	0.015	12.5	0.017	22.6	0.016	26.4
τ	0.813		-0.019		0.226		1		0.667	
p	< 0,05		> 0,05		> 0,05		< 0,05		< 0,05	
Trend	↑		↔		↔		↑		↑	

p-level = 0.1

$\tau_{0.05} = 0.436$; $\tau_{0.1} = 0.359$

↑ increasing trend, ↓ decreasing trend, ↔ no trend (i.e. no statistical significance detected)

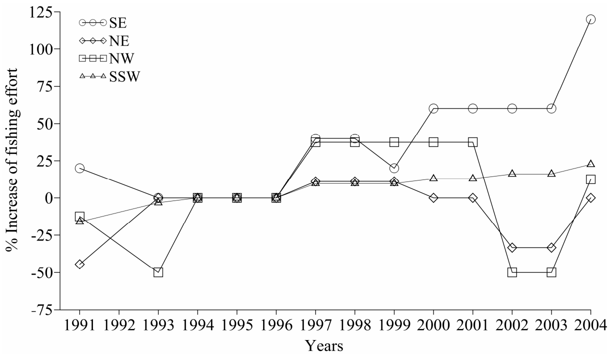

Fig. 2. – Percentage increase in fishing effort (1994 is considered the reference year)

3.2 Rarefaction

Rarefaction curves differ in length along the horizontal axis, shape and in relative position to each other (Fig. 3). Overall, almost all curves reach an asymptote and is noteworthy that samples with large numbers of specimens are not the richer in species. Because of the wider trawling area, the SSW sub-zone is the most rich in species. For this area, rarefaction curves for all years reach an asymptote and most of them exceed the threshold of 70 species. On the other hand, the narrow depth strata related to red shrimp for the NW coast, together with the rocky bottoms and the presence of canyons of the eastern coast (SE and NE sub-zones), resulted in a less quantity of species caught for those sub-zones. Moreover, due to the sampling protocol strategy, less sampling weight has been given to sub-zones with less trawling surface.

However, the comparison between rarefied species richness and sampled based species richness, display that the latter have higher values then estimates ones.

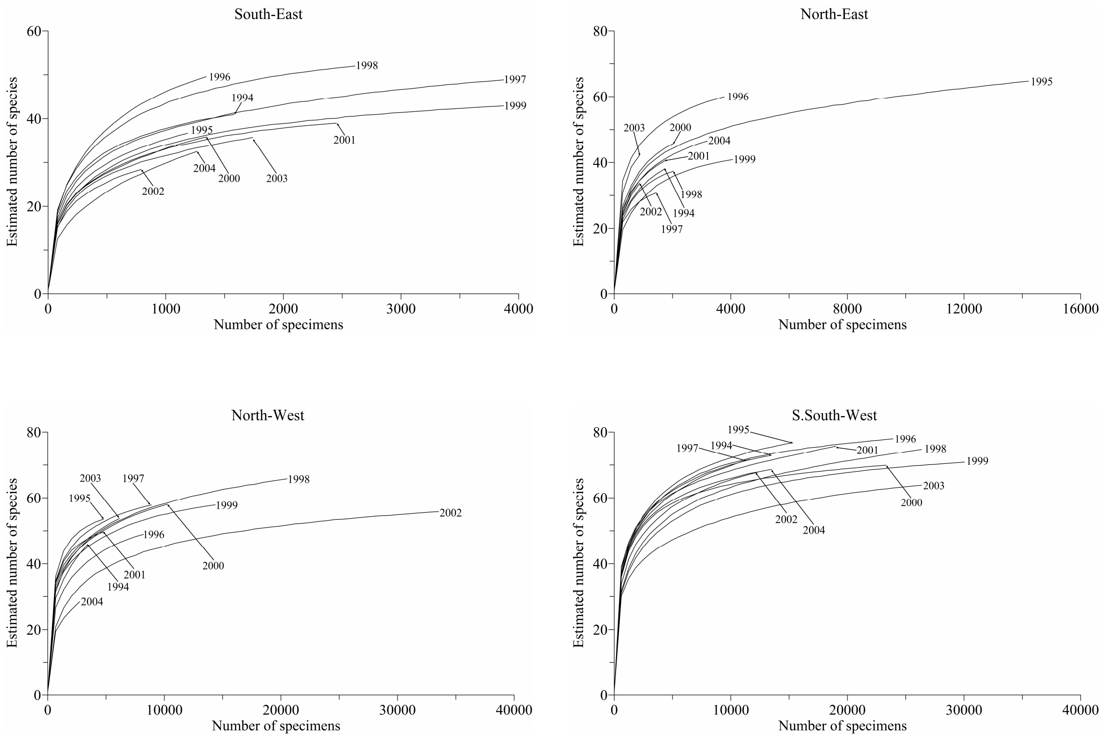

Fig. 3. – Species richness estimates in the investigated sub-zones per each year, using rarefaction curves based on all taxa.

3.2. Species caught

According to the selection criteria adopted, 154 species were collected. Teleosteans were dominant (75), followed by Crustaceans (38), Molluscs (24) and Selachians (17). Of the 154 species captured, 40 alone represented about 98% of the total number of specimens (Table 2).

Crustaceans comprised 50% of all the catches and were thus the most abundant systematic class (N h^{-1}). *Aristaeomorpha foliacea* (Risso, 1827) and *Aristeus antennatus* (Risso, 1816) are the two most highly commercial and target species for fisheries that operate at the depths considered. They represent an important slice of the total catch (13.1%). However the most abundant single species was *Pasiphaea sivado* (Risso, 1816) (12.7% of the total catches), of little commercial value, prevalently caught in NW and SSW zones (Table 2).

Teleosteans comprised 43.1% of total catches. Of these, *Chlorophthalmus agassizii* Bonaparte, 1840; *Gadiculus argenteus* Guichenot, 1850; *Phycis blennoides* (Brunnich, 1768) and *Hymenocephalus italicus* Giglioli, 1884 constituted about 36% of total captures of bony fish (Table 2).

Catches of Selachians and Molluscs were fewer (respectively 5,5% and 1,3%) than those of Teleosteans. All species accounted for under 1% of the total, with the exception of *Galeus melastomus* Rafinesque, 1810 (4.5 %) (Table 2).

Table 2. – Caught species, divided by zones and total studied area.

	South-East		North-East		North-West		S.South-West		Total	
	N/h	%	N/h	%	N/h	%	N/h	%	N/h	%
Teleosteans										
Argentina sphyraena	1	0	74	0.2	87	0.1	590	0.3	752	0.2
Chlorophthalmus agassizii	3040	13.5	4507	11.8	8567	7.0	31481	14.4	47595	11.9
Coelorhynchus coelorhynchus	115	0.5	183	0.5	433	0.4	3850	1.8	4581	1.1
Epigonus denticulatus	118	0.5	29	0.1	9	0	584	0.3	740	0.2
Epigonus telescopus	127	0.6	30	0.1	170	0.1	551	0.3	878	0.2
Gadiculus argenteus	159	0.7	5305	13.9	11776	9.7	19771	9.0	37011	9.2
Glossanodon leioglossus	0	0	0	0	0	0	742	0.3	742	0.2
Helicolenus dactylopterus	83	0.4	236	0.6	1370	1.1	1194	0.5	2883	0.7
Hoplostethus mediterraneus	132	0.6	122	0.3	762	0.6	849	0.4	1865	0.5
Hymenocephalus italicus	2517	11.2	3014	7.9	8029	6.6	15423	7.1	28983	7.2
Lampanyctus crocodilus	626	2.8	721	1.9	1135	0.9	2328	1.1	4810	1.2
Lepidorhombus boscii	75	0.3	115	0.3	241	0.2	436	0.2	867	0.2
Merluccius merluccius	24	0.1	19	0	592	0.5	1342	0.6	1977	0.5
Micromesistius poutassou	26	0.1	319	0.8	134	0.1	272	0.1	751	0.2
Mora moro	277	1.2	9	0	167	0.1	601	0.3	1054	0.3
Nettastoma melanurum	91	0.4	35	0.1	79	0.1	323	0.1	528	0.1
Nezumia sclerorhynchus	265	1.2	293	0.8	296	0.2	2727	1.2	3581	0.9
Notacanthus bonapartei	130	0.6	40	0.1	63	0.1	460	0.2	693	0.2
Phycis blennoides	2365	10.5	2123	5.6	6433	5.3	18472	8.5	29393	7.3
Others Teleosteans (56)	400	1.8	462	1.2	558	0.5	1731	0.8	3151	0.8
Total	10571	46.9	17636	46.3	40901	33.6	103727	47.5	172835	43.1
Selachians										
Etmopterus spinax	239	1.1	822	2.2	735	0.6	1492	0.7	3288	0.8
Galeus melastomus	2069	9.2	4649	12.2	4224	3.5	6936	3.2	17878	4.5
Scyliorhinus canicula	3	0	370	1.0	68	0.1	30	0	471	0.1
Others Selachians (14)	23	0.1	191	0.5	51	0	238	0.1	503	0.1
Total	2334	10.4	6032	15.9	5078	4.2	8696	4.0	22140	5.5
Molluscs										
Eledone cirrhosa	12	0.1	123	0.3	59	0	197	0.1	391	0.1
Illex coindetii	17	0.1	27	0.1	433	0.4	73	0	550	0.1
Sepietta oweniana	15	0.1	1004	2.6	924	0.8	254	0.1	2197	0.5
Others Molluscs (21)	142	0.6	411	1.1	775	0.6	745	0.3	2073	0.5
Total	186	0.8	1565	4.1	2191	1.8	1269	0.6	5211	1.3
Crustaceans										
Aristaeomorpha foliacea	1849	8.2	1217	3.2	4328	3.6	24454	11.2	31848	7.9
Aristeus antennatus	2442	10.8	1485	3.9	3899	3.2	13178	6.0	21004	5.2
Geryon longipes	128	0.6	434	1.1	35	0	56	0	653	0.2
Nephrops norvegicus	221	1.0	1848	4.9	1953	1.6	1486	0.7	5508	1.4
Parapenaeus longirostris	43	0.2	86	0.2	3735	3.1	2905	1.3	6769	1.7
Pasiphaea multidentata	74	0.3	50	0.1	3814	3.1	3978	1.8	7916	2.0
Pasiphaea sivado	628	2.8	2063	5.4	26034	21.4	22346	10.2	51071	12.7
Plesionika acanthonotus	366	1.6	185	0.5	268	0.2	1532	0.7	2351	0.6
Plesionika antigai	230	1.0	219	0.6	734	0.6	2075	0.9	3258	0.8
Plesionika edwardsii	27	0.1	53	0.1	1793	1.5	1701	0.8	3574	0.9
Plesionika gigliolii	754	3.3	144	0.4	258	0.2	1603	0.7	2759	0.7
Plesionika heterocarpus	1	0	56	0.1	1222	1.0	408	0.2	1687	0.4
Plesionika martia	1963	8.7	3334	8.8	15594	12.8	23161	10.6	44052	11.0
Polycheles typhlops	646	2.9	1436	3.8	185	0.2	780	0.4	3047	0.8
Sergestes corniculum	0	0	0	0	9415	7.7	4535	2.1	13950	3.5
Others Crustaceans (23)	81	0.4	207	0.5	192	0.2	677	0.3	1157	0.3
Total	9453	41.9	12817	33.7	73459	60.4	104875	48.0	200604	50.0
Total N (154 Species)	22544		38050		121629		218567		400790	

3.3. Temporal evolution of biodiversity indexes

Biodiversity indexes showed an oscillating and widely ranging variation in trends, except for species richness (ln*S*), which showed less extensive variations compared with other indexes (Fig. 4). Shannon's entropy values ranged between a maximum of 2.922 (NE, 1996) and a minimum of 1.672 (NW, 2004). Higher peaks were recorded during the first half of the period (1994-1999), whereas more stable values were observed in the subsequent years. The NW area differs slightly from the others in that its oscillating trend was maintained for the whole period of investigation (Fig. 4).

Simpson's index (*D*) values were analogous but sometimes more pronounced than those in Shannon's entropy. In Eastern areas (NE and SE), 1999 registered the highest values. The absolute minimum was recorded for North Eastern Sardinia ($D = 0.084$, 1996), whereas the absolute maximum was in the NW sub-zone ($D = 0.323$, 1995) (Fig. 4).

For Western Sardinian seas, Pielou's index (*J'*) also behaved in a similar manner as that observed for Shannon's entropy. Slight differences can be found in the SE and NE sub-zones. In the SE sub-zone, a narrow and regular variation range was documented, interrupted by only two decreases in 1997 and 1999. The NE shows a wider variation range mainly owing to the maximum index value recorded for 2002 (0.720), the highest for the whole period investigated. The minimum value was recorded in the NW zone (0.469, 1995) (Fig. 4).

Species richness (ln*S*) varied from between 4.357 (SSW, 1996 and 2001) and 3.401 (SE, 2002 and NW, 2004). Annual variations were smoother than previous indexes. The first three years of the period investigated showed higher values than the following years for the NE area. Furthermore, the NW sub-zone registered a significant loss of species richness in 2004 (Fig. 4).

Temporal trend analysis (*MK* test) found no statistically significant variation in Northern areas (NW e NE) (Table 3). On the other hand, significant variation was observed in the SE zone, for all biodiversity indexes, with the exception of evenness which recorded a slight decrease, not though confirmed by *MK* test (Table 3). Due probably to a sustained fall starting from 1998, species richness registered a statistically significant decrease with $p<0.1$. Such a decrease in richness affected Shannon's entropy, which, during the study period, showed a constant drop (*MK* test $p<0.05$). On the other hand, Simpson's index showed an increase in dominance for the SE area, with $p<0.1$ (Table 3).

The SSW area reported variation only in the species richness, which underwent a constant decrease (*MK* test $p<0.05$), although values here continued to be higher than in the other zones (Table 3). Decreases in Shannon's entropy and Pielou's evenness (both $\tau = -0.236$) as well as dominance increase, were not statistically significant trends (Table 3).

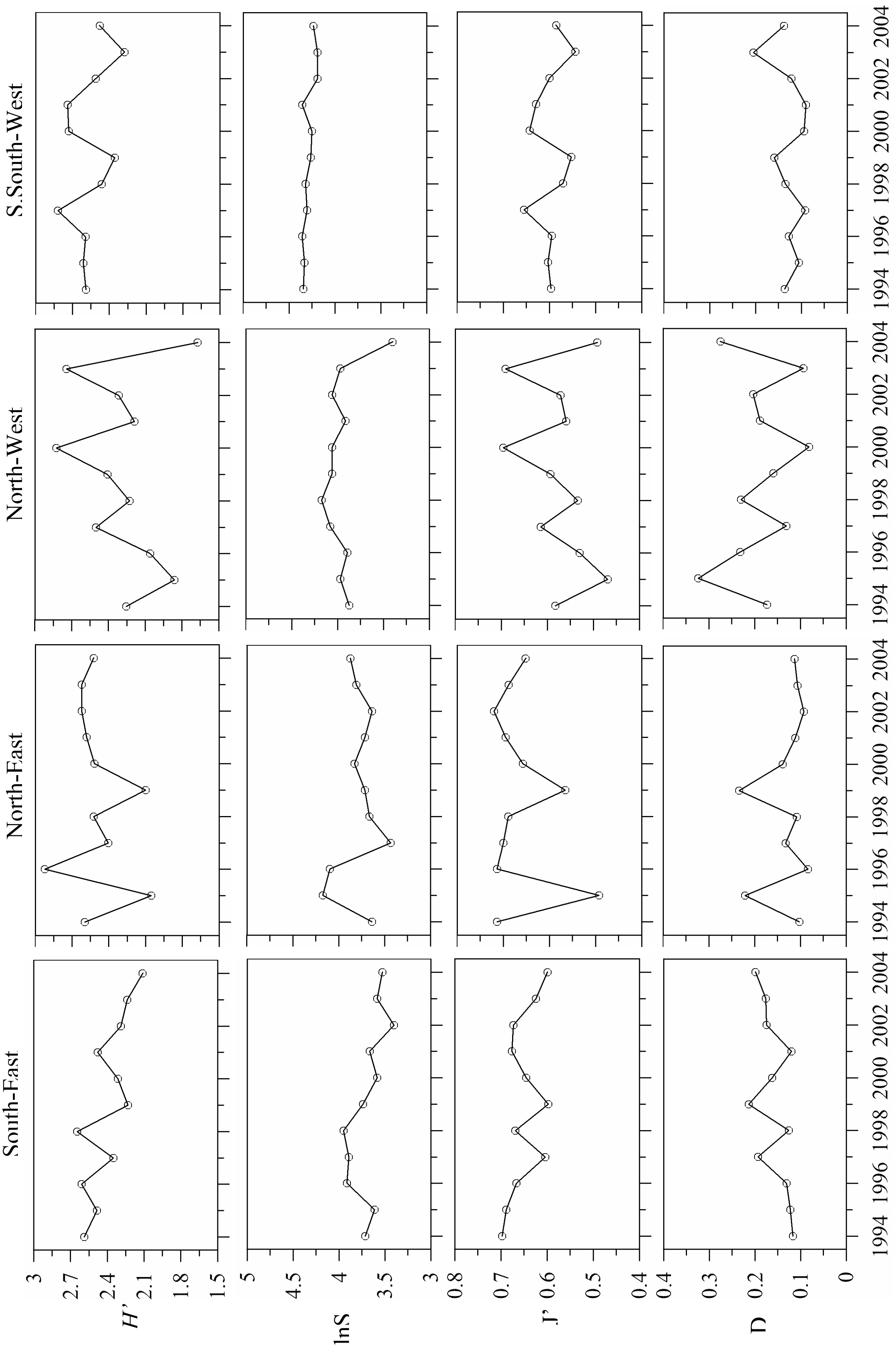

Fig. 4. – Variation of the biodiversity indexes during time: trajectories of Shannon (*H'*), species richness (ln*S*), evenness (*J'*) and dominance (*D*) indexes are presented for each sub-zone.

Table 3. – Biodiversity indexes: temporal trend

Years		1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	τ	p	Trend
SE	*H'*	2.589	2.486	2.608	2.350	2.646	2.233	2.313	2.478	2.290	2.238	2.112	-0.564	< 0,05	↓
	ln*S*	3.714	3.611	3.912	3.892	3.951	3.738	3.584	3.664	3.401	3.584	3.526	-0.481	< 0,1	↓
	J'	0.697	0.689	0.667	0.604	0.670	0.598	0.645	0.676	0.673	0.625	0.599	-0.382	> 0,1	↔
	D	0.117	0.122	0.131	0.193	0.126	0.214	0.163	0.120	0.175	0.177	0.199	0.455	< 0,1	↑
NE	*H'*	2.595	2.054	2.922	2.404	2.524	2.099	2.518	2.581	2.619	2.620	2.523	0.200	> 0,1	↔
	ln*S*	3.638	4.174	4.094	3.434	3.664	3.714	3.829	3.714	3.638	3.807	3.871	0.094	> 0,1	↔
	J'	0.713	0.492	0.714	0.700	0.689	0.565	0.658	0.695	0.720	0.688	0.652	-0.127	> 0,1	↔
	D	0.102	0.221	0.084	0.132	0.108	0.233	0.139	0.111	0.092	0.106	0.113	-0.018	> 0,1	↔
NW	*H'*	2.254	1.861	2.061	2.503	2.227	2.408	2.825	2.186	2.317	2.743	1.672	0.127	> 0,1	↔
	ln*S*	3.871	3.970	3.892	4.078	4.174	4.060	4.060	3.912	4.060	3.970	3.401	-0.059	> 0,1	↔
	J'	0.582	0.469	0.530	0.614	0.533	0.593	0.696	0.559	0.571	0.691	0.492	0.164	> 0,1	↔
	D	0.173	0.323	0.231	0.130	0.229	0.159	0.081	0.188	0.203	0.093	0.275	-0.127	> 0,1	↔
SSW	*H'*	2.590	2.610	2.592	2.819	2.462	2.355	2.731	2.741	2.512	2.277	2.478	-0.236	> 0,1	↔
	ln*S*	4.344	4.331	4.357	4.304	4.317	4.263	4.248	4.357	4.190	4.190	4.234	-0.585	< 0,05	↓
	J'	0.596	0.603	0.595	0.655	0.570	0.552	0.643	0.629	0.600	0.544	0.585	-0.236	> 0,1	↔
	D	0.136	0.106	0.127	0.092	0.135	0.159	0.094	0.090	0.122	0.204	0.138	0.127	> 0,1	↔

p-level = 0.1

$\tau_{0.05} = 0.491$; $\tau_{0.1} = 0.418$

↑ increasing trend, ↓ decreasing trend, ↔ no trend (i.e. no statistical significance detected)

The Fisher test carried out to analyse correlation between fishing effort and biodiversity, confirmed what was previously observed by means of biodiversity index analysis (Table 4). While no significant trends were detected for the NE and NW sub-zones, the South Eastern seas did evidence negative correlation for Shannon indexes (p-value < 0.05) and species richness (p-value ≈ 0.1). Variation for species richness index in the SSW sub-zone was highly significant (p-value=0.01).

