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Changes in the trophic structure, abundance and species diversity of exploited fish assemblages in the artisanal fisheries of the northern coast, Senegal, West Africa

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This work investigates the effects of changes in both fishing pressure and the environment on the trophic dynamics, abundance and diversity of species in the artisanal commercial fisheries off the northern coast of Senegal. Using artisanal commercial fishing data (provided by the Centre for Oceanographic Research of Dakar-Thiaroye [CRODT] in Senegal), we identify changes in the catch per unit effort, mean trophic level, biomass trophic spectrum and species diversity between two fishing periods (1990–1999 and 2000–2009). Decreases in mean trophic level, the biomass of high trophic level species and indices of species diversity between 1990 and 2009 were observed in commercial catches. These decreases were then related to changes in fishing pressure, fishing strategy and the combined effects of fishing and environmental factors (as derived from satellite observations). This paper helps to better inform the management of fisheries resources by providing decision makers with more effective biological indicators that incorporate the effects of fishing pressure and environmental change and that are applicable at local, regional and global scales.

Keywords: artisanal fishery, ecosystems indicators, trophic level

Introduction

Changes in fishing practices, exploitation levels and climate affect the steady state of marine ecosystems (Garcia et al. 2001), with fishing being a key human impact (Jennings and Kaiser 1998; Jackson et al. 2001). Fisheries activities affect fish communities directly by removing target (Zwanenburg 2000) and bycatch species (Stevens et al. 2000) and indirectly through habitat modification (Moran and Stephenson 2000). In addition, fishing indirectly alters biotic interactions, a phenomenon which is spread through food webs via trophic cascades (Cury et al. 2003). In conjunction with climate change, fishing can cause changes in the structure of fish assemblages in terms of biomass, species richness, trophic structure (Pauly et al. 1998; Albaret and Laë 2003) and size spectra (Blaber et al. 2000).

The need to respond effectively to these multi-faceted impacts has led to the increasing adoption of ecosystem-based approaches to fisheries management (Garcia et al. 2003; Cury et al. 2005; Coll et al. 2008). Commonly, ecosystem indicators, including those based on size, trophic structure, relative abundance and species diversity (e.g. Bianchi et al. 2000; Trenkel and Rochet 2003), are used to inform such approaches (Cury and Christensen 2005). When changes to these indicators occur, managers are provided with an enhanced picture of how fishing pressures and/or environmental changes may be impacting the evolution of the structure and functioning of ecosystems.

Few studies in Africa have integrated the effects of both fishing and environmental change in the assessment of fisheries resources (Domain 2000; Erzini et al. 2005; Shannon et al. 2009). Typically, they are assessed separately (e.g. fishing effects: Gascuel and Ménard 1997; Jouffre et al. 2004; Laurans et al. 2004; or environmental effects: Cury and Roy 1987; Diatta et al. 2010; Koranteng 2001). This study aims to integrate both fishing effort and environmental factors when assessing the fisheries resources of a heavily exploited, but poorly documented, marine ecosystem. By investigating changes in biological indicators based on trophic structure, relative abundance and species diversity between two fishing periods (1990–1999 and 2000–2009), this study aims to better understand the responses of fish assemblages to the effects of both environmental change and fishing pressure.

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Materials and methods

Data

Commercial fishing data

Artisanal commercial fishing data were obtained from the Centre for Oceanographic Research of Dakar-Thiaroye (CRODT), Senegal. The catch data were based on daily surveys of landings in Saint-Louis and Kayar on the northern coast (Grande Côte) of Senegal between 1990 and 2009 (Figure 1). This database includes catch by species (kg), fishing effort (i.e. number of fishing trips), fishing season and fishing gear. The data collection method is based on a hierarchical three-tier stratification scheme based on: (i) the fishing port; (ii) the month during which fishing occurred; and (iii) the type of gear used. This method incorporates the spatio-temporal variations of each fishing method.

The trophic levels of the species considered in this study were extracted from FishBase (Froese and Pauly 2012).

Environmental data

Sea surface temperatures and chlorophyll *a* concentrations (1990–2009) were extracted from the websites of the National Oceanic and Atmospheric Administration (NOAA; http://las.pfeg.noaa.gov) and the French Research Institute for the Exploitation of the Sea (Ifremer; http://www.brest.ird.fr/us191/valorisation/obssat/aoos.php). A coastal upwelling index was extracted from http://las.pfeg.noaa.gov/las6in?var=1638_5/servlets/constra.

