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Long-term evolution (1988–2008) of *Zostera* spp. meadows in Arcachon Bay (Bay of Biscay)

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Abstract:

The spatial variability of seagrass meadows in Arcachon Bay, was studied between 1988 and 2008 using a combination of mapping techniques based on aerial photographs for intertidal dwarf-grass (*Zostera noltii)* beds and acoustic sonar for permanently submerged eelgrass (*Zostera marina*) populations. The results show a severe decline over the period for both species, as well as an acceleration of the decline since 2005 for *Z. noltii.* The total surface regression over the studied period is estimated to be 22.8 km^2 for *Z. noltii* and 2.7 km^2 for *Z. marina*, which represent declines of 33 and 74% respectively.

Environmental data time series spanning the same period were investigated in order to seek the causes for such a decline. The calculated inter-annual trends for temperature, salinity, nitrate plus nitrite, ammonia, suspended sediment and chlorophyll *a* did not identify any clear environmental change capable of explaining the observed seagrass regression. For instance, no evident sign of eutrophication was observed over the study period. On the other hand, we suggest that the observed variations of ammonia in the inner part of the lagoon are a symptom of the seagrasses' disappearance and thus, a first sign indicating a change of the Arcachon Bay ecosystem towards more instability and vulnerability. Several hypotheses to explain the observed seagrass decay are proposed.

Keywords: seagrass; temporal evolution; aerial photography; acoustic sonar; *Zostera noltii*; *Zostera marina*; Arcachon Bay

1. Introduction

Seagrass beds represent a fundamental biological compartment of littoral ecosystems (Hemminga and Duarte, 2008 ; Larkum et al., 2007). Numerous biotic as well as abiotic environmental factors control their variability, either facilitating their expansion or contributing to their decay. Seagrass habitats are moreover the centre of a growing interest, not only for their fundamental ecological function, but also because they represent the indirect resource for numerous human activities such as fishing and tourism (Hemminga and Duarte, 2008 ; Larkum et al., 2007 ; Frederiksen et al., 2004b).

The health status of seagrass beds is considered as a bio-indicator of the quality of tidal ecosystems (Duarte, 1999). When considering a continuum between pristine and highly-anthropised areas, it is nowadays admitted that healthy seagrass meadows stand for ecosystems of good quality, shifting towards the pristine side of the continuum. However, the difficulty is to assess the proper parameters and appropriate thresholds that can separate a healthy bed from an unhealthy one (Krause-Jensen et al., 2005 ; Frederiksen et al., 2004a ; Van Katwijk, 2003 ; Olesen and Sand-Jensen, 1994). Measuring the expansion or decrease rate of a seagrass bed surface area, within spatio-temporal variability studies, would give useful information on the evolution of the ecosystem and its quality (Frederiksen et al., 2004a ; Frederiksen et al., 2004b ; Van Katwijk, 2003).

To study this surface area variability, photo-interpretation of aerial photography is commonly used and permits a good assessment of the leaf-cover (Godet et al., 2008). Another method consists in field study, using a sub-metric differential Global Positioning System (dGPS), which generates a precise spatial positioning of the inventoried surfaces. In subtidal areas, a towed underwater video camera can be used, as described in Haag et al. (2008) or in Kendall et al. (2005). More recently, the use of acoustic sonar has been developed, since it usually permits covering wider areas in a shorter time (Komatsu et al., 2003 ; Pasqualini et al., 2000 ; Pasqualini et al., 1998 ; Siljeström et al., 1996).

These various methods provide data which are set inside a geo-referenced space. In terms of analysis, the Geographical Information System (GIS) is the most powerful tool for describing the spatial characteristics of benthic environments and is therefore particularly well suited for monitoring seagrass beds (Fergusson and Korfmacher, 1997 ; Robbins, 1997 ; Remillard and Welch, 1993).

Arcachon Bay is a tidal ecosystem sheltering European largest seagrass bed of dwarfgrass (*Zostera noltii*) (Larkum et al., 2007; Auby and Labourg, 1996). Next to it, located in the subtidal zone, a smaller bed of eelgrass (*Zostera marina*) grows. Although both species are perennial in the Bay, *Z. noltii* displays important seasonal variations in density and biomass (Auby and Labourg, 1996), as is probably also the case for *Z. marina* if we refer to similar temperate North-eastern Atlantic eelgrass meadows (Jacobs, 1982).

Recently, Waycott et al. (2009) reported the ongoing decline of seagrasses worldwide and the induced threat to coastal ecosystems as for instance loss of habitat, loss of biodiversity, sediment erosion, degraded water quality, lower primary production and carbon sequestration. In this context, the object of this study was to characterize for the first time the spatial variability of the seagrass communities of Arcachon Bay over the last 20 years. For that purpose, a combination of techniques were used at the scale of the whole Bay : GIS mapping of aerial photographs and acoustic sonar maps, underwater video imaging and field validation. With this in view, environmental data from a perennial monitoring survey (ARCHYD) were used in order to investigate the causes that could eventually explain the seagrass beds variability.

2. Materials and methods

1.1 Study area

Arcachon Bay is a triangular shaped meso-tidal lagoon (Fig. 1), located on the French Atlantic coast (44°40 N, 1°10 W). It communicates with the Atlantic Ocean through two narrow passes. The tide is semi-diurnal and the tidal amplitude varies from 0.8 to 4.6 m. Mean temperature is 22.5°C and 6°C in summer and winter respectively, and fluctuation in fresh water contributions (rivers and rainwater) influence water salinity between 22 and 35.
The total lagoon surface (174 km²) can be divided in two parts: the channels (57 km²), and the intertidal area (117 km²). The main channels have a maximum depth of 25 m and are extended by a secondary network of shallower channels. The intertidal area comprises sandy to sandy-mud substrata (mudflat). Part of this mudflat is settled by dwarfgrass (*Z. noltii*). An important oyster farming activity also takes place on the intertidal mudflats with a total annual production of about ten thousand tons of Japanese oyster (*Crassostrea gigas*). In the immediate continuity of the mudflats, and lining the channels, eelgrass (*Z. marina*) occupies the subtidal sector.
Many little streams run into the lagoon, but the two main rivers, the Eyre and the Porges, contribute more than 95% (73% and 24% respectively) of the total annual freshwater inflow (813 million cubic meters).

1.2 Seagrass mapping

The ecological features of each of the seagrass species require implementing various techniques in order to determine their extension. The successive methods for mapping *Z. noltii* and *Z. marina* meadows are presented hereafter.