Table 4. – Fishing effort vs. Biodiversity correlation

	South-East			**North-East**			**North-West**			**S.South-West**		
	p-value	**b**		**p-value**	**b**		**p-value**	**b**		**p-value**	**b**	
H'	0.02	-33.97	↓	0.30	-50.89	↔	0.91	-2.84	↔	0.45	-41.78	↔
ln*S*	0.12	-23.83	↓	0.86	-7.75	↔	0.78	4.39	↔	0.01	-46.23	↓
J'	0.12	-5.13	↔	0.40	-12.02	↔	0.79	-1.42	↔	0.77	-3.46	↔
D	0.13	4.85	↑	0.25	11.49	↔	0.93	-0.51	↔	0.56	6.59	↔

p-level = 0.1

↑ increasing trend, ↓ decreasing trend, ↔ no trend (i.e. no statistical significance detected)

K-dominance curves analysis, considered to be particularly robust to the affection of sample size, as compared to other diversity indices (Zhou et al. 2007), provides us an additional point of view regarding the biodiversity time variation. Although under normal annual fluctuations, a temporal trend for Southern areas should be underlined. In fact for SE and SSW sub-zones can be noticed that the last years of the study (2002-2004), resulted less diverse then the starting years (Fig. 5). For the Northern seas a clear temporal pattern couldn't be underlined, thus not showing evidences in time changes.

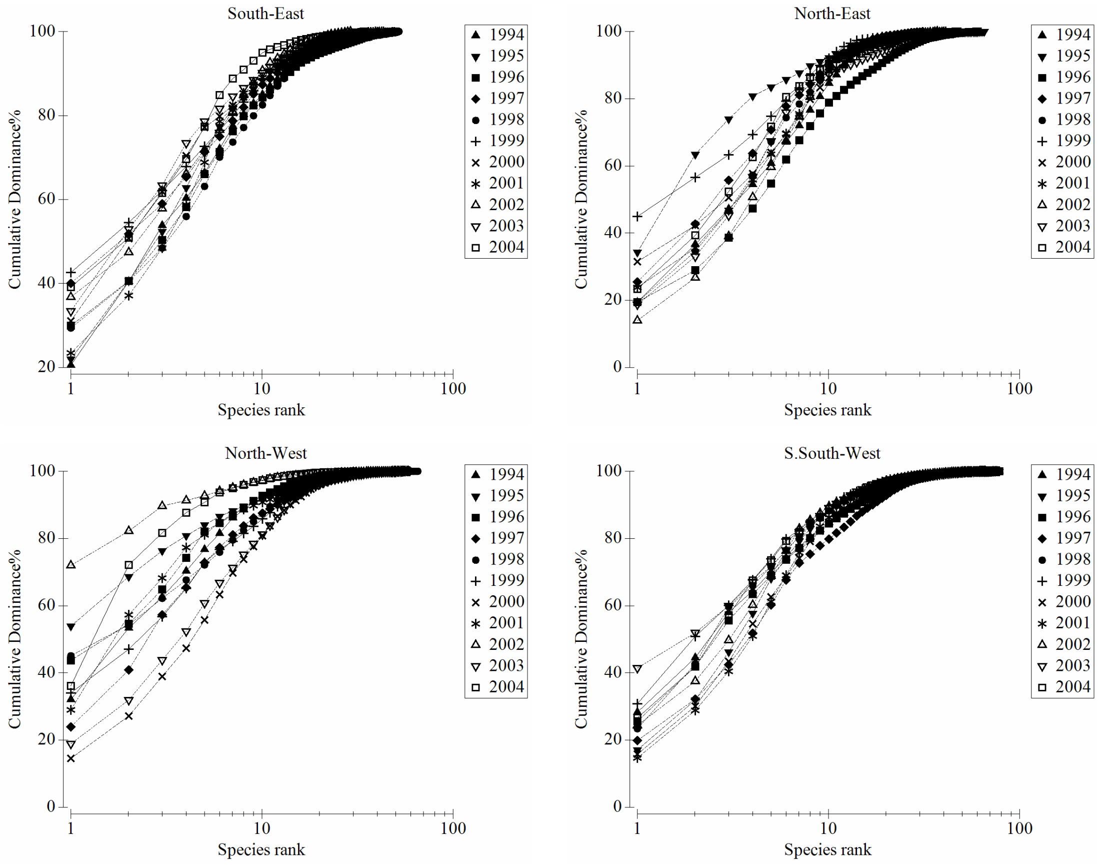

Fig. 5. – Comparison of *k*-dominance curves for the 400-800 m demersal fish communities related to red shrimp, between investigated years from the SE, NE, NW and SSW sub-zones.

4. Discussion

The use of biodiversity indexes as a instrument for assessing marine resources is a subject of continued debate (Caddy & Sharp 1986; Caddy & Griffiths 1995; Ungaro et al. 1998; Lobry et al. 2003). Several studies have been carried out (Blanchard 2001; Ungaro et al. 1998; D'Onghia et al. 2003; Lobry et al. 2003; Colloca et al. 2003; Labropopoulou et al. 2004) but they have not brought to light clear evidence of the effects of fishing effort on biodiversity.

Our analysis on biodiversity, based on data from 11 annual trawl surveys, showed different temporal trends in the four sub-zones considered.

The studied demersal assemblages, specifically related to red shrimp at depths of between 400 m – 800 m, showed that only about 1/4 of the 154 species caught had a high degree of dominance, representing 98% of total abundances (N h^{-1}). Bony fish species were the most numerous, followed by crustaceans, which were more abundant in terms of specimen numbers. These results concur with those reported in other studies from the Eastern and Central Mediterranean (D'Onghia et al. 2003; Colloca et al. 2003).

There are several causes and factors that can induce variations in biodiversity (Lobry et al. 2003; Caddy & Sharp, 1986; Caddy & Griffiths, 1995; Ungaro et al. 1998). Marine traffic accounts for just a small part of anthropic pressure on these assemblages. In Sardinian seas, it is limited to main navigation routes and areas in proximity to major ports. Although climatic changes are usually directly responsible for important environmental variations, they apparently affect only surface layers and can only affect the deeper ones indirectly (IFREMER, 1997). Moreover Mediterranean sea in particular is characterised by high degree of environmental stability, being homeothermic below 200 - 300 m depth, with warm deep waters (13-14 °C) reaching the abyssal seafloor down to 5.050 m depth (Hopkins et al. 1989; Tyler 2003; Hopkins 2006). A temporal series trend analysis carried out on sea temperature data, estimated by the ECCO-JPL project (ECCO-JPL Ocean Data Assimilation Project 1999), confirms what stated above, not revealing any significant variation in temperature for our bathymetric range over the time period considered. Fishing activity seems thus to be the main source of anthropic pressure on these assemblages. Temporal variation of biodiversity indexes (entropy, species richness, evenness and dominance), in fact, seem to be clearly "fishing effort" ($N°_{boats}\ km^{-1}$) related.

The renewal of fishing fleets in Sardinia also corroborates our analysis, as it coincided with our period of investigation, which saw the total fishing effort increase by 26.4%, though it has affected the 4 studied sub-zone in different ways.

The varying intensity of exploitation described, enabled us to identify the effects that such disturbance has had on biodiversity and on the population structures exploited, both positively correlated with fishery yield (Ungaro et al. 1998; Worm et al. 2006). "Undisturbed" ecological systems (i. e. those subject only to natural dynamics), are generally in a non-equilibrium state (Huston 1979), in which environmental variation maintains a certain level of diversity between species (Huston 1979). Non-equilibrium states depend on population growth rates and frequencies of population reduction, which are closely correlated (Huston 1979). In conditions of low to intermediate frequency of population reduction, biodiversity increases, whilst high frequency of population reduction results in a decrease in diversity (Huston 1979). This suggests that fishing activity may well play a part in modifying relations between competing species, i. e. competitive relationships may be altered due to changes in space or food availability, which bring about important changes in the communities (Jennings et Polunin 1996).

The increased fishing pressure in SE Sardinia, due to fleet renewal (+120%), has caused a reduction in red shrimp biomass, resulting for some years in overexploitation (Sabatini et al. 2006). Our results show that other species associated with red shrimp have also been affected in this process, as indicated by the decrease in species richness, which has generated the heavy drop in diversity shown by the index. Put simply, the high frequency of stock reduction in this sub-zone, does not allow exploited populations to react to fishing pressure with a growth rate increase (Huston 1979). Furthermore both, the dominance index increase and the higher degree of dominance displayed in *k*-dominance curves for the last years, would indicate a rapid increase of those species with a low life cycle and high growth rate, which adapt better to high frequency of population reduction (Huston 1979; Suvapepun 1991). Such an alteration in the biodiversity suggests that a significant variation occurred in species composition in the sub area analyzed.

Also SSW Sardinia shows a higher dominance gradient for the last years of the investigated period, but only a statistically significant drop in species richness can be underlined. This data would seem to indicate that, due to an increased frequency of population reduction, there was an initial stage of overexploitation in this sub-zone (confirmed by the data registered for red shrimp) which, although less than the previous sub-zone, reveals a decrease in biomass and an increase in fish mortality (Sabatini et al. 2006). Nevertheless, a wider trawling area than that of the SE and the characteristic turnover of gulf waters make this sub-zone more resilient to the fishing effort increase (+ 22.6%) that has occurred in Sardinian seas over the last 10 years.

Northern Sardinia fleets (NE and NW) show no temporal variation, keeping a constant level during the whole period of investigation (NE) or a slight increase (+12.5 NW). No statistically significant temporal trend for biodiversity indexes was observed for these areas, leading us to surmise that slight variations observed were due to chance or to common environmental dynamics.

Thus, according to our observations, species richness index seems to be the most sensitive to fishing effort. This is particularly evident for SE and SSW sub-zone. For SE sub-zone, the most important increase in fishing effort, started in 1999, coincided with an important decrease in ln*S* index. For SSW sub-zone a constant and slow increase in fishing effort resulted in a constant decrease in species richness. While no statistical trend have been found for NE sub-zone, a correspondence in data behaviour between fishing effort and species richness can be observed: a decrease in fishing effort for 2001, resulted in a tendential decrease in ln*S* index.

Our results highlight that more pronounced and significant variations in biodiversity indexes are concentrated where fishing effort was more intense (SE and SSW Sardinia). Correlation analysis of fishing effort versus biodiversity indexes confirm as much, showing statistically significant variations for the SE (*H'*, ln*S* e *D*), and SSW sub-zones, but no evidence for the NE and NW sub-zones.

5. Conclusion

In the light of the above, the diversity index together with its components of species richness and evenness as well as dominance index, seem to differ in their reaction sensitivity to perturbation and are able to indicate changes in exploited ecosystems. Although the best method for measuring biodiversity is still under debate (Caddy & Sharp 1986; Bianchi et al. 2000; Izsàk & Papp 2000; Ricotta 2000; Magurran 2003; Rochet & Trenkel 2003), as is its reliability as a tool for ecosystem evaluation (Caddy & Giffiths 1985; Lobry 2003), it is however well documented that commercial fishery affects marine ecosystems, due to the persistent modifications on species composition, dynamics and the structure of exploited populations, which affect related ecosystems (Sharp & Csirke 1984; Pauly 1988; Fogarty and Murawski 1998; Haedrich and Barnes 1997; Bianchi et al. 2000; Loreau et al. 2001; Luck et al. 2003; Ormerod 2003, Worm et al. 2006).

According to our results, biodiversity levels, evaluated by the simultaneous estimation of all its "expressions" (entropy, species richness, evenness and dominance), allow us to detect early warnings of degrees of modifications in resilience that otherwise would pass unnoticed but enable us to predict more considerable and deleterious changes. The peculiarity of the situation in Sardinia, characterized by varied geomorphology, and monitored constantly during fleet renewal, provided favourable conditions for this type of study. Even given the limitations of the single physical - environmental context of each analyzed sub-zone, it was possible to evidence the influence of anthropic pressure on such zones and understand the extent to which it may modify ecosystem dynamics.

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Chapter 2

Statistical model for the prediction of fishery-induced changes in biodiversity of Sardinian demersal assemblages.

Statistical model for the prediction of fishery-induced changes in biodiversity of Sardinian demersal assemblages.

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Abstract – Marine biodiversity is considered as a key factor for ecosystem functioning. While is well proved that the commercial fishing affects overall biomass, size and species structure of demersal fish communities, few studies considered effects of fishery on species diversity. In the present contribution a Shannon's entropy (*H'*) temporal trend were simulated using a multiple linear regression, in order to investigate about the relationships between fishing effort and species diversity. Were considered demersal assemblages, related to red shrimp, in specific locations of Sardinian seas (Central-Western Mediterranean), which have recently undergoes increase in fishing pressure. Data come from annual MEDITS trawl surveys carried out from 1994 to 2004, concomitant to a progressive Sardinian fleet renewal. *H'* was considered as response variable, while biodiversity components and different fishing effort expressions were incorporated and tested as potential explanatory variables. The best model ($R^2 = 28.58\%$; F-test p-value = 0.0017) incorporate species richness and fishing effort expressed as number of boats per trawling area.

The fishing effort influence in shaping demersal red shrimp-related biodiversity is finally discussed.

Key words: Biodiversity / Multiple linear regression / Middle-slope assemblage / Mediterranean sea

Introduction

The high concentration of human population near coasts and the oceans' productive coastal margins, strongly affects marine ecosystems and its resources (Vitousek et al. 1997; Halpern et al. 2008). Industrial fishing in particular can have both direct and indirect consequences on marine systems, including habitat destruction, populations biomass and size modifications (Pauly 1979, 1988; Fogarty et al. 1998; Greenstreet and Hall 1996; Bianchi et al 2000), incidental mortality of non-target species, evolutionary shifts in population demographic, cascading effects on food web and changes in the function and structure of ecosystems (Botsford et al. 1997; Rogers and Ellis 2000; Worm and Myers, 2003). Moreover the exploitation process can also results in changes in marine biodiversity (Worm et al, 2005, 2006; Lotze et al. 2006; Jackson, 2001, Dulvy et al. 2003; Heip, 2003).

Species diversity, is considered a basic ecological aspect in marine ecosystems. Several studies have underlined the role of biodiversity in ecosystem functioning (Chapin et al. 2000; Loreau et al. 2001; Worm et al, 2006; Dubois et al, 2006; Hector and Bagchi 2007). Biodiversity is, in fact, considered as responsible for enhancing ecosystem stability and resiliency to disturbance and for preserving a wide variety of goods and services provided by marine ecosystems, including vital food resources for millions of people (Naeem and Li 1997; Heip, 2003; Worm et al, 2006).

In the light of the above, come out the importance of biological diversity, and the development of further tools for monitoring its status should be considered imperative. Moreover, due to its extremely sensitiveness to degradation, biodiversity can also be useful as tool to assess the health of ecosystems, indicating deleterious changes that escape analytical methods (Heip, 2003; Beever, 2006). In fact, monitoring biological diversity rather than simply few "charismatic" ecosystem components acknowledges the multi-scale nature and complexity of ecosystems and envisions a proactive rather than reactive approach to species conservation (Beever, 2006).

The great variety in the geomorphology and natural environments of Sardinian sea beds (Cau et al. 1994), together with the increased intensity of exploitation of the last ten years, which has been constantly monitored since 1994 by trawl surveys (MEDITS) (Bertrand et al. 2002), make Sardinian seas a reliable testing ground for evaluating the varying consistency in biodiversity in relation to fishing effort.

Our aim is to built a statistical model of Shannon's diversity index temporal behaviour (*H'*) via a two-explanatory variable multiple linear regression model, in order to evaluate the dependence of biodiversity on fishing effort and to provide a tool that can allow us to forecast the biodiversity behaviour in relation to one of its components and fishing effort itself. We consider data from capture depths of red shrimp in specific locations of Sardinian seas that have been subject to increasing fishing pressure over the past eleven years.

Materials and methods

The study area

Sardinia is one of the most interesting areas of Mediterranean sea Central western basin, both for geographical position and the extent of fishing bottoms (Fig. 1) (Cau et al. 1994).

Geomorphology of Sardinian sea bottoms is various. The South Western and Western seafloor is characterized by a wide continental shelf with a gradual decline that ends between 150-200 m depth and a gradual continental slope. The South Eastern and Eastern bottoms are characterized by a narrow continental shelf that terminates at about 50-100 m of depth and a really steep continental shelf (Fig. 1).

For our investigation were considered a depth interval of 400-800 m and of about 4220 km^2 (Cau et al. 1994). At the studied depths, Western and South Western sea floors are characterized by sandy muds (until 450 m) and deep muds (>500 m depth), whereas the South Eastern and Eastern bottoms are rocky, irregular and spaced out by active canyons, that allow transportation of sediments and nutrients to deeper regions. After its estimation, the model were applied on 4 sub-zones (North Western 595 km^2, North Eastern 855 km^2, South Eastern 502 km^2 and South-South Western 2268 km^2), in which Sardinian seas were subdivided, due to differences on geomorphology of sea bottom and fishing habits (Fig. 1). The Northern seas were excluded due to the lack of deep strata.

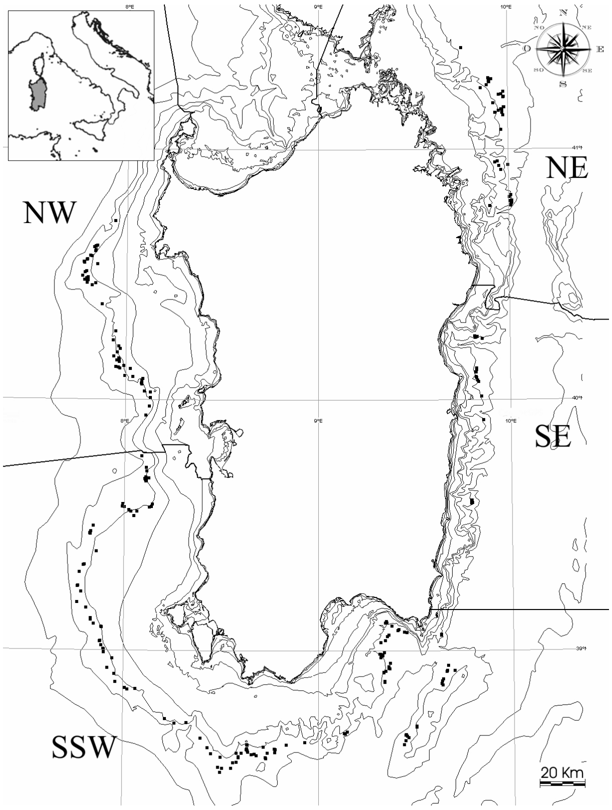

Fig. 1. – Seas around Sardinia. Location of the trawl surveys during the period 1994-2004. Depth from between 400-800 m were considered.

Data collection

Data were collected over 1994-2004 period, during eleven trawl surveys carried out during summer months, within the framework of the international MEDITS research programme. Samples were collected by means of a trawler of 168 GT, using a stratified random strategy. Fishing gear is a GOC 73, a bottom trawl (Fiorentini et al. 1999) designed for experimental fishing with scientific purpose. Its codend mesh size is 20 mm (stretched mesh) (Bertrand et al. 2002).

Were considered a total of 294 hauls (Fig. 1) of the duration of one hour each, at depths of between 400 and 800 m. In order to evaluate species strictly associated to red shrimps, were selected exclusively those hauls where Aristeus antennatus and Aristaeomorpha foliacea were present. Abundance data (N) were successively standardized per 1 h (N h-1), in order to homogenize data.

Teleosts, Chondrichthyes, Decapod crustaceans and Cephalopods captured in each haul were identified, then counted and weighed.

Fishing fleet data, used for the estimation of fishing effort, were collected thanks to Regione Autonoma della Sardegna fleet archives. These data regarded the number and the tonnage of trawlers. Were selected those boats of a tonnage >30 GT, that operate during the investigated period.

Variables.

In order to estimate multiple regression model coefficients, was necessary the computation of biodiversity and fishing effort indexes that would have used for build the model.

From abundance data, fish diversity (*H'* dependent variable) was calculated for each year and sub-zone according to the Shannon index formula (Shannon, 1948):

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$

The quantity p_i is the proportion of individuals found in the *i*th species (Pielou 1969).

Were then considered two different sets of independent variables. The first set (biodiversity set) comprised: species richness, estimated by the maximum Shannon's entropy (Ganis 1991):

$$H_{\max} = \ln S$$

evenness, computation of which has been carried on by Pielou's *J'* index (Pielou 1969), presented as the ratio of observed diversity (*H'*) and maximum entropy ($H_{\max}$):

$$J' = \frac{H'}{H_{\max}} = \frac{H'}{\ln S}$$

And dominance (*D*) that have been evaluated by the Simpson index:

$$D = \sum_{i=1}^{S} p_i^2 = \sum_{i=1}^{S} \left(\frac{n_i [n_i - 1]}{N[N-1]} \right).$$

The second set of explanatory variables (namely, fishing effort set) were estimated by developing 4 indexes of fishing effort from fishing fleet number and tonnage data. Two of them used just the number and tonnage row data and were defined respectively as N° and GT. The other two were the same indexes weighted by surface potentially available for trawling of each area, and defined as: GT Area^{-1} and N°_{boats} Area^{-1}.

Prior the multiple regression models estimation, an exploratory correlation analysis regarding all variables considered, was performed by means of Spearman rank correlation, in order to have a first idea on significant and non-significant relationships existing between all variables.

