

Chapter 11

Impact of Eutrophication on the Seagrass Assemblages of the Mondego Estuary (Portugal)

Marina Dolbeth, Patrícia Cardoso and Miguel Ângelo Pardal

Abstract Human population has for long been attracted to live on the shores, imposing major pressures on transitional waters (including estuaries, lagoons) and adjacent coastal areas. A wide array of human impacts may be expected, colliding with the ecological function of these ecosystems and threatening their long-term integrity. Among major threats, eutrophication may be considered as a global ecological problem, affecting several worldwide coastal areas. The Mondego estuary (Portugal) is a coastal system, which has suffered eutrophication over the last three decades leading to major changes in environmental quality. Accordingly, this study addresses the responses of macrobenthic community and different key species to eutrophication, providing an insight on potential impacts for the whole ecosystem integrity. In the late 1990s a restoration plan was implemented in the system to control the eutrophication process and its main effects. A review on these major changes, occurring from 1993 to 2002, will be presented focusing on (1) nutrient dynamics; (2) seagrass and macroalgal dynamics; and (3) macrobenthic community biodiversity, density, biomass, production and feeding guilds composition, evaluating both the type and time of the response to the eutrophication effects. Additionally, six species will also be studied in more detail, which are representative of taxa commonly found at estuaries and other transitional waters, and important for the estuarine foodwebs: *Hydrobia ulvae* (Gastropoda), *Cyathura*

carinata (Isopoda), *Scrobicularia plana* (Bivalvia), *Hediste diversicolor*, *Alkmaria romijni* and *Capitella capitata* (Polychaeta).

Keywords Eutrophication · Macrobenthic assemblages · Seagrass · Macroalgal Bloom · Multiple Stressors · Restoration

11.1 Introduction

Human development and the associated increasing population growth in watershed areas underlie many of the environmental problems occurring in freshwater, transitional (e.g. estuaries, lagoons) and coastal ecosystems. Nutrient enrichment (N and P) is one of the most prominent consequences directly related to the human activities (e.g. Kennish 2002, Paerl 2006), with eutrophication now considered as a global and worldwide problem in several aquatic ecosystems (Cloern 2001, Breitburg et al. 2009, Fox et al. 2009). The consequences of eutrophication may vary considerably, due to the integrating effect of the physical and biological characteristics of the ecosystem and climate combined with the nutrient loading itself (Cloern 2001, Rabalais et al. 2009). Several impacts on the biota have been studied, most of them addressing the effects on the structure (Raffaelli et al. 1998, Kennish 2002), and, to a lower extent, on the production (Dolbeth et al. 2003) and composition and stability of recipient food webs (Fox et al. 2009).

Similar to several transitional waters worldwide, the Mondego estuary, in the central Portugal (North Atlantic Ocean), has suffered from an ongoing cultural eutrophication process over the last 20 years. Several impacts were observed in the quality and

M. Dolbeth (✉)
Department of Life Sciences, IMAR-CMA; Institute of Marine Research, University of Coimbra, Apartado 3046, 3001-401, Coimbra, Portugal
e-mail: mdolbeth@ci.uc.pt

quantity of the primary and secondary production levels, with huge increases in macroalgal productivity, concomitant with decreases in the macrophyte one (Lillebø et al. 2005, 2007, Leston et al. 2008), and general impoverishment of the biotic communities (macrobenthic communities: Dolbeth et al. 2003, 2007, Cardoso et al. 2008a, b; birds: Lopes et al. 2006; fishes: Leitão et al. 2007). In the meantime, a restoration programme was implemented in the system to control and reverse eutrophication and its main effects, which included experimental mitigation measures such as changes in hydrology to increase circulation and diversion of nutrient-rich freshwater inflow to the estuary. All these changes occurring in the Mondego estuary have been studied since the last 15 years, providing a large database and a comprehensive information background on the eutrophication event and on the efficiency of the measures to control eutrophication and restore the original seagrass community. Accordingly, this study will present a review on these changes, integrating information on nutrient dynamics, seagrass and macroalgal dynamics and macrobenthic

fauna density, biomass, production and trophic structure. This information will be used to evaluate:

- type and time of the response to the eutrophication effects and
- how effective are being the measures to control eutrophication.

Additionally, the response of six estuarine species, representative of *taxa* usually found in estuarine systems, will be studied in more detail.

11.2 Case Study: The Mondego Estuary

The Mondego estuary (Portugal) locates in a warm temperate region, on the Atlantic coast of Portugal (40°08'N, 8°50'W), near to Figueira da Foz city. It is a small estuary (8.6 km² area), with two arms (north and south) of distinct hydrologic characteristics, separated by the Murraceira Island (Fig. 11.1). The north arm is deeper (4–10 m during high tide, tidal range 1–3 m) and constitutes the main navigation channel

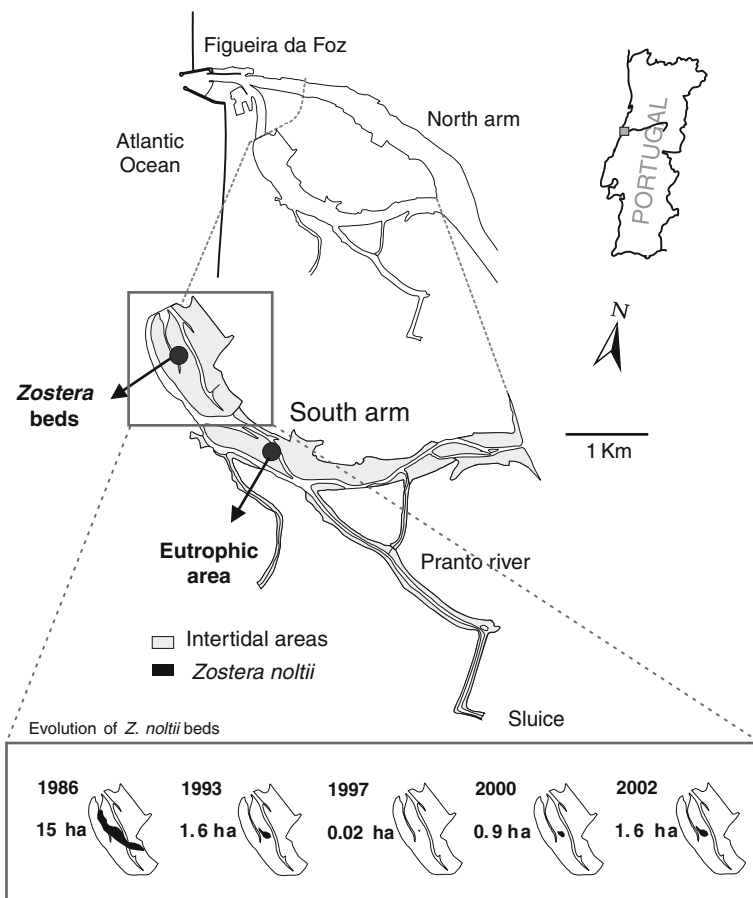


Fig. 11.1 The Mondego estuary, with indication of intertidal area, salt marshes and the seagrass bed (*Zostera noltii*) evolution in the south arm since 1986 until 2002 (box). Mapping of benthic vegetation is based in field observations, aerial photographs and GIS methodology (ArcView GIS version 8.2)

and the location of the Figueira da Foz harbour. The main freshwater inputs to the north arm are from the Mondego River (Fig. 11.1). The south arm is shallower (2–4 m during high tide, tidal range 1–3 m) and characterized by large areas of exposed intertidal flats during low tide (about 75% of total area). Before the implementation of the mitigation measures, water circulation in the south arm mostly depended on the tides and on the freshwater input from the Pranto River (Fig. 11.1), as the upstream areas were almost silted up, with only a small connection with north arm. The downstream areas of the south arm exhibit *Spartina maritima* marshes and *Zostera noltii* beds (Fig. 11.1). More details are available in Cardoso et al. (2008a) and Teixeira et al. (2008).

11.2.1 Anthropogenic Pressures

The Mondego estuary has a high regional value, sustaining several industries (mostly cellulose- and paper-related industries), aquaculture (several old salt ponds transformed into semi-intensive aquacultures), some traditional salt extraction, the location of the mercantile harbour of Figueira da Foz city and a marina (Ribeiro 2002). There is also an increasing human pressure in the area, with more than 60,000 inhabitants in Figueira da Foz city (Ribeiro 2002). Upstream the estuary, in the lower Mondego valley, there are agriculture fields (more than 15,000 ha of cultivated land), producing mainly rice (Lillebø et al. 2005), which drain nutrient-rich waters to the estuary. These anthropogenic activities have been the cause of high environmental pressure on the estuary mostly due to physical disturbance and to the high input of nutrients from agricultural fields and aquacultures (Lillebø et al. 2005, Cardoso et al. 2007, 2008a).

11.2.2 Eutrophication in the South Arm

The high input of nutrients from agriculture fields and aquaculture affected mostly the south arm. In fact, until the end of 1998, the south arm of the Mondego estuary was almost silted up in the upstream areas. Water circulation was mainly dependent on tides and on the freshwater input from the Pranto River (Fig. 11.1). This was artificially controlled by a sluice, according to the rice fields irrigation needs in the lower

Mondego valley. The occasional freshwater flow to the estuary caused the high input of nutrients (discussed below) and high water residence time (about 5–7 days) (Lillebø et al. 2005). Consequently, since the 1980s, clear eutrophication symptoms were observed. The downstream areas maintained *S. maritima* marshes and the *Z. noltii* beds, but in the inner parts of the estuary the seagrass bed completely disappeared and blooms of the opportunistic macroalgae *Ulva* spp. were common (Lillebø et al. 2005, Cardoso et al. 2007, Dolbeth et al. 2007, Ferreira et al. 2007). The occurrence of macroalgal blooms was pointed out as one of the major reasons for *Z. noltii* decline, which was reduced from 15 ha in 1986 to 0.02 ha in 1997 (Fig. 11.1) (Dolbeth et al. 2007, Cardoso et al. 2008a).

11.2.3 Management Measures to Control Eutrophication

In late 1998, experimental mitigation measures were taken in order to control eutrophication process and restore the original seagrass beds of the Mondego's estuary south arm (Lillebø et al. 2005, Dolbeth et al. 2007, Cardoso et al. 2008a). These included the reduction of nutrient loading and the water residence time (from 5–7 days to 1 day) through:

- reduction of Pranto River sluice opening (most of the nutrient-enriched freshwater was then diverted through the north arm by another sluice located more upstream) and
- improving the hydraulic regime, by enlarging the upstream connection between the two arms and allowing water to flow from the north arm at high-tide conditions (Lillebø et al. 2005, Cardoso et al. 2008a).

11.3 Materials and Methods

11.3.1 Sampling Programme and Laboratory Procedures

This study integrates the information taken from the south arm of the estuary from 1993 to 2002. Sampling occurred fortnightly from February 1993 to June 1994 and monthly during the rest of the study period, at

low tide in two contrasting sites regarding an eutrophication gradient: (a) a non-eutrophic *Z. noltii* bed – *Zostera* area and (b) a sand flat where macroalgal blooms occurred – eutrophic area (Fig. 11.1). On each sampling occasion, 6–10 cores (141 cm²) were taken to a depth of 20 cm for the study of the macrobenthic community and evaluation of the flora (algae and macrophyte) dynamics. Samples were washed in 500 µm mesh sieve bags. In the laboratory, the biological material was separated and preserved in a 4% buffered formalin solution. Plant material was sorted and separated into green algae and *Z. noltii*. For both faunal and plant material the ash-free dry weight (AFDW) was assessed, after combustion for 8 h at 450°C. In the same time, salinity was measured in the intertidal pools, while water samples were collected for analysis of dissolved inorganic nitrogen and phosphorus. At the laboratory, samples were immediately filtered (Whatman GF/F glass-fibre filter) and stored frozen, until analysis following standard methods described in Limnologisk Metodik (1992) for ammonia (NH₃-N) and phosphate (PO₄⁻P) and in Strickland and Parsons (1972) for nitrate (NO₃⁻N), and nitrite (NO₂⁻N).