1.2.1 *Zostera noltii*

Mapping methods

For the tidal zone, three sets of aerial photographs were used : summer 1989, July and September 2005 and August 2007. These high resolution photographs (50 cm resolution for 1989 and 2007, 20 cm for 2005) were supplied in 1989 and 2007 by the *Institut Géographique National* (IGN) and in 2005 by the *Syndicat Intercommunal du Bassin d'Arcachon* (SIBA). They respect optimal ortho-photography acquisition conditions, i.e. flight during spring tide and low water ± 1 hour, the cloud layer was moderate and the sun position was as close as possible to the zenith in order to avoid shadows (appropriate window: 11 h - 16 h, local time). Photographs were all taken during annual maximum of biomass, i.e. between June and September according to Auby and Labourg (1996). Polygons of vegetalized surfaces were made by subjective photo-interpretation (2005 and 2007, digital photographs) or stereo-photo-interpretation (1989, silver photographs, paper prints), under GIS (ArcGis© 9.2). The minimum mapping unit surface was 75 m^2, and maps were produced at 1:25,000 scale for 1989 and 1:5,000 scale for 2005 and 2007, under the extended Lambert II coordinates system.

Field validation

Field validation was performed in September and October 2007. First, limits of seagrass beds were geo-referenced by means of a dGPS (Trimble™ geoXT) during a pedestrian walk

over the mudflats. During the progress, the dGPS recorded one location per second. Secondly, a sub-metric precision was obtained by adjusting the recorded position, relative to a network of fixed beacons. Post-treatments were done by means of the software GPS Pathfinder© 3.1x. Geo-referenced data were converted into polygons, exported in shape files compatible with ArcGis© 9.2. Comparison of photo-interpreted contours with true extent limits, allowed the improvement of subjective interpretation, as well as the rectification of photo-interpreted polygons in areas travelled on foot, to produce a seagrass map as accurate as possible. About 36% of year 2007 subjective polygons were proceeded on foot for part or all of their contours.
Moreover, a leaf cover percentage was assessed using quadrats and landscape photographs during the progress on foot. Each time a change in the seagrass cover was noticed , several pictures were taken, geo-referenced and stocked within a database. Using the Alloncle et al. (2005) works as a reference, three covering classes (expressed as a percentage) were individualized as shown in Tab. 1. Comparing the information collected in the field and the aerial photographs for areas travelled on foot, allowed us to estimate the leaf cover in the rest of the Bay.

1.2.2 *Zostera marina*

Mapping methods

The first spatialized data on *Z. marina* beds in the Arcachon lagoon dates back to 1988. At that time, the eelgrass extension for the whole Bay was outlined using photo-interpretation and field sampling by Bouchet (unpublished data). The map was produced at the 1:25,000 scale.
During July 2008, the *Z. marina* map was actualized using a method implementing different acoustic techniques. Acoustic sonar scan has been previously used in numerous studies for mapping permanently submerged vegetation (Sabol et al., 2002 ; Siljeström et al., 1996). In this study, we coupled the following two different techniques : on one hand, a side scan sonar (GeoSWATH™, frequency 250 kHz), and on the other hand, a mono frequency echo sounder (Simrad™ ES60). Both systems were operated from the same boat, together with a dGPS (Leica™ GX1230 RTK) giving the boat's geographical position with sub-metric precision.
The backscattered signal of the echo sounder is very different depending on whether the seafloor is covered or not by vegetation, and permits discrimination between bare sediment and phanerogam canopies. Moreover, the echo sounder provides a bathymetry survey, which is a useful addition of information for seagrass beds demarcation. Contrary to the echo sounder that provides information on a relatively narrow band just below the boat (about 1 meter wide, for a depth up to 10 meters), the side scan sonar allows the definition of homogeneous acoustic reflectivity areas on a relatively wide band (30 m on both sides of the ship). All information were simultaneously transferred to onboard computer, in order to make real time interpretation and modify the boat course if necessary. Lastly, putting together all different kinds of data allowed the production of an ArcGis© 9.2 map (scale, 1:5,000).

Field validation

As mentioned above, the method for analysing the acoustic signals depends on the type of underwater vegetation and the physical structure of the environment (Siljeström et al., 1996). Arcachon lagoon is a site of intense Japan oyster production, and wild oysters reefs are widespread across the basin. Consequently, in cases of very low shoot densities, the sonar cannot discriminate with certainty between seagrasses, or another type of substrate (e.g.

oyster beds). Thus, an important part of the present work was to calibrate the method for this type of tidal system with non-uniform benthic relief.
In order to discriminate precisely between the different surfaces mapped by the sonar and to confirm eelgrass presence, a video camera was towed behind a second boat, as was described in other works aiming to map benthic aquatic vegetation (Kendall et al., 2005 ; Sabol et al., 2002). The towed video system (Ifremer-MOOGLI©) is composed of a Plexiglas frame containing the video camera (Sony™ model XC 555), removable ballasts and the tow cable. Images of the sectors for which there was the greatest uncertainty were then inspected visually on board by means of LCD screens and recorded on a computer together with depth and geographical position provided by a Humminbird™ 917c Combo equipment. Software (Ifremer-Videonav©) automatically records the different sensors data on a video time code. The operator has also the possibility to tag short information on images during the process. The boat speed during the hauling operation was too low (1 knot maximum) to inspect large areas. Thus, the images were sufficient to discriminate between benthic substrata in all doubtful areas, but were insufficient to provide coverage information for all *Z. marina* meadows. Thus, no information on covering percentages for *Z. marina* will be reported.

1.3 Environmental data

Environmental data were twice-monthly collected at two stations (InnerS and OuterS, refer to Fig. 1 for locations) during high tide, for the whole study period (1989-2008). Temperature and salinity were measured at 1 meter depth, using a WTW LF 197-S conductimeter (accuracy 0.5%). Seawater was collected at 1 meter depth using a Niskin sampling bottle, for nutrients, suspended sediment and chlorophyll *a* analysis. Samples for ammonium were fixed immediately on board, and subsequently analysed following the indophenol coloration method (Koroleff, 1973), whereas samples for other nutrients (nitrate plus nitrite) were frozen for further analysis on a segmented flow analyser according to the technique of Tréguer and Le Corre (1975). Chlorophyll *a* was determined after filtration of 100 mL of seawater (Whatman GF/F), using a spectrofluorimetric method after extraction in 90% acetone (Neveux and Panouse, 1987). Particulate matter samples (1 L) were filtered through pre-combusted (500 °C for 3 hours) and pre-weighed Whatman GF/F filters. After the filtration, filters were rinsed with isotonic ammonium formate to remove salts. Then, filters were dried for 24 hours at 60 °C and weighed for total particulate matter. Inorganic matter was given by mass of ash remaining after burning at 500 °C for 5 h (suspended sediment, mg ash free L^{-1}).

1.4 Data analysis

The spatial analyses (mapping and surface area calculations) were realized by geo-treatment under ESRI™ ArcGis 9.2 software.
Environmental data inter-annual trends were obtained by decomposition of the time series into seasonal, trend and residual components, using LOESS smoothing (Cleveland, 1979) after regularisation. All statistical treatments were performed using the R software (R development team, 2008) and the Pastecs library (Ibanez et al., 2009).