Multiple linear regression (MLR).

The multiple regression procedure was used to estimate the regression coefficients β_0, β_1, β_2, of the linear equation:

$$Y_j = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \varepsilon$$

where the regression coefficients β_0, β_1, β_2, represent the independent contributions of each independent variables X_1 X_2 and to the prediction of the dependent variable Y.

Were estimated several MLR models by combining a dependent variable of the biodiversity set (species richness, evenness and dominance) together with a variable of the fishing effort set. The selection criteria for the best model choice were: the statistical significance of the intercept and the dependent variable included in the model (t-test); the proportion of the total variability in the dependent variable that is accounted for by the regression equation (R^2); the p-values of the analysis of variance which test the whole model (F-test). The dependence of residuals were evaluated by means of the Durbin-Watson statistic (D-W statistic) (Durbin and Watson, 1951 I-II).

The model coefficients has been evaluated on the whole investigated area and then applied to single zone model estimation. Such procedure, that implied the a priori choice of intercept and regressor coefficients, allowed us to mitigate the influence of environmental variability within analysed areas in order to highlight as possible only the influence of fishing effort on examined assemblages. Moreover this allowed us to describe the diverse reactions of Shannon's entropy sub-zone by sub-zone with only one model.

The different degree of fitting between the predicted data and the single sub-zones observed data, were evaluated by Root mean squared error (RMSE), a common model performance index which gives a global idea of the difference between the observed and modelled values (Sousa et al. 2006).

Results

A total of 400790 individuals belonging to 84 families and, 154 species were collected. Teleosteans were dominant (75 species), followed by Decapod Crustaceans (38 species), Cephalopods (24 species) and Chondrichthyes (17 species) (Fig. 2). Of the 154 species captured, 40 alone represented about 98% of the total number of specimens. Crustaceans comprised 50% of all the catches and were thus the most abundant systematic class (N h^{-1}). *Aristaeomorpha foliacea* and *Aristeus antennatus* represent an important slice of the total catch (13.1%). However the most abundant single species was *Pasiphaea sivado*

(12.7% of the total catches), prevalently caught in NW and SSW sub-zones (Table 2). Teleosteans comprised 43.1% of total catches. Of these, *Chlorophthalmus agassizii*, *Gadiculus argenteus*, *Phycis blennoides* and *Hymenocephalus italicus* constituted about 36% of total captures of bony fish. Catches of Chondrichthyes and Cephalopods were fewer (respectively 5,5% and 1,3%) than those of Teleosteans. All species accounted for under 1% of the total, with the exception of *Galeus melastomus* (4.5 %).

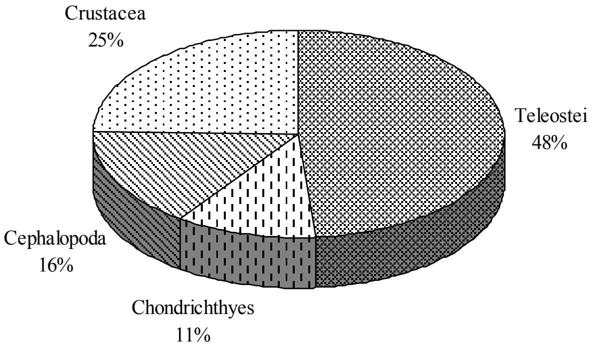

Fig. 2. – Composition of catches for middle-slope assemblage in Sardinian seas (400-800 m).

Descriptive statistics carried on both dependent and independent variables showed that the highest coefficients of variation were recorded for fishing effort set (GT and N° variables with 89% and 78.5% respectively). Regarding the biodiversity set the highest CV is for Dominance (31%). Shannon index displayed an average of 2.474 ± 0.2090 SD, with a maximum of 2.922 (NE sub-zone; 1996) and a minimum of 1.762 (NW sub-zone; 2004) (Table 1).

Table 1. – Average, standard deviation (SD), maximum, minimum and coefficient of variation of independent variables (ln*S*, *J'*; *D*, GT; N°; GT Area^{-1}; N°Area^{-1}), and Shannon index (*H'*).

TOT	*H'*	lnS	*J'*	*D*	GT	N°	GT Area^{-1}	N° Area^{-1}
Average	2.474	3.931	0.631	0.142	1055.120	15.098	0.900	0.013
SD	0.2090	0.2803	0.0559	0.0442	939.4210	11.8529	0.2935	0.0036
Min	2.061	3.401	0.530	0.081	240.480	4	0.404	0.007
Max	2.922	4.357	0.720	0.233	2996.380	38	1.571	0.022
CV(%)	8.44	7.13	8.86	31.02	89.03	78.51	32.60	26.76

The exploratory correlation analysis showed that several correlation coefficient were significant (for $P<0.05$, $P<0.01$ and $P<0.001$) (Table 2). Only the variable *J'* (evenness) showed a significant correlation with all other variables. The highest values of correlation were found mainly for analogous variables, in particular for the fishing effort set (0.94 for GT vs N°). Significant correlation values between the two different sets, were found between ln*S* and N (0.62 $P<0.001$), ln*S* and GT (0.57 $P<0.01$). The dependent variable (*H'*) showed significant correlations with ln*S*, *J'*, *D* (0.35, 0.65 and -0.93 respectively) and N°$_{boats}$ Area^{-1} (-0.34).

Table 2. – Correlation matrix between Shannon index (*H'*) and independent variables.

	N° Area^{-1}	GT Area^{-1}	N°	GT	*D*	*J'*	LnS
GT Area^{-1}	0.9016***						
N°	0.6065***	0.6541***					
GT	0.6896***	0.8111***	0.9417***				
D	0.2856	0.2317	-0.0541	0.0035			
J'	-0.4378*	-0.4603*	-0.4217*	-0.4566*	-0.7357***		
LnS	0.2016	0.2975	0.6214***	0.5711**	-0.1745	-0.452*	
H'	-0.3391*	-0.2698	0.0333	-0.0345	-0.9265***	0.6493***	0.3468*

***=P<0.001; **=P<0.01; *=P<0.05

Multiple regression analysis

Via multiple regression analysis were generated 12 different forecasts of Shannon's index *H'* time trend behaviour (Table 3). The variables from time to time tested, were species richness (ln*S*), evenness (*J'*) and dominance (*D*) (biodiversity set), together with fishing pressure expressions: GT, N°, GT Area^{-1}, N°$_{boats}$ Area^{-1} (fishing effort set). As expected the biodiversity set contribution were statistical significant in all models. Statistical significance for fishing effort set was found for GT Area^{-1} and N°$_{boats}$ Area^{-1} when in association with ln*S* and for GT and N° when in association with *J'*. None contribution from fishing effort was found when it was in association with dominance.

According to the selection criteria, the best model resulted to be #4 (Table 3), summarized by the following equation:

$$\hat{H}' = 1.470 + 0.335 \cdot \ln S - 23.146 \cdot \frac{N°_{boats}}{Area} \text{. } R^2=28.58\%$$

This is to say that for a given species richness, Shannon's entropy decrease of 23.146 for each unit increase in fishing effort. The model also underline that while species richness is positively correlated with Shannon's entropy fishing effort is inversely proportional to biodiversity.

The statistical significance of the model was confirmed by the F-test (P-value = 0.0017). The explanatory variables, species richness ln*S* (t-test P-value = 0,0028) and fishing effort N°$_{boats}$ Area^{-1} (t-test P-value = 0,0069) together with the intercept (t-test P-value = 0,0008), provide a statistical significant contribution on the Shannon's entropy estimate (Table 3). The residuals check reveals that there is no indication of serial autocorrelation in the residuals (D-W stat=2.05607 p-value=0.3405)

Besides other models (5-12) showed higher R^2 values then model #4, underlining a better explanatory power, were however excluded because at least one of the coefficient estimated is not statistical significant (NS), invalidating the suitability of those models for *H'* estimate. Models 1 and 2 have also been excluded because F-test condition has not been satisfied (Table 3).

Another model that can successfully used for estimating of Shannon's entropy is the #3 (R^2=26,58%). This model is built with the species richness as #4, and another expression of fishing effort (GT $Area^{-1}$). The whole model and the single contribution of each regressor resulted statistical significant.

Table 3. – Multiple regression model selection. Are reported: t-test of the intercept and the dependent variable included in the model (***=P<0.001; **=P<0.01; *=P<0.05; NS=P>0.05); coefficient of determination (R^2); analysis of variance F-test p-values (F).

	H'																							
Model #	***1***		***2***		***3***		***4***		***5***		***6***		***7***		***8***		***9***		***10***		***11***		***12***	
R^2	14.35%		14.27%		26.58%		28.58%		68.36%		69.74%		43.82%		43.10%		85.92%		85.96%		86.00%		85.95%	
F	3.18	NS	3.16	NS	6.88	**	7.60	**	41.06	***	43.79	***	14.82	***	14.39	***	115.97	***	116.33	***	116.67	***	116.27	***
Intercept	1.09963	NS	1.10106	NS	1.31215	**	1.4701	***	0.280973	NS	0.241323	NS	0.755622	NS	0.915049	*	3.08358	***	3.0804	***	3.12331	***	3.12552	***
ln*S*	0.359109	*	0.359895	*	0.358146	**	0.335115	**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
J'	-	-	-	-	-	-	-	-	3.27086	***	3.29981	***	2.62481	***	2.46332	***	-	-	-	-	-	-	-	-
D	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-4.34198	***	-4.33705	***	-4.32721	***	-4.31348	***
GT	-3.537E-05	NS	-	-	-	-	-	-	0.00012195	***	-	-	-	-	-	-	9.2371E-06	NS	-	-	-	-	-	-
N°	-	-	-0.0027715	NS	-	-	-	-	-	-	0.00993913	***	-	-	-	-	-	-	0.0008101	NS	-	-	-	-
GT $Area^{-1}$	-	-	-	-	-0.273259	*	-	-	-	-	-	-	0.0686261	NS	-	-	-	-	-	-	-0.035631	NS	-	-
N° $Area^{-1}$	-	-	-	-	-	-	-23.1463	**	-	-	-	-	-	-	0.317424	NS	-	-	-	-	-	-	-2.6773	NS

The model performance, i.e. its describing power, resulted to be higher in the SE sub-zone (RMSE=0.11), followed by SSW sub-zone (RMSE=0.15). Although statistical significant, the describing power resulted to be less efficient for NE and NW sub-zone (RMSE=0.23 and 0.30 respectively) (Fig. 3).

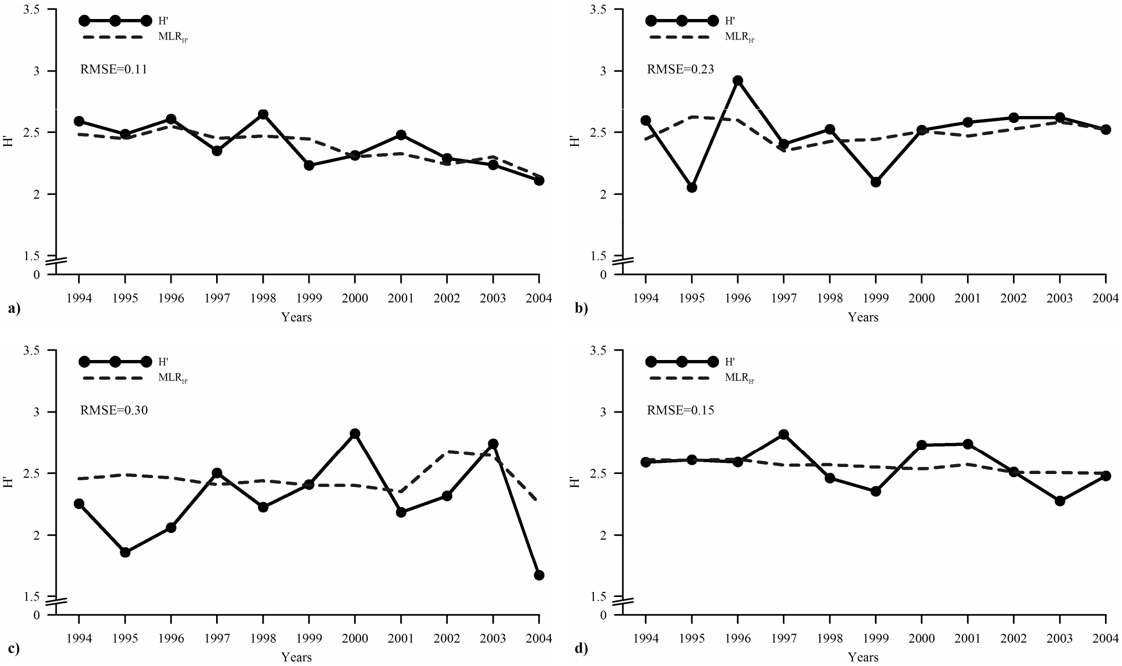

Fig. 3. – Observed (solid line) and predicted (dashed line) biodiversity a) SE, b) NE, c) NW d) SSW sub-zones

Discussion

It is now widely accepted that commercial fisheries reduces the abundance of target populations, affects life-history parameters such as growth rate and age at maturity and, in extreme circumstances, results in the local extirpation of species (Casey and Myers, 1998; Jennings and Kaiser, 1998; Pauly et al., 1998).

Trawling fisheries in particular can affect the seabed habitat (Kaiser et al. 2002) and can bring to a reduced biodiversity and a change in the community structure (Ball, Munday, & Tuck, 2000; Jennings & Reynolds, 2000; Worm et al, 2005, 2006; Lotze et al. 2006; Jackson, 2001, Dulvy et al. 2003; Heip, 2003). The diversity-stability hypothesis suggests that biodiversity provides a general insurance policy that minimizes the chance of large ecosystem changes in response to global environmental change (Heip, 2003, Ives and Carpenter, 2007). Therefore, the research will require and develop rapid assessment techniques for monitoring of marine biodiversity.

This study investigates on relationships existing between fishing effort and biodiversity, and on the possibility of carrying out a prediction of the Shannon diversity index (*H'*), starting solely from one of its components and fishing effort. The Shannon's entropy temporal trend simulation, developed by a multiple linear regression, actually displayed statistical significant relationship between species richness and a fishing effort expression used to develop the model. This result strengths what stated in a previous work by Sabatini et al. (submitted), in which due to the results obtained were hypothesized that species richness may be the most sensitive biodiversity feature (component) to the fishing effort.

The studied demersal assemblages, specifically related to red shrimp at depths of between 400 m – 800 m, showed that only about 1/4 of the 154 species caught had a high degree of dominance, representing 98% of total abundances (N h^{-1}). Bony fish species were the most numerous, followed by crustaceans, which were more abundant in terms of specimen numbers. These results concur with those reported in other studies from the Eastern and Central Mediterranean (D'Onghia et al. 2003; Colloca et al. 2003).

The high values of the coefficient of variation found for fishing effort expressions may be generated by the high variability of those data, mainly due to the renewal of fishing fleets that, last decades increase the fishing effort in seas around Sardinia of 26.4%, reaching a maximum of 120% for SE sub-area (Sabatini et al. submitted).

The exploratory correlation analysis shows a statistical significant relationship between Shannon's index and both ln*S*, and the best fishing effort regressor $N°_{boats}$ $Area^{-1}$. While the former relationship (*H'* vs. ln*S*) should be expected and explained by the tight connection between biodiversity components and entropy measure *H'* (Magurran 2004) (including also dominance), the latter (*H'* vs $N°_{boats}$ $Area^{-1}$) we think may provide a first evidence on a correlation between fishing effort and biodiversity.

Multiple regression analysis reflected and confirmed what previously observed. For all models were in fact displayed that, besides species richness, also evenness and dominance can give a significant contribute in Shannon's entropy description, furthermore with more accuracy (expressed by a better p-value). Regarding fishing effort variables set, the multiple regression analysis showed that only two explanatory variables (GT $Area^{-1}$ and $N°_{boats}$ $Area^{-1}$) of the four considered, gave a statistical contribution in describing biodiversity, furthermore only jointly to species richness.

Coefficient of determination (R^2), showed that the fitted model explains the 28.58% of the *H'* variability, underlining that changes in species richness or in fishing effort may generate, acting jointly, more than 1/4 of the change observed in Shannon's diversity. The models with a higher R^2 were, on the other side, not suitable due to their non statistical significant p-values. So the “better” describing power is actually ineffective, due to a non significant intercept (models #5 and #6) or because were generated by only one of the explanatory variables considered (models #7-#12).

The application of the model on single sub-zones and its performance measurement by means of RMSE, displayed that biodiversity of southern Sardinian seas (SE and SSW sub-zones) is better described than the northern seas one. This may highlight that for these sub-zones there should be a tighter accordance between fishing effort species richness and the statistical model can provide a better describing power of biodiversity. Therefore, the high fishing effort that occurred in these areas (Sabatini et al. submitted), should have a non negligible influence in shaping demersal red shrimp-related assemblages (Daan and Richardson 1996; O’ Brien et al. 2000). Otherwise in the Northern sub-zones, where increasing in fishing effort were not statistical significant (Sabatini et al. submitted), species richness and fishing effort showed less describing power (i.e higher values of RMSE) of Shannon’s entropy.

On the other side, a high proportion of unexplained variability of biodiversity still remains. Is hypnotizable that unexplained proportion of biodiversity variation should be due to natural environmental stocasticity. There are several causes and factors that can induce variations in biodiversity (Lobry et al. 2003; Caddy & Sharp, 1986; Caddy & Griffiths, 1995; Ungaro et al. 1998). Marine traffic accounts for just a small part of anthropic pressure on these assemblages. In Sardinian seas, it is limited to main navigation routes and areas in proximity to major ports. Also climatic changes are usually directly responsible for important environmental variations, but they apparently affect only surface layers and can only affect the deeper ones indirectly (IFREMER, 1997). Moreover Mediterranean sea in particular is characterised by high degree of environmental stability, being homeothermic below 200 - 300 m depth, with warm deep waters (13-14 °C) reaching the abyssal seafloor down to 5.050 m depth (Hopkins et al. 1989; Tyler 2003; Hopkins 2006).

Although the influence of local environmental processes cannot be excluded (local up-welling, bottom streams, active canyons) (Ricklefs, 1987; 2004; Huston 1999; Gristina et al. 2006 Hoeinghaus et al. 2007), we tried as possible to design the model specifically to isolate the effect of fishing effort, by connecting it with an environmental feature (species richness) that we consider highly sensitive to exploitation (Sabatini et al. submitted). For this reason we think that, in particular for SE and SSW zones, the fishing effort may had a non negligible influence in shaping demersal red shrimp-related biodiversity and its effect may overcame the contribution of other environmental process in describing the temporal variation of biodiversity.

Conclusions

The presence of *A. foliacea* and *A. antennatus* makes the depths ranging from 400 to about 800 m of particular commercial interest. Because of the fishing technique characteristics, trawl catches are composed not only by red shrimps, but of a highly diversified mix of fish (Colloca et al. 2003). In relation to these features, management of trawling fishing should require a multispecies approach, which considers the community and not the single species as the basic unit of the analysis (Pitcher 2000). The traditional approach, based on single stock assessment, represents an inadequate management strategy for such multispecies fishery because it does not consider the effect of interspecific relationships on the species abundance and distribution (Pitcher, 2000). In this context, we think that information on species diversity may be useful to identify those strategies that may improve fishery management.

In the light of the results, it is necessary to frame the usefulness of the model (its forecast capacity) in the context of a short–medium time period. In this framework, the model may be used to estimate the possible diversity value generated by a variation of fishing effort and a certain value of species richness and thus, due to the tight connection between biodiversity and ecosystem functioning (Worm et al, 2006; Dubois et al, 2006; Hector and Bagchi 2007), to forecast the health of marine ecosystem subject to fishery exploitation.

Although nonlinearity of fish and fishery processes is a distinctive feature of models in fisheries research, linear modelling techniques have a long and distinguished history in quantitative fisheries science (e.g. Beverton and Holt, 1957; Ricker, 1973, 1975). Our results showed that the mechanisms for diversity variations in the Sardinian red shrimp-related community have been explained, at least in part, by the multiple linear regression model we used. Our future aim may be to increase as possible the knowledge of the “unexplained part”. First of all by analysing biodiversity behaviour using other kinds of model (e.g. non linear), including also the environmental variables that mostly affect demersal biodiversity in Sardinian seas.

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Chapter 3

Assemblages in a submarine canyon: influence of depth and time.

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BIODIVERSITY IN ENCLOSED SEAS

Assemblages in a submarine canyon: influence of depth and time

**A. Sabatini · M. C. Follesa · I. Locci ·
A. A. Pendugiu · P. Pesci · A. Cau**

Abstract In this work we have studied the assemblages from Quirra canyon (Sardinia). We analysed data from 25 trawl samples from the canyon, made at different times and depth. A total of 71 demersal species (38 teleosts, 5 cartilaginous fishes, 13 molluscs and 15 crustaceans) were examined. We found four groups using cluster analysis; each group is characterised by a certain depth or time. In particular we have shown that the time of the day appears to have a role in the movement of shrimps and other species. This phenomenon seems to be linked to trophic need and by the consequent different food availability into the Canyon.

Keywords Cluster analysis · Multidimensional scaling · Submarine canyon · Fish assemblages · Depth · Mediterranean sea

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Introduction

Submarine canyons are areas rich in nutrients, due to both their geomorphology and their material and hydrographic flows; they are areas with strong turbidity currents that cause morphology changes (Shepard et al., 1974).