11.3.2 Macrobenthic Feeding Guild Assignments

The feeding guild composition of macrobenthic assemblages was also analysed. Each *taxon* was assigned to a feeding guild according to its food type. Feeding guilds used in this study were carnivores (C), herbivores (H), omnivores (O) and detritivores or deposit feeders (D), with this last group divided into subsurface-deposit feeders (SsDF), surface-deposit feeders (SDF) and suspension feeders (SuF), according to the following literature: Gaston and Nasci (1988, 1995), Sprung (1994), Oug et al. (1998), Mancinelli et al. (1998). Preliminary analysis included the snail *Hydrobia ulvae*, but it was also decided to analyse trophic structure omitting this species, since it occasionally occurred in very high numbers and its inclusion masked changes in other species.

11.3.3 Secondary Production

The secondary production of the macrobenthic community was evaluated, as described in Dolbeth et al. (2007). The methods used were cohort

increment summation method (see below) for the dominant species, Brey (2001) method version 4-04 (worksheet provided in Brey 2001, www.awi-bremerhaven.de/Benthic/Ecosystem/FoodWeb/Handbook/main.htm) for other representative species and for the species with lower densities and biomasses, production was estimated by summing the increases in biomasses from one sampling date to the other. For more details, see Dolbeth et al. (2007). Each species production was then cumulated into community production and also analysed per feeding guild. Additionally, six intertidal benthic species, usually found in estuarine systems, were studied in more detail: *H. ulvae* (Gastropoda), *Cyathura carinata* (Isopoda), *Scrobicularia plana* (Bivalvia), *Hediste diversicolor*, *Capitella capitata* and *Alkmaria romijni* (Polychaeta). For *H. ulvae* and *C. carinata*, all individuals were measured and production was estimated by the increment summation method, after definition of cohorts through size-frequency distribution analysis of successive sampling dates (described in Ferreira et al. 2007, Cardoso et al. 2008b), according to:

$$P_{cn} = \sum_{t=0}^{T-1} \left(\frac{N_t + N_{t+1}}{2} \right) \times (\bar{w}_{t+1} - \bar{w}_t) \text{ and } P = \sum_{n=1}^N P_{cn}$$

where P_{cn} is the growth production of cohort n , N is the density (ind m⁻²), \bar{w} is the mean individual weight (g WW m⁻²) and t and $t+1$, consecutive sampling dates. Population production estimates correspond to the sum of P_{cn} (each cohort production).

For the other species, annual production was computed using the empirical method of Brey (2001) version 4-04, without computing depth, following the recommendations of Dolbeth et al. (2005).

For the species whose production was computed by cohort increment summation method, the fortnightly production dynamics during the bloom will also be presented; for remaining species, whose production was assessed by an empirical method, the fortnightly biomass dynamics will be used.

11.4 Results

11.4.1 Climate

In the Mondego estuary there was a clear seasonal pattern of rainfall over the 10-year period, with the highest precipitation values in the winter (Fig. 11.2a).

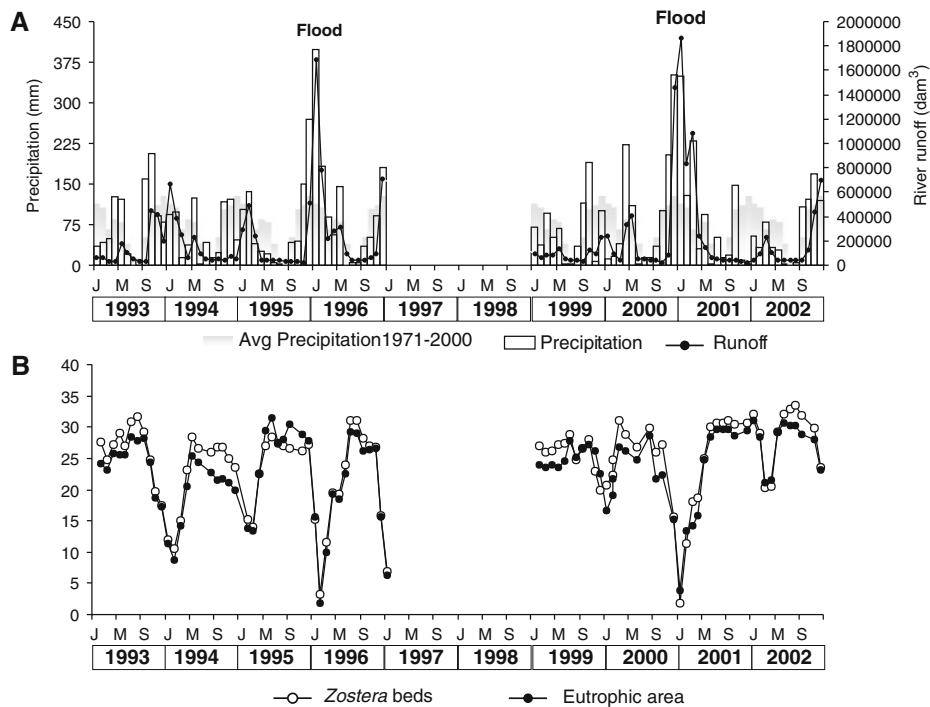


Fig. 11.2 Long-term variation in the (a) measured precipitation during the study period and mean precipitation for central Portugal during the period of 1971–2000 and (b) mean salinity for both study areas

Yet, taking into consideration the mean precipitation regime for central Portugal during the period of 1940–1997 (winter: 418 mm, spring: 265 mm, summer: 62 mm, autumn: 285 mm, INAG – <http://snirh.inag.pt>) some above-mean precipitation periods were evident (Fig. 11.2a). The hydrological years of 1993/94 (autumn: 593 mm) and 1995/96 (winter: 670 mm) were atypical, recording floods, and 2000/01 was even more atypical, with severe flooding occurring (winter: 767 mm) (INAG – <http://snirh.inag.pt>).

The seasonal pattern of rainfall and flooding reflected in the monthly and inter-annual variation of salinity in the south arm. During periods of intense rainfall, salinity declined severally (Fig. 11.2b), occasionally reaching <5 (Feb 96, Jan 97 and Jan 01). During these times of high rainfall there was an extensive opening of the Pranto River sluice, further contributing to the salinity decline (Lillebø et al. 2005).

11.4.2 Nutrient Dynamics

Regarding nutrient concentrations, there are two different scenarios (before and after the application of the mitigation measures) all over the study period.

Concerning the dissolved inorganic nitrogen, there was a strong decline after the implementation of the mitigation measures, at both study sites (Fig. 11.3a). On the other hand, for the dissolved inorganic phosphorus, the pattern of variation was the opposite, with an increment after the management, especially in the eutrophic area (Fig. 11.3b). The *N/P* ratio showed a decline after the implementation of the management measures for both study sites (Fig. 11.3c).

11.4.3 Primary Producers

In the seagrass beds, *Z. noltii* showed an abrupt decline in total biomass during the pre-management period (Fig. 11.4a, $R^2 = 0.84$). After 1998, there seems to be a gradual recovery of its total biomass (Fig. 11.4a, $R^2 = 0.60$), reaching in 2002 similar biomass values to the ones registered in 1994 (Fig. 11.4a). The *Z. noltii* total biomass in 2002 corresponded to almost 50% of 1993 biomass. The green macroalgae *Ulva* spp. were more abundant in the eutrophic area, especially during the pre-management period (1993–1995). In the post-mitigation period, the biomass of green macroalgae tended to decrease, however, a small

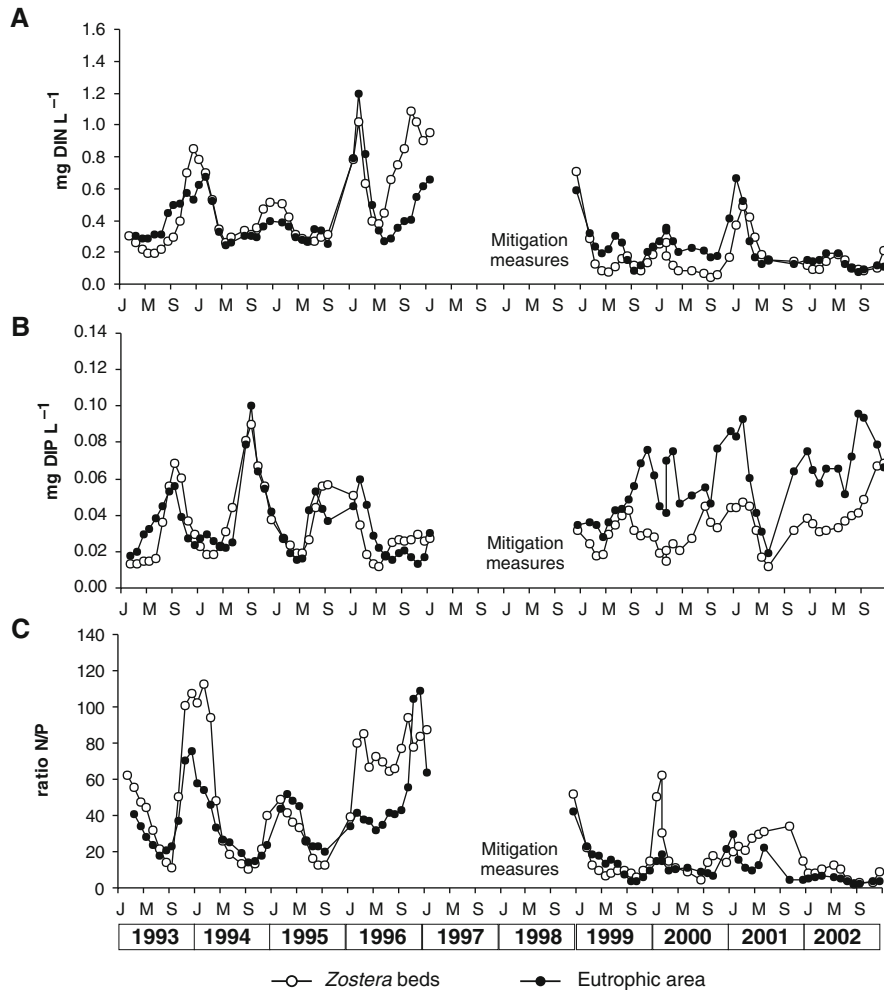


Fig. 11.3 Long-term variation in the nutrient concentrations: (a) dissolved inorganic nitrogen (DIN), (b) dissolved inorganic phosphorous (DIP); and (c) N/P atomic ratio, for both study areas

increase was recorded in the spring of 2000 (Fig. 11.4b). In the opposite, the red macroalgal *Gracilaria* sp. was more abundant in the *Zostera* beds and tended to increase all over the study period. In the eutrophic area, the biomass of *Gracilaria* sp. was vestigial (Fig. 11.4c).

11.4.4 Macrofauna Community General Trends

11.4.4.1 Changes in Diversity

Eighty different *taxa* were recorded over the 10-year period. The seagrass beds generally supported more

species than the eutrophic area, with this difference more pronounced in the pre-mitigation period (Fig. 11.5a). Nevertheless, evenness was higher in the eutrophic area (Fig. 11.5b), mainly due to the dominance of *H. ulvae* in the *Z. noltii* beds, as detailed by Cardoso et al. (2008a).