3. Results

The dwarfgrass meadows spread over the whole bay, on intertidal mudflats, between 0.3 and 3.1 m above lowest astronomical tide (Fig. 2). However, our results show that the total

surface of dwarfgrass underwent an important regression over the whole period (1989 to 2007) with an acceleration during the last two years (Tab. 2). *Z. noltii* total surface area declined by 11.1% between 1989 and 2005, and by 25% between 2005 and 2007. The total surface regression over the studied period is estimated to be 22.8 km², ca. 33 % of year 1989 total surface.
The decay of *Z. noltii* beds was not spatially homogeneous. Fig. 3 shows that the surface drop concerned mainly the eastern sector of the Bay, and specially the Eyre river outlet area. On the contrary, the meadows located in the western part and in the north of the bay remained stable along the study period. In the south end of the lagoon, near the Eyre river outlet, only scattered spots of seagrasses with low densities subsisted. The areas where *Z. noltii* disappeared were not, at least until the pedestrian survey in October 2007, colonized by any other macrophyte, leaving bare sediment uncovered instead. *Z. noltii* bed progression took also place in some rare spots.
A small *Z. noltii* meadow was also mapped on Arguin Bank in 2005 and 2007 (refer to Fig. 1 for location). No information was available before year 2005 on this area, it seems nevertheless that this meadow resisted better, exhibiting a stable total surface during the period 2005-2007 (Tab. 2).

The *Zostera marina* meadows are located on the edges of the main channels (Fig. 4). A small eelgrass bed was also found in the Arguin Bank area. The calculated *Zostera marina* total extensions for years 1988 and 2008 are reported in Tab. 2. Over the study period, *Z. marina* underwent regression that was more important than for *Z. noltii*, with a 73% loss (2.7 km²) in 19 years. In the same way as *Z. noltii,* the loss of *Z. marina* was not spatially homogeneous. Eelgrass decline was observed in the whole lagoon, but not to the same extent everywhere. The south-eastern and the northern sectors of Arcachon Bay exhibited the most severe eelgrass loss (Fig. 5), with total disappearance in those two sites. This has been confirmed by underwater video images. As opposed to *Z. noltii*, the eelgrass population located on Arguin Bank also declined during the study period.

The temporal evolution of environmental data differs widely among the parameters, as well as between stations (Figs. 6 and 7). Logically, the influence of the open ocean is stronger at the outer station (OuterS, 17.5 m deep), whilst the inner station (InnerS, 7 m deep) is under the direct influence of the Eyre river, the intertidal mudflats and the salt marshes. Thus, the amplitude of variation for temperature, salinity and nutrients was higher at InnerS than at OuterS. Temperature maximum and minimum were measured respectively during summer 2006 (26.9°C) and winter 1992 (4.9 °C) at InnerS. Lowest salinity (17.9) was measured at InnerS during February 1991, and the highest salinity was observed at OuterS (35.5) during summer 2007. Nitrate plus nitrite maximum (78.2 µmol L^{-1}) was measured at InnerS during February 2003. Highest ammonia concentrations were measured at InnerS, with a 6.8 µmol L^{-1} maximum reached in October 2006. Suspended sediment highest level was obtained in 1989 with 28.4 mg L^{-1} at OuterS, and chlorophyll *a* maximum and minimum were respectively 0.2 and 10.4 mg L^{-1} during June and March 1994 at station OuterS.
For both stations no clear trend for temperature was obtained over the whole study period. This is not the case for salinity, which has slowly increased at both stations during the past 20 years. Over the study period, nitrate plus nitrite did not show a clear tendancy either to increase or decrease, at station InnerS. On the contrary, at OuterS, the regression line decreases at the beginning of the time series, and seems to stabilize from 1995 and onwards. This is probably due to the very high concentrations (ranging 20 to 30 µmol L^{-1}) measured at the end of year 1990, that have never been reached again since. Suspended sediment concentrations at the inner station showed a constant slight decrease over the study period. At OuterS, the suspended sediment levels remained stable until year 2000 and started to decrease slowly afterwards. Moreover, organic particulate matter (data not shown)

exhibited roughly the same temporal pattern as suspended sediment. For both stations, the same general trend was found for Chlorophyll *a* concentrations : an initial slight decrease followed by an increase at the end of the study period. The difference between the two stations lies in that at the outer station the early decrease lasted until 2004, whilst at InnerS, chlorophyll *a* concentrations were found more or less stable between 1994 and 2004. Among the monitored parameters, ammonia showed the strongest variations over the study period despite a few differences between the stations. At the inner station, the calculated trend clearly shifted from a slight decrease prior to 1997 towards a strong increase until the end of the time series. At OuterS the same pattern was observed, however, the increase was much less pronounced.

4. Discussion

The variation of seagrass spatial distribution in Arcachon Bay during summer, was studied by means of aerial photographs (*Zostera noltii* on intertidal areas) and acoustics sonars (*Z. marina* on subtidal areas). Seagrass beds were mapped in 1989, 2005 and 2007 for *Z. noltii* and in 1988 and 2008 for *Z. marina*, and inter-annual comparisons of total extensions have shown an important decline for both species over the study period (33% total loss for *Z. noltii* and 74% total loss for *Z. marina*). Mean rate of change in *Z. noltii* area were respectively -0.7% year^{-1} between 1989 and 2005 and -12.5% year^{-1} between 2005 and 2007. For *Z. marina*, mean rate of change was -3.8% year^{-1} between 1988 and 2008. Our results are unfortunately comparable to the accelerating loss of seagrasses recently reported by Waycott et al. (2009), who estimated to -5 % year^{-1} the median rate of declining seagrass meadows after 1980, at the global scale.
This reduction cannot be attributed to seasonal variations since all maps were made during the same season, in summer, when the seagrass biomass is supposed to be at its maximum (Auby et al., 1996). One could object that the mapping method was based on subjective delineation made on aerial photographs, but the high precision of photographs and the important work of field validation make us reasonably sure of the validity of the maps produced as well as of the reproducibility of the method. For large areas such as Arcachon Bay (total surface is 174 km^2), covering the whole surface by foot would be too time consuming, and aerial photographies offers an acceptable alternative (Frederiksen et al., 2004a, Kendrick et al., 2002). The use of aerial photographs is nevertheless not always possible for permanently submerged vegetation, particularly when high suspended matter loads stop light penetration. In the Arcachon Bay waters, the somewhat high light attenuation (mean suspended sediment concentration is 7.6 mg L^{-1}) does not permit investigation below one or two meters deep. Then, the use of acoustic techniques offers a satisfactory alternative that permits covering large areas within a reasonable time. Again, it is capital to stress the importance of the validation operations (performed here using a towed video camera), in order to take the relevant decision in areas where the presence of eelgrasses is doubtful.
It is also important to remember that the precision of mapping was not equal over the whole study period. As mentioned before, the older maps were produced at the 1:25,000 scale whilst photographs taken during 2005 and 2007 allowed more accurate maps at the 1:5,000 scale. Thus, the surfaces calculated in the present studies, specially those for 1988 and 1989, have to be taken as orders of magnitude. In the same way, reduction in the *Z. marina* surface calculated as the difference between 1988 and 2008 maps also has to be taken with caution, due to the absence of intermediate data. However that may be, the reduction in area is such for both species, that the reported ongoing general decline of seagrasses in Arcachon Bay, seems to us relevant.
Eelgrass populations have already suffered drastic variations in their history. Indeed, eelgrass populations were practically wiped out from Arcachon Bay in the 1930's, during the