Although less deep than in the Atlantic Ocean, submarine canyons are also present in the Mediterranean Sea. In the seas around Sardinia a number of submarine canyons with highly variable morphology divide the upper slope, mainly along the eastern coast (Western part of Central Tyrrhenian Sea). One of these is Quirra Canyon; it slopes steeply down from a depth of 120 m to 1400 m in only 5 nautical miles, perpendicular to the coast. Fishing grounds are very irregular in this canyon, because of its morphology.

Submarine canyons are complex environments, described as unstable (Thorne-Miller & Catena, 1991) and species that live there have greater mobility than those of typical deep-sea assemblages (Rowe, 1971). Their influence on the movements of a number of species has been widely studied (Tudela et al., 2003; Sardà et al., 1997; Tursi et al., 1996, among many), but information concerning day-night movements in submarine canyons is scarce.

In this work we have studied the structure of demersal assemblages in Quirra Canyon. This canyon has not been widely studied till now and there are no works about its macro-faunal communities. This paper represents the first description of its demersal assemblages. We have also determined the diversity of assemblages related to depth and time and highlined day-night movements of the species which live in the canyon.

Materials and methods

The present work is based on data collected between April 1996 and October 1997 in the Quirra Canyon. This canyon is located in South-Eastern Sardinia (Western part of Central Tyrrhenian Sea) (Fig. 1) and is surrounded by a continental slope characterized by a marked variation in depth.

We analysed 25 samples, taken at different hours, between depths of 160 and 600 m with an 'Italian type' traw used by local fishermen; the hauls lasted between 30 and 100 min. We classified each haul according to time and depth; those made during daylight hours (from about 7 a.m. to 6 p.m.) were classified 'L' ('light'), those during the first part of the night (about 6 p.m. to 1 a.m.) were 'N1' ('night 1') while those made in the second part of the night (1 to 7 a.m.) were classified 'N2' ('night 2'). In terms of depth, 'C' is the code for hauls made down to 350 m, 'D' for those between 350 m and 500 m, and 'E' for hauls beyond 500 m.

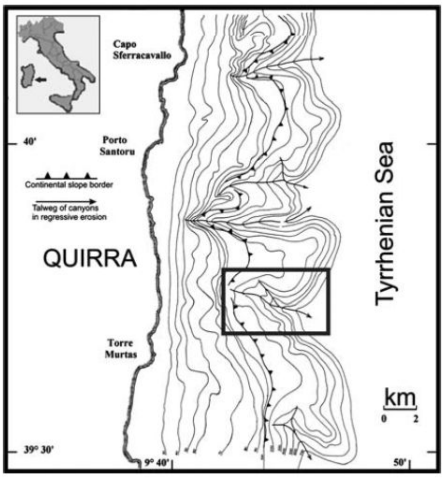


Fig. 1 Map of study area: rectangle indicates Quirra Canyon; lines represent depth curves; arrow in upper left rectangle indicates the geographical position of Quirra Canyon

All species caught in each haul were counted and weighed; these data were used to obtain biomass indexes standardised per hours of haul. Data were checked to reduce the potentially confusing effects of rare species and occasional catches of pelagic ones (Biagi et al., 2002). We accepted for the analysis only the species with an abundance of at least 10 individuals and that had occurred in a minimum of 2 hauls. In consequence, 71 species were selected for the cluster analysis.

Cluster analysis was carried out using biomass data for each species caught, according to depth (C, D and E) and time of day (L, N1 and N2). The data was 'root-root' transformed (Field et al., 1982) and subsequently the similarity between hauls was calculated using the Bray-Curtis measure (Bray & Curtis, 1957; Field et al., 1982). Samples were classified by hierarchical agglomerative cluster analysis using the group average linking method. To show the percentage contribution of each taxon to the average dissimilarity between samples of the various groups' pair combinations, we used the SIMPER (similarity percentage) analysis (Clarke, 1993; Clarke & Warwick, 1994). This procedure indicates the average contribution (%) of each species to the similarity (typifying species) and dissimilarity (discriminating species) between groups of samples. Ordination of samples was then made using multidimensional scaling techniques (MDS) (Kruskal, 1964). MDS was applied to the distances matrix generated by clustering, in order to obtain a 2-dimensional representation of hauls.

Statistical analyses were all performed using the PRIMER package (release 4.0) (Clarke & Warwick, 1994).

Results

About 12 of the 25 analysed hauls were performed in daylight, 10 during the first part of the night (N1) and 3 in the second (N2). In terms of depth range, the hauls were made between 166 m and 639 m. Specifically, 14 hauls were made in the C stratum (<350 m), 2 in the D (between 350and 500 m deep) and 9 in the E stratum (>500 m) (Table 1).

We analysed 71 species, 38 of which were teleosts, 5 cartilaginous fishes, 13 molluscs and 15 crustaceans. Most species were caught at only one particular depths, both during the day and at night, revealing their typical depth range in the Quirra Canyon (see Electronic Supplementary Material). Species such as *Aspitrigla cuculus* or *Mullus barbatus* were found in the canyon only at a depth of less than 350 m throughout the whole day. Another group is formed by those species captured beyond 500 m; they were never, at any time, found at different depth ranges (for example *Paromola cuvieri* and *Polycheles typhlops*). The other two groups identified are characterized by a wider extension: they were caught at depth up to 500 m (such as *Capros aper* or *Sepia orbignyana*) or from 350 to over 500 m (like *Stomias boa* or *Epigonus telescopus*). None of these were ever found at different depth ranges.

Moreover, there are also ubiquitous species; these were caught at all the depths examined, both during the day and at night, for example *Phycis blennoides* and *Helicolenus dactylopterus*.

More interesting, however, are those species which, although spending their lives at a specific depth range, move at a certain time of the day. In Canyon Quirra we found 9 species (including the giant red and rose shrimps, *Aristaeomorpha foliacea* and *Aristeus antennatus*) which, during the first part of the night only, are found in the C stratum, moving from a depth of more than 500 m; in the second part of the night, we find these species at depths of more than 500 m, therefore returning to their usual depth range. Furthermore, we observed that two species (*Lophius budegassa* and *Pasiphaea multidentata*) are found down to 350 m all through the day and go deeper only in the night, to all appearances moving in the opposite direction to the previous group.

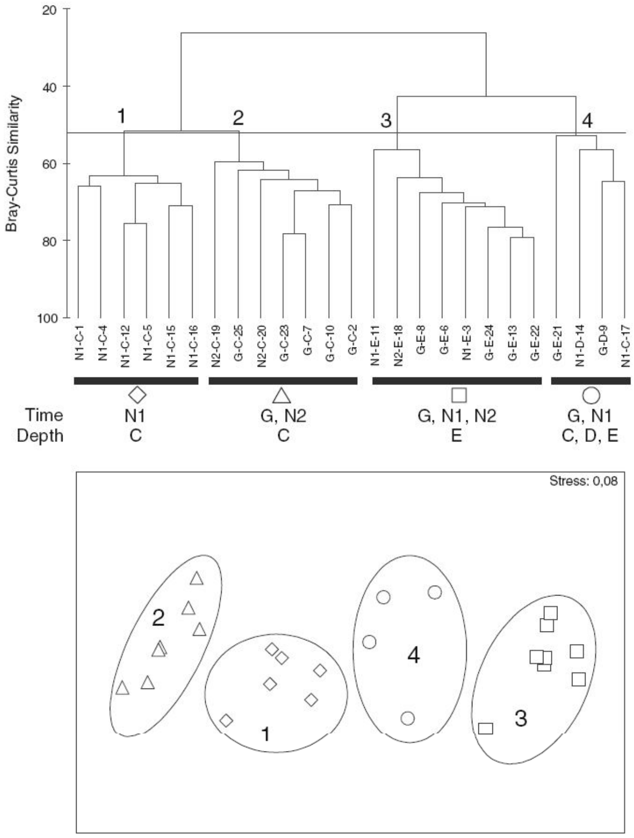
To evaluate what relation there was, in terms of depth and time of day, between the analysed hauls, we have used cluster analysis. We found four main groups (Fig. 2). The first two are composed of the hauls made in the Canyon at depths of less than 350 m: the first group (1) comprises the hauls made during the first part of the night only (N1), in which the absence of hauls made during daylight or in the second part of the night should be noted; the second (2) is made up of hauls made during the second part of the night and in daylight. The third group (3) brings together the hauls made at depths of over 500 m irrespective of time of day, while the fourth groups four hauls (4), two made in daylight, in the D and E strata (G-D-9 and G-E-21), and two others made during the first part of the night (N1) at lesser depths (N1-C-17 and N1-D-14). The four groups gather together hauls made in different seasons, so that seasonality doesn't seem to be relevant in this analysis. The ordination of the 25 hauls with MDS was in agreement with the clustering, showing the same aggregation of samplings previously observed.

SIMPER analysis revealed that the main indicator species for the first group is *Merluccius merluccius* (7.24%), for the second *Glossanodon leioglossus* (13.44%), for the third *A. antennatus* (8.84%) and for the fourth *Chlorophthalmus agassizi* (10.24%). The discriminating species between group 2 and all the others is *G. leioglossus* (with% variable from 5.76, group 2 versus group 1, to 7.24, group 2 versus group 4) (Table 2). For group 4, the discriminating species is *C. agassizi*, with 5.99% versus group 1 and 6.53% versus group 3. For the third group versus group 1 the discriminating species is *G. leioglossus* (4.86%); for this group only, the discriminating species does not coincide with that of the indicator. The analysis also shows that about 50% of the contribution to the total dissimilarity is accounted for by a minimum of 16 to a maximum of 20 species.

Discussion and conclusion

Submarine canyons are very different areas from the typical continental slope (Haedrich et al., 1975, 1980). They generally have higher productivity and a regular influx of organic matter that influences the life of species living in. It has been noted that submarine canyons in the Mediterranean

Fig. 2 Classification (cluster analysis) and ordination (multidimensional scaling ordination analysis) of species assemblages in Quirra Canyon



Sea (Stefanescu et al., 1994) exert an influence on fish assemblages, just as in the Atlantic Ocean (Haedrich et al., 1975); we studied this phenomenon in Quirra Canyon.

The analysis carried out in this paper clearly distinguishes between the fauna of the deeper waters (E stratum) and that living at less depths (C stratum). Each association appears to be characterized by an indicator species which typifies the group. The first group is characterised by hauls made in the C stratum during the first part of the night only; indicator species for this group appeared to be *M. merluccius*. In fact, we found this species at all the depths analysed but, above all, in the C stratum. As regards the second group, the indicator species is *G. leioglossus*. This group brings together the hauls made at depths of <350 m during the day only or in the second part of the night. This species was found in the C stratum at all hours of testing. The third group gathers together the hauls made at greater depths, in the E stratum. The species characterising this group are those which typically live at greater depths, hence the indicator species for this group is *A. antennatus*. The fourth group collects together different kind of hauls, some made in the E stratum during the day and others made at lesser depths during the night only. The indicator species for this group is *C. agassizi*. Other species that characterize this group are *A. antennatus* and

Table 1 Code, date, season, start and end time (solar time), last and depth for each hauls studied

Code	Date	Season	Start sampling	End sampling	Last (minutes)	Depth (m)	Belonging group	
N1-C-1	22/04/1996	Spring	21,48	22,28	40	198	N1	C
G-C-2	08/05/1996	Spring	16,05	17,30	85	206	G	C
N1-E-3	08/05/1996	Spring	18,50	19,50	60	595	N1	E
N1-C-4	08/05/1996	Spring	22,00	23,18	78	212	N1	C
N1-C-5	29/07/1996	Summer	0,01	1,03	62	200	N1	C
G-E-6	29/07/1996	Summer	17,30	18,28	58	590	G	E
G-C-7	29/07/1996	Summer	19,36	20,07	31	170	G	C
G-E-8	27/08/1996	Summer	13,55	15,10	75	586	G	E
G-D-9	27/08/1996	Summer	16,15	17,10	55	420	G	D
G-C-10	27/08/1996	Summer	18,05	19,20	75	231	G	C
N1-E-11	27/08/1996	Summer	20,30	21,45	75	587	N1	E
N1-C-12	27/08/1996	Summer	22,50	23,59	69	208	N1	C
G-E-13	05/11/1996	Autumn	13,35	14,34	59	583	G	E
N1-D-14	11/02/1997	Winter	19,55	20,31	36	411	N1	D
N1-C-15	11/02/1997	Winter	21,24	22,30	66	205	N1	C
N1-C-16	11/02/1997	Winter	23,26	24,10	44	166	N1	C
N1-C-17	20/05/1997	Spring	22,34	24,00	86	308	N1	C
N2-E-18	21/05/1997	Spring	0,56	1,50	54	599	N2	E
N2-C-19	21/05/1997	Spring	3,18	4,05	47	173	N2	C
N2-C-20	21/05/1997	Spring	4,50	6,30	100	198	N2	C
G-E-21	21/05/1997	Spring	7,42	9,00	78	519	G	E
G-E-22	24/06/1997	Summer	8,25	9,25	60	580	G	E
G-C-23	24/06/1997	Summer	10,38	11,08	30	188	G	C
G-E-24	13/10/1997	Autumn	8,50	10,02	72	639	G	E
G-C-25	13/10/1997	Autumn	11,25	12,05	40	247	G	C

A. foliacea. Movement patterns for these species have been examined by other authors (Tudela et al., 2003; Sardà et al., 1997; Tursi et al., 1996) who have noted the ability of deep-water shrimps to adapt their life cycle to the morphology of a canyon. In this work we have shown that shrimps and other species inside Quirra Canyon move during a day-night cycle, probably with light as a parameter indirectly related to their activity, as proposed by Tobar & Sardà (1992). These species are mainly predators (Gristina et al., 1992; Fischer et al., 1987) moving during the night to catch prey. They find the canyon a suitable place for finding food, in which they can reach different depths in a relative short distance; this is what probably happens in Quirra Canyon.

In fact, this Canyon is characterised by a regressive erosion (Palomba & Ulzega, 1984) that is gradually modifying its morphology, bringing the limit of its borders closer and closer to the coast. A great movement of materials inside Quirra Canyon makes it, in a certain way, an ''active canyon''. This probably causes transportation of sediments rich in organic matter from the shelf to deeper regions, and even diel migratory plankton from upper levels, in the same way seen for other canyons (Macquart-Moulin & Patriti, 1993).

This phenomenon, together with the fact that in a canyon it is possible to reach different depths with minimum distances, determines the variations in species assemblages with time and depth observed in this paper. Observed movements do not seem to be primarily linked to seasonality in Quirra Canyon, although a number of nocturnal movements have been attributed primarily to hydrographical phenomenon due to seasons (Sardà et al., 1997; Cartes et al., 1994; Sardà et al., 1994; Cau & Deiana, 1982; Bombace, 1975, Maurin, 1960). The results of our work show that movements of species inside the Canyon, and so the different species' compositions observed, seems to be linked to a day-night cycle, and probably by the consequent different food availability, as suggested by Cau & Deiana (1982) and Stefanescu et al. (1994).

Table 2 Between group comparison, results of SIMPER analysis

Species	Group	Group	Cum. %	Species	Group	Group	Cum. %
Av. diss. = 48.53	2	1		Av. diss = 52.17	4	1	
Glossanodon leioglossus	2.73	1.58	5.76	*Chlorophthalmus agassizi*	1.88	0.30	5.99
Aristeus antennatus	0.00	1.17	10.67	*Raja clavata*	0.11	1.16	9.91
Aristaeomorpha foliacea	0.00	1.16	15.48	*Glossanodon leioglossus*	0.77	1.58	13.47
Zeus faber	0.99	0.29	19.09	*Galeus melastomus*	1.03	0.33	16.77
Plesionika edwardsii	0.00	0.8	22.46	*Aspitrigla cuculus*	0.00	0.83	19.92
Gnathophis mystax	0.18	0.77	25.5	*Gnathophis mystax*	0.00	0.77	22.82
Mullus barbatus	0.74	0.18	28.24	*Capros aper*	0.76	0.63	25.69
Raja clavata	1.69	1.16	30.93	*Merluccius merluccius*	0.66	1.36	28.45
Calappa granulata	0.00	0.61	33.49	*Plesionika martia*	0.80	0.10	31.11
Dipturus oxyrinchus	0.37	0.57	35.97	*Scyliorhinus canicula*	0.94	0.93	33.67
Scyliorhinus canicula	1.49	0.93	38.29	*Nephrops norvegicus*	0.75	0.06	36.22
Trigla lyra	0.74	0.65	40.47	*Dipturus oxyrinchus*	0.48	0.57	38.65
Lepidorhombus boscii	0.21	0.57	42.53	*Trigla lyra*	0.16	0.65	40.91
Argentina sphyraena	0.32	0.61	44.53	*Plesionika edwardsii*	0.51	0.80	43.12
Loligo forbesi	0.19	0.42	46.52	*Gadiculus argenteus*	0.91	0.47	45.20
Scaeurgus unicirrhus	0.52	0.07	48.49	*Micromesistius poutassou*	0.69	0.41	47.25
Lepidotrigla cavillone	0.41	0.37	50.45	*Macrorhamphosus scolopax*	0.00	0.54	49.29
Av. diss = 88.1	2	3		Av. diss = 69.34	2	4	
Glossanodon leioglossus	2.73	0.09	7.24	*Glossanodon leioglossus*	2.73	0.77	5.83
Raja clavata	1.69	0.05	11.79	*Chlorophthalmus agassizi*	0.09	1.88	11.20
Scyliorhinus canicula	1.49	0.00	15.89	*Raja clavata*	1.69	0.11	15.95
Galeus melastomus	0.00	1.30	19.49	*Galeus melastomus*	0.00	1.03	19.13
Aristeus antennatus	0.00	1.28	23.04	*Zeus faber*	0.99	0.00	22.10
Phycis blennoides	0.08	1.12	25.90	*Aspitrigla cuculus*	0.93	0.00	24.92
Zeus faber	0.99	0.00	28.63	*Gadiculus argenteus*	0.00	0.91	27.64
Aristaeomorpha foliacea	0.00	0.97	31.34	*Aristaeomorpha foliacea*	0.00	0.90	30.35
Aspitrigla cuculus	0.93	0.00	33.92	*Scyliorhinus canicula*	1.49	0.94	32.92
Lophius budegassa	1.02	0.11	36.45	*Aristeus antennatus*	0.00	0.85	35.49
Illex coindetii	1.00	0.22	38.69	*Coelorhynchus coelorhynchus*	0.00	0.84	38.02
Capros aper	0.81	0.00	40.91	*Merluccius merluccius*	1.46	0.66	40.47
Macrorhamphosus scolopax	0.80	0.00	43.11	*Macrorhamphosus scolopax*	0.80	0.00	42.87
Lampanyctus crocodilus	0.00	0.75	45.18	*Plesionika martia*	0.00	0.80	45.25
Sepia orbignyana	0.75	0.00	47.23	*Phycis blennoides*	0.08	0.85	47.56
Eledone cirrhosa	0.79	0.07	49.27	*Capros aper*	0.81	0.76	49.84
Av. diss = 57.25	4	3		Av. diss = 70.76	3	1	
Chlorophthalmus agassizi	1.88	0.33	6.53	*Glossanodon leioglossus*	0.09	1.58	4.86
Scyliorhinus canicula	0.94	0.00	10.29	*Raja clavata*	0.05	1.16	8.40
Gadiculus argenteus	0.91	0.00	14.02	*Galeus melastomus*	1.30	0.33	11.78
Capros aper	0.76	0.00	17.23	*Scyliorhinus canicula*	0.00	0.93	14.78
Glossanodon leioglossus	0.77	0.09	20.32	*Aspitrigla cuculus*	0.00	0.83	17.46
Lampanyctus crocodilus	0.00	0.75	23.41	*Plesionika edwardsii*	0.00	0.80	20.06
Lepidorhombus whiffiagonis	0.73	0.07	26.22	*Lophius budegassa*	0.11	0.89	22.62
Galeus melastomus	1.03	1.30	28.95	*Gnathophis mystax*	0.00	0.77	25.09
Eledone cirrhosa	0.73	0.07	31.67	*Phycis blennoides*	1.12	0.37	27.48
Argentina sphyraena	0.59	0.00	34.18	*Etmopterus spinax*	0.72	0.00	29.79
Etmopterus spinax	0.16	0.72	36.63	*Lepidorhombus whiffiagonis*	0.07	0.79	32.09
Illex coindetii	0.78	0.22	39.04	*Lampanyctus crocodilus*	0.75	0.07	34.29
Merluccius merluccius	0.66	0.84	41.35	*Trigla lyra*	0.00	0.65	36.38
Helicolenus dactylopterus	0.66	0.64	43.55	*Capros aper*	0.00	0.63	38.40
Micromesistius poutassou	0.69	0.46	45.74	*Calappa granulata*	0.00	0.61	40.37
Plesionika edwardsii	0.51	0.00	47.91	*Argentina sphyraena*	0.00	0.61	42.34
Paromola cuvieri	0.00	0.52	50.01	*Plesionika martia*	0.66	0.10	44.23
				Nephrops norvegicus	0.63	0.06	46.12
				Hoplostethus mediterraneus	0.59	0.00	48.01
				Merluccius merluccius	0.84	1.36	49.89

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SECOND PART

Land-based influences on the Atlantic Bluefin Tuna

Chapter 4

Effects of local scale perturbations in the Atlantic bluefin tuna (Thunnus thynnus L.) trap fishery of Sardinia (W. Mediterranean).