For both areas, there was a clear decline in the number of species during the pre-mitigation period. Following the introduction of the management plan in 1998, there was a tendency for a species richness increment in both study areas. However in 2000/01, during the high rainfall event, there was a decline in species richness. After this extreme event, species richness only started to recover again in 2002 for both areas. Evenness recovery was more pronounced in the

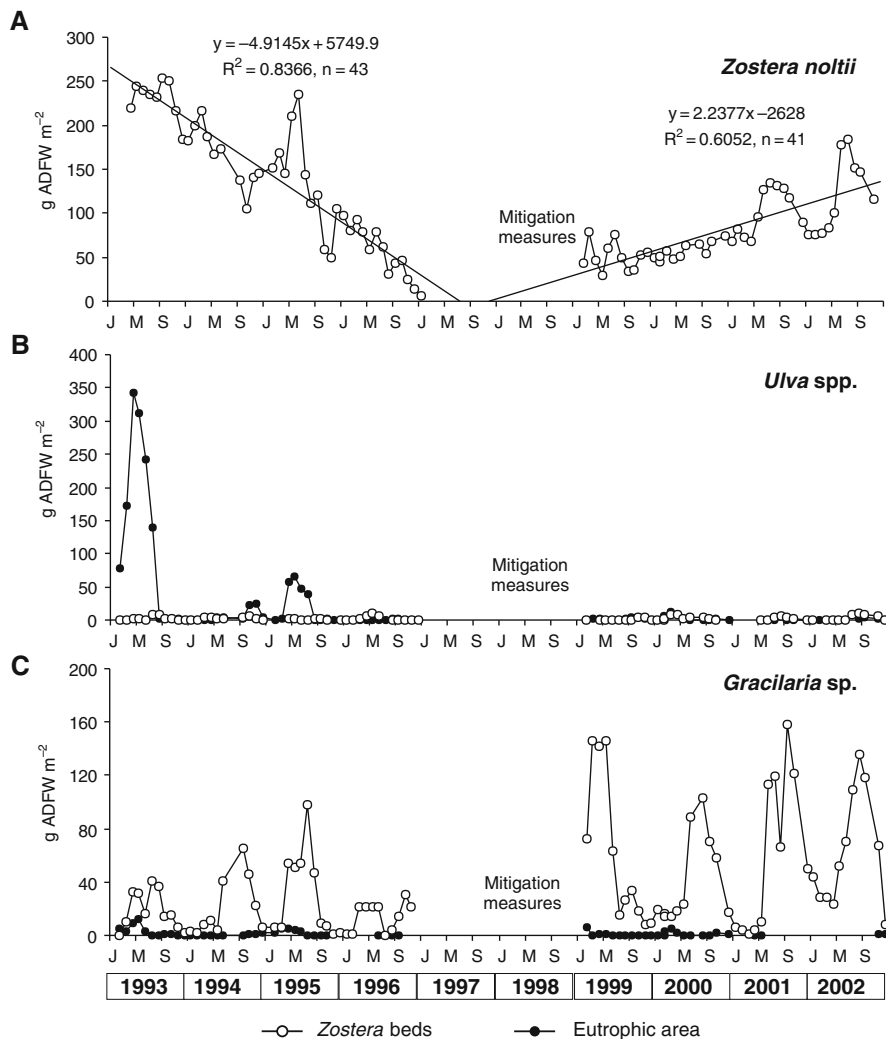


Fig. 11.4 Long-term variation of the seagrass and macroalgal biomass for both study areas: (a) *Z. noltii*, (b) *Ulva* spp. and (c) *Gracilaria* sp

recent years in the eutrophic area (Fig. 11.5b), while in the *Zostera* beds it remained quite stable all over the time.

11.4.4.2 Changes in Density, Biomass and Production

A similar tendency to the diversity changes occurred with density, biomass and annual production. In general, these parameters were higher in the *Zostera* beds than in the eutrophic area (Fig. 11.5c, d, Table 11.1), except in 1999, when higher annual production was obtained in the eutrophic area. Also, there was a general decline in density, biomass and production

when comparing pre-mitigation and post-mitigation periods (Fig. 11.5c, d, Table 11.1).

Mean densities showed a clear seasonal pattern in the *Zostera* beds, with higher values in spring/summer throughout the study period, with the highest value observed in the spring of 1994 (Fig. 11.5c). In the eutrophic area, there was a general decline in total density all over the time (Fig. 11.5c), while total biomass increased considerably in the post-management period (Fig. 11.5d). Contrarily, in the *Zostera* beds, mean biomass declined in the beginning of post-management period. Afterwards it showed an increasing pattern until the occurrence of the flood in 2000/01 (Fig. 11.5d). In the eutrophic area alone,

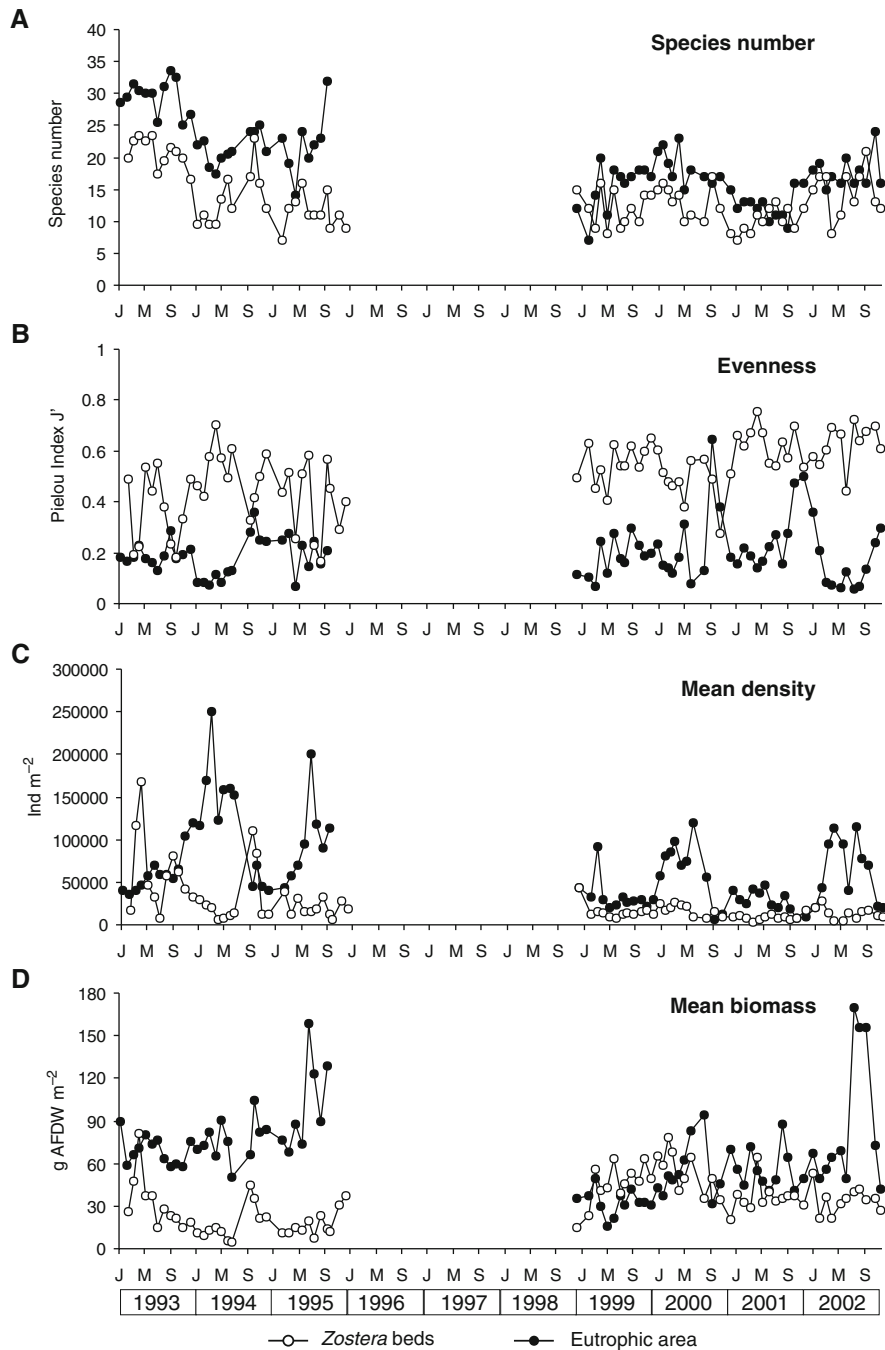


Fig. 11.5 Monthly long-term variation of: (a) species richness; (b) evenness; (c) mean biomass; and (d) mean density, for both study areas

mean density and biomass were higher during spring 1993 and spring 1995 (Fig. 11.5c, d), matching with the periods of higher macroalgal biomass (Fig. 11.2d). The highest value of annual production was obtained

in 1993, the bloom year (Table 11.1). In the post-mitigation period, both mean biomass and annual production (P) were within similar values in all years, appearing to maintain a relatively stability, only with

Table 11.1 Community annual production (*P*) and mean annual biomass (*B̄*) for *Zostera* beds and eutrophic area, during the 10-year study period

	<i>Zostera</i> beds		Eutrophic area	
	<i>P</i> (g AFDW m ⁻² y ⁻¹)	<i>B̄</i> (g AFDW m ⁻²)	<i>P</i> (g AFDW m ⁻² y ⁻¹)	(g AFDW m ⁻²)
1993	148.0	75.1	89.3	31.4
1994	222.9	84.6	45.8	15.8
1995	137.9	105.8	32.8	19.7
1999	45.9	34.9	57.0	45.0
2000	121.3	60.1	60.6	62.3
2001	88.4	54.4	51.7	37.8
2002	199.5	94.8	58.2	35.0

a slightly decrease in 2001, following the highest flood event.

The greatest annual production was obtained for the *Zostera* beds, in 1994 and 2002 (Table 11.1), matching the years when the area covered with *Z. noltii* was practically the same (1.4% of the intertidal area, Fig. 11.1).

11.4.4.3 Feeding Guilds Relative Composition

The analysis of the community feeding guilds relative composition including *H. ulvae* showed that for the *Zostera* beds the most representative groups, both in terms of density and production, were detritivores

and herbivores (Fig. 11.6a). This result was due to the dominance of *H. ulvae* (considered as both as detritivore and herbivore) in the community in this area, with other groups comprising only a small fraction. For the eutrophic area, the herbivores percentage was not so high, with most of the community abundance and production represented by detritivores (Fig. 11.6b). Yet, it is worth to notice that about 25% of the community production was represented by carnivores, and that after the mitigation measures most of the detritivores production in the community increased (Fig. 11.6b). Analysing in detail the detritivore assemblage, surface-deposit feeders (SDF) were the dominant group in both study areas in terms of density, while the relative production contribution of SDF for

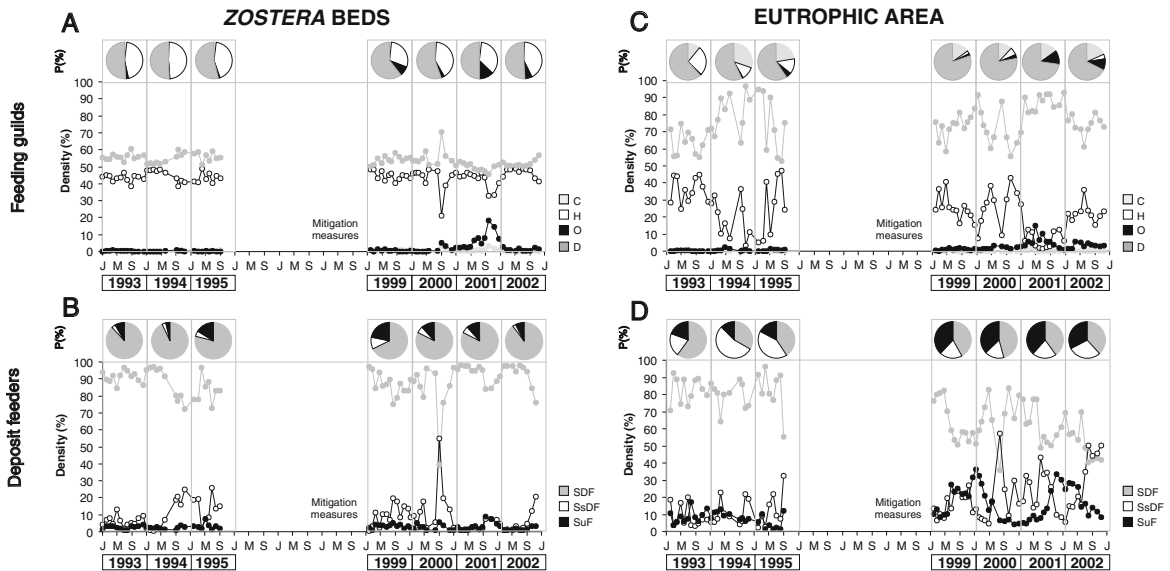


Fig. 11.6 Long-term variation of feeding guilds relative composition regarding density and production in the presence of *Hydrobia ulvae* for the (a, b) *Zostera* beds and (c, d) eutrophic area of the Mondego estuary, with indication of the detailed

relative composition of the detritivores/deposit feeders (c, d). Legend: C, carnivores; H, herbivores; O, omnivores; D, detritivores/deposit feeders; SDF, surface-deposit feeders; SsDF, subsurface-deposit feeders; SuF, suspension feeders