global wasting disease (Sigalas, 1935). This period was followed by a slow recolonization till the mid-1950's, when Lubet (1956) described *Z. marina* meadows spreading in almost all Bay's channels. Our data state that eelgrass has severely declined again during the last 20 years, without being able to say if this phenomenon was constant or if it has accelerated in recent years. It nonetheless appeared that surface loss concerned principally the vicinity of the two major rivers outlets, the Eyre to the South-East and the Porges to the North.
For *Z. noltii,* intermediate maps reveal that the decline strongly increased recently, between 2005 and 2007. As for eelgrasses, the surface loss was not homogeneous, and the more impacted areas were found in the oriental part of the lagoon and specially near the Eyre river outlet. Moreover, the year 2007 map shows that the dwarfgrass beds are also in much better shape (cover percent over 75%) in the western part than in the eastern border, where heterogeneous beds and scattered shoots (cover percent below 50%, and even often below 25%) were principally reported.
Decline of seagrass beds has been reported worldwide, and this phenomenon is at the present time considered as a global crisis (Orth et al., 2006). Eutrophication deriving from human activities, has often been identified as responsible for such decay (Burkholder et al., 2007). Castel et al. (1996) reported for Arcachon Bay an increase of nitrogen fluxes between the 1970's and the early 1990's, due to intensive agriculture development on the Eyre river catchment area. Concomitantly, macroalgal blooms (*Monostroma obscurum*) have been reported, but they did not affect the seagrass meadows that have remained almost constant over this period (Castel et al., 1996). Nowadays, eutrophication does not seem of major concern. In fact, the riverine nutrient inputs are stable and moderate (on average the areal N-loading is 460 mmol m^{-2} y^{-1}, De Wit et al., 2001), due to a catchment area remaining mainly dedicated to forestry. Moreover, an important tidal flushing avoids water confinement (Plus et al., 2009). Our data also show that neither nitrate nor chlorophyll *a,* displayed significant increase during the last twenty years. Organic particulate matter did not increase either, and no extensive macro-algal bloom have been reported recently. The major changes that we measured during the study period, concern salinity and ammonia concentrations. The increase in salinity can probably be explained by the lower Eyre river flows recorded from 2001 onwards (data not shown, DIREN - Direction Régionale de l'Environnement - Water Directorat). This might also explain the decrease in suspended sediment at InnerS during the last years of the time series.
According to those results, eutrophication was probably not the principal cause of the seagrass decline in Arcachon Bay between 1989 and 2008. In the same way, oyster farming activity seems not to be relied to this recent seagrass decline. *Crassostrea gigas* is reared in Arcachon Bay since the 1970's (and other European species before that) and no major change in this activity took place during the study period. Indeed, the production as well as the total exploited surfaces are slowly decreasing since 1993 according to the French national comity for shellfish farming (Comité National de la Conchyliculture, pers. comm.). Moreover, the seagrass greatest decline took place in the internal parts of the lagoon, where oyster farming activities are quite rare. Nevertheless, other hypotheses can be listed, and hopefully serve as ideas for future research.
Dredging activities take place permanently in the Bay for harbour and channels maintenance as well as for supplying beaches with sand and to fight the coastal erosion. These activities often increase the seawater sediment loads, thus lowering the available light for the seagrass canopy (Erftemeijer and Lewis, 2006). This impact will logically affect subtidal species in the first place, and secondarily intertidal species. Despite the fact our results did not show any increase of suspended sediment concentrations, a seagrass decline due to a drop in available light cannot be discarded. Indeed, it has to be remembered that our measurements were always done at high tide, when the lowest currents are recorded, and near the surface. Hence, the suspended sediment time series presented may not be the best way to investigate the sediment dynamics and its impact on the seagrasses. In addition to induced

light reduction, burial by fine sediment has been also pointed out as a possible cause for *Z. noltii* decay (Cabaço et al., 2007).
Several wildfowl species consume seagrass leafs and/or rhizomes and roots in the Arcachon Bay : the dark-billed brent goose (*Branta bernicla bernicla)*, the mute swan (*Cygnus olor*), the common coot (*Fulica atra*), the northern pintail (*Anas acuta*), the eurasian wigeon (*Anas penelope*), the gadwall (*Anas strepera*) and the mallard (*Anas platyrhynchos*). Between them, only two species, the brent goose and the swan, present densities likely to play a significant role in the seagrass meadows dynamics. Annual counts have shown that the number of brent geese wintering in Arcachon Bay, has increased significantly in recent years (Fig. 8), the Bay becoming one of the major wintering site in Europe for that species. Contrary to the brent geese, the swans are present all year long in Arcachon Bay and their number also seem to have increased recently (Fouque et al., 2007), with important intra-annual variations according to their life cycle (reproduction, moulting and wintering periods). Mathers et al. (1998) and Tinkler et al. (2009) showed that in Northern Ireland, consumption of *Zostera* leaves and rhizomes by brent geese can have a predominant impact on *Zostera* biomass reduction. Thus, the grazing pressure should be studied as well as its spatial repartition within Arcachon Bay, in order to compare with the present study results.
As mentioned before, the wasting disease caused a general seagrass decay in temperate regions during the 1930's. Even if *Z. marina* was more affected, both *Zostera* species suffered from the marine protist *Labyrinthula zosterae*, and we know that the pathogen continues to affect *Zostera* meadows in North America and Europe. Moreover, evidence of *L. zosterae* inhibition at low salinities were recently reported by McKone and Tanner (2009). Within a context of salinity increase in Arcachon Bay, it would be interesting to investigate if a pathology is at the origin of the seagrass decline.
Recent studies have revealed the presence of herbicide molecules in the Arcachon Bay waters (Auby et al., 2007, Devier et al., 2005), of both agricultural (pesticides) and nautical (hull anti-fouling paints) origin. Amongst the listed molecules, Atrazine, Metolachlor, Acetochlor (of terrestrial origin) as well as Diuron, Irgarol, tributyl tin (TBT) and copper (of nautical origin), were found at concentrations that could maybe have an impact on seagrass development. Irgarol for example, used as a booster for the effectiveness of anti-fouling copper based paints, has been reported as rapidly accumulating in eelgrass leaves. Significant effects on eelgrass photosynthesis have been found at concentrations of 0.18 µg L^{-1} (Scarlett et al., 1999), and interactive effects on eelgrass photosynthetic efficiency for low concentrations of Irgarol plus Diuron, were reported by Chesworth et al. (2004). Up to 0.07 µg L^{-1} of Irgarol have been measured in the Arcachon Bay waters (Auby et al., 2007), and in the context of a rapid increase of the number of boats frequenting the lagoon, it seems important to study the impact of these molecules, separately as well as in synergy.