Extrapolation of catch

The catch from commercial fisheries (Table 1) was extrapolated from the number of daily fishing trips according to the following equations:

$$C_j = (C_{sur} / S_{sur}) \times S_j \quad (1)$$

$$C_k = (C_j \times 365) \quad (2)$$

where C_j is the estimated catch (kg) per day j; S_j is the fishing effort (i.e. number of fishing trips); C_{sur} is the catch that is surveyed (kg); S_{sur} is the fishing effort (i.e. number of fishing trips) represented by the survey catch; and C_k is the estimated annual catch (kg), assuming each year has 365 days.

Catch per unit effort

A generalised linear model (GLM) was applied (Gaussian distribution) to standardise the catch per unit effort (CPUE) (i.e. the abundance index [AI]) per year for each trophic class (i.e. classes based on trophic levels binned by 0.25, range: 2 to 4.75). A benefit of GLMs is that they enable an equal weight distribution to be given to each category of the explanatory variables selected, thereby balancing the sampling plan. They can also simultaneously process different years of data by taking into account information that is acquired each year (Laurans et al. 2004). The variables included in the GLM were the fishing port (Kayar and Saint-Louis), year (1990 to 2009), season (cold and hot season) and gear type (lines, longlines, gillnets, driftnets, trammelnets, purse-seines, beach-seines and castnets). First, a logarithmic transformation was applied to the data to homogenise the variances due to the multiplicative effects of the variables. We assumed that the response, ln(CPUE), follows a linear model of the form:

$$\ln(\mathrm{CPUE}) = \mu + P + Y + S + G + \varepsilon \quad (3)$$

where ln(CPUE) is the natural logarithm of the CPUE by trophic class, μ is the intercept, P is the effect of fishing port, Y is the year effect, S is the season effect, G is the effect of fishing gear category and ε is the residual. The individual and interaction effects were tested. Only variables

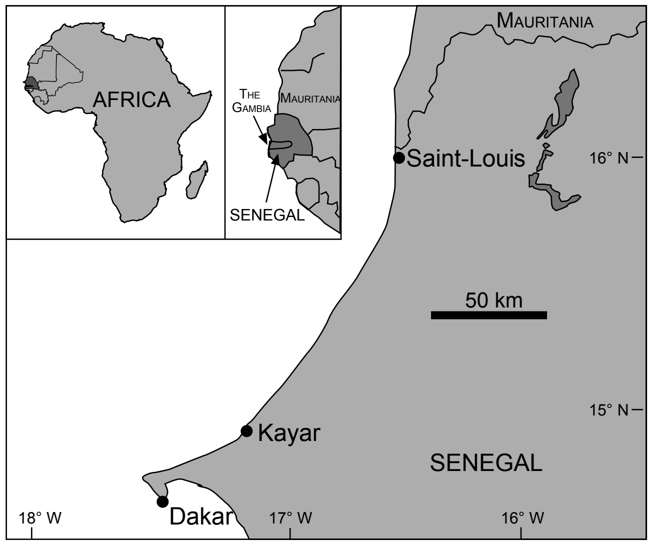

Figure 1: Map showing the study areas along the northern coast of Senegal

Table 1: Catch (kg) by artisanal fisheries at two fishing ports on the northern coast of Senegal from 1990 to 2009

Year	Kayar	Saint-Louis	Total
1990	3 561 052.3	2 529 927.0	6 090 979.2
1991	2 316 671.9	1 178 099.7	3 494 771.6
1992	11 592 230.0	2 338 425.1	13 930 655.1
1993	11 865 580.4	8 801 335.0	20 666 915.4
1994	7 881 408.8	10 119 916.4	18 001 325.1
1995	6 078 383.4	7 823 386.1	13 901 769.6
1996	6 645 038.4	4 914 459.4	11 559 497.8
1997	6 492 308.5	1 282 712.8	7 775 021.3
1998	6 486 377.3	840 252.7	7 326 630.0
1999	6 506 720.7	5 513 389.1	12 020 109.7
2000	9 788 529.6	4 820 661.1	14 609 190.7
2001	10 498 139.0	4 362 525.1	14 860 664.1
2002	7 988 503.9	11 483 161.7	19 471 665.7
2003	11 468 148.8	19 280 855.0	30 749 003.9
2004	17 920 390.5	30 776 076.4	48 696 466.8
2005	17 579 236.3	65 603 975.9	83 183 212.1
2006	13 448 114.6	42 653 701.8	56 101 816.4
2007	187 620.8	49 616 469.9	49 804 090.7
2008	18 469 208.8	53 708 312.8	72 177 521.6
2009	21 784 868.0	30 377 233.8	52 162 101.8
Total	198 558 531.9	358 024 876.6	556 583 408.5