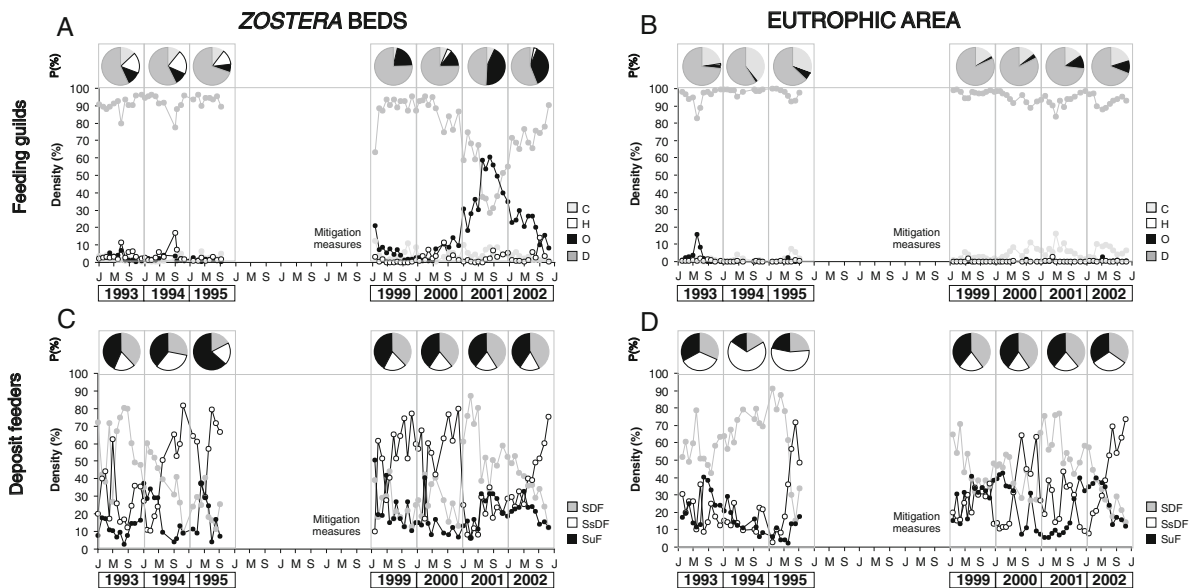


Fig. 11.7 Long-term variation of feeding guilds relative composition regarding density and production in the absence of *H. ulvae* for the (a, b) *Zostera* beds and (c, d) eutrophic area of the Mondego estuary, with indication of the detailed relative composition of the detritivores/deposit

feeders (c, d). Legend: C, carnivores; H, herbivores; O, omnivores; D, detritivores/deposit feeders; SDF, surface-deposit feeders; SsDF, subsurface-deposit feeders; SuF, suspension feeders

the *Zostera* beds was considerably higher than for the eutrophic area (Fig. 11.6c, d). In addition, in the *Zostera* beds after the flood peak (December 2000) there was a large decline in subsurface-deposit feeders (SsDF) abundance and an increase in SDF (Fig. 11.6c). In the eutrophic area, there was a greater variability in the deposit feeders abundance composition from 1998 to 2002 compared to 1993 to 1995 period, which was mostly dominated by SDF abundance. This was evident for suspension feeders (SuF) abundance, which decreased by the large floods of 2000/01 (Fig. 11.6d). Yet an analysis of the production composition reveals a different scenario: while in the pre-mitigation period, the production had a higher percentage of SDF and SsDF, in the post-mitigation a higher percentage of SDF and SuF production contributing to the community was observed (Fig. 11.6c, d).

The analyses including *H. ulvae* were difficult to interpret, due to the masking effect of the large abundance of this species. Thus, an analysis of the feeding guilds composition was also performed for the community excluding *H. ulvae*. These analyses showed that the macrofaunal communities continue to be dominated by detritivores, which together accounted for more than 90% of the total macrobenthic abundance

in both areas (Fig. 11.7a, b), with exception to the *Zostera* beds in 2001. In this year, it was observed a marked decline in the detritivores percentage, followed by a large increase in abundance of omnivores (e.g. *H. diversicolor*) (Fig. 11.7a). When analysing the community production, higher percentages of the other feeding guilds were observed, yet the highest percentage was still due to detritivores production in both areas (Fig. 11.7a, b).

Within the detritivore assemblages in the *Zostera* beds, SDF abundance declined from 1993 to 1999, following the decline of the seagrass *Z. noltii*, and started to increase again in 2001/2002 (Fig. 11.7c). In contrast, SsDF (mainly small polychaetes, such as *C. capitata*) showed the opposite pattern, dominating the community abundance in 1994–2000 and declining abruptly after the floods of 2000/2001 to start to increase again in 2002 (Fig. 11.7c). Yet, again the production analysis showed a slightly different picture, since the community was mostly dominated by SuF production from 1994 to 1995 and showed a similar pattern in the remaining years, being dominated by SuF and SDF production (Fig. 11.7c).

In the eutrophic area, in 1993 and 1994 there was an increase of the SDF abundance, accompanied

by a gradual decrease of SsDF and SuF abundance. However, the production was not dominated by SDF (Fig. 11.7d). Instead, in 1993 similar percentages were obtained by the three deposit feeding guilds, while in 1994 the production was dominated by SsDF (Fig. 11.7d). By the end of the pre-mitigation period, in late autumn 1995, SsDF abundance increased, with this increasing tendency maintained over the 10-year period, except for 2001 (Fig. 11.7d). A similar trend was observed with SuF abundance, yet with lower percentage and with a marked reduction during the floods of 2000/2001, recovering over the following year (Fig. 11.7d). In the post-mitigation period, the relative production of the deposit feeders was similar, dominated by SuF and SDF, even in 2000/01, when the abundance of SuF decreased considerably (Fig. 11.7a).

11.4.5 Species-Specific Responses

11.4.5.1 *Hydrobia ulvae* (Gastropoda)

H. ulvae was the dominant species within the *Z. noltii* beds, where it attained the highest values of density, biomass and production. From 1993 to 1995, *H. ulvae* biomass was relatively constant (range 50–100 g m⁻²), but had two density peaks in 1994 and 1995 (Fig. 11.8a). Following the decline of *Z. noltii*, biomass decreased considerably by approximately an order of magnitude (Fig. 11.8a), which implies a loss of a large number of reproductive adults during this time. *H. ulvae* population appeared to start to recover, until the fall/winter of 2000/01, when the period of prolonged and heavy rainfall was associated with a decline in both density and biomass (Fig. 11.8a). The population seemed to recover again in 2002 (Fig. 11.8a). The growth production followed the density and biomass pattern showing the highest values during the pre-mitigation period, especially in 1994, and in 2002 (Fig. 11.8a). This species was highly productive within the community and in some of the years its production alone represented a major part of the total *Zostera* beds annual community production (Table 11.1).

In the eutrophic area, the pattern of change was completely different. Density and biomass of *H. ulvae* declined throughout the study period, with no obvious indication of recovery in the post-mitigation period

(Fig. 11.8b). Production reached the highest value in 1993, when the macroalgal bloom occurred. In fact, a closer view on the short-term dynamics of algae biomass and *H. ulvae* fortnightly production enabled to detect substantial increases in the production during the bloom occurrence (Fig. 11.8c). This increase in production occurred less than 1 month after the increase of algal biomass (when algae biomass attained ± 300 g AFDW m⁻²; Fig. 11.8c). In fact, 1 month production (April production) corresponded to 75% of the whole 1993 annual production. Afterwards, with the first signs of the macroalgae decline and following crash, the production also decreased considerably (Fig. 11.8c).

During the post-mitigation period, density, biomass and annual production were considerably lower than in the pre-mitigation period, with the highest increases in 2000 and 2002, coincident with the appearance of greater biomasses of green macroalgae (Fig. 11.8b).

The P/\bar{B} ratio of *H. ulvae* was much higher in the eutrophic area (between 2.7 and 4.8), where the population is mainly composed of juveniles, than in the *Z. noltii* beds (between 1.3 and 3.0), where it presents a more structured population (Table 11.2).

11.4.5.2 *Cyathura carinata* (Isopoda)

In the *Zostera* beds, *C. carinata* population was more unstable, showing a great variability all over the time than in the eutrophic area (Fig. 11.9a). At the beginning of 1993, *C. carinata* was absent from the *Z. noltii* beds; however, its population sprouted until the end of 1994 and 1995, reaching to more than 700 ind m⁻² and with increases in biomass and annual production (Fig. 11.9a). Afterwards, the population decreased considerably by 1999, yet seemed to recover in the following 2000 and 2001, until a new decline by the end of 2002, being on the verge of disappearing from this area (Fig. 11.9a). The annual production of this species was much lower in the *Zostera* beds, reaching the highest value in 1994 (Fig. 11.9a). *C. carinata* presented a stable and consistent population in the eutrophic area all over the study period, exhibiting a characteristic annual pattern of variation. Density increased during summer, achieving maximum values in autumn/winter and then declined until late spring (Fig. 11.9b). Annual production was within similar values during the study period, with exception to a slightly higher peak in 1994 (Fig. 11.9b). During the