Seagrass decline consequences are numerous, such as, for instance, loss of habitat, decrease of biodiversity or modification of sediment bio-geochemistry. The role of seagrasses in stabilizing the Arcachon Bay ecosystem has been stressed by De Wit et al. (2001). This role is in fact highly dependent on the seagrass “buffering capacities” thanks to its persistent autotrophy, nutrient uptake, role in nitrogen fixation and denitrification processes, biomass export etc. It is therefore clear that the decline of seagrass habitats will result in a depletion of the buffering capacity in the short or mean-term, that may probably shift the ecosystem towards more instability and vulnerability. In fact, a first sign of such a shift could lie in the increase of ammonia concentrations recorded from 2000 onwards at station InnerS. The capacity for seagrasses to oxygenate the upper sediment layers through oxygen release by rhizomes and roots, has been already established (Jensen et al., 2005 ; Viaroli et al., 1996) and Deborde et al. (2008) have also shown that, for Arcachon Bay, benthic iron and phosphorus dynamics are modified within the *Z. noltii* rhizosphere. Hence, it is likely that the dwarfgrass beds' disappearance in the inner part of the Bay, switched the sediment towards more reduced forms, indirectly increasing the benthic fluxes of reduced nutrients such as ammonia towards the water column. In addition, the nutrient uptake by rhizomes, roots and leaves probably hardly dropped when the seagrasses disappeared.

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Tables

Table 1. Classification of *Zostera noltii* coverage (adapted from Alloncle et al., 2005).

Coverage classes	< 25%	25 to 75%	> 75%
Characteristics	shoots scattered in weak density or very heterogeneous seagrass bed	heterogeneous seagrass bed presenting an alternation of covered spots and patches of bare substratum.	Continuous and homogeneous seagrass bed presenting a strong leaf cover.
Landscapes			
Quadrats **(30 x 30 cm)**			

Table 2. *Zostera marina* and *Z. noltii* beds calculated surfaces (km²) for years 1989, 1989, 2005, 2007 and 2008. When available, the different coverage classes were reported.

Location	Species	Coverage classes	Years				
			1988	1989	2005	2007	2008
Arcachon Bay	*Z. noltii*	< 25%				13.6	
		25-75%				14.7	
		> 75%				17.4	
		Total		68.5	60.8	45.7	
	Z. marina	Total	3.7				1.0
Arguin Bank	*Z. noltii*	< 25%				0.01	
		25-75%				0.04	
		> 75%				0.00	
		Total			0.05	0.05	
	Z. marina	Total	0.09				0.02

Figures

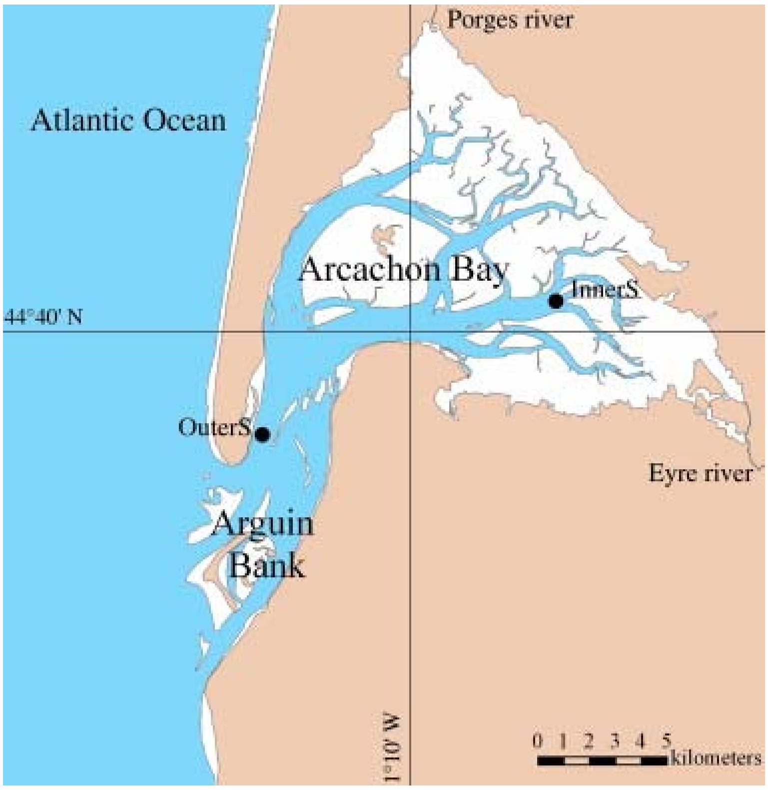

Figure 1. General view of Arcachon Bay at low tide. Intertidal mudflats are presented in light grey.

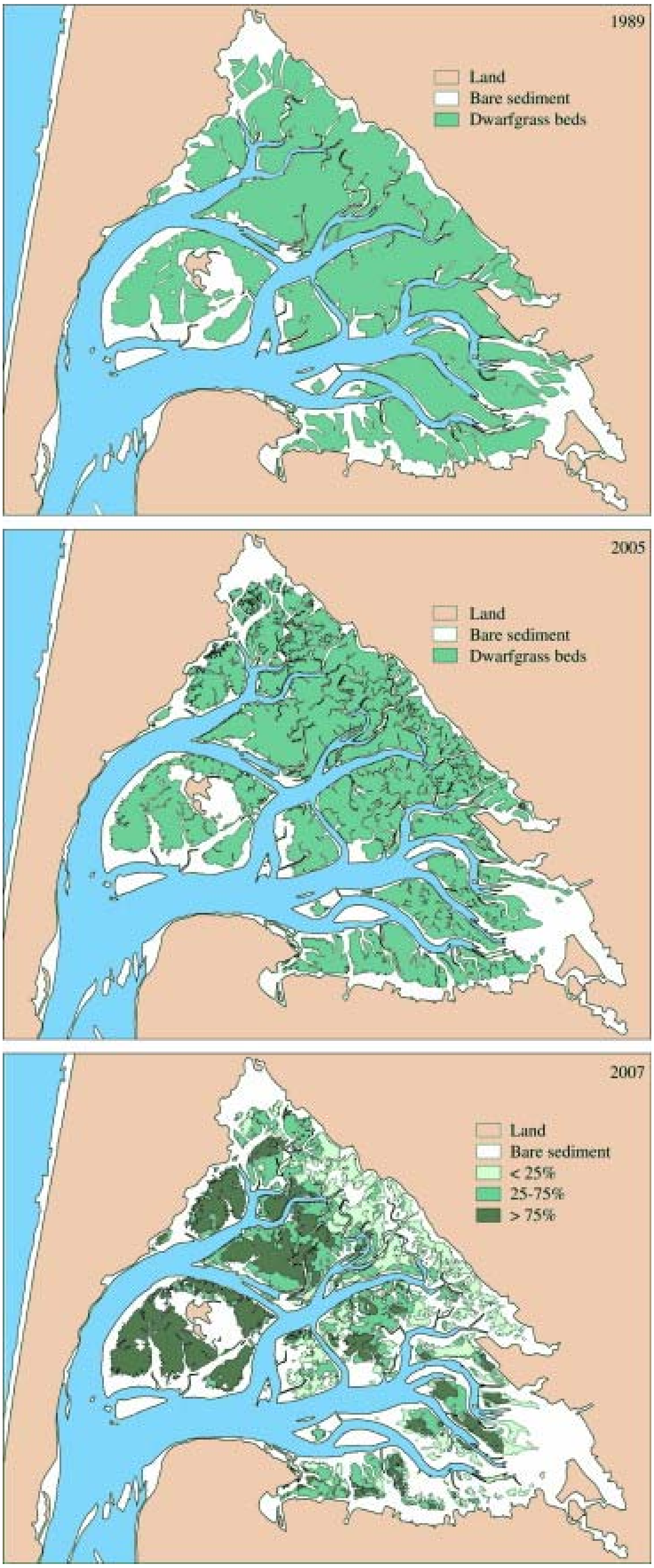
1989
Land
Bare sediment
Dwarfgrass beds
2005
Land
Bare sediment
Dwarfgrass beds
2007
Land
Bare sediment
< 25%
25-75%
> 75%