that significantly explained the observed deviance were retained (Table 2). CPUE values were then retransformed to their original scale using an exponential transformation with the Laurent correction (Laurent 1963), which provides unbiased values of the expected AI:

$$AI_i = e^{\ln(CPUE_i)} \times e^{(\sigma^2(\ln(CPUE_i)))/2} \quad (4)$$

where σ^2 is the variance estimator associated with the observation.

Mean trophic level

The mean trophic level (TL_m) was calculated from the commercial fishing catches as follows:

$$TL_m = \sum_{i=1}^{S} Y_{ik} TL_i \Big/ \sum_{i=1}^{S} Y_{ik} \quad (5)$$

where Y_{ik} is the landing of species i for year k, TL_i is the trophic level of the species and S is the number of species surveyed during the year. To verify the relationship between year and the mean trophic level, a linear correlation was used.

Diversity indices

Species diversity was compared between the two fishing periods using the commonly used Shannon diversity index H' (Lobry et al. 2003; Munyandorero 2006), the Simpson index λ, the equitability index J and the dominance index D. In the present study, H' was calculated from the proportions of species in the catches (Whitfield 1986; Jin and Tang 1996; Emery et al. 2007), according to the following equation:

$$H' = -\sum_{i=1}^{S} p_i \times \ln p_i \quad (6)$$

where p_i is the proportion of species i and S is the total number of species in the sample. The Shannon index is often accompanied by the equitability index J (Pielou 1966):

$$J = H' \big/ \ln S \quad (7)$$

The Simpson index λ (Simpson 1949) measures the probability that two randomly selected individuals belong to the same species:

$$\lambda = \sum_{i=1}^{S} p_i^2 \quad (8)$$

A determination of the dominance parameters allowed the taxonomic structure of the fish assemblages to be defined (Grall and Coïc 2006):

$$D = 1 - \sum_{i=1}^{S} p_i^2 \quad (9)$$

Statistical analysis

The CPUE, trophic level and species diversity indices were tested between the two fishing periods (1990–1999 and 2000–2009) using bootstrapping. The Aspin–Welch non-parametric test, which is equivalent to the Student's *t*-test for samples that possibly have unequal variances, was performed to compare the means of the indicators between the two periods. A one-way ANOVA was used to test the effects of environmental factors and fishing pressure – separately and together – and the combined effects of both on changes in the indicators.

Results

Environmental variation and dynamics of exploitation

Significant differences were found for both fishing pressure and environmental factors between the two fishing periods (Table 3). The means of the fishing effort were greater during the second period (2000s: 270 688 fishing trips; SD 43 658) than the first (1990s: 153 386 fishing trips; SD 57 870). In contrast, the mean concentration of chlorophyll *a* was greater in the 1990s (6.15 mg m^{-3}; SD 2.10) than in the 2000s (4.30 mg m^{-3}; SD 0.59).

CPUE abundance index

The AI represented by the commercial fishing CPUE showed a gradual decline from 1990 to 2009 (Figure 2). The mean CPUE differed significantly (Table 3) between the two fishing periods, with an output of 1 008.8 kg fishing trip^{-1} y^{-1} in the 1990s as compared with 948.4 kg fishing trip^{-1} y^{-1} in the 2000s (Figure 2).

Mean trophic level

The TL_m of the commercial catches showed a significant decrease between 1990 and 2009 (Figure 3), with a mean decrease of 0.01 trophic level year^{-1}. It differed significantly between the two periods (Table 3) with a TL_m of 3.50 (1990–1999) and 3.42 (2000–2009; Figure 3).