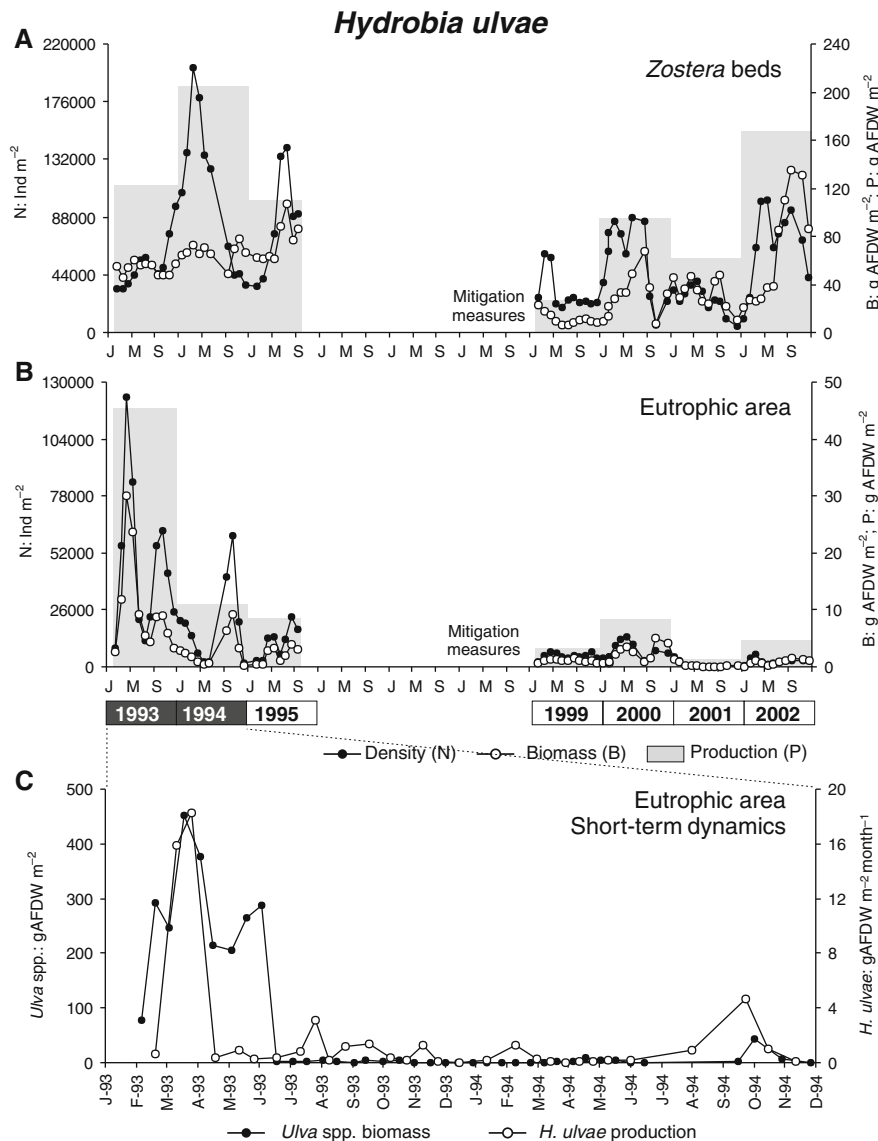


Fig. 11.8 Long-term variation of the monthly mean density, mean biomass and annual production of *H. ulvae* for (a) the *Zostera* beds and (b) eutrophic area, with indication of (c) the

detailed short-term dynamics of *Ulva* spp. biomass and *H. ulvae* fortnightly production during the macroalgal bloom (1993) and the following year (1994)

macroalgal bloom, it was not clear whether *C. carinata* production increased due to the presence of the algal cover. As soon as the algal biomass started to increase in initial March (after ± 80 g AFDW m⁻² of algae), *C. carinata* production in the following sampling was high (less than 15 days) (Fig. 11.9c). Yet, during the highest algal biomass, the production decreased considerably and maintained low in remaining 1993. In 1994, with almost no algae, higher production peaks were obtained in March, similar to the previous year, and in August and November (Fig. 11.9c). The P/\bar{B}

ratio of *C. carinata* was around similar values for both areas and slightly higher in the pre-mitigation period than in the post-mitigation one (Table 11.2).

11.4.5.3 *Scrobicularia plana* (Bivalvia)

S. plana density, biomass and annual production were higher in the eutrophic area than in the *Zostera* beds (Fig. 11.10a, b). In both sites, an important increase in all parameters was observed after the introduction of mitigation measures (Fig. 11.10a, b, Table 11.1).

Table 11.2 P/\bar{B} ratios (P/\bar{B} : y^{-1}) estimates for the main species for *Zostera* beds and eutrophic area, during the 10-year-study period

		1993	1994	1995	1999	2000	2001	2002
Zostera beds	<i>Hydrobia ulvae</i>	2.1	2.8	1.3	2.4	3.0	2.0	2.5
	<i>Scrobicularia plana</i>	0.9	1.8	1.3	0.7	0.8	0.6	0.8
	<i>Cyathura carinata</i>	4.6	2.9	3.0	2.4	2.0	2.2	2.2
	<i>Hediste diversicolor</i>	2.0	1.3	1.3	1.1	1.4	1.7	1.4
	<i>Capitella capitata</i>	6.9	9.2	8.9	5.8	7.0	8.1	5.5
	<i>Alkmaria romijni</i>	8.9	10.1	9.2	11.3	9.8	6.2	5.7
Eutrophic area	<i>H. ulvae</i>	4.8	4.5	2.4	3.1	2.9	4.1	2.7
	<i>C. carinata</i>	1.6	1.6	1.0	0.6	1.0	0.9	1.0
	<i>S. plana</i>	2.9	3.4	1.7	1.9	1.6	1.5	2.0
	<i>H. diversicolor</i>	2.0	1.8	1.5	1.1	1.4	1.9	1.9
	<i>C. capitata</i>	9.5	10.9	10.2	7.6	7.0	8.4	10.6
	<i>A. romijni</i>	13.4	11.8	10.3	9.0	8.5	10.6	9.5

For this bivalve, the biomass increase was more relevant than the density one (biomass increment: *Z. noltii* bed – 500%, eutrophic area – 250%) (Fig. 11.10a, b), which translated into lower P/\bar{B} ratios in the post-mitigation period (Table 11.2). Like *H. ulvae*, after the extreme flood in 2000/01, there was a decrease in density, biomass and annual production, which increased again in the following year (Fig. 11.10a, b). During the macroalgal bloom, there were no specific increases in *S. plana* biomass, since the fortnightly variation pattern maintained similar in both 1993 and 1994, showing only a slight tendency to decrease with time (Fig. 11.10c).

11.4.5.4 *Hediste diversicolor* (Polychaeta)

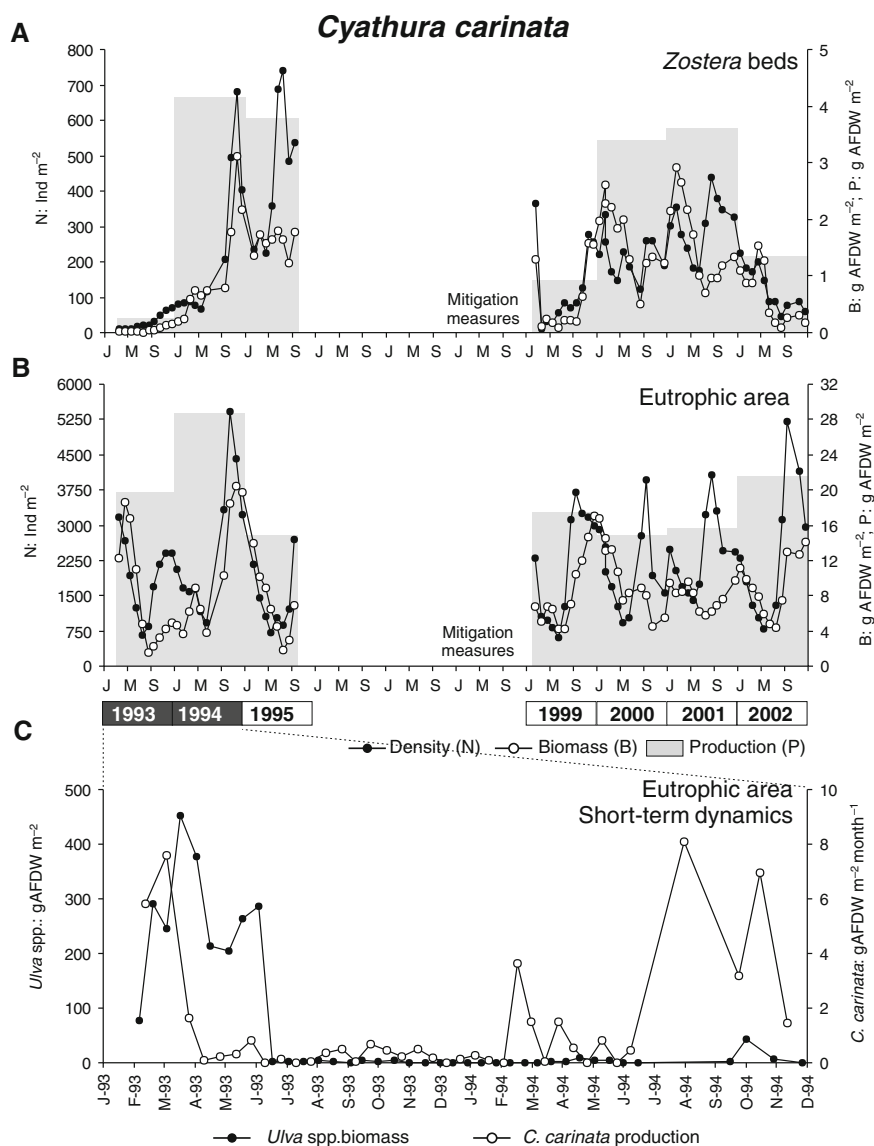
In the pre-mitigation period, *H. diversicolor* had slightly higher density, biomass and annual production in the eutrophic area than in the *Zostera* beds (Fig. 11.11a, b). After the mitigation measures, it presented a huge increment for both areas, especially for the *Zostera* beds. In fact, in this period, density, biomass and annual production were almost the double in the *Zostera* beds in comparison to the eutrophic area (Fig. 11.11a, b). During the macroalgal bloom, no specific increases in *H. diversicolor* biomass were observed; on the contrary, the biomass was nearly null

and only increased after the bloom in the remaining period (Fig. 11.11c). The P/\bar{B} ratios varied within similar values for the all study period, being slightly higher in the eutrophic area (Table 11.2).

11.4.5.5 *Alkmaria romijni* and *Capitella capitata* (Polychaeta)

A. romijni and *C. capitata*, together with *H. diversicolor* constitute the most abundant polychaete species in the Mondego estuary. Both had higher density, biomass and annual production in the eutrophic area (Figs. 11.12, 11.13), especially *A. romijni*, whose difference between areas is huge (Fig. 11.13). Over the study period, there was an important decline in all parameters in the post-mitigation period for both species and areas. This decline was quite high in 2001 for both species, after the extreme flood, with a slight increase for *C. capitata* in the following year 2002 (Fig. 11.12a, b). For *C. capitata*, the highest annual production was obtained in 1993 in the eutrophic area, during the macroalgal bloom (Fig. 11.12), while for *A. romijni*, the highest values were obtained in both 1993 and 1994, with similar annual production values (Fig. 11.13). A closer view on the macroalgal bloom short-term dynamics enables to detect *C. capitata* biomass increases about 1 month after the highest

Fig. 11.9 Long-term variation of the monthly mean density, mean biomass and annual production of *Cyathura carinata* for (a) the *Zostera* beds and (b) eutrophic area, with indication of (c) the detailed short-term dynamics of *Ulva* spp. biomass and *C. carinata* fortnightly production during the macroalgal bloom (1993) and the following year (1994)



increases of the algal biomass (Fig. 11.12c). A considerable decrease in production was observed about 1 month after the first signs of the macroalgae decline (Fig. 11.12c), suggesting that the algal biomass somewhat influenced the polychaete biomass. Regarding *A. romijni*, the pattern was not as clear, since there seems to be an increase following the algal biomass increases, yet the biomass did not decrease with the algal crash, and there were other biomass peaks following 1993 and in 1994 (Fig. 11.13c). These species had the highest P/\bar{B} ratio values of the all the six species studied, with most values varying among 7–11 (Table 11.2). In general, these were higher in the pre-mitigation period