Figure 2. Total extent of *Zostera noltii* beds in 1989, 2005 and 2007. Coverage classes are given for year 2007.

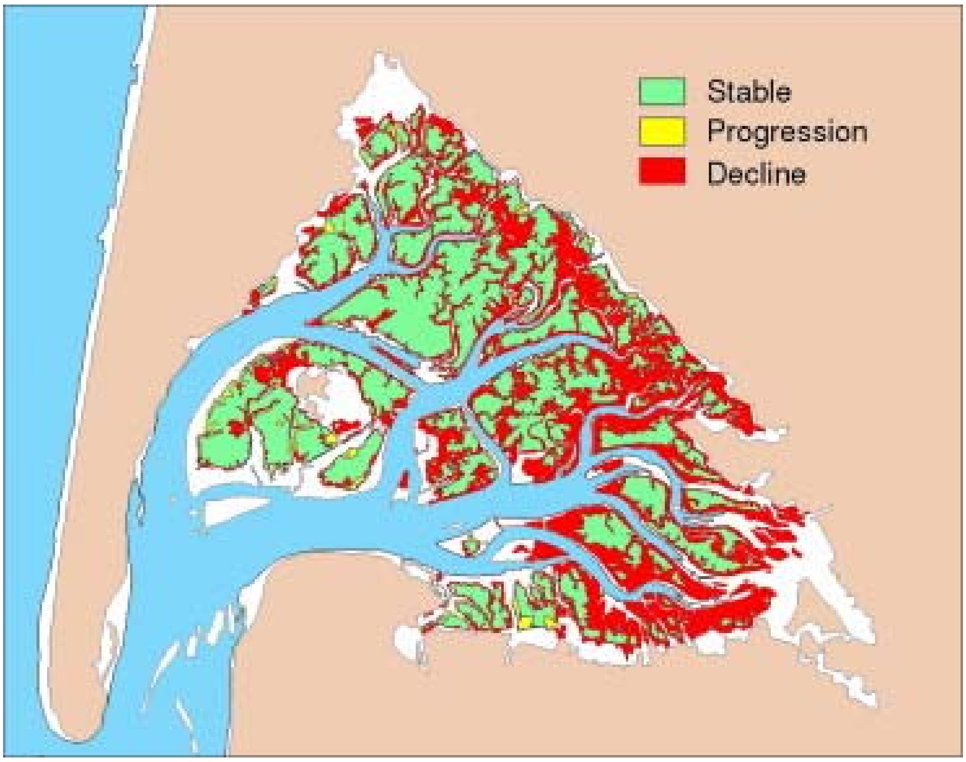

Figure 3. Evolution of *Zostera noltii* meadows between 1989 and 2007. All year 2007 coverage classes were put together.