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Effects of local scale perturbations in the Atlantic bluefin tuna (*Thunnus thynnus* L.) trap fishery of Sardinia (W. Mediterranean)

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Abstract

Time series (1825–1973) of the Atlantic bluefin tuna, *Thunnus thynnus*, landings from the trap fishery of Sardinia (W. Mediterranean) were analyzed to identify (i) patterns of catch variability in the historical traps of Isola Piana, Portoscuso and Porto Paglia, and (ii) effects of three local environmental variables: Mining, Run-off and Sea Surface Temperature.

We applied different time series analysis techniques: auto-correlation functions, spectral analysis and Dynamic Factor Analysis (DFA). Moreover, an asymmetrical design using a Before/After and Control/Impact location approach was used to test the perturbations due to mining processes that have characterized the fishery area.

A sharp decreasing trend of Porto Paglia trap landings beginning in 1880, coincided with mechanized mining. This result was confirmed by the ANOVAs contrasting Before vs. After ($P < 0.01$) and Controls vs. Impact location ($P < 0.01$).

We found a slight cyclical behaviour in the same time series. Spectral analysis highlighted that pattern synchronized with the Run-off cycles with periods corresponding to 7 years. We hypothesized that the effect of Run-off from watershed mine tailing dams creates a pulsing boundary 'reflective' effect for tuna migration schools on the in-shore trap of Porto Paglia, resulting in a periodic oscillation of captures.

We have identified (i) a reversal of trends in captures between Portoscuso and Isola Piana traps and (ii) significant differences of the monthly distribution of capture events and timing of landings ($P < 0.01$) with a higher proportion now in May, rather than in June as in the past. We assume there is a relationship with climatic changes because warmer waters can induce precocity in gonad maturation. DFA did not show any significant differences among environmental factors and landings probably because the cyclical component in Run-off is masked. We emphasize the importance of Mediterranean traditional traps as 'sentinels' of the homing reproductive behaviour of bluefin tuna and we argue the case for a plan to link the remaining traps in a common monitoring network.

Keywords: Atlantic bluefin tuna; *Thunnus thynnus*; Trap fishery; Time series analysis; Mediterranean Sea

1. Introduction

Atlantic bluefin tuna, BFT (*Thunnus thynnus* L.), undertake a feeding and spawning migration, covering distances of several thousands of miles primarily over the northern Atlantic Ocean, Gulf of Mexico, and Mediterranean Sea (Mather et al., 1967, 1973; Farber and Lee, 1980; Lutcavage et al., 1999; Block et al., 2001, 2005). These migrations are genetically controlled, but the timing and routes can be conditioned by a large variety of factors: environmental conditions, bottom topography and oceanic currents, availability of food, population density, and anthropogenic perturbations (Cort, 1990; Sharp, 2001; Sarà et al., 2007). Knowledge and understanding of what controls the behaviour, spatial distribution and movements of tuna schools is scant, but there are direct observation and anecdotal data on migration patterns from professional fishermen and scientific studies (Mather et al., 1967; Sarà, 1998), the use of pop-up and archival tags (Lutcavage et al., 1999; De Metrio et al., 2002; Block et al., 2005; Wilson et al., 2005) and descriptive models (Royer et al., 2005; Gutenkunst et al., 2006). Bluefin tuna show steady 'homing behaviour' prior to spawning when schools move in the same general direction, but this is weak or non-

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existent in the feeding grounds when schools move in apparently random directions searching for prey (Lutcavage et al., 1999).

Traditional traps (known as 'tonnara' in Italy) serve as passive gears that have documented the 'natal homing' of BFT from the 12th century to today (Cury, 1994; Sarà, 1998; Cury et al., 1998; Doumenge, 1998). Information from the traps has been the basis of many scientific studies that have resulted in a better understanding of the biology and ecology of BFT (ICCAT, 2006). Prior to the end of the 1960s, when purse seines and long-lines were introduced, nearly all Mediterranean BFT landings data were provided by trap fisheries, while information on the interactions of the population with other fishing gears was basically absent (ICCAT, 1997). Traditional traps harvest the ancestral migratory flow of tunas at a fixed site and therefore, in the absence of interference of other gear types, trap landings represent a good proxy for variation in population abundance through time (Addis et al., 1997; Fromentin et al., 2000a; Ravier and Fromentin, 2001).

There is variability in trap landings data; therefore, it is important to identify the sources. The variability has been evaluated on large geographical scales with some evidence of synchronous fluctuations over periods of about 100–120 years which are considered directly related to global environmental variables (Ravier and Fromentin, 2004).

On a smaller geographical scale, historical documentation based on anecdotal data, evidence such as environmental conditions may account for trap landing variability (Parona, 1919; Neuparth, 1923; Sella, 1929; Lozano Cabo, 1958; Rodriguez-Roda, 1966). The documentation of human perturbations on landings in traditional traps is rather scant, with the exception of the micro scale effect of boat noise (commercial shipping) on BFT behaviour shown by Sarà et al. (2007). According to Lemos and Gomes (2004), local weather and coastal ocean variability are the probable sources of short-term oscillations in trap catches. Another possible explanation of variability might be so called boundary effects, or the well-recognized fact that tunas tend to concentrate on boundary fronts in the ocean (Murphy, 1959; Blackburn, 1965; Sund et al., 1981; Laurs et al., 1984; Fielder and Bernard, 1987). It is also possible that they are negatively impacted by other interfaces in the coastal zone. According to Hubbard et al. (2004), environmental boundaries can affect pelagic migration patterns. Their definition of boundaries included specific isotherms, confines between warm and cool water masses, specific depth contour lines and bodies of water with different physical and chemical characteristics.

We think this knowledge gap should be addressed in order to better understand the relationship between this important fishery resource and the rapidly developing coastal zone environment. It is imperative to protect ancient fishing practices with a fisheries management and environmental policy based upon the application of rigorous science.

Fig. 1. Historical map of Sardinia with the location of 23 traps (T, active; †, stopped) in the 19th century (Angotzi, 1901). The black arrow specifies the trap sites studied.

Sardinian trap fisheries have a long history (Dean et al., 2003) and flourished in the 15th century during the Spanish occupation (Cetti, 1777). The historical position of 20 Sardinian traps (Fig. 1) confirms that BFT migration occurred only along the western coast (Angotzi, 1901). There have been no captures using experimental gear sets along the eastern and southern coasts in the past (Angotzi, 1901). The south-western region of Sardinia has been the location of one of the most important Italian trap fisheries since the late 15th century and fishing continues to this day with three active traps (Addis et al., 2006).

For at least 2000 years this region was also historically important for lead, silver and copper mining (Manconi, 1986). The area was prized as a colony by Carthage and Rome due to its metal resources, including zinc and barium. Manconi (1986) documents the history and evolution of industrial development, including shipping, based upon the mineral resources of the region and also accounts for its demise in the mid-20th century. South-western Sardinia's mining areas produced about 15 Mton of metal, which makes the region comparable to Ireland in total metal production (Salminen et al., 2005). The cumulative impacts of environmental hazards in south-western Sardinia resulting from mining and industrial activities is well documented: on land (Leita et al., 1989; Caboi et al., 1993; Boni et al., 1999; ISS, 2002), in coastal areas (Caredda et al., 1999; Pagano et al., 2002), and in human health risk factors (Cardia et al., 1989; Sanna et al., 2002; Cocco et al., 1996). However, there is little information available on the impact of such factors on pelagic marine fisheries. The migration of the Atlantic bluefin tuna in coastal waters is genetically controlled. As the bluefin make their spawning migration past Sardinia, it is reasonable to consider local perturbations, generated by social and economic events and environmental changes, as disruptive to the pathways of bluefin schools and thus account for variability in the Mediterranean trap captures.

The aim of the present paper is (i) to investigate patterns of catch variability in three historical traps of south-western of

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Sardinia, (ii) to test for mining perturbation effects on impact and control trap sites using a BACI experimental design (Green, 1979; Underwood, 1994), and (iii) to analyze the interactions between trap landings and multiple environmental variables using a Dynamic Factor Analysis (DFA) (Zuur et al., 2007).

2. Materials and methods

2.1. Study sites and gear description

The traps examined in this study were Portoscuso (PS) (39°14′N; 08°22′E), Isola Piana (IP) (39°11′N; 08°18′E), and Porto Paglia (PP) (39°16′N; 08°25′E) (Fig. 2).

The fishing gear used is classified as 'tonnara di corsa' (arrival trap) because BFT are captured along their pre-spawning migration route and with ripening gonads (Rodrıguez-Roda, 1967; Olla, 2000). The traps consist of nylon nets with a stretched mesh of 25 cm × 25 cm, arranged from east to west in 5 rooms or chambers: the 'grande', the 'bordonaro', the 'bastardo', the 'camera' and the 'camera della morte' (death chamber: DC) (Fig. 3). Only the DC has a net mesh 'floor'. The series of chambers together form what is called the 'castle', and the tail is perpendicular to the trap and almost reaches the coastline. The traps are deployed at similar water depth and bottom topography, from 10 m at the end of the tail to maximum of 42 m at the castle. All traps have a similar layout with regard to the nets, tail and chamber orientation and management organization. The sea bottom consists of a wide, gently sloping shelf in the case of the traps at Porto Paglia and Portoscuso, and is narrow and less gently sloping at the Isola Piana trap. The fact that the traps share common features is particularly significant because this avoids having to standardize data for unit of effort due to gear differences. For at least 1000 years, these traps have been fished in the same site and the only variation in gear is the use steel anchors and artificial fibres for the ropes and nets.

2.2. Response variables

Our response variable, *Catch*, consists of annual landings from 1825 to 1973 of BFT captured in each trap (Table 1).

2.3. Explanatory variables

Three explanatory variables were used to test the effects on BFT landings. The ***Mining*** (M) variable used yearly mining production data for the period 1850–1973 in the high activity mining area or Mining Hot Spot, as shown in Fig. 2. The ***Run-off*** variable (RO) used the overall run-off in the winter months for the period 1918–1973 as an index of the hydrological processes that drain the tailing reservoirs, dams and other water output from mining activities feeding into local streams that in turn flow into the shore face of the trap area. The ***Sea Surface Temperature*** (SST) was used to index anomalies in monthly temperature variations from 1880 to 1973. We used the mean of monthly SST (April–May–June) based on data available in the area between latitude 38°N and longitude 08°E with a spatial resolution of 3°. Data were obtained from ICOADS (http://www.dss.ucar.edu). Detailed characteristics of the overall data sets used and their sources are reported in Table 1.

2.4. Data exploration

The shape of *Catch* and explanatory series measured over time is expressed as lattice graphs (Zuur et al., 2007). Data auto-

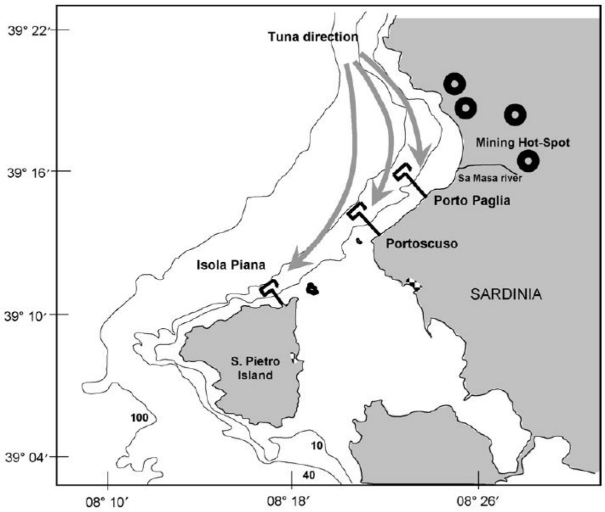

Fig. 2. South-western region of Sardinia where the traps are deployed (Black circles = major mine sites according the Geologic Service of Italy).

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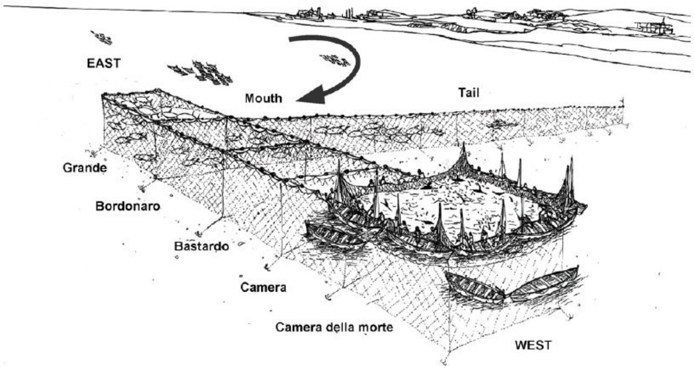

Fig. 3. Scheme of the traditional trap of Isola Piana ('arrival trap') composed by 5 chambers (modified with permission by J.L. Cort).

correlation functions (ACF) and Partial ACF were calculated to obtain insight into the strength of the relationship over time for each time series (Zuur et al., 2007). ACF gives an indication of the amount of association between variable Y_t and Y_{t-k}, where the time lag k takes the values 1, 2, 3, etc. Thus, with a time lag of k years, the ACF represents the overall association between time points that are k years apart. The correlation coefficient is used to quantify the association, and always consists of values between −1 and 1 (Diggle, 1990). A slow moving auto-correlation function indicates the presence of a trend, whereas an oscillating auto-correlation is evidence of a cyclical pattern in the time series (Zuur et al., 2007). In such cases, the patterns of data were studied using single spectral analysis (Platt and Denman, 1975) which is a method that cannot account for missing values and requires that the series be stationarized through a sequence of transformation steps (Legendre and Legendre, 1998). Periodograms were smoothed with a Parzen window to identify spectral densities with the highest significance of contribution to the cyclical events (Parzen, 1961).

We also investigated the monthly catch distributions found in historical archives data (Table 1) (April = 4; May = 5; June = 6) in three separate time series (1 = 1829–1844; 2 = 1950–1973; 3 = 1991–2004.) Data were initially explored with a box-and-whiskers plot (Month × Period), after which Generalized Linear Modelling (GLM) was used to analyze variance to test the effect of factor Month and the interaction Month × Period (Ortiz de Urbina and de la Serna, 2003).

2.5. *BACI design: detection of mining perturbation*

According to Underwood (1994) an impact is more readily detected as a different pattern of statistical interaction from Before to After it starts, between the Impact and Control locations than it would be by merely examining data from the Control locations. This experimental design, Before–After, Control–Impact (BACI) enabled us to test whether the long-term effect of industrial mining was a potential source of perturbation that correlated with trap landings. The 'Before–After' time split of the environmental perturbation was chosen on the basis of historical documentation from which we know that large mechanized mining operations were established after 1870 (Sella, 1871) and that pumping stations to prevent mine gallery flooding were installed in the late 1880s. We therefore selected the period 1825–1887 as 'Before' and 1888–1960 as 'After' (Fig. 4). We chose the Porto Paglia trap as the putatively disturbed 'Impact' location because of its position close to the mining "Hot Spot" and where there was riverine output from the mines. Portoscuso and Isola Piana were selected as "test Control locations" because of their increasing distance from the riverine output and the Porto Paglia trap (Fig. 1).

A 2-way analysis of variance was performed using a modelled set of data to verify spatial-temporal effects on BFT landings. Data were transformed with a $\ln(x+1)$ to normalize distribution and equalize variances (Snedecor and Cochran, 1967). Prior to performing the ANOVA, we tested for heterogeneity of variance

Table 1
List of the response and explorative variables utilized

Variable	Period	*n*	Remarks	Source
IP-PS-PP	1825–1973	149	Number of BFT × year	Addis et al. (1997)
IP-PS	1829–1844	15	Number of BFT × month	Private records, Ligure Sarda Spa
IP-PS	1950–1973	24	Number of BFT × month	Private records, Ligure Sarda Spa
IP-PS	1991–2004	13	Number of BFT × month	Addis et al. (2006)
M	1832–1979	147	Tons of lead + zinc	Mining Archive, Iglesias
RO	1918–1973	56	Run-off index	Cao-Pinna (1998)
SST	1880–1973	93	Sea Surface Temperature	http://dss.ucar.edu

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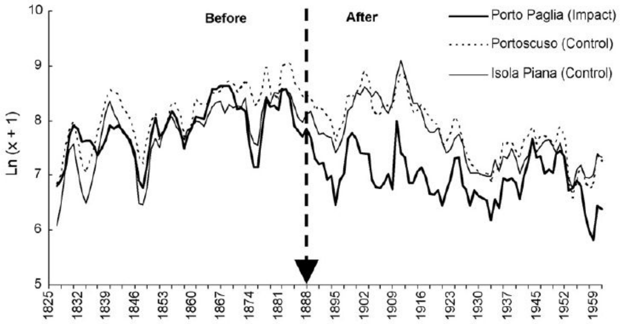

Fig. 4. Data modelling for the BACI design with two controls and one impacted site.

with Cochran's *C*-test, while Scheffé post hoc test allowed us to identify the interaction effects. Statistica 7.0 (StatSoft, Inc.) was used to perform ANOVAs.

2.6. Relationships: Catch vs. explanatory variables

Our data consists of a response variable, *Catch* (PS, IP, PP), and multiple explanatory variables (M, RO and SST), and the question is whether there is a relationship between the two and which of the explanatory variables shows significance. We used the dimension reduction technique of Dynamic Factor Analysis (Zuur et al., 2003a,b), which is useful to analyze short, non-stationary time series containing missing values. The main advantage in this method is that time-series are modelled as a function of (i) a linear combination of common trends, (ii) a constant level parameter, (iii) one or more explanatory variables, and (iv) noise (Zuur and Pierce, 2004).

We used four sets of dynamic factor models (Table 2) that considered the three trap series modelled (i) as linear combinations of one common trend, (ii) with or without explanatory variables, and (iii) with a diagonal or non-diagonal error covariance matrix R.

We applied DFA to the longer range of the available *Catch* and explanatory variables series from 1880 to 1973 ($n = 93$). Because the RO series begins in 1918 and the SST series has many missing values in the 1880–1910 time frame, we also conducted the DFA in the 1918–1973 ($n = 56$) data set. In that period there are only seven missing values in the SST series and the *Catch* series has a gap in 1943 because of WW II. Results from the DFA were interpreted in terms of the estimated parameters factor loadings and match between model estimation and observed values. The smallest AIC indicates the most appropriate model (Zuur et al., 2003a,b). The AIC tries to find a balance between a measure of fit (maximum likelihood) and the number of parameters (number of trends, explanatory variables and structure of R). If more parameters are used (i.e. trends or number of explanatory variables), the model fit is better, but the penalty for extra parameters is higher as well (Zuur et al., 2003a,b). The influence or weight of each explanatory variable on the response series is given by the regression parameters.

A detailed description and application of DFA to univariate and multivariate fishery data series is given by Erzini (2005), Erzini et al. (2005), Zuur and Pierce (2004), Zuur et al. (2003a,b). Data were processed using the software package Brodgar v. 2.0.6 (http://www.brodgar.com).

Table 2
Different dynamic factor models applied to the BFT landings of Sardinia trap fishery

N	Model	R
1	3 trap series = constant + M common trends + noise	Diagonal
2	3 trap series = constant + M common trends + explanatory variable + noise	Diagonal
3	3 trap series = constant + M common trends + noise	Non-diagonal
4	3 trap series = constant + M common trends + explanatory variable + noise	Non-diagonal

3. Results

3.1. Time series analysis

The three traps used in our study harvested more 1.14 million bluefin tuna in the period 1825–1973. Numerical captures [log $(x + 1)$ transformed] between periods revealed that during 1825–1973 Portoscuso was the most productive trap with 7.84 ± 0.77 (S.D.) followed by Isola Piana with 7.66 ± 0.71 (S.D.) and Porto Paglia with 7.26 ± 0.82 (S.D.). A reverse pattern was observed for the recent period, 1991–2004, since Isola Piana with 6.64 ± 0.82 (S.D.) had a higher yield than Portoscuso with 5.08 ± 1.46 (S.D.). Porto Paglia is not included in this analysis because it stopped operating in 1973 due to a lack of fish which made it economically unviable.

The overall BFT catch series varied considerably, and also showed clear differences in patterns over short time periods between traps. A general overview of data revealed a common decreasing trend for Isola Piana and Portoscuso traps starting very late in the 19th century. A similar trend but starting earlier,

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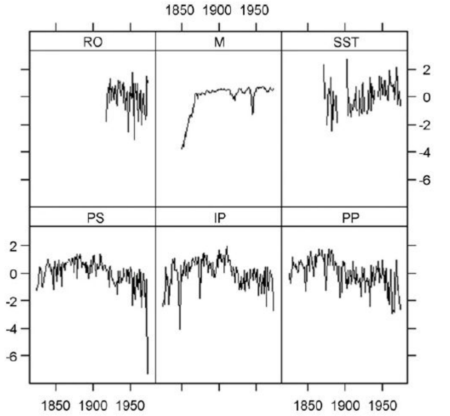

Fig. 5. Time series for three trap landings (PS, IP, PP) and the explorative (M, RO and SST) variables standardized.

exists for the Porto Paglia trap (Fig. 5). The explorative variables showed considerable variability, mainly for *Run-off* and SST, while *Mining* is characterized by an increasing trend in the period 1832–1870 and consistent level, with minor fluctuations in succeeding periods of production.