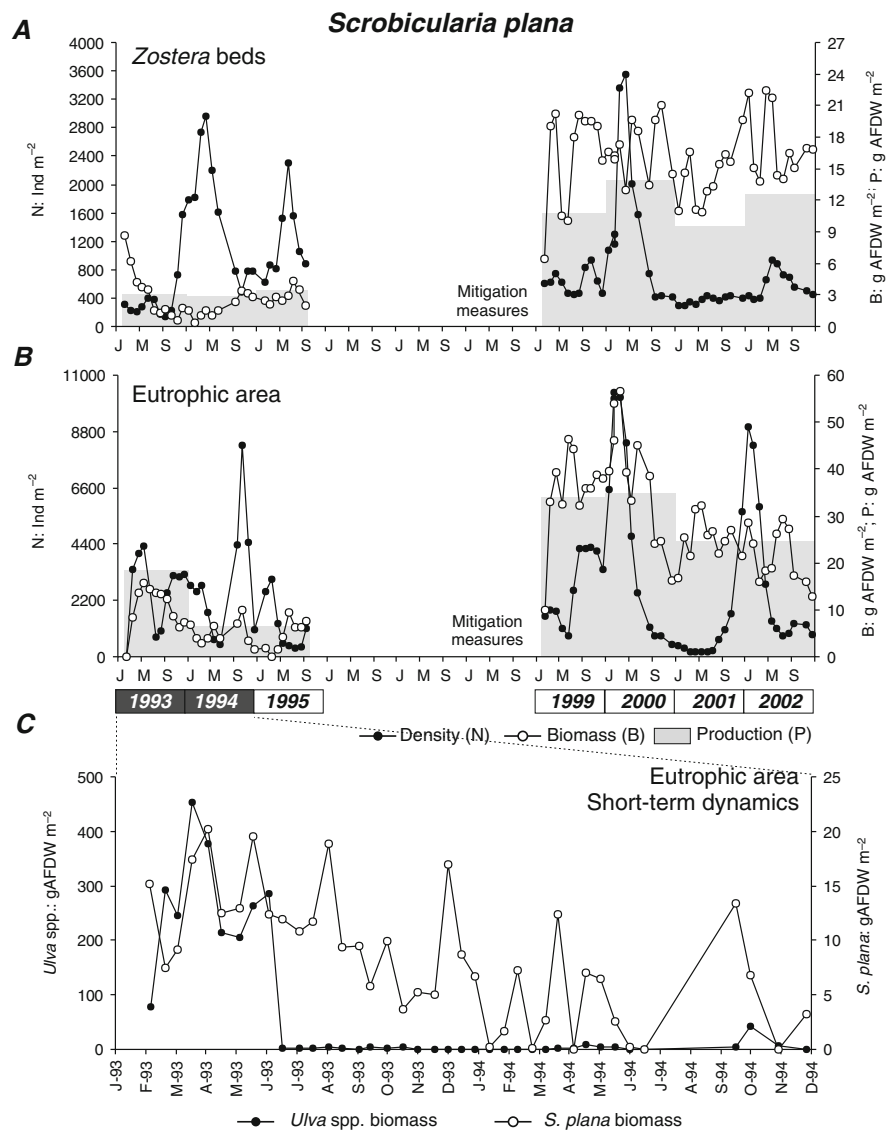
for both species and both areas and were slightly higher for *A. romijni* (Table 11.2).

11.5 Discussion

11.5.1 Eutrophication Effects

Eutrophication may be defined as “the process of changing the nutritional status of a given water body by increasing the nutrient resources” (Jørgensen and Richardson 1996). Due to this nutrient enrichment,

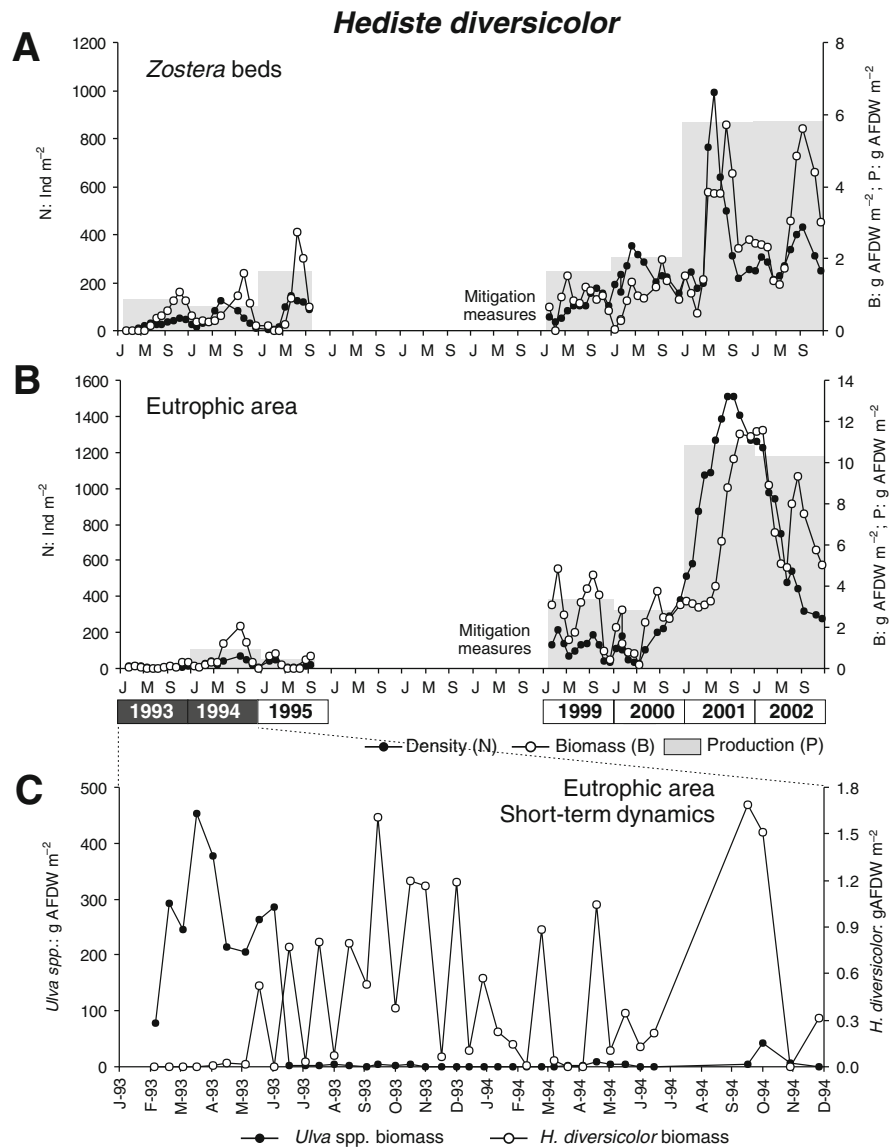
Fig. 11.10 Monthly long-term variation of the mean density, mean biomass and annual production of *Scrobicularia plana* for (a) the *Z. noltii* beds and (b) eutrophic area



most in the form of N and P, several impacts on the aquatic ecosystem may occur affecting primary and secondary production (Flindt et al. 1999, Dolbeth et al. 2003, Paerl 2006). The magnitude of the eutrophication effects depends on several aspects, from the hydrological characteristics of the system itself to climate (Cloern 2001, Paerl 2006, Rabalais et al. 2009). The Mondego estuary has a history of eutrophication, starting from the 1980s, and constitutes an important case study on this matter, since it has been monitored from 1993 to nowadays, and several hidden effects of eutrophication may only be revealed with long-term monitoring programmes. As discussed by Lillebø et al.

(2005), nitrogen, in the form of ammonia, appears to be the limiting nutrient in the Mondego estuary, similar to other estuarine systems, and an increasing pattern of this nutrient was indeed observed in the pre-mitigation period. An important short-term effect of this nutrient enrichment, associated with high water residence time, was the development of macroalgal blooms, mainly *Ulva* spp., as also occurred in several other coastal ecosystems suffering from eutrophication (Raffaelli 1998, Prins et al. 1999, Feuerpfeil et al. 2004, Fox et al. 2009, Pravoni et al. 2008). Another consequence in the system was the replacement of *Z. noltii* beds by *Ulva* spp., which in turn affected the entire trophic

Fig. 11.11 Monthly long-term variation of the mean density, mean biomass and annual production of *Hediste diversicolor* for (a) the *Z. noltii* beds and (b) the eutrophic area



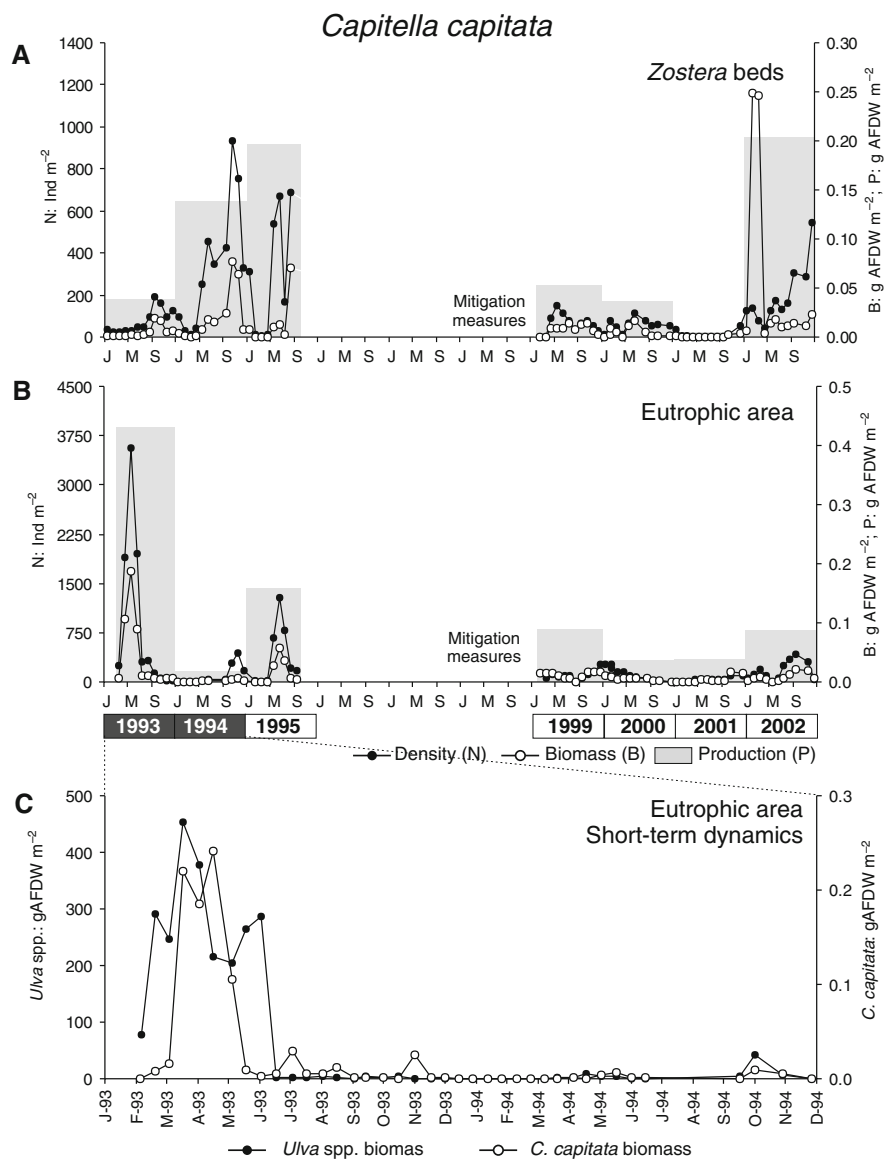
structure of the associated benthic community. These effects have been reported worldwide, mainly in highly populated coastal areas, where the nutrient over enrichment has led to excessive production of organic matter in the form of algal blooms (fast-growing opportunistic algae such as epiphytic green macroalgae and/or phytoplankton), conducting to the gradual replacement of perennial benthic macrophytes (see review from Flindt et al. 1999, Cloern 2001), hypoxia and anoxia scenarios (Breitburg et al. 2009), in turn affecting the aquatic heterotrophic organisms depending on that primary production and living in

the hostile generated environment (Oviatt et al. 1986, Raffaelli et al. 1998, Prins et al. 1999, Edgar and Barrett 2002, Dolbeth et al. 2003, Breitburg et al. 2009, Fox et al. 2009).