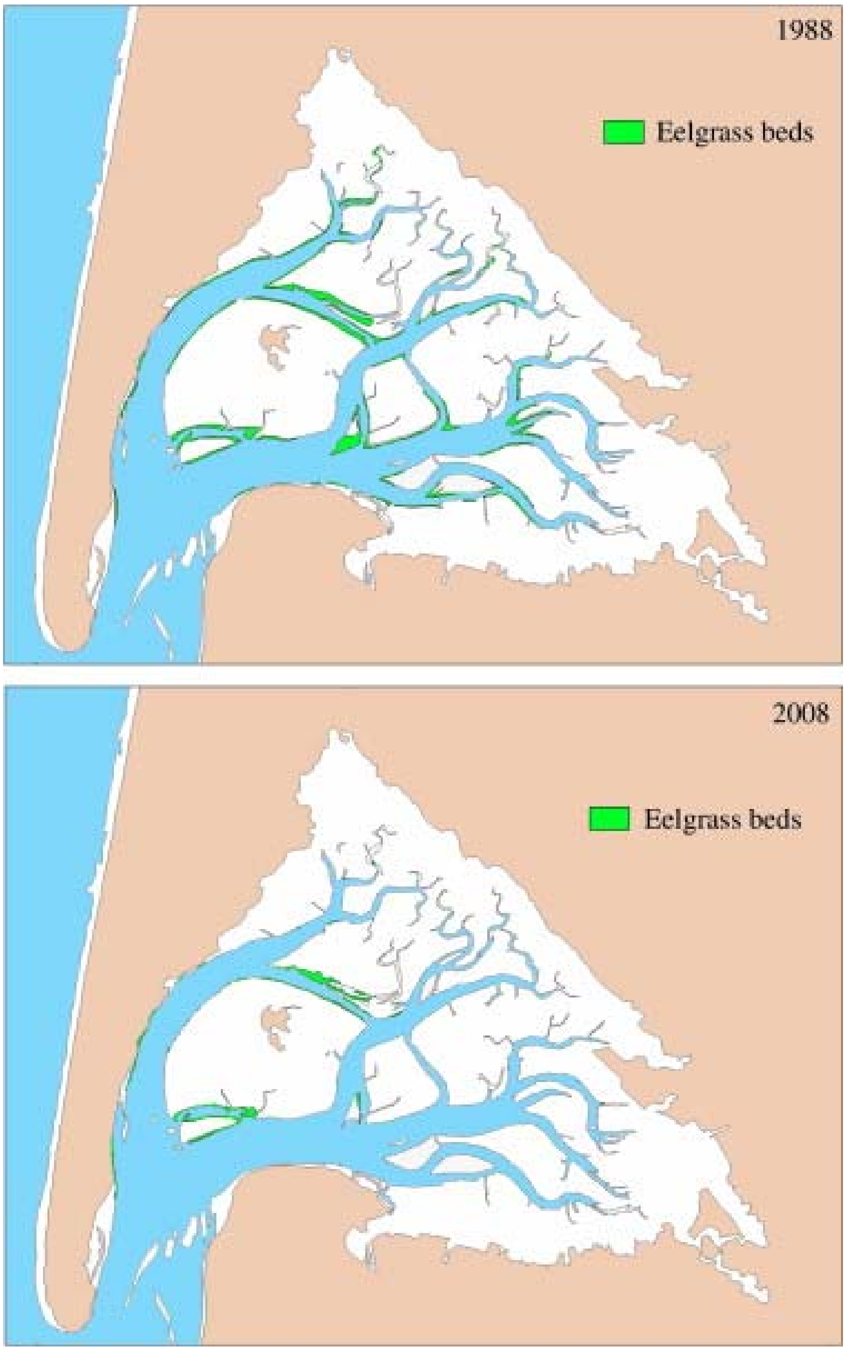

Figure 4. Total extent of *Zostera marina* beds in 1988 (Bouchet, unpublished data) and 2008.

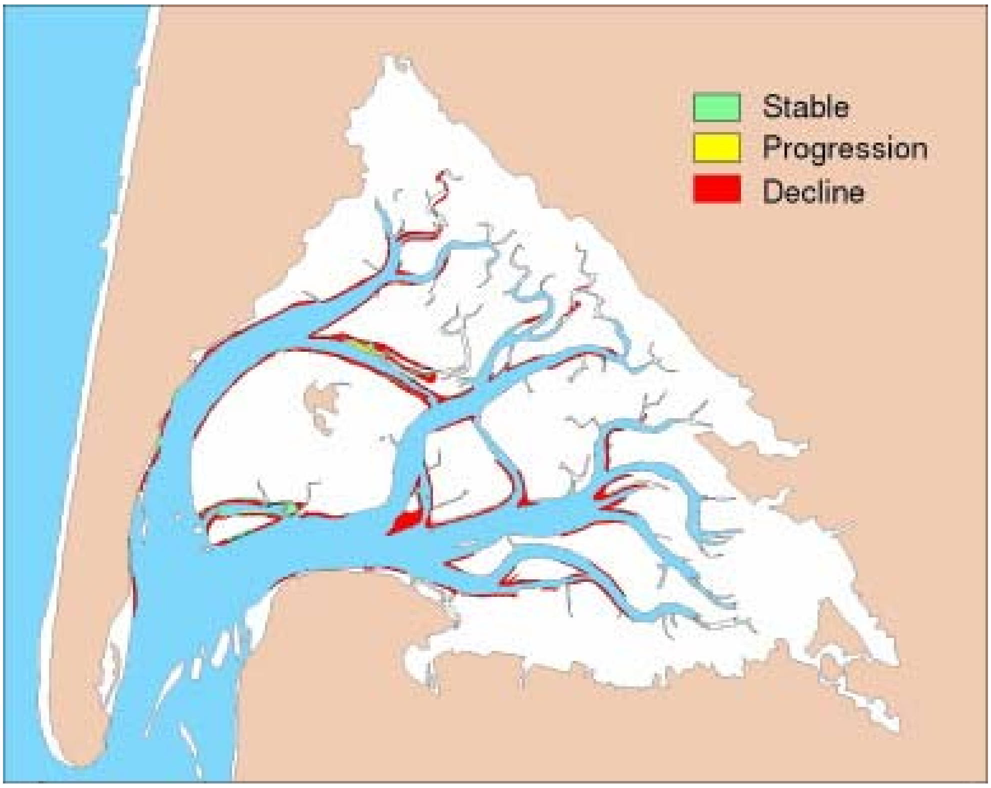

Figure 5. Evolution of *Zostera marina* meadows between 1988 and 2008.

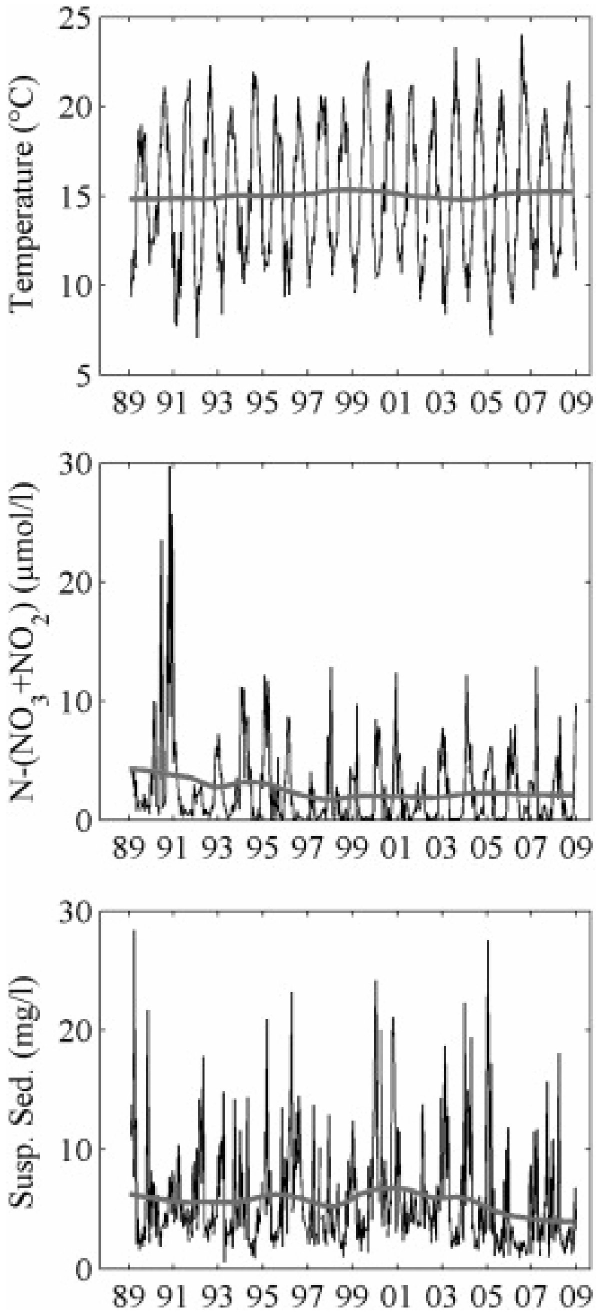

Figure 6. Evolution of surface temperature, salinity, nitrate plus nitrite, ammonia, suspended sediment and chlorophyll *a* at station OuterS between 1989 and 2008. For each graph, the LOESS regression line (gray line) is reported.