The ACF plots (dotted lines in each figures represent 95% confidence intervals) shows a slow moving shape and the presence of a high value (trend component) for the first time lags *k* for all series, except for the Porto Paglia landing series and the explanatory variables *Run-off* and SST (Fig. 6).

In detail, Partial ACF (calculated on 30% of total number of observations) for the trap series showed a significant autocorrelation with time lag of 7 years for Porto Paglia and nearly significant for Portoscuso with time lag of 8 years, and no-significant for the Isola Piana trap.

The prior results indicate the presence of a well-defined trend pattern for the series of Isola Piana, and cyclic signal for Porto Paglia the *Run-off* and SST variables. Finally, Portoscuso seems to be characterized mainly by a linear trend with a slight cyclic pattern signal appearing.

We applied a single spectral Fourier analysis to PP, PS and *Run-off*, SST series with the aim of identifying the spectral density that contributes most to the overall periodic behaviour of the series. The single periodograms for PP and PS showed a high spectral density for low frequency that can be interpreted as the basal 'noise' due to the trend pattern, and a second lower peak corresponding to time intervals of 7 years for PP. Because this result showed a very close value with *Run-off* spectral density (period = 6) and SST (period = 7.5), we then used a cross-spectrum analysis (bivariate Fourier analysis) to test for periodicity in the catch data. The results show that landings of PP and PS are consistent with the *Run-off* and SST cycles. Cross-

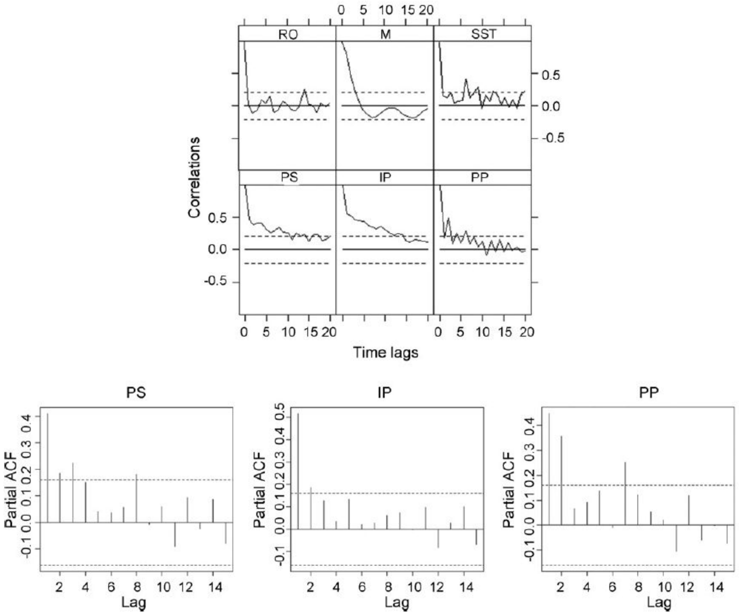

Fig. 6. ACF plots for the trap landings and the explorative variables and Partial ACF for the three trap landings.

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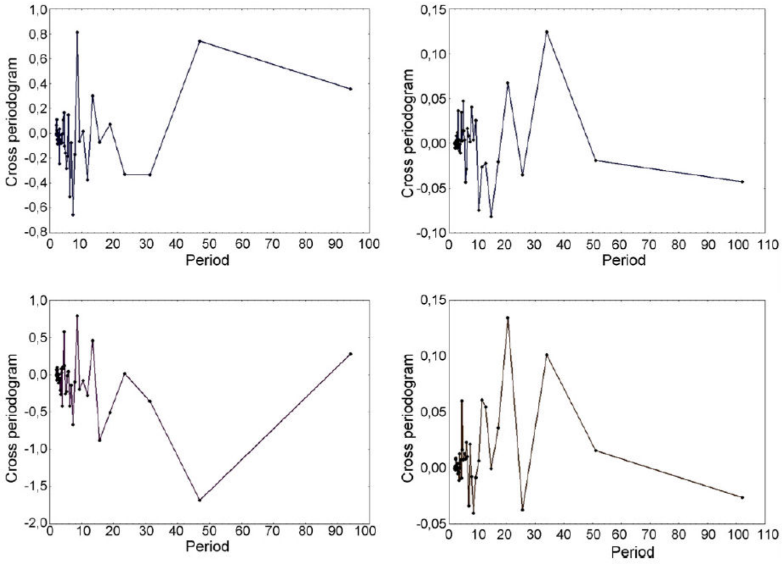

Fig. 7. Plots of the cross-peridogram between the Porto Paglia series vs. Run-off and SST variables (above); Portoscuso vs. Run-off and SST (below).

periodograms (Fig. 7) possess a high frequency component for the combination of PP–*Run-off* (0.8) at the period of 8 years. The PS–*Run-off* combination does not revealed a well-defined peak. Cross-periodograms between the trap series of PP, PS and SST did not show a synchronized component as frequency values were very low.

Analysis of monthly catch distribution with box-and-whiskers plots yielded the following patterns: (i) there were no catches in April during the periods 1829–1844 and 1950–1973 but there were catches in the most recent period 1991–2004, (ii) there was a higher catch in June in the period 1829–1844, (iii) there was an equal distribution of catches in the May–June period of 1950–1973 and finally (iv) there was a higher catch proportion in May in the period 1991–2004 (Fig. 8). Results of the Generalized Linear Modelling showed statistically significant differences between Periods and the interference effect of Period × Month (Table 3). In the long term, the three data series analyzed led us to conclude there was a shift from June to May in the majority of captures and a further shift to the end of April over the last 15 years.

Table 3
Results of GLM testing the effects of monthly distribution of captures between periods

Source	Estimate	S.E.	*z* Value	*Pr*(>|*z*|)
Intercept	−1.621	1.511	−1.072	0.283
Period	1.482	0.639	2.320	0.020
Month	0.545	0.272	2.001	0.048
Period × Month	−0.275	0.115	−2.381	0.017

3.2. *BACI design*

The analysis of spatial-temporal effects on BFT landings, modelled according to the BACI design, highlights the impact on trap captures upon the establishment of mechanized mining industries. The effect is evident in the contrast between Before vs. After ($P<0.01$) and Test Controls vs. Impact ($P<0.01$) (Table 4). Specifically, Scheffé's post hoc test highlighted no differences in the mean landings in the testing Control location

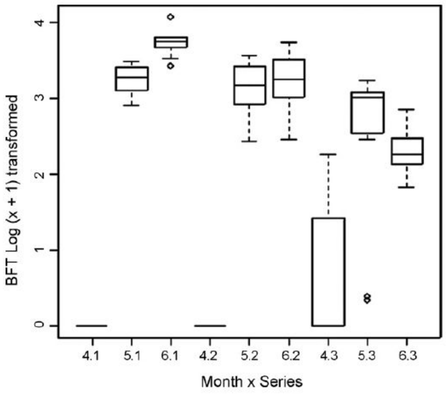

Fig. 8. Box-plots of trap captures distributed by month (4 = April; 5 = May; 6 = June) of three time series (1 = 1829–1844; 2 = 1950–1974; 3 = 1991–2004).

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Table 4
Results of the ANOVA of model data from a single Impact (Porto Paglia) and two Controls locations (Isola Piana and Portoscuso) sampled Before and After mechanized mines processes (ns = no significant difference; $^{*}P<0.05$; $^{**}P<0.01$; $^{***}P<0.001$)

Source of variation	d.f.	MS	F-ratio	P-value
Before vs. After = B	1	0.480331	56.284	**
Control vs. Impact = C	2	0.310184	36.346	**
B × C	1	0.212898	24.946	**

Cochran's C-test: n.s.

Table 5
Results of the post hoc test

Source of variation		{1}	{2}	{3}
Before vs. Control	{1}			
Before vs. Impact	{2}	0.9185		
After vs. Control	{3}	0.1955	0.7862	
After vs. Impact	{4}	***	***	***

"Before" compared to "After" the disturbance began (Table 5). There were highly significant differences on the landings at the Impact location during the same time period. Results clearly showed that in the late 1880s, some source of disturbance affected the landings of BFT trap captures at the location of Porto Paglia.

3.3. Relationships: Dynamic Factor Analysis

In the first model (Table 2), the three trap series were modelled as constant plus a linear combination of one and two common trends and a noise term. We subsequently added explanatory variables which tested all combinations with common trends. For the last two models we used the same approach described above, except that the covariance was modelled as symmetric non-diagonal matrix.

All AIC values of all models containing a non-diagonal matrix R were smaller than those with a diagonal error covariance matrix. Thus, there is some information in the residuals of trap series that cannot be explained with the explanatory variables, nor with the common trends.

The application of DFA to the longest series (period 1880–1973) resulted in the best fit model without explorative variables and two common trends (AIC = 512).

The two smoothing trends are reported in Fig. 9 which included the cross-correlations parameters between the trap series and the common trends. The elements of the error covariance matrix showed the highest value for PP (0.64) followed by similar values for PS (0.33) and IP (0.35). Large values for the PP time series indicated a pattern that is not fitted by the model.

The addition of explorative variables did not improve the goodness-of-fit as the lowest AIC value was the model with RO (AIC = 521).

DFA was successively applied to the series of 1918–1973, which has the fewest number of missing values. Of the models with no explanatory variables, the best fit was for those models with one common trend (AIC = 400) (Table 6). The best fit model with only one explanatory variable was that with RO (1 common trend, AIC = 399; 2 common trends, AIC = 403) followed by that with M (1 common trend, AIC = 404; 2 common trends, AIC = 408). The smallest AIC values with the combination of two variables were found in the RO and M models (1 common trend, AIC = 404; 2 common trends, AIC = 408). As the AIC values show, the contribution of RO in the model with explanatory variables is only one unit less than model with only response variables. Although the choice could be considered speculative,

Table 6
Period 1918–1973, values of Akaike's information criterion (AIC) for DFA models with two common trends and three different sets of explanatory variables

	Non-diagonal matrix[a]		Diagonal matrix[a]	
M number of trends	1	2	1	2
Common trends + noise				
PS-PP-IP	400	404	411	414
Common trends + explanatory variable(s) + noise				
RO–M–SST	409	413	421	426
M–SST	410	414	420	424
RO–SST	405	409	417	420
RO–M	404	408	417	421
RO	399	403	412	416
M	404	408	416	420
SST	405	409	419	415

[a] Model.

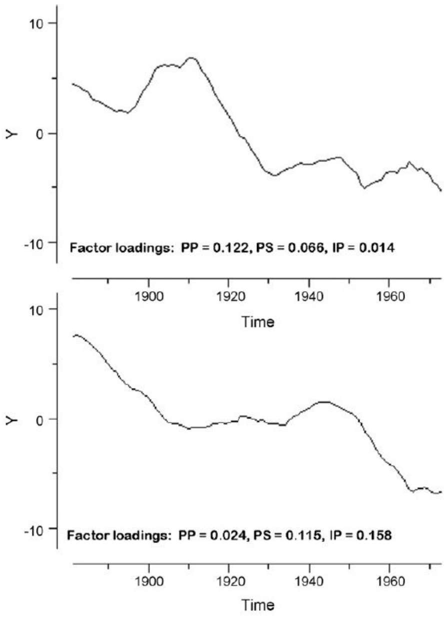

Fig. 9. Period 1880–1973, two main common trends for the dynamic factor model without explanatory variables.

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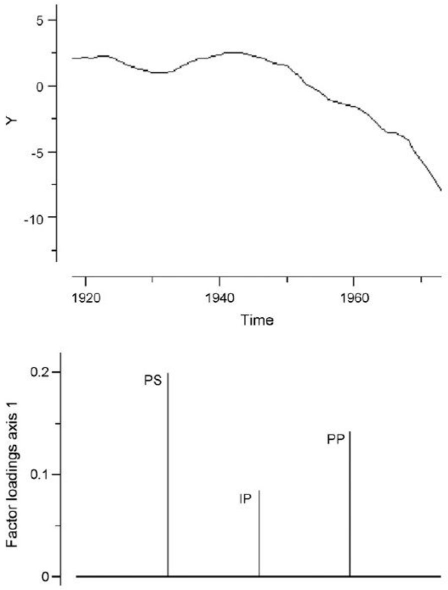

Fig. 10. Period 1918–1973, common trend and factor loadings for the dynamic factor model containing the Run-off variable.

we analyzed different validation elements of the model with one common trend and the combination of the variables RO.

The estimated common trend (Fig. 10) is characterized by a steady shape in the 1918–1925 years, a small decrease in the period 1926–1930, followed by an increase to a maximum in 1942 and a progressive decrease later.

The bi-plot of factor loadings shows that all series are positively correlated with the common trend but PS (0.20) followed by PP (0.14) have the best fit because of their relatively larger values which are higher than IP (0.084) (Fig. 10).

The estimated *t*-values (in absolute sense) for the regression for each trap series and RO variables, were respectively -0.97 ± 0.13 for the PP series, -1.59 ± 0.14 for the PS, and -2.66 ± 0.09 for the IP.

Elements of the error covariance matrix were relatively large for PS (0.90) and PP (0.80), indicating that those time series did not fit well. The smallest value was for the IP series (0.45). Large diagonal elements of the error covariance matrix R indicate that the time series do not have a good fit and the common pattern obtained is "forced" by the model.

4. Discussion

Bluefin tuna on their spawning migration pass through the western basin of the Mediterranean along the waters of the western coast of Sardinia in late spring in a north, north-west → south direction. They swim at a depth of 20 m or below, as confirmed by the historical trap sites (Angotzi, 1901) and the occurrence of specimens caught in the trap-tails.

One of the main questions that motivated this study was to test if the environmental variables available on a local scale interfered with trap fishery landings over time.

From current literature, three main patterns are focused on the theoretical approach used to explain the variability of fish abundances: (i) a trend or basal component, which is the most marked change over time; (ii) an irregular component that is random with no constant cause or exceptional events, which can however modify trends in the short and long term; (iii) (quasi-) periodic signals, seasonal or cyclical, repeated at regular and constant intervals (Caddy and Gulland, 1983).

The trap landings data we analyzed with the ACF and Partial ACF approaches showed a clear trend for all of the traps. While the Porto Paglia series indicated a cyclical behaviour, there was less evidence for any cyclical response for the Portoscuso trap. When we analyzed trap series and explorative variables together using bivariate spectral analysis, the Porto Paglia and *Run-off* series showed good fit, as indicated by the cross-periodograms. That response enabled us to hypothesize and test for a 'pulsing' pattern in the captures of Porto Paglia as a result of the *Run-off* effect. Furthermore, we assume that the *Run-off* effect on trap landings was the result of water draining from mine tailing drainage into local rivers in the coastal watershed close to the trap site. The small cyclic signal highlighted for the Portoscuso series could be explained as a residual *Run-off* effect due to the spatial relationships with the mining Hot Spot.

We found evidence that the perturbation due to tailing discharges from abandoned mines continues to exist today because indices of contamination, mainly heavy metals, are found both in the soils and waters in the watershed, in foreshore sediments and in the seagrass *Posidonia oceanica* (Caredda et al., 1999; Di Gregorio and Massoli-Novelli, 1992).

According to a long-term Italian program on environmental impact assessment the abandoned mines, tailing dams and industrial manufacturing processes, have created an environmental hazard in the waters on the south-western coast of Sardinia (ISS, 2002). Part of the study area was classified as 'a high risk environmental crisis' (Decree 349/1986) and 'a high technological risk' (Decree 334/1999). The area has subsequently been incorporated into a National Restoration Plan by the Italian Ministry for the Environment.

The work carried out on the Environmental Plan should benefit the Southwester Sardinian trap fishery, particularly for the Portoscuso and the Porto Paglia traps. Success will be verified in the near future because the Porto Paglia trap will be deployed again in renewed fishing tests.

Also, the asymmetrical BACI design provided consistent evidence that the effects of mechanized mining can be presumed to be 'exceptional events' that sharply modified the landings trend of Porto Paglia captures from the end of 1880 onwards.

We also tested the effects of the three explorative variables with the Dynamic Factor Analysis technique. The model including three *Catch* series and *Run-off* gave the lower AIC, but because that value is very close to the model with only the *Catch* variables, its effect seems to be negligible. It is possible

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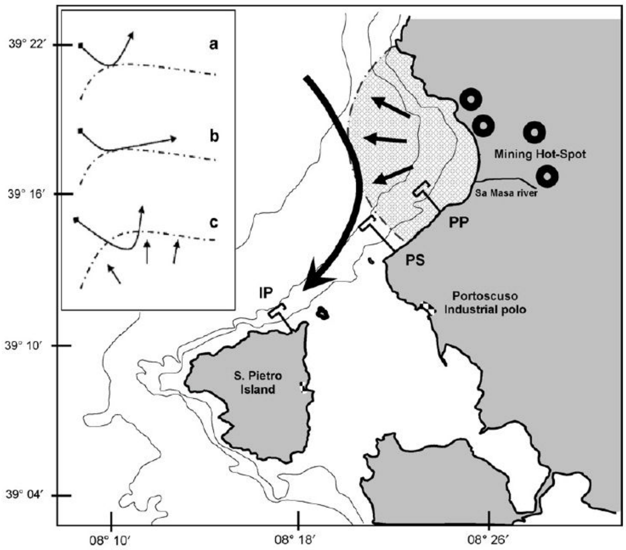

Fig. 11. Scheme of the reflective boundary condition supposed in the in-shore trap of Porto Paglia.

that due the high variability in *Run-off* over the long time period the model is confounded. It is worth noting that with dynamic factor models, if seasonal or cyclic components are present in the time series, they will be masked and included in the trend component.

Our conclusion is that a modification of the water masses and sediment occurred in the area where the Porto Paglia trap was deployed, which resulted in the formation of a 'pulse boundary' with an intra-decadal fluctuation of approximately 7-years, within the periods analyzed (Fig. 11).

The phenomenon of the intra-decadal fluctuations of traps yields and their sources is a process described in the Portuguese trap of "Medo das Casas" by Neuparth (1923) and it was subsequently investigated in the trap at "Barril" by Lemos and Gomes (2004). According to those authors, the 8-year signal found by a regression model was probably a response to local meteorological and hydrographical variability. It must be pointed out that they underline the weakness of the model estimates and conclusions due to the fact that only a single trap was analyzed.

Our 'pulse boundary' hypothesis has more fidelity because we analyzed three traps deployed close to each other with standardized gear, harvesting practices and management. Moreover, there is historical documentation on the mine pollution occurrences and documents on conflicts between trap and mine managers (Mazzarelli, 1917; Grassi, 1913).

According to Hubbard et al. (2004) boundary effects on fish migration can result from three types of conditions: (a) tangential, (b) reflective, and (c) repulsion conditions (Fig. 11a–c).

We think that the recurring formation of that 'pulse boundary' led to a 'reflective or pushed back condition' of bluefin tuna migration schools (Fig. 11). Tuna migration pathways were progressively altered as fish avoided the boundary and moved out of the path causing a progressive decrease in the catch yield in the in-shore trap or Impacted site (located wholly inside the boundary, the tangential condition), namely Porto Paglia. Coincidentally with that shift in behaviour, there was no effect on the Isola Piana trap that was deployed in an off-shore isle (S. Pietro Island). Portoscuso and Isola Piana are the test sites for this analysis, but while Isola Piana can be considered a traditional experimental control site (the no impact site), Portoscuso, due to its middle position, seems to have a similar response, but at a reduced level, as those environmental insults that affect the trap at Porto Paglia. That interpretation is also strengthened by the reversal of the trend in captures between Portoscuso and Isola Piana. In the period 1992–2007 Isola Piana has a higher and steady production than Portoscuso, in contrast to the historical record.

We also documented a change in the monthly distribution of captures by testing periodicity of landings in three historical phases: (i) (1829–1844) a high percentage of captures in June and longer fishing season in the past, (ii) (1950–1973) an equal distribution of captures between May and June, (iii) (1991–2004) a high percentage of landings in May than in June and a shorter fishing season. We can assume this to be an indicator of phenological modification. Phenological changes in the migration of pelagic fish as a consequence of climate change are not easy to define, due mainly to the obvious difficulties of

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direct observation. However, there is evidence that higher sea-surface temperatures in spring contribute to earlier movements of migratory marine fish species to spawning grounds (Ware and Tanasichuk, 1989; Carscadden et al., 1997). Analyses of long-term phenological patterns have shown climatic warming accounts for earlier dates of migration to breeding sites and earlier egg-laying for birds, and also early flowering of plants (Crick et al., 1997; Forchhammer et al., 1998; McCleery and Perrins, 1998; Crick and Sparks, 1999; Inouye et al., 2000; Both and Visser, 2001; Post et al., 2001).