11.5.1.1 Macroalgal Bloom Dynamics in the Eutrophic Area

During the occurrence of the *Ulva* spp. bloom in the Mondego estuary in 1993, several species were able to take advantage of the extra food resources, habitat heterogeneity and protection against predation,

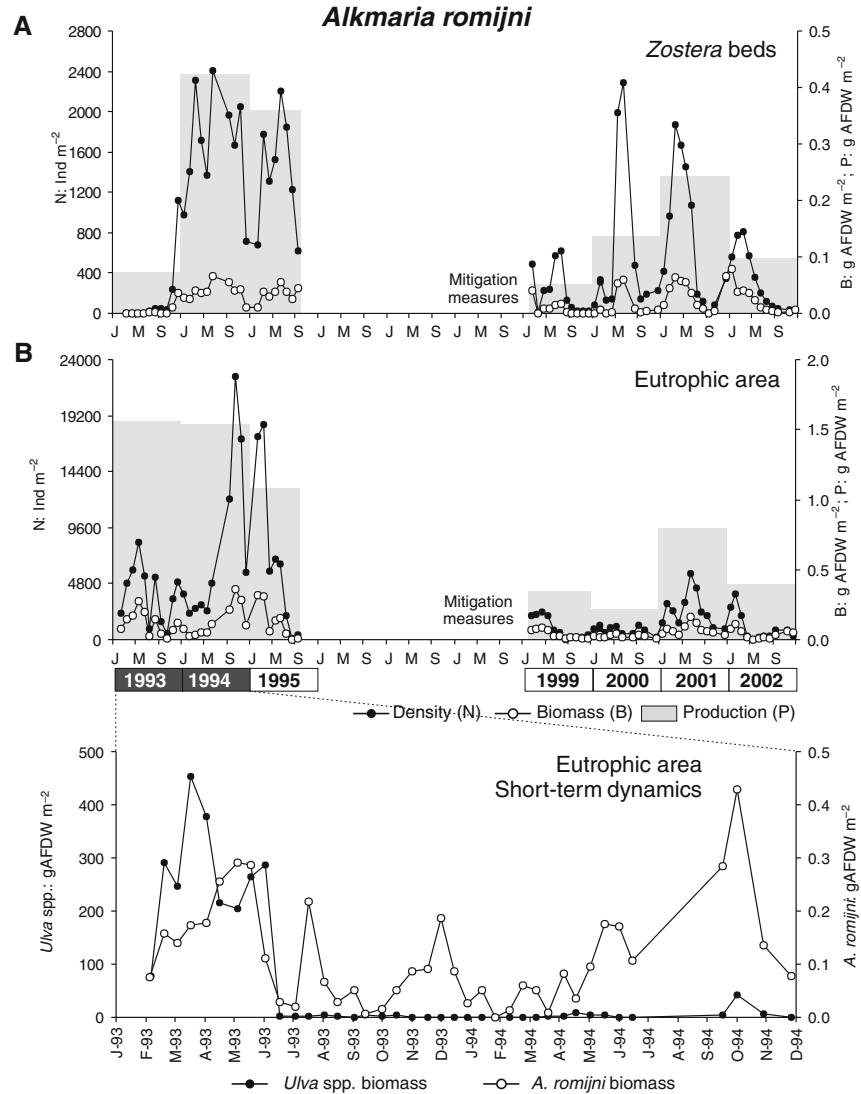
Fig. 11.12 Monthly long-term variation of the mean density, mean biomass and annual production of *Capitella capitata* for (a) the *Z. noltii* beds, and (b) eutrophic area



which the algae represent, as also discussed in other studies (Norkko et al. 2000, Dolbeth et al. 2003, 2007). In fact, mean annual density/biomass and production of the associated fauna from the eutrophic area, where the bloom took place, increased considerably. However, the response to the algal bloom is highly species specific (Dolbeth et al. 2003, Grilo et al. 2009), as also demonstrated experimentally (Cardoso et al. 2004). The link between changes in the primary and secondary production to the nutrient loadings in aquatic ecosystems is quite difficult, due to the interference of several other environmental

factors (Cloern 2001, Edgar and Barrett 2002, Nixon and Buckley 2002). Yet, the analysis of the six species responses to the algal blooms enabled to set some hypotheses regarding which species could benefit from the algae. The gastropod *H. ulvae* and the polychaete *C. capitata* were the species that seemed to be influenced by the presence of the algal cover, with important positive changes in the density, biomass and production during the bloom year and with fastest response to the presence of algal cover (less than 1 month). These species are highly opportunist and seemed to optimize their population

Fig. 11.13 Monthly long-term variation of the mean density, mean biomass and annual production of *Alkmaria romijni* for (a) the *Z. noltii* beds and (b) eutrophic area



growth at the eutrophic area, following the presence of the additional resource, as also observed by Fox et al. (2009). *H. ulvae* is a dominant species in the *Z. noltii* beds, where it presents a well-structured population (Cardoso et al. 2008b), with lower turnover ratios (P/\bar{B} ratios), reflecting higher stability in this area. So, the eutrophic area, during the presence of the algal cover, probably worked as an alternative habitat, similar to the conclusions of Norkko et al. (2000) and Cardoso et al. (2004). *C. capitata* was generally more abundant in the bare sand flat eutrophic area. Yet, during the bloom, the increased habitat complexity and available organic matter probably worked as a driver for the observed increased biomass

(reflected into higher annual biomass and production), emphasizing its *r*-strategist behaviour. Nevertheless, as also stated by Norkko et al. (2000), the effects of drift algal mats on infauna depend on the spatial and temporal extent of algal coverage. Once the algae grew beyond a certain threshold, the system collapsed into an algal crash, and both species suffered a strong decline in density, biomass and production, presenting low values in the following pre-mitigation period. This tendency was also observed for the community patterns, with declines in the community mean annual density, biomass and production in the following 1994 and 1995. These decreases were associated to the negative effects of macroalgal blooms and associated crash,

which have been well documented. Among described effects are changes in sediment chemistry, nutrient release, rates of oxygen consumption and accumulation of metal sulfides and H₂S in sediment pore waters (Jorgensen and Richardson 1996, Cloern 2001), which in turn cause the invertebrate assemblages to collapse, as also reported elsewhere (Raffaelli et al. 1998), and ultimately affect the higher trophic levels depending on that invertebrate production (Dolbeth et al. 2008).

For the other studied species, the parallelism between algal growth and fauna-positive responses was not as clear or did not seem to occur at all. For instance, *S. plana*, *C. carinata* and *A. rominji* had increases in density and biomass during the bloom, yet the highest peaks in density and biomass were observed afterwards. So, the algae did not seem to have worked as an additional resource for those species populations development. *H. diversicolor* did not seem to be influenced by the presence of the algal cover at all. On the contrary, its biomass only increased after the bloom, although other studies have referred this species as being tolerant to hypoxia, anoxia and commonly found under algal mats (Pearson and Rosenberg 1978, Norkko and Bonsdorff 1996). In fact, during the bloom alone, *H. diversicolor* presented the lowest biomass values of the pre-mitigation period, which increased considerably after the bloom, suggesting that the algae did not act as a driver for the population development. For the Mondego estuary in particular, *H. diversicolor* was associated to a post-algae and recover phase (discussed below), rather than as an indicator of the organic matter enrichment, as also discussed by Cardoso et al. (2007).

The percentage of herbivores abundance and production at the eutrophic area was higher in the bloom year, in expected since the algae are a readily available food resource. However, it is worth to mention that most of this percentage is due to *H. ulvae* increase, which behaves both as herbivore and surface-deposit feeder. Also, a higher percentage of surface-deposit feeders was observed in 1993, and afterwards in the following pre-mitigation period, subsurface-deposit feeders dominate the community production. These differences are probably due to the fact that surface-deposit feeders prefer to feed on newly deposited organic matter, probably with highest amounts in 1993.

11.5.2 Differences Between Sites

The *Z. noltii* beds had in general higher species richness, density, biomass and production than the eutrophic area during the whole study period. These differences are in line in several studies elsewhere (e.g. Sprung 1994, Heck et al. 1995, Fox et al. 2009, Pranovi et al. 2008), comparing sediments covered with rooted macrophytes with bare and eutrophic sandflats, such as the eutrophic area, which has no seagrass cover for more than 15 years and has lower organic matter content. Nevertheless, the seagrass area presented lower evenness, mainly due to the dominance of *H. ulvae*, a common *taxon* in several estuarine and coastal areas (Norkko et al. 2000). Regarding the feeding guild composition, the dominance of this species reflected into the almost identical and dominant percentages of herbivores and detritivores abundance and production in seagrass area, especially in the pre-mitigation period; while in the eutrophic area the community was dominated essentially by detritivores. When excluding *H. ulvae* from the analysis, most of the community abundance and production was dominated by the detritivores alone (with few exceptions), which is a common feature in estuarine systems (Sprung 1994). This suggests that a great part of the energy/biomass enters the system via the detritus food chain, with only a small contribution from the grazing generated food chains. Differences in the detritivore composition were also registered among areas and throughout the study period (discussed below). When *H. ulvae* is accounted, the deposit feeders composition was more heterogenic in the eutrophic area; when not accounted, the pattern was similar in the two areas.

11.5.3 Pre-mitigation versus Post-mitigation Periods

A general conclusion that may be taken is that the macroalgal blooms temporarily increased production of specific *taxa*, contributing the overall increase in community production. Yet, this temporary gain is quite short lived, since the long-term effects of eutrophication and associated algal blooms necessarily imply a loss of the faunal production associated to the seagrass, conducting to an overall decrease in whole

estuarine production and integrity, as also discussed by Dolbeth et al. (2003, 2007). In fact, the competition with the fast-growing algae has been considered responsible for the disappearing of the macrophyte beds, as also referred to occur in several other coastal areas worldwide (Flindt et al. 1999, Cloern 2001). So, if no mitigation measures would be taken in the Mondego estuary, the system would most probably collapse.

When comparing both areas, before and after the implementation of mitigation measures, several differences are noticeable in the structure and function of the benthic communities. While the nitrogen loading into the estuary was effectively reduced (lowering N/P ratio) since 1999, the benthic communities, both flora and fauna, took longer time to recover. The lowest biomass and extent of *Z. noltii* was observed in 1999, concomitant with the lowest values of species richness, density, biomass and production obtained for the *Z. noltii* area. Afterwards, the benthic community seemed to recover, until the occurrence of an extreme major flood, when the benthic community slowed down the recovery process, to re-start the recovery again after the flood. The eutrophic area, as a simple bare sandy/muddy habitat, appeared less variable over time in species richness, density, biomass and production. Similar to the *Z. noltii* area, during the extreme flood, the benthic community slowed down the recovery process. This occurrence highlights climate variability as an additional source of stress, whose frequency is becoming higher, and may intensify the course of eutrophication in estuarine and coastal waters, as discussed by Rabalais et al. (2009). In fact, ecosystems are being subjected more frequently to multiple stressors, which may act synergistically to lower the natural resistance and resilience of their inhabitants to disturbance, as observed for the Mondego estuary (Cardoso et al. 2008a, b, Dolbeth et al. 2007). Regarding the trophic organization, most of the community was dominated by deposit feeders, emphasizing the role of the detritus in the food chain. Yet, some differences were observed comparing pre-and post-mitigation periods for both areas. For the eutrophic area, generally, higher heterogeneity of the feeding guilds in the pre-mitigation period was observed, while in the post-mitigation period the community was essentially dominated by detritivores. These differences are probably associated to the higher diversity of food sources in the pre-mitigation period,

while in the post-mitigation one the energy sources are probably more stable. The *Z. noltii* bed supported, in general, higher percentage of carnivores, herbivores and omnivores than the eutrophic area, which makes it functionally richer. For this area, there was an increase in omnivores in the post-mitigation period, especially during the flood year. This tendency was also observed for the eutrophic area, though in lower percentages. An increase in omnivores dominance may be regarded as an advantage, especially when subjected to a stress source, since omnivores have flexible generalist diets, being able to change their function in the benthic food web, according the available resource (Fox et al. 2009).