Relationship between catch and TL_m

A significant negative relationship was found between TL_m

Table 2: Generalised linear model (GLM) results for testing the effects of variables on CPUE by trophic class

	Df	Deviance	Resid. df	Resid. dev	*p*
Null			29 792	56 724	
Year	19	2 127.83	29 773	54 596	<2.2E−16
Port	1	990.96	29 772	53 605	<2.2E−16
Season	3	42.97	29 769	53 562	1.4E−05
Gear	7	2 814.81	29 762	50 747	<2.2E−16

Table 3: Results of statistical analyses (Aspin–Welch test) of differences between environmental parameters, fishing effort, catch per unit effort (CPUE) and mean trophic level (TL_m) during two fishing periods (1990–1999 and 2000–2009) on the northern coast of Senegal

	t-value	df	*p*
Chlorophyll *a* concentration	7.17	10.39	0.0225
Sea surface temperature (°C)	5.479	11.9	0.0375
Fishing effort	26.18	16.74	0.0001
CPUE	14.19	13.74	0.0021
TL_m	19.71	11.18	0.001

and catch volume ($r = -0.52$; $p = 0.020$; Figure 4a), as although TL_m decreased in the commercial catch, the total volume of catch increased (Figure 4b). Catch volumes of demersal species were observed to decline from 1990 to 2009, while volumes of pelagic species were observed to increase (Figure 4c).

Trophic spectra of catch

Higher catch rates of low trophic level species (TL < 3.00) were observed in the 2000s but the catch rates between the two periods were relatively similar for high (TL > 3.50) and intermediate (3.00 ≤ TL ≤ 3.25) trophic level species (Figure 5).

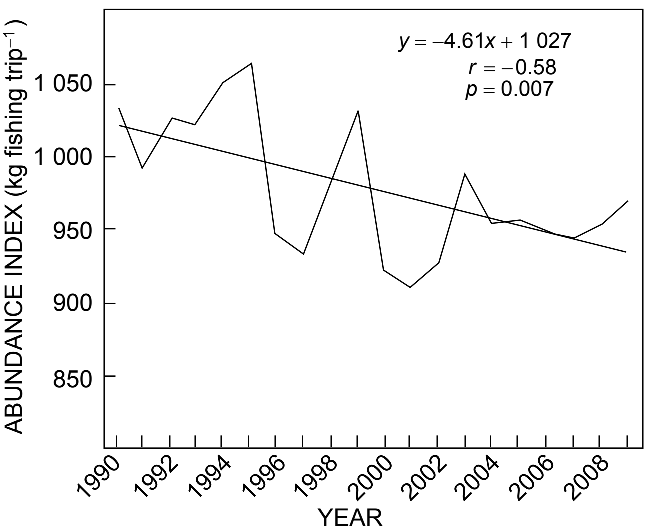

Figure 2: Trend in the abundance index of fish communities estimated from the catch per unit effort of commercial fishing on the northern coast of Senegal between 1990 and 2009

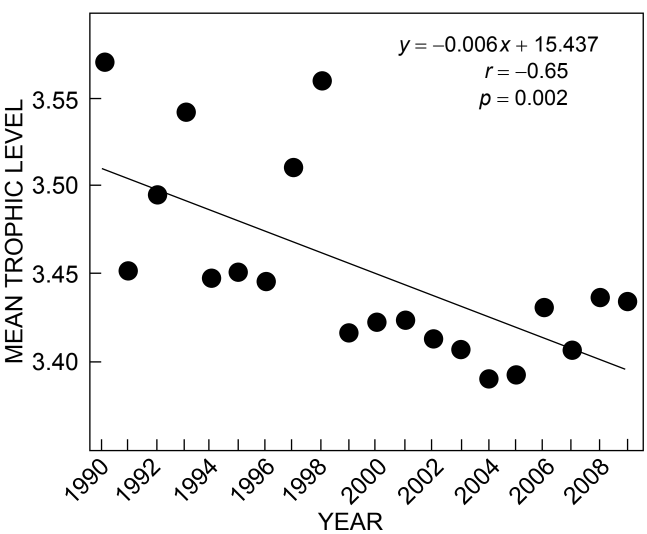

Figure 3: Trend in mean trophic level of fish communities estimated from commercial fishing catches on the northern coast of Senegal between 1990 and 2009

Diversity indices

The Shannon species diversity index was found to be greater in the 1990s (2.14) than the 2000s (1.44). In contrast, the dominance index was stronger in the 2000s (0.51) than the 1990s (0.32). The equitability index was