Because sea temperatures have risen in the oceans over the past three decades, and recent predictions indicate a 0.5–4 °C increase in North Atlantic sea-surface temperatures over the next century (Hulme et al., 2002), it is possible that the timing and location of peak abundance of fish can be altered (Sims et al., 2004). Tuna and other fishes have temperature-dependent gonadal development (Ware and Tanasichuk, 1989) and so become ready to spawn, and hence migrate, earlier in warmer years. These facts probably help explain the shift in the periods of trap capture we have observed. Peak landings 100 years ago took place in late May and early June. Today, all the landings are in May and recently there have even been captures in April. We found some fragmentary documentation on trap income and captures in historical archives that document landings also occurring in early July during the early 19th century. In the last 15 years, during which we have carried out a long-term investigation on BFT ecology (Cau, 2000), we have found a very high percentage of bluefin of both sexes fully mature in early May and some are mature at the end of April (Olla, 2000).

Following the closure of the mining industries after 1970, a new industrial area was opened near Portoscuso trap (Fig. 11), which represented another source of anthropogenic perturbation in the coastal zone. In 1975, 'red sludges' from the aluminium manufacturing retention basin residue were released into the surrounding coastal waters. As a result, the trap fishery collapsed for that year and the following 3 years. The trap operator and fishermen brought legal action against the manufacturing company.

From available literature we found further historical documentation on anthropogenic perturbations in the Mediterranean trap fisheries. Specifically, Mazzarelli (1917) reported no captures of bluefin tuna occurring in the Tunisian trap of Sidi Daoud during excavation work in Tunis harbor as a result of the increase in the turbidity in coastal waters from sediments. Negative effects were frequently recorded in Libyan traps during seasons with a strong Ghibli (wind from south-east) that transports sand from the desert (Grassi, 1913). It is well known that bluefin tuna are very sensitive to sediments in coastal waters (Lemos and Gomes, 2004) and large mortality of tuna occur in the "tuna farming" industry when sediments occur in the grow out pens (Munday and Hallegraeff, 1997).

According to Sarà et al. (2007) acoustic noise due to shipping traffic cause tuna schools to alter their schooling behaviour in the trap at Favignana (north-west Sicily), historically a highly productive trap (Sarà, 1998). We did not directly measure a negative effect on tuna landings in Sardinia due to ships transiting in the area. We did document an increase in ship traffic from the industrial harbor of Portoscuso from port records. Before industrialization began (in the period 1950–1968) a mean of 97 ships during April–June entered and left the port, which meant approximately 1 ship per day. In recent years shipping traffic to and from the harbor of Portoscuso has progressively increased. Subsequently, in the period 1969–2005, the average number of ships recorded was 230 (or 2.5 ships per day), with a maximum of 385 ships in 1989 (4.3 ships/day). Over 80,000 vessels pass through the Straits of Gibraltar and enter the Mediterranean Sea each year (De Stephanis et al., 2000). The expected growth of trade and exchange with new economy countries (i.e. China, India, Brazil), gas and oil exploration would markedly increase the traffic of vessels movements and thus increase the impact due to noise pollution on fish communities. In 2008, a gas pipeline from Algeria to Sardinia will cross very close to the trap area. This will in all probability dramatically increase the traffic of vessels and will impact fish due to noise pollution affecting fish communities. There is a significant lack of information on this topic and it needs more and immediate research.

During more than 15 years of direct observations we noted other local sources of variation in the annual and daily captures by traps. These sources of disturbance can have small or large spatial scales, for some of which no investigation has been conducted. For example, we have little knowledge of the impact of the purse seine fleets on the migration routes of BFT or the vulnerability of tuna schools that appear to be distressed by the purse seine nets. In the past, trap fisherman raised many concerns about the presence of drift net vessels, long-line gear and purse seine fleets off Sardinia because the industrial fishing boats fish very aggressively along the bluefin migration routes.

Traditional traps are an excellent example of an ecologically compatible fishing technology because they have provided a sustainable bluefin tuna fishery for over 2000 years. It is only in the very recent past (the last 100 years) that the population has suffered significant declines (Dean et al., 2003). The traps should be utilized as bluefin tuna 'sentinel' gear and as a source of significant ecological information.

The employment of the few remaining traps to achieve research objectives (i.e. a long-term monitoring program) would lead to a better understanding of the behaviour, physiology and ecology of the bluefin tuna, the status of the stock and the relationship of this great fish to future climatic changes.

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Chapter 5

Body temperature of the Atlantic bluefin tuna (Thunnus thynnus L.) in the Western Mediterranean.

Body temperature of the Atlantic bluefin tuna (*Thunnus thynnus* L.) in the Western Mediterranean

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Abstract – This study documents body temperature in the Atlantic bluefin tuna (*Thunnus thynnus* L.) in the Mediterranean Sea and temperature variability caused by the stress of capture. The investigation was carried out in the traditional trap (*tonnara*) of Isola Piana (Sardinia, W Mediterranean) where body temperature recording were conducted on free-swimming bluefin confined in the system of nets known as "*camere*" or chambers. We tracked the body temperature of two bluefin tuna (214 and 191cm CFL) using a commercial data logger (HOBO U12, Onset Computer Corporation), under two conditions: the *pre-fishing* phase, when specimens confined in the "*camera di ponente*" are undisturbed and the *fishing* phase when bluefin tuna are trapped in the "*camera della morte*" and undergo the stress of confinement and capture (*mattanza*). Body temperature increased by about 2 °C during the "*mattanza*", whereas no temperature variation was exhibited during the *pre-fishing* phase. The heat transfer coefficient (K), calculated for both bluefin tuna during the "*mattanza*", revealed a rapid increase in heat transfer. Additional data on ambient temperature T_a, white muscle T_w (n=65; 110 - 287 cm CFL) and red muscle temperature T_r, (n=249; 107 - 287 cm CFL) were obtained from live fish during angling operations, and excess body temperature (T_x=T_r-T_a) was calculated. Mean red muscle temperature was 27.6 ± 1.48 °C in an ambient temperature of 18.9 ± 0.84 °C. The excess red muscle temperature T_x was 8.21 - 9.10 °C, and the red muscle was 2.4 ± 1.78 °C warmer than white muscle.

Key-words*:* *Thunnus thynnus*, bluefin tuna, body temperature, harvesting condition, Mediterranean sea

Introduction

The Atlantic bluefin tuna, *Thunnus thynnus* L. has the capacity to conserve metabolic heat via vascular counter current heat exchangers (*retia mirabilia*), and can maintain a steady-state body temperature that is greater than ambient water temperature (Carey and Teal, 1969; Carey and Gibson, 1983). This physiological specialization, known as regional endothermy, is hypothesized to enhance tuna performance (Brill, 1996), independent of environment temperature, and enables bluefin tuna to expand their vertical and latitudinal thermal range (Graham and Dickson, 2001).

The basic anatomical and physiological features of regional endothermy of the tribe *Thunnini* (family Scombridae), such as yellowfin and bluefin tunas, have been discussed by Carey and Teal (1966; 1969), Carey et al. (1971; 1984), Carey and Lawson (1973), Graham (1975), Stevens et al. (2000). Carey's pioneering research carried out in the Western Atlantic, was based on temperatures measured with a thermistor in dead fish, and demonstrated the capacity of giant bluefin to thermoregulate through significant ranges in ambient temperature. Moreover, using acoustic tags placed in the stomach, Carey also made the first observations of significant visceral warming following a meal, and on the basis of this data he suggested that the elevated temperature would serve to speed digestion and allow tuna to feed frequently. Since then, other studies on free-swimming wild fish, using electronic tags, have provided more comprehensive information on the eco-physiology and behaviour of bluefin tuna, including data on thermoregulatory biology over a large geographical scale (e.,g., Lutcavage et al., 2000; Marcinek et al., 2001; Brill et al., 2002) and evidence that bluefin tuna maintain muscle temperature at about 24 °C, significantly higher than the ambient temperatures of their high latitude feeding grounds in summer months. Large individuals with archival tags placed in the peritoneal cavity experienced a wide range of environmental temperatures (2.8° to 30.6°C) but maintained relatively constant internal peritoneal temperature (25°C) and a thermal excess up to 21°C above ambient temperature Block et al. (2001). Significant differences in body temperature were also documented over years, monitoring, including periods when tagged fish were present in the Gulf of Mexico, a known spawning area (Teo et al. 2007). Most of the literature available concerns the Atlantic area; the study of body temperature for bluefin tuna in the Mediterranean sea only regards juvenile fish tagged in the Adriatic sea (Yamashita and Miyabe, 2001) while data for adults is still lacking.

In the present study, part of a broader research project on the ecology of *T. thynnus*, we provide data on body temperature obtained from free-swimming bluefin tuna , tagged with a temperature logger in the chambers of a traditional Mediterranean trap, or "*tonnara*". Trap fisheries in Sardinia (W Mediterranean, Italy) have a long history and flourished in the 15th century during the Spanish occupation (Dean et al., 2003). At the present time Sardinian traps provide an annual yield of about 100 tons (± 58) of bluefin tuna (Addis et al., 2006), of which approximately 70% are exported to the Japanese market.

The Mediterranean trap is a passive capture method that relies upon the natural swimming behaviour of the tuna on their reproductive migration routes along the Mediterranean coast. Information from tuna studied in these traps has increased understanding of the biology and ecology of bluefin tuna (ICCAT, 2006). Traps have proved very useful in conducting tag and release operations (De Metrio, 2001), in investigating tuna behaviour (Sarà, 2007) and time series analyses of catches (Ravier and Fromentin, 2001, 2004; Lemos and Gomes, 2004; Addis et al., 2008). Once the bluefin enter into the chamber they tend to adjust quickly to captivity, and display low levels of activity, and are thus considered to be in a low stress condition. In contrast, during fishing operations, or the "*mattanza*", the tuna are withdrawn from the water by raising the bottom of the net mesh floor and are then hoisted up onto the fishing vessel *via* hooks. The act of confining the fish in a small space to be killed presumably produces a high degree of stress, and a cascade of metabolic processes that deplete the oxygen reserves in the muscles. The normal aerobic glycolysis of glycogen sugar cannot take place to allow the release of energy resources in the form of ATP molecules, and muscles resort to anaerobic glycolysis for the production of ATP. This process leads to an accumulation of lactic acid with a reduction in the pH of the muscles, and a drop in the energy resources thereof (Soto et al., 2006). Moreover, the increase in swimming speed, a consequence of a high tail beat frequency observed during fishing operations, results in increased metabolic heat within the slow-twitch muscle (red) and contributes to body temperature elevation (Blank et al., 2007).

In this paper we have endeavoured to use body temperature as an indicator of stress, ranging from the presumed "minimal stress" condition of free swimming bluefin tuna in the chambers to the high degree of stress associated with fishing operations in a traditional trap in Sardinia. We also provide data on white (T_w) and red muscle (T_r) temperature and the excess of mean red muscle temperature compared to mean ambient water temperature (T_a) (T_x=T_r-T_a) for individual bluefin tuna.

Material and Methods

Description of the tonnara

Our investigation was carried out at the "Tonnara of Isola Piana" (39° 11'N - 08° 18'E), western Sardinia. This trap is classified as an "arrival trap" because bluefin tuna are captured during their migration along the pre-spawning route when individuals typically have ripening gonads (Corriero et al., 2005). The gear consists of nylon nets arranged in 5 chambers known as the "*grande*", the "*bordonaro*", the *"bastardo*", the "*camera di ponente*" and the "*camera della morte*" (death chamber), all connected by vertically-moving "net doors" (Fig. 1). Bluefin tuna enter naturally through the mouth of the trap (a one-way system of nets) located in the "*grande*" and subsequently cross naturally from east to west chambers when fishermen open the net doors. Only the death chamber has a net mesh "floor" used to draw up bluefin tuna during the last phase of fishing ("*mattanza*"). The series of chambers together form what is known as the "castle"; the tail is perpendicular to the trap and almost reaches the coastline. The trap is deployed at a depth of 10 m at the end of the tail to a maximum of 42 m at the castle. Our experiments were carried out in the "*camera di ponente*" (45 x 36 m) and the "*camera della morte*" (45 x 40 m).

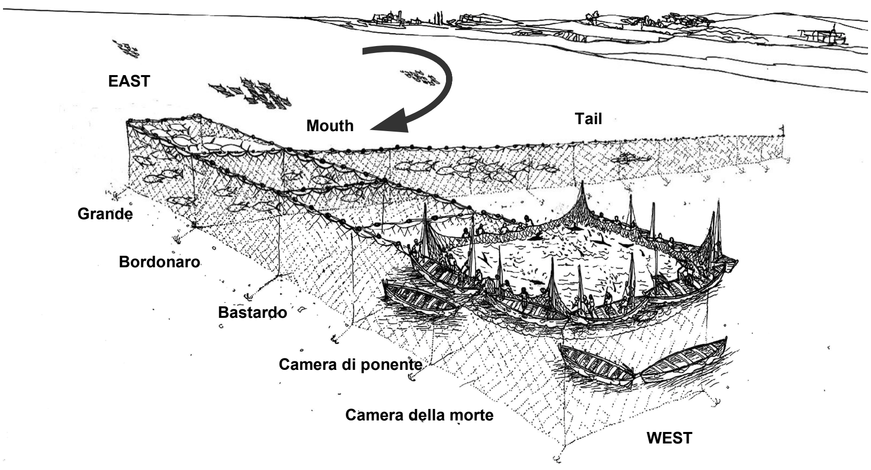

Figure 1. – Scheme of the traditional trap of Isola Piana composed by 5 chambers deployed from East to West, and a tail perpendicular to the chamber system that almost reaches the coast line (with permission by J.L. Cort).

Body temperature measurements

We monitored body temperature (T_b) continuously in two free-swimming giant bluefin (160 kg, 214 cm CFL; 117 kg, 191 cm CFL) from the "*camera di ponente*" to the "*camera della morte*", under two experimental conditions chosen in accordance with the timing of the fishing operations:

- *Pre-fishing* phase, or the time between the tagging operations (i.e. the placing of the temperature data logger) and the start of net raising operations, in which a steady state heat balance is assumed (Graham and Dickson 2001).
- *Fishing* phase, or the time between raising the net mesh floor and the actual "*mattanza*", in which tunas react vigorously to capture and confinement, where a non-steady state heat balance is assumed (Graham and Dickson 2001).

We used a commercial temperature data logger shot into the fish from a distance of about 2 m. In order to estimate the amount of heat generated during the non-steady state condition, we calculated the body heat transfer coefficient (K) of each tagged bluefin assuming that tunas maintain a mean swimming speed constant. We applied a heat budget model which considers the relationship between body temperature (T_b) and ambient temperature (T_a) (Holland et al., 1992; Brill, 1994b):

$$\frac{dT_b}{dt} = K(T_a - T_b) + \overset{g}{T}_m$$

where T_m, the rate if temperature change due to internal heat production can be neglected because mean swimming speed has been assumed constant.

In addition, red muscle temperature, T_r (n=249; 107 - 287 cm CFL) and white muscle temperature, T_w (n=65; 110 - 287 cm CFL) of live fish were measured in the core of the medial red muscle and in the dorsal musculature, respectively, with digital thermometer equipped with a 12.7 cm probe on submerged fishes in the "*camera della morte*", once the mesh floor was drawn up. T_r measurements were monitored over 2005-2007 period, while T_w data are only available for 2007.

Water temperature (T_a) was measured with a multi-probe sonde (Hydrolab DataSonde® 4a) within the depth range of the "*camera della morte*". The elevation of T_r above ambient temperature, or excess temperature (T_x), was calculated for each individual as T_r-T_W. Differences in data were evaluated for the three years investigated for T_r and T_a, using a Kruskal-Wallis test. The relationship between T_r and T_a was established by means of a linear regression model and the differences between T_r and T_w by a paired t-test.

Logger features

We used a modified stainless steel temperature data logger (HOBO® U12 Stainless Temp Data Logger, U12-015-02) inserted into the dorsal musculature by a diver, then extracted, and its exact location recorded once the fish were processed. The data logger consists of a cylindrical body closed by an end-cap (101.6-mm length, 17.5 mm OD), and a 124 mm spike probe (4 mm OD) housing the temperature sensor (Fig. 2A). The entire device is fabricated of high-strength corrosion resistant steel (AISI 316L), and its housing sealed by an EPDM O-ring positioned between the cylindrical body and the end-cap. It can withstand pressure equivalent to 1500 m.

The device contained a single channel temperature logger with a 12-bit resolution, a real-time clock and a non volatile-memory (43,000 measurements), and a glass bead thermistor with a resolution of 0.025 °C at 25 °C. Each data record included temperature, time of each reading and the battery voltage. In order to overcome placement challenges, we modified the original device with a Tahitian shaft speargun (21.7 mm x 112.6 mm) for seating the data logger (Fig. 2B), and applied two Teflon barbs to the probe with epoxy resin.

In insertion operations, once the assembly reaches the target, the probe penetrates the tuna's muscle, while the modified Tahitian shaft falls to the sea bottom, where it can be recovered by the diver.

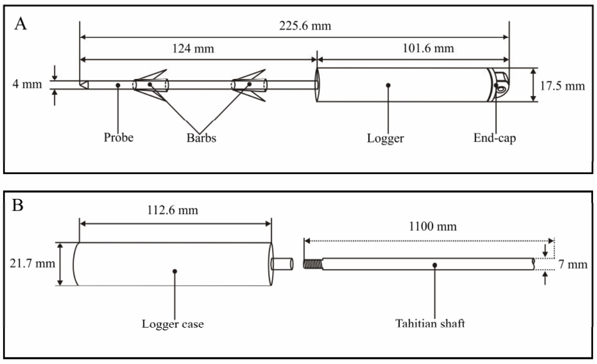

Figure 2. – Temperature Data Logger implantation system. A) HOBO Data Logger specifications. B) Tahitian shaft modification. Modifications included addition of two Teflon barbs on data logger probe.

Results

Acute temperature measurements

Bluefin tuna used in the study (n = 249) were 107 - 287 cm CFL ($\underline{x}$=169 ± 40.5 cm SD mode =154 cm CFL). Annual mean T_a, values were 17.9 °C to 20 °C ($\underline{x}$=18.9 ± 0.84 °C) with significant differences in T_a among the three years investigated ($H = 224.69$, $P < 0.05$ Kruskal-Wallis). Red muscle temperature (T_r) values ranged from 23.5 °C (~ 280 cm CFL) to $T_r = 31.0$ °C (in three individuals of ~ 225 cm CFL) with a mean value of 27.6 ± 1.48 °C. T_r measurements differed significantly between years ($H = 19.28$, $P < 0.05$ Kruskal-Wallis). There was a significant positive correlation between red muscle temperature and water temperature across the study period ($F = 6.41$, $P < 0.05$) (Fig. 3), and $T_x=T_r-T_a$ was 8.21 and 9.10 °C. The comparison between red muscle temperature and white muscle temperature showed a significant difference (2.4 ± 1.78 °C), by paired t test ($t = 10.68$; $P < 0.05$) (Fig. 4).

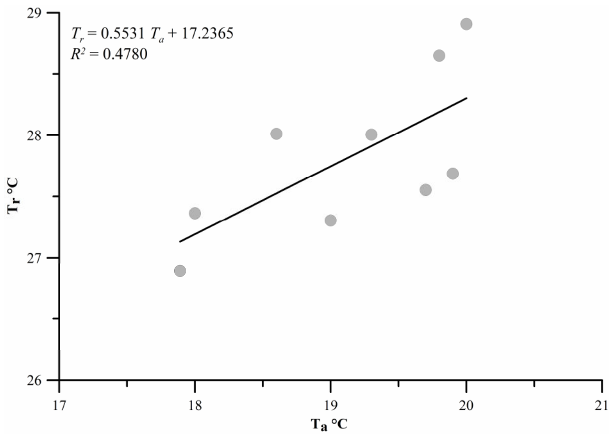

Figure 3. – Relationship between the red muscle temperature (T_r) and water temperature (T_a) for bluefin tuna studied in the chambers of a traditional trap. Mean values of T_r and T_a for each "*mattanza*" were used (2005-2007).

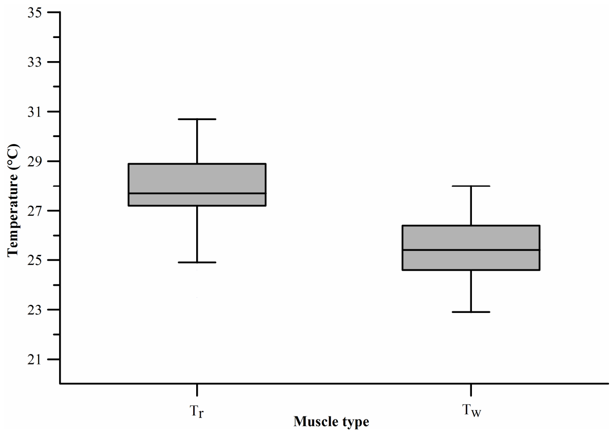

Figure 4. – Comparison between red muscle (T_r) and white muscle (T_w) temperature distributions. Measurements of 65 bluefin tuna for the 2007 fishing season are reported.

Trap Measurements

We inserted a data logger on two giant bluefin free swimming within the chambers of the trap at Isola Piana. Continuous tracks were monitored for about 1 h 30 min during which fish spent much of their time in the medium-shallow waters. Observations by professional divers confirmed that data logger implantation did not noticeably affect swimming behaviour. During the *pre-fishing* phase, divers noted that the fish tended to swim at a steady slow speed (speed was not evaluated). When fishing operations were underway, the tuna were observed to become increasingly active, and increased their swimming speed.