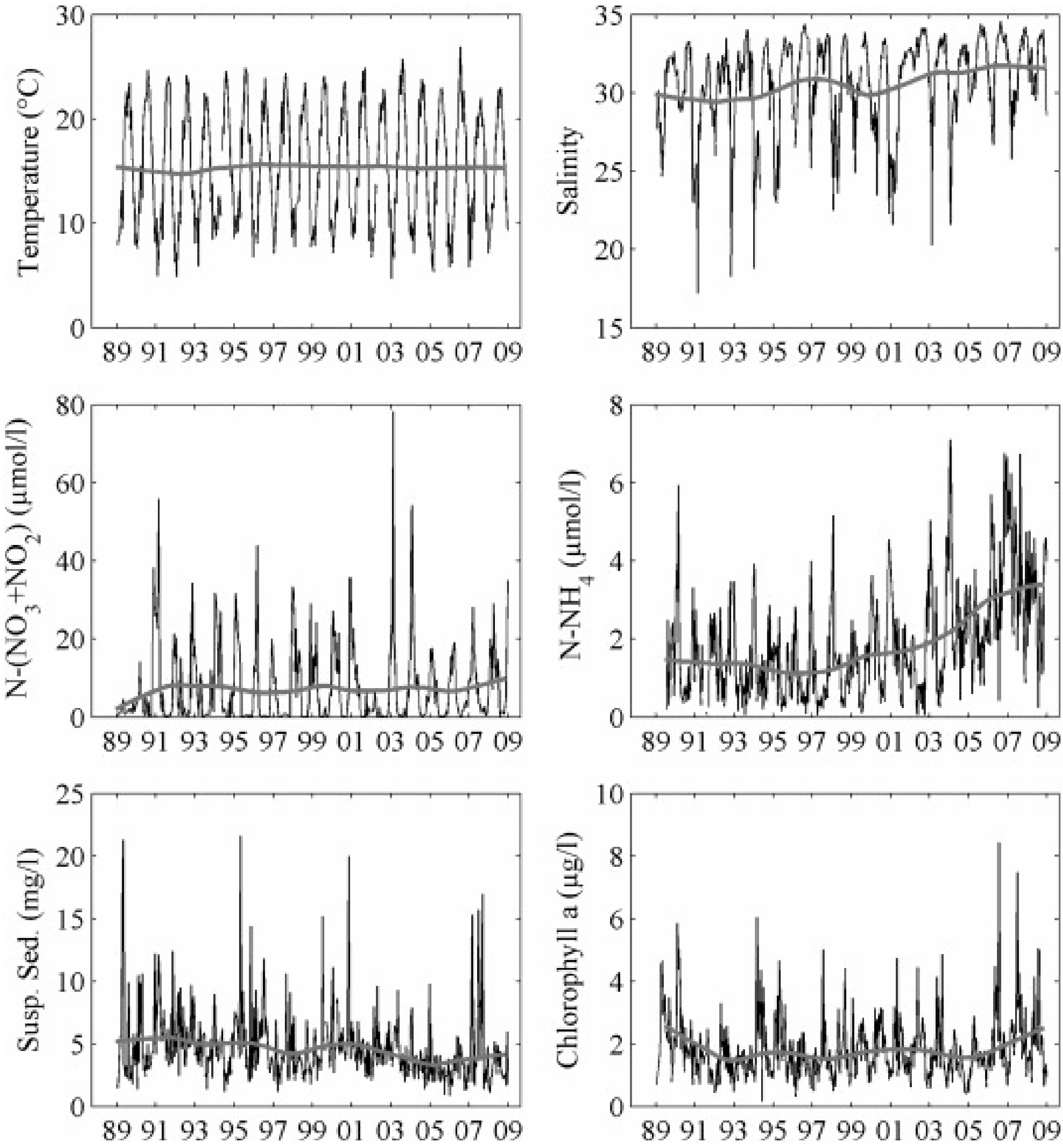

Figure 7. Evolution of surface temperature, salinity, nitrate plus nitrite, ammonia, suspended sediment and chlorophyll *a* at station InnerS between 1989 and 2008. For each graph, the LOESS regression line (gray line) is reported.

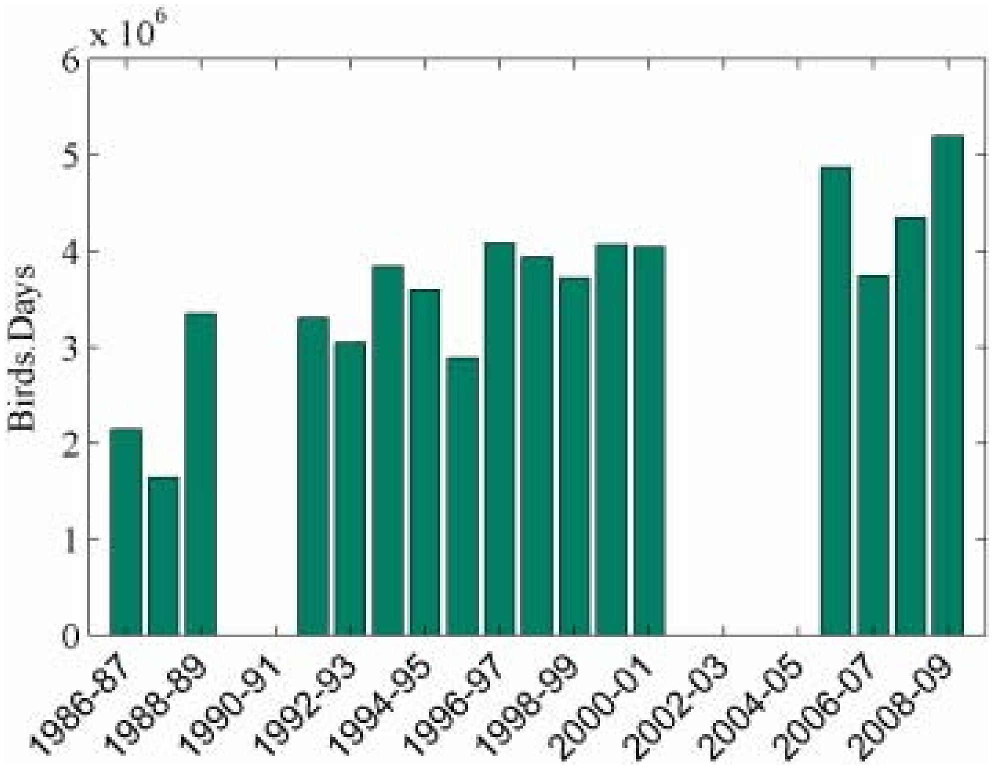

Figure 8. Numerical trend (expressed as the number of birds multiplied by the number of days of presence) of dark-billed brent geese population wintering in the Arcachon Bay between 1988 and 2009. Source : Game and Wildlife National Agency (ONCFS), Departmental (Gironde) Hunting Associations, environmental protection organisations (LPO, Sepanso).