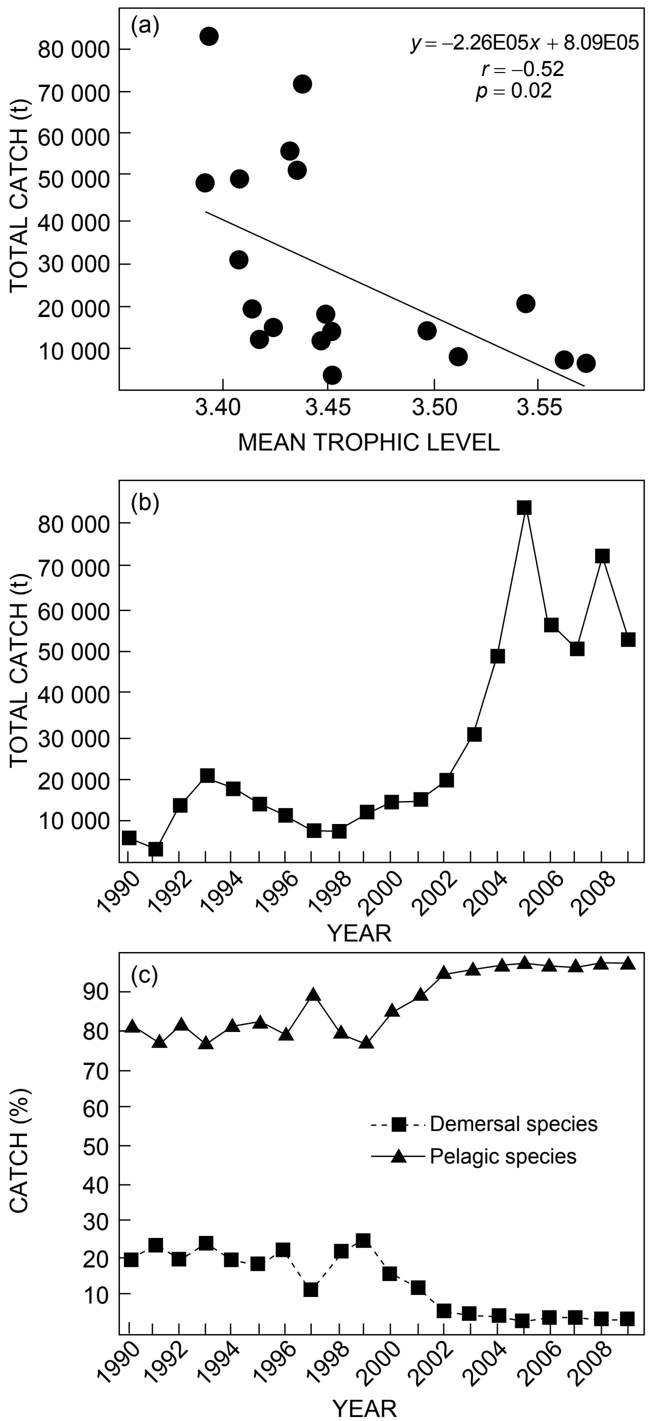

Figure 4: (a) Relationship between the mean trophic level and total catch estimated from commercial fishing; (b) time-series of the estimated total catch from commercial fishing; (c) percentage of the catch of pelagic and demersal species, on the northern coast of Senegal from 1990 to 2009

determined to be greater in the 1990s (0.41) than the 2000s (0.27) (Table 4). During the period 1990–1999 the dominant species caught, by biomass, were *Sardinella aurita* (57.6%), *Sardinella madarensis* (15.1%), *Pagellus bellottii* (4.7%) and *Chloroscombrus chrysurus* (2.9%), and during the period 2000–2009 *S. aurita* (70.6%), *S. madarensis* (11.4%), *C. chrysurus* (6.4%) and *Scomber japonicus* (1.8%) (Table 5).