Compelling with the changes in the community in the latest years of the study was the evidence of succession from *r*-strategists towards *K*-strategists species in both study areas. *S. plana*, typically a slow growing species, increased considerably in the estuary after the mitigation measures, especially in the eutrophic area, probably responsible by the increase of the estuarine mean biomass in the estuary. *H. diversicolor* showed a similar development as *S. plana*, increasing considerably in estuary after the introduction of the mitigation measures, together with an overall decrease of the opportunist polychaetes *A. romijni* and *C. capitata*, considered as indicators of nutrient enrichment and pollution of estuarine systems (Cardoso et al. 2007). As discussed by Cardoso et al. (2007), in the Mondego estuary, *H. diversicolor* is not considered as an indicator of organic enrichment, but instead represents a measure of the ecosystem trajectory into a more nutrient-controlled system.

11.5.4 Evaluation of the Ecosystem Recovery

The measures undertaken in Mondego estuary south arm in the late 1990s included a set of restoration measures, such as the controlled use of fertilizers in the agriculture fields located upstream, the improvement of water circulation by enlarging the connection between the two arms, a better management of sluice openings, which led to strong modifications of the physico-chemical features of the estuary (lower turbidity, lower suspended organic matter and lower re-mineralization). These seemed to have some

success, reflected in the dynamics of the primary producers and macrofauna. Nutrient loading was significantly reduced and no further macroalgal blooms were recorded, as also discussed by Lillebø et al. (2005, 2007). In addition, *Z. noltii* beds are gradually recovering, both in biomass and extent (4.7 ha in 2006), starting to re-colonize the upstream areas (personal observation). The biomass and production of the macrozoobenthic community also increased, together with a slight increase in biodiversity, as discussed in the previous section. Yet, although the cover extent of the *Z. noltii* is the same at the beginning and end of the study period (i.e. 1993 and 2002), biodiversity levels have still not recovered to the observed in 1993, which might have compromised the macrobenthic community after the extreme floods in 2000/01, as discussed by Dolbeth et al. (2007).

At the present time, the *Z. noltii* beds are recovering, but the eutrophic area restoration is still far away from the original habitat similar to the *Z. noltii* beds. Quite recently, some very small patches of *Z. noltii* appeared in the eutrophic area (personal observation), but the plant recovery is not fully taking place. Potentially this area may be reaching a new steady state community for bare sand/mud, which is consistent with the increase of the *K*-strategists species (e.g. *S. plana*) in this area.

References

- Breitburg DL, Davias DW, Hondorp LA, Diaz RJ (2009) Hypoxia, nitrogen, and fisheries: integrating effects across local and global landscapes. *Annu Rev Mar Sci* 1:329–349
- Brey T (2001) Population dynamics in benthic invertebrates. A virtual handbook. Version 01.2. <http://www.awi-bremerhaven.de/Benthic/Ecosystem/FoodWeb/Handbook/main.html>, Alfred Wegener Institute for Polar and Marine Research, Germany
- Cardoso PG, Bankovic M, Raffaelli D, Pardal MA (2007) Polychaete assemblages as indicators of habitat recovery in a temperate estuary under eutrophication. *Estuar Coast Shelf Sci* 71:301–308
- Cardoso PG, Pardal MA, Lillebø AI, Ferreira SM, Raffaelli D, Marques JC (2004) Dynamic changes in seagrass assemblages under eutrophication and implications for recovery. *J Exp Mar Biol Ecol* 302:233–248
- Cardoso PG, Raffaelli D, Lillebø AI, Verdelhos T, Pardal MA (2008a) The impact of extreme flooding events and anthropogenic stressors on the macrobenthic communities' dynamics. *Estuar Coast Shelf Sci* 76:553–565
- Cardoso PG, Raffaelli D, Pardal MA (2008b) The impact of extreme weather events on the seagrass *Zostera noltii* and related *Hydrobia ulvae* population. *Mar Pollut Bull* 56:483–492
- Cloern JE (2001) Our evolving conceptual model of coastal eutrophication problem. *Mar Ecol Prog Ser* 210: 223–253
- Dolbeth M, Cardoso PG, Ferreira SM, Verdelhos T, Raffaelli D, Pardal MA (2007) Anthropogenic and natural disturbance effects on a macrobenthic estuarine community over a 10-year period. *Mar Pollut Bull* 54:576–585
- Dolbeth M, Lillebø AI, Cardoso PG, Ferreira SM, Pardal MA (2005) Annual production of estuarine fauna in different environmental conditions: an evaluation of the estimation methods. *J Exp Mar Biol Ecol* 326:115–127
- Dolbeth M, Martinho F, Viegas I, Cabral H, Pardal MA (2008) Estuarine production of resident and nursery fish species: conditioning by drought events? *Estuar Coast Shelf Sci* 78:51–60
- Dolbeth M, Pardal MA, Lillebø AI, Azeiteiro UM, Marques JC (2003) Short and long-term effects of eutrophication on the secondary production of an intertidal macrobenthic community. *Mar Biol* 143:1229–1238
- Edgar GJ, Barrett NS (2002) Benthic macrofauna in Tasmanian estuaries: scales of distribution and relationships with environmental variables. *J Exp Mar Biol Ecol* 270:1–24
- Ferreira SM, Brandão A, Baeta A, Neto JM, Lillebø AI, Marques JC, Jensen KT, Pardal MA (2007) Effects of restoration management on the estuarine isopod *Cyathura carinata*: mediation by trematodes and habitat change. *Mar Biol* 151:109–118
- Feuerpfeil P, Rieling T, Estrum-Youseff SR, Dehmlow J, Papenfuß T, Schoor A, Schiewer U, Schubert H (2004) Carbon budget and pelagic community compositions at two coastal areas that differ in their degree of eutrophication, in the Southern Baltic Sea. *Estuar Coastal Shelf Sci* 61: 89–100
- Flindt MR, Pardal MA, Lillebø AI, Martins I, Marques JC (1999) Nutrient cycling and plant dynamics in estuaries: a brief review. *Acta Oecol* 20(4):237–248
- Fox SE, Teichberg M, Olsen YS, Heffner L, Valiela I (2009) Restructuring of benthic communities in eutrophic estuaries: lower abundance of prey leads to trophic shifts from omnivory to grazing. *Mar Ecol Prog Ser* 380:43–57
- Gaston GR, Brown SS, Rakocinski CF, Heard RW, Summers JK (1995) Trophic structure of macrobenthic communities in northern Gulf of Mexico estuaries. *Gulf Res Rep* 9(2):111–116
- Gaston GR, Nasci JC (1988) Trophic structure of macrobenthic communities in the Calcasieu estuary, Louisiana. *Estuaries* 11(3):201–211
- Grilo TF, Cardoso PG, Dolbeth M, Pardal MA (2009) Long-term changes in amphipod population dynamics in a temperate estuary following ecosystem restoration. *Hydrobiologia* 630:91–104
- Heck KL, Able KW, Roman CT, Fahay MP (1995) Composition, abundance, biomass, and production of macrofauna in a new England estuary – comparisons among eelgrass beds and other nursery habitats. *Estuaries* 18(2): 379–389
- Jørgensen BB, Richardson K (1996) Eutrophication in coastal marine ecosystems. American Geophysical Union, Washington, DC

- Kennish MJ (2002) Environmental threats and environmental futures of estuaries. *Environ Conserv* 29: 78–107
- Leitão R, Martinho F, Cabral HN, Neto JM, Jorge I, Pardal MA (2007) The fish assemblage of the Mondego estuary: composition, structure and trends over the past two decades. *Hydrobiologia* 587:269–279
- Lleston S, Lillebø AI, Pardal MA (2008) The response of primary producer assemblages to mitigation measures to reduce eutrophication in a temperate estuary. *Estuar Coastal Shelf Sci* 77:688–696
- Lillebø AI, Neto JM, Martins I, Verdelhos T, Lleston S, Cardoso PG, Ferreira SM, Marques JC, Pardal MA (2005) Management of a shallow temperate estuary to control eutrophication: the effect of hydrodynamics on the system's nutrient loading. *Estuar Coastal Shelf Sci* 65: 697–707
- Lillebø AI, Teixeira H, Pardal MA, Marques JC (2007) Applying quality status criteria to a temperate estuary before and after the mitigation measures to reduce eutrophication symptoms. *Estuar Coastal Shelf Sci* 72:177–187
- Limnologisk Metodik L (1992) Ferskvandsbiologisk Laboratorium. Københavns Universitet (ed), Akademisk Forlag, København
- Lopes RJ, Pardal MA, Múrias T, Cabral JA, Marques JC (2006) Influence of macroalgal mats on abundance and distribution of dunlin *Calidris alpina* in estuaries: a long-term approach. *Mar Ecol Prog Ser* 323:11–20
- Mancinelli G, Fazi S, Rossi L (1998) Sediment structural properties mediating dominant feeding types patterns in softbottom macrobenthos of the Northern Adriatic Sea. *Hydrobiologia* 367:211–222
- Nixon S, Buckley BA (2002) “A strikingly rich zone” – nutrient enrichment and secondary production in coastal marine ecosystems. *Estuaries* 25:782–796
- Norkko A, Bonsdorff E (1996) Rapid zoobenthic community responses to accumulations of drifting algae. *Mar Ecol Prog Ser* 131:143–157
- Norkko J, Bonsdorff E, Norkko A (2000) Drifting algal mats as an alternative habitat for benthic invertebrates: species responses to a transient resource. *J Exp Mar Bio Ecol* 248:79–104
- Oug E, Næsm K, Rygg B (1998) Relationship between soft bottom Macrofauna and polycyclic aromatic hydrocarbons (PAH) from smelter discharge in Norwegian fjords and coastal waters. *Mar Ecol Prog Ser* 173:39–52
- Oviatt CA, Keller AA, Sampou PA, Beatty LL (1986) Patterns of productivity during eutrophication: a mesocosm experiment. *Mar Ecol Prog Ser* 28:69–80
- Paerl HW (2006) Assessing and managing nutrient-enhanced eutrophication in estuarine and coastal waters: interactive effects of human and climatic perturbations. *Ecol Eng* 26:40–54
- Pearson TH, Rosenberg R (1978) Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanogr Mar Biol Ecol* 20:1–41
- Pranovi F, Da Ponte F, Torricelli P (2008) Historical changes in the structure and functioning of the benthic community in the lagoon of Venice. *Estuar Coastal Shelf Sci* 76:753–764
- Prins TC, Escaravage V, Wetssteyn LPMJ, Peeters JCH, Smaal AC (1999) Effects of different N- and P-loading on primary and secondary production in an experimental marine ecosystem. *Aquat Ecol* 33:65–81
- Rabalais NN, Turner RE, Diaz RJ, Justic D (2009) Global change and eutrophication of coastal waters. *ICES J Mar Sci* 28:1–10
- Raffaelli DG, Raven JA, Poole LJ (1998) Ecological impact of green macroalgal blooms. *Oceanogr Mar Biol Annu Rev* 36:97–125
- Ribeiro JLGS (2002) Conflicts in the Mondego estuary: sustainability of natural resources and of traditional land use. Littoral 2002, The changing coast. EUROCOAST/EUCC, Porto – Portugal Ed. EUROCOAST – Portugal, ISBN 972–8558–09–0
- Sprung M (1994) Macrobenthic Secondary Production en the Intertidal Zone of Ria Formosa – a lagoon in Southern Portugal. *Estuar Coastal Shelf Sci* 38:539–558
- Strickland JDM, Parsons TR (1972) A practical handbook of seawater analysis, 2nd edn. Fisheries Research Board of Canada, Ottawa
- Teixeira H, Salas F, Borja A, Neto JM, Marques JC (2008) A benthic perspective in assessing the ecological status of estuaries: the case of the Mondego estuary (Portugal). *Ecol Indic* 8:404–416