Bluefin 1

In the first experiment, which started 22.05.07 at 12:20 h, a giant bluefin (160 kg – 214 CFL) was tracked for 1 h 25 min, while free swimming in the "*camera di ponente*" and in the "*camera della morte*" before being hooked after the "*mattanza*" (Fig. 5A). In the *pre-fishing phase,* the data logger was inserted into the fish at 12:25. After about 20 minutes, the tagging device stabilized at a mean temperature of 22.3 ± 0.01 °C (SD), which remained constant until 13:28. From 13:28 to 13:38, preliminary fishing operations began, and fishermen began to raise the net mesh floor from the sea floor, and body temperature increased from 22.3 to 22.5 °C .

Starting from 13:38 until the end of "*mattanza*" body temperature increased from 22.5 °C to 24.3 °C. Assuming a mean value for T_e (the body temperature at steady state) of 22.3 °C and $T_b(0)$ (t=0 when *mattanza* started) of 22.5 °C, values of 2.40 10^{-3} of K is required for the best fit regression.

Bluefin 2

A second giant bluefin (117 kg, 191 cm CFL) was tracked for 1h 10 min on 07.06.07, starting at 14:55 h (Fig. 5B).

Body temperature was recorded beginning at 15:01, and device stabilization required almost 20 minutes, after which a relatively stable mean temperature of 22.3 ± 0.01 °C was recorded, until 15:52. Once the net floor was raised, body temperature increased from 22.3 °C to 22.5 °C between 15:52 and 16:01, revealing a slight increase in the tuna's metabolic heat production, presumably due to faster swimming.

Fishing operations began at 16:01, and the subject's body temperature increased from 22.5 °C to 24 °C (+1.5 °C). Heat budget model fitting returned a K value of 2.88 10^{-3}, for a T_e of 22.3 °C and a $T_b(0)$ of 22.5 °C, with a clear linear relationship in both phases. Calculation of regression parameters indicated a higher slope value for the *fishing* phase (b = 0.009) than for the *pre-fishing* phase (b = 0.0002) underlining a factor of 10^2 in the elevation of T_b presumably related to the stressful *fishing* phase. The difference between *pre-fishing* and *fishing* phase regression slopes was statistically significant ($P < 0.001$) for both bluefin tuna analyzed.

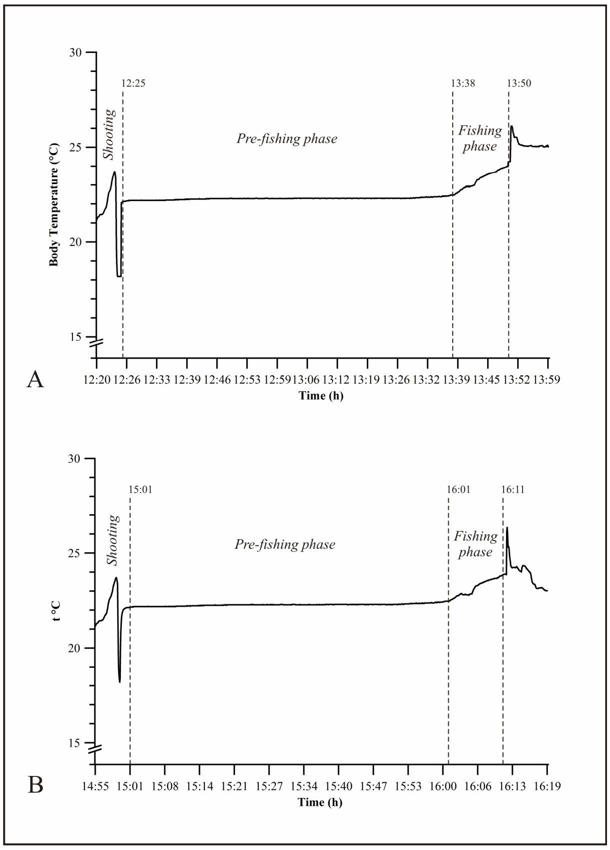

Figure 5. – A) Temperature records for Bluefin 1. Measurements were recorded every 4 seconds (n=1,272); B) **t**emperature records for Bluefin 2. Measurements were recorded every 4 seconds (n=1,061).

Discussion

This study is the first to examine body temperature of bluefin tuna within the chambers of a traditional Mediterranean trap and also to consider the role of temperature-related stress occurring during the fishing operations known as "*mattanza*".

Atlantic bluefin tuna swim constantly in order to maintain hydrostatic equilibrium and are ram ventilators (Magnuson, 1973; Roberts, 1978). Tunas are also capable of rapid bursts and acceleration in pursuit of prey and during rapid depth changes (Carey and Teal, 1966; 1969; Block et al., 1992). By continuously swimming bluefin tuna generate high metabolic heat within the red muscle which affects the body temperature. Our results confirm the ability of bluefin tuna confined in traps to elevate and maintain muscle temperature above seawater temperature by about 10 °C when water temperature was 19 °C, conditions which occur during the breeding migration along the south-western coast of Sardinia. The excess of temperature found is similar to values obtained for individuals in the western Atlantic (Stevens et al., 2000), even though the range of water temperature was dissimilar, and our measurements were conducted on smaller individuals.

The model of heat balance, developed in laboratory studies of tuna endothermy, has shed light on the mechanism of thermal conservation in bluefin tuna (Graham and Dickson, 2001). Heat balance models describe that a tuna swimming at steady state generates an equal rate of heat production and loss (Graham and Dickson, 2001), while a non-steady state heat transfer applies when heat production and loss are not equal, and body temperature is changing. This occurs during vigorous swimming, which can greatly increase the differences between body temperature and ambient temperature. With regard to heat balance, our assumption is that the experimental conditions considered here i.e., *pre-fishing* and *fishing* phase, reflect steady state and non-steady conditions respectively.

Underwater observations clearly show the transition from customary swimming during *pre-fishing* phase where heat production and heat loss should be equal (confirmed by a constant $T_{b,}$ i.e. a steady state condition), to a condition in which entrapment drives the tuna to swim faster in an effort to seek a way out. In our instrumented tunas, this rapidly brought about an increase in body temperature of about 2 °C (non-steady state condition), which corresponds to K values greater than values calculated in literature (Kitagawa et al., 2001; Teo et al., 2007). In our experimental conditions the ambient temperature was constant, so body temperature variation found during the "*mattanza*" was generated only by the variation in metabolic heat.

The frenetic behaviour displayed by tunas is typical during the final *fishing* phase. As well as an increase in body temperature, capture struggles also generates varying degrees of stress (Skomal, 2007). During the "*mattanza*", tunas are subject to physical trauma from collisions with other tunas, and also to physiological stress due to the high level of anaerobic activity and muscular fatigue (Skomal, 2007). This can be highly detrimental for bluefin tuna because intense muscular activity can result in death (Black, 1958). Measurements of the degree of stress is usually carried on by checking for changes in blood chemistry, by analysing levels of plasma electrolytes, osmolarity, blood metabolites (glucose, lactate), stress hormones, plasma enzymes and hematocrit (Skomal and Chase, 2002; Skomal, 2007). However, blood sampling may cause even further stress for the fish. Since we believe we have shown a relationship between body temperature and the degree of stress, we suggest that body temperature may be an appropriate indicator for stress magnitude. We realize that this is a first qualitative, limited attempt to gather information on the connection between body temperature and stress. Linking blood chemistry stress indicators with body temperature is our goal in forthcoming studies.

Bluefin tuna have had an important socio-economic role for many centuries. Local fisheries in southwestern Sardinia (Carloforte and Portoscuso) have long centered on seasonal tuna fishing, and this ancient tradition of tuna fishing has remained practically unchanged for five centuries. Because of the high quality standards for raw fish required by the Japanese market, quality control of the final product is of utmost importance. These factors include freshness, fat content, meat colour and absence of Yake, a Japanese term for Burnt Tuna Syndrome (BTS) (Mateo et al. 2006). Yake may be the result of the high body temperature and stress-related physiological disturbance that speeds up the onset and progression of bacterial growth and biochemical deterioration (Lehane and Olley, 2000; Mateo et al. 2006). It has been demonstrated that vigorous activity during the capture of bluefin affects the meat quality (Jerret et al., 1996). Consequently, it is generally accepted that a stress-free, or perhaps, more strictly ‘exercise-free’, harvest will improve flesh quality by ensuring that the fish are not exhausted during harvesting (Wells et al. 1986). Electroslaughtering fishing techniques in tuna farms can actually improve meat quality because it provides exercise-free fish (Soto et al., 2006), and electric harpoons are used routinely to produce high quality product in the New England giant bluefin fishery. Current practice for most trap fishermen is, to chill the tuna as soon as possible after harvesting, thus reducing the initial temperature of the tuna, rather than seek stress-free capture.

Bluefin tuna captured in Sardinian traps are valuable because of their high quality, but the flesh colour, flavour and shelf-life are all greatly influenced by harvesting and handling practices. Reducing the level of hyperactivity during the harvesting process will be a key aspect of future research, as will examining the combined effects of pre-harvest fasting and changes to on-board handling practices.

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CONCLUSIONS

The analysis conducted in Chapters 1 and 2, can be put in the context of indicators for assessing the health of marine ecosystem subject to fisheries exploitation. The use of biodiversity index is widely recognised (Blanchard 2001; Ungaro et al. 1998; D'Onghia et al. 2003; Lobry et al. 2003; Colloca et al. 2003; Labropopoulou et al. 2004; Gristina et al. 2006), but not widely applied.

The analysis on biodiversity underlined remarkable variations on biodiversity of middle-slope assemblages and showed different temporal trends in the four sub-zones considered. Among the several causes and factors that can induce observed variations (Lobry et al. 2003; Caddy & Sharp, 1986; Caddy & Griffiths, 1995; Ungaro et al. 1998), climatic changes are usually directly responsible, but they apparently affect only surface layers and can only affect the deeper ones indirectly (IFREMER, 1997). Moreover Mediterranean sea in particular is characterised by high degree of environmental stability (Hopkins et al. 1989; Tyler 2003; Hopkins 2006). Thus the temporal variation of biodiversity indexes seem to be "fishing effort" related.

The increased fishing pressure in SE Sardinia, due to fleet renewal (+120%), has caused a reduction in red shrimp biomass, resulting for some years in overexploitation (Sabatini et al. 2006). The observed decrease in species richness, which has generated the heavy drop in diversity index show that also other species associated with red shrimp have also been affected in this process. According to ecological theory (Huston 1979), this means that the high frequency of stock reduction in this sub-zone, does not allow exploited populations to react to fishing pressure with a growth rate increase. More evidences are given by, the dominance analysis (Simpson's index and *k*-dominance curves), that would indicate a rapid increase of those species which adapt better to high frequency of population reduction (Huston 1979; Suvapepun 1991).

SSW Sardinia shows a higher dominance gradient for the last years of the investigated period, but only a statistically significant drop in species richness can be underlined. This data would seem to indicate that, due to an increased frequency of population reduction, there was an initial stage of overexploitation in this sub-zone which, reveals a decrease in biomass and an increase in fish mortality (Sabatini et al. 2006). Nevertheless, a wider trawling area than that of the SE and the characteristic turnover of gulf waters make this sub-zone more resilient to the fishing effort increase (+ 22.6%) that has occurred in Sardinian seas over the last 10 years.

Northern Sardinia fleets (NE and NW) show no remarkable temporal variation and no statistically significant temporal trend for biodiversity indexes, leading us to surmise that slight variations observed were due to chance or to common environmental dynamics.

Thus, according to our observations, species richness index seems to be the most sensitive to fishing effort. This is particularly evident for SE and SSW sub-zone.

Our results highlight that more pronounced and significant variations in biodiversity indexes are concentrated where fishing effort was more intense (SE and SSW Sardinia).

In the light of the above, the diversity index together with its components of species richness and evenness as well as dominance index, seem to differ in their reaction sensitivity to perturbation and are able to indicate changes in exploited ecosystems. According to the results, biodiversity levels, evaluated by the simultaneous estimation of all its "expressions" (entropy, species richness, evenness and dominance), allow us to detect early warnings of degrees of modifications in resilience that otherwise would pass unnoticed.

The Chapter 2 provides additional proofs on the relationships existing between fishing effort and biodiversity and the possibility of carrying out a prediction of the Shannon diversity index (*H'*), starting solely from one of its components and fishing effort.

The Shannon's entropy temporal trend simulation, developed by a multiple linear regression, actually displayed statistical significant relationship between species richness and a fishing effort expression used to develop the model.

Coefficient of determination (R^2), showed that the fitted model explains the 28.58% of the H' variability, underlining that changes in species richness or in fishing effort may generate, acting jointly, more than 1/4 of the change observed in Shannon's diversity. The application of the model on the single sub-zones and its performance values (RMSE), displayed that biodiversity of southern Sardinian seas (SE and SSW sub-zones) is better described than the northern seas one. This may highlight that for these sub-zones there should be a tighter accordance between fishing effort species richness, thus the statistical model can provide a better description of biodiversity. Therefore, the high fishing effort that occurred in these areas (Sabatini et al. submitted), should have a non negligible influence in shaping demersal red shrimp-related assemblages, (Daan and Richardson 1996; O' Brien et al. 2000). Otherwise in the Northern sub-zones, where increasing in fishing effort were not statistical significant (Sabatini et al. submitted), species richness and fishing effort showed less describing power (i.e higher values of RMSE) on Shannon's entropy.

Regarding the high proportion of unexplained variability of biodiversity is hypnotizable that it should due to natural environmental stochasticity that, considering the high degree of environmental stability of the Mediterranean (Hopkins et al. 1989; Tyler 2003; Hopkins 2006), may be generated mainly by local environmental processes (Ricklefs, 1987; 2004; Huston 1999; Hoeinghaus et al. 2007) such as local up-welling, bottom streams and active canyons (Gristina et al. 2006). However, by the results obtained, we think that the fishing effort may had for some zones a non negligible influence in shaping demersal red shrimp-related biodiversity and its effect may overcame the contribution of others environmental process in describing the temporal variation of biodiversity.

Our results showed that the mechanisms for diversity variations in the Sardinian red shrimp-related community have been explained, at least in part, by the multiple linear regression model we used.

Although further investigations are necessary, biodiversity indexes have revealed to be high sensitive to fishing pressure and could be an important support to classical stock assessment methodologies.

The analysis carried out in Chapter 3 underlined the influence of a peculiar area such as submarine canyon on the structure of the middle-slope assemblages. In particular distinctions can be made between the fauna of the deeper waters (>500 m) and that living at less depths (<350 m). Each association appears to be characterized by an indicator species which typifies the group.

The first group is characterised by hauls made at depths <350 m during the first part of the night only and is characterized by the species *M. merluccius*. As regards the second group, which assemble hauls made at depths of <350 m during the day only or in the second part of the night, the indicator species is *G. leioglossus*. The third group gathers together the hauls made at greater depths (>500 m). The species characterising it is *A. antennatus*. The fourth group collects together different kind of hauls, some made ad depths >500 m during the day and others made at lesser depths during the night only. The indicator species for this group is *C. agassizi*. Other species that characterize this group are *A. antennatus* and *A. foliacea*.

This results highlights that shrimps and other species inside Quirra Canyon move during a day-night cycle, probably with light as a parameter indirectly related to their activity, as proposed by Tobar & Sardà (1992). These species are mainly predators (Gristina et al., 1992; Fischer et al., 1987) moving during the night to catch prey. They find the canyon a suitable place for finding food. In fact, the ability to reach different depths in a relative short distance together with the mass transportation of sediments rich in organic matter from the shelf to deeper regions, determines the variations in species assemblages with time and depth observed. The results show that movements of species inside the Canyon, and so the different species' compositions observed, seems to be linked to a day-night cycle, and probably by the consequent different food availability, as suggested by Cau & Deiana (1982) and Stefanescu et al. (1994).

One of the main questions that motivated the study developed in Chapter 4 is to test if the environmental variables available on a local scale interfered with trap fishery landings over time.

The trap landings data we analyzed with the ACF and Partial ACF approaches showed a clear trend for all of the traps. While the Porto Paglia series indicated a cyclical behaviour, there was less evidence for any cyclical response for the Portoscuso trap. When we analyzed trap series and explorative variables together using bivariate spectral analysis, the Porto Paglia and Run-off series showed good fit, as indicated by the cross-periodograms. That response enabled us to hypothesize and test for a 'pulsing' pattern in the captures of Porto Paglia as a result of the Runoff effect. Furthermore, we assume that the Run-off effect on trap landings was the result of water draining from mine tailing drainage into local rivers in the coastal watershed close to the trap site. The small cyclic signal highlighted for the Portoscuso series could be explained as a residual Run-off effect due to the spatial relationships with the mining Hot Spot.

Also, the asymmetrical BACI design provided consistent evidence that the effects of mechanized mining can be presumed to be 'exceptional events' that sharply modified the landings trend of Porto Paglia captures from the end of 1880 onwards. We also tested the effects of the three explorative variables with the Dynamic Factor Analysis technique. The model including three Catch series and Run-off gave the lower AIC, but because that value is very close to the model with only the Catch variables, its effect seems to be negligible. It is possible that due the high variability in Run-off over the long time period the model is confounded. It is worth noting that with dynamic factor models, if seasonal or cyclic components are present in the time series, they will be masked and included in the trend component.

Our conclusion is that a modification of the water masses and sediment occurred in the area where the Porto Paglia trap was deployed, which resulted in the formation of a 'pulse boundary' with an intra-decadal fluctuation of approximately 7-years, within the periods analyzed. The recurring formation of that 'pulse boundary' led to a 'reflective or pushed back condition' of bluefin tuna migration schools. This caused a progressive decrease in the catch yield in the in-shore trap or Impacted site Porto Paglia. Coincidentally with that shift in behaviour, there was no effect on the Isola Piana (control site) trap that was deployed in an off-shore isle. Portoscuso (another control site), due to its middle position, seems to have a similar response of Porto Paglia, but at a reduced level.

Bluefin tunas are very sensitive to the modification of water masses and sediments (Lemos and Gomes, 2004). Evidences came out both for human-related perturbations (Mazzarelli 1917) and natural-related perturbations (Grassi, 1913). Another important source that interference with tuna's behaviour is acoustic noise due to shipping traffic (Sarà et al. 1998; 2007) We did not directly measure a negative effect on tuna landings in Sardinia due to ships transiting in the area, but we did document an increase in ship traffic from the industrial harbor of Portoscuso due to the growth of trade and exchange with new economy countries. Among the other possible local sources of variation in the captures by traps, purse seine fleets may also play an important role.

Traditional traps are an excellent example of an ecologically compatible fishing technology because they have provided a sustainable bluefin tuna fishery for over 2000 years. It is only in the very recent past (the last 100 years) that the population has suffered significant declines (Dean et al., 2003). The traps should be utilized as bluefin tuna 'sentinel' gear and as a source of significant ecological information.

The employment of the few remaining traps to achieve research objectives (i.e. a long-term monitoring program) would lead to a better understanding of the behaviour, physiology and ecology of the bluefin tuna, the status of the stock and the relationship of this great fish to future climatic changes.

The investigation carried out in Chapter 5 is the first to examine body temperature of bluefin tuna within the chambers of a traditional Mediterranean trap and also to consider the role of temperature-related stress occurring during fishing operations (“mattanza”). The results first of all, confirm the ability of the bluefin tuna confined in traps to elevate and maintain muscle temperature above seawater temperature by about 10 °C.

Heat balance models describe that normally tunas swim at steady state and generates an equal rate of heat production and loss (Graham and Dickson, 2001), while during vigorous swimming a non-steady state heat transfer applies and heat production and loss are not equal bringing to a change in body temperature. In the case examined in Chapter 5 has been assumed that the experimental conditions considered (pre-fishing and fishing phase), reflect steady state and non-steady conditions respectively.

Underwater observations clearly show the transition from customary swimming during pre-fishing phase where heat production and heat loss should be equal, to a condition in which entrapment drives the tuna to swim faster in an effort to seek a way out. In examined tunas, this rapidly brought about an increase in body temperature of about 2 °C (non-steady state condition), which corresponds to K values greater than values calculated in literature (Kitagawa et al., 2001; Teo et al., 2007). In our experimental conditions the ambient temperature was constant, so body temperature variation found during the “mattanza” was generated only by the variation in metabolic heat.

As well as an increase in body temperature, capture struggles also generates varying degrees of stress (Skomal, 2007). During the “mattanza”, tunas are subject to several physical traumas and also to physiological stress (Skomal, 2007) that can also result in death (Black, 1958). Measurements of the degree of stress is usually carried on by checking for changes in blood chemistry (Skomal and Chase, 2002; Skomal, 2007). However, blood sampling may cause even further stress for the fish. Since we believe we have shown a relationship between body temperature and the degree of stress, we suggest that body temperature may be an appropriate indicator for stress magnitude.

Moreover due to the high economic importance of bluefin tuna for South-Western Sardinia fisheries, a high quality standards of the final product is of utmost importance. It has been demonstrated that vigorous activity during the capture of bluefin affects the meat quality. Consequently, the monitoring of the stress level and the consequent development of a “stress-free” harvesting technique, is important and will improve flesh quality (Wells et al. 1986).

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