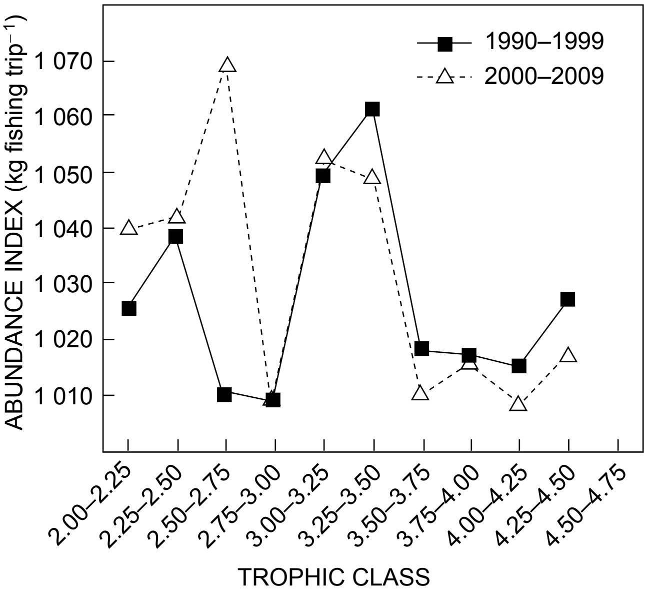

Figure 5: Trophic spectra of fish communities estimated from commercial fishing catches for two fishing periods (1990–1999 and 2000–2009) on the northern coast of Senegal

Table 4: Dominance and species diversity indices estimated from commercial fishing catches between 1990–1999 and 2000–2009 on the northern coast of Senegal (*t*: bootstrapping)

	1990–1999	2000–2009	*t*
Total no. of species *S*	178	178	ns
Sample weight (kg)	114 784	441 729	***
Dominance *D*	0.32	0.51	***
Shannon index *H*′	2.14	1.40	***
Equitability index *J*	0.41	0.27	***

ns = not significant
*** = significant at 0.001

Analysis of impact factors

The results showed significant statistical associations or potential effects of both fishing effort and chlorophyll *a* concentration on CPUE and TL_m (Table 6). There was also a significant potential effect of fishing effort on the Shannon diversity index (Table 6).

Discussion

Several studies have been conducted to develop indicators that identify variations and changes in the structure and health of communities; this goal has been desirable so that principal threats acting on marine ecosystems may be detected more easily and with a greater understanding of their impacts (Nicholson and Jennings 2004; Jennings and Dulvy 2005; Greenstreet and Rogers 2006). However, the management and/or mitigation of these threats requires an understanding of their relative importance, and cumulative effects, in complex ecosystems (Blanchard et al. 2010; Bundy et al. 2010; Shin and Shannon 2010). Biological indicators are a synthetic simplification of this information and can thus act as relevant tools for management purposes (Elliott et al. 2007; Harrison and Whitfield 2008).

The decrease in the commercial CPUE from 1990 to 2009 can be explained by the decrease in the catches of larger species and the increase of smaller species, which are both related to increasing fishing pressure (as also found by Sibert et al. [2006]), changes in environmental factors (e.g. chlorophyll *a* concentration) and a change in the targeted species in this fishery. The study of Erzini et al. (2005) in Mauritania also found significant relationships between CPUE and the environment, and linked the general decline in CPUE over time to environmental changes.

The change in the TL_m of catches toward a higher biomass of low trophic level species for the 2000–2009 period may be due to fishing down marine food webs (Pauly et al. 1998). In combination with this hypothesis, our results suggest that the observed decrease in TL_m found here may be related to effects of changing fishing effort and environment on the trophic structure of fish assemblages. We observed a change in fishing strategies of these fisheries to target the smaller, more abundant species (e.g. use of small mesh, mixing of fishing gear, etc.) as larger species become

Table 5: Percentage of dominant species in the catches and their trophic levels (TL) for the two fishing periods (1990–1999 and 2000–2009) on the northern coast of Senegal

1990–1999 Species	Catch (%)	TL	2000–2009 Species	Catch (%)	TL
1. *Sardinella aurita*	57.6	3.4	1. *Sardinella aurita*	70.6	3.4
2. *Sardinella maderensis*	15.1	3.2	2. *Sardinella maderensis*	11.4	3.2
3. *Pagellus bellottii*	4.7	3.6	3. *Chloroscombrus chrysurus*	6.4	3.2
4. *Chloroscombrus chrysurus*	2.9	3.2	4. *Scomber japonicus*	1.8	3.1
5. *Decapterus rhonchus*	2.6	3.6	5. *Decapterus rhonchus*	1.4	3.6
6. *Brachydeuterus auritus*	1.9	3.0	6. *Sardina pilchardus*	0.9	3.1
7. *Pteroscion peli*	1.7	3.6	7. *Pagellus bellottii*	0.9	3.6
8. *Dentex angolensis*	1.3	3.5	8. *Brachydeuterus auritus*	0.7	3.0
9. *Cynoglossus canariensis*	1.2	3.6	9. *Pteroscion peli*	0.5	3.6

Table 6: Results of statistical analyses of the effects of chlorophyll *a* concentration and of fishing effort on the biological indicators using a one-way ANOVA test. *H'* = Shannon diversity index ; TL_m = mean trophic level; CPUE = catch per unit effort

Factor	Variable	df	Sum sq.	Mean sq.	*F*-value	*p*
Chlorophyll *a*	TL_m	1	0.01484	0.014842	5.233	0.0345
	CPUE	1	0.0141	0.014097	9.076	0.00748
	H'	1	0.767	0.7665	3.814	0.0666
Fishing effort	TL_m	1	0.02494	0.024942	10.96	0.00389
	CPUE	1	0.00907	0.009067	4.948	0.0392
	H'	1	1.146	1.146	6.37	0.0212
Chlorophyll *a* : Fishing effort	TL_m	1	0.000122	0.000122	0.065	0.80266
	CPUE	1	0.002124	0.002124	1.758	0.20349
	H'	1	0.0438	0.0438	0.266	0.6134

more scarce in the ecosystem (Caddy and Garibaldi 2000; Laë et al. 2004). Sosa-Lopez et al. (2005) also observed changes in the trophic structure of a coastal lagoon in Mexico due to several simultaneous and interacting factors, supporting our observations.

Indeed, as low trophic level species usually occur in schools, fishing effort is expected to have an important influence on the amount of biomass in the catches. Similar results have been found in the Celtic Sea (Pinnegar et al. 2002) and in the Mauritanian exclusive economic zone (Abdellahi 2010), where an overall decline in the TL_m of artisanal fisheries catch was observed over a 12-year period. The decline in TL_m shown here is linked to the shift in fishing effort toward small-sized species, which then become the predominant proportion of the catch. The increased catches of a high volume of low trophic level species landed by the artisanal fisheries then affect the abundance of these species in the ecosystem. This is likely to have a critical impact on ecosystem health as these species are an important component of the food web and help to maintain the trophic structure of the ecosystem (Cury et al. 2003).

Several studies on artisanal fisheries have shown that as fishing effort increases, the diversity index decreases (Albaret and Laë 2003; Kantoussan et al. 2010). The decline in the Shannon diversity index from the 1990s to the 2000s can be explained, in part, by the significant differences in fishing effort and strategies between the two periods (Daan et al. 2005; Jouffre and Inejih 2005). In fact, higher fishing pressure is commonly associated with lower diversity and the higher dominance of a few species in the ecosystem (Gislason and Rice 1998; Pinnegar et al. 2000).

Decreases in the biological indicators used in this study (i.e. CPUE, TL_m and the Shannon diversity index) are significantly linked to the effects of intense fishing pressure, as found in other studies. For example, Lobry et al. (2003) observed that decreases over time in the abundance and species richness of fish on the continental shelf of Guinea was a result of increased fishing effort. Moreover, recent studies have shown that different trophic-level-trend scenarios can correspond to specific sequences of the evolution of a fishery (Foley 2013).

The present results indicate that fishing pressure is affecting the composition of catches off the Senegalese coast. Therefore, our results indicate that changes in fishing pressure may play an important role in changing the trophic structure, diversity and abundance of fish species in the Senegalese coastal ecosystem, consistent with the findings of several studies (Munyandorero 2006; Coll et al. 2010; Link et al. 2010). However, fishing effort is not the sole factor affecting ecosystem dynamics, as we also found that chlorophyll *a* concentrations significantly influenced the trophic structure and abundance of fish species.

Our study tested the response of three indicators (TL_m, the Shannon diversity index and CPUE as an abundance index) to the effects of both fishing effort and environmental changes, with the aim of better estimating how biological indicators can be used in fisheries management. The results of this paper identify the importance of simultaneously assessing the individual and cumulative effects of fishing effort and environmental factors on biological indicators of ecosystem health. This paper supports their inclusion in resource and fisheries management, as has been requested by other recent studies (Link et al. 2010; Shannon et al. 2010).

Conclusion

This study suggests that environmental changes and fishing pressure are both influencing the fish assemblage dynamics along the northern coast of Senegal. These results show that integrating environmental effects is an important consideration for fisheries management. The results of this study also demonstrate the urgent need to again investigate the potentially significant role of coastal artisanal fisheries on the degradation of fishery resources. This paper helps to better inform the management of fisheries resources by providing decision makers with more effective biological indicators that incorporate the effects of fishing pressure and environmental change and that are applicable at local, regional and global scales.

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