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Impact of mercury contamination on the population dynamics of *Peringia ulvae* (Gastropoda): Implications on metal transfer through the trophic web

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ABSTRACT

The effects of mercury contamination on the population structure and dynamics of the gastropod *Peringia ulvae* (also known as *Hydrobia ulvae*) and its impact on the trophic web were assessed along a mercury gradient in Ria de Aveiro (Portugal). The gastropod was revealed to be a tolerant species to the contaminant, since the highest densities, biomasses and growth productivity values were recorded at the intermediate contaminated area followed by the most contaminated one and finally the least contaminated area. *P. ulvae* was however negatively affected by mercury in terms of growth and life span. So, in the most contaminated area the population was characterised mainly by the presence of juveniles and young individuals. The intermediate contaminated area showed a greater equilibrium in terms of groups' proportion, being the adults the dominant set. The least contaminated area presented intermediate values. *P. ulvae* life spans were shortest in the most contaminated area (7–8 mo), followed by the least contaminated area (10–11 mo) and finally, the intermediate one (11–14 mo).

P. ulvae revealed to be an important vehicle of mercury transfer from sediments to the trophic web, incorporating approximately 15 g of Hg, annually, in the inner area of the Laranjo Bay (0.6 Km^2). Therefore, despite *P. ulvae* being revealed to be not a good bio-indicator of mercury contamination, since it did not suffer profound modifications in its structure and functioning, it is a crucial element in the mercury biomagnification processes throughout the food web.

1. Introduction

Mercury (Hg) is widely distributed and has been listed as a high priority pollutant by many international agencies because of its persistence in environment and high toxicity to organisms. The fate of Hg distributed in sediments, has received great attention since they can act both as a major sink for or a potential source of Hg in aquatic environments (Yu et al., 2012 and references therein). In these systems ingestion of sediments may account for up to 100% of the total amount of metal accumulated in some deposit-feeding invertebrates (Chong and Wang, 2000 and references therein).

In contaminated environments, mercury (in the organic and inorganic forms) may be transferred from the abiotic to the biotic compartment (e.g. Baeyers et al., 2003; De Marco et al., 2006; Donkor et al., 2006). Once in biota other processes may occur, and the bioaccumulation and biomagnification throughout the food web (macroalgae, shellfish, fish) is a matter of great concern. It is recognised that anthropogenic sources of mercury are responsible for the highest environmental impacts (EPA, 1997), having deleterious effects on biota, including humans (Pan and Wang, 2011) and ecosystem functions (Boening, 2000).

Benthic invertebrates have become well established as useful bio-indicators of ecological quality in coastal systems. Numerous studies have demonstrated that benthic macrofauna respond in a predictable and relatively rapid manner to a variety of natural and anthropogenic stress (Calabretta and Oviatt, 2008 and references therein). Their sedentary condition and continuous exposure to stress conditions make them useful as bio-indicators (Calabretta and Oviatt, 2008; Dauvin, 2008).

The gastropod *Peringia ulvae* (Pennant 1777) also known as *Hydrobia ulvae* is a deposit-feeder that lives in intertidal mudflats and muddy/sandy sediments. It is also found in a wide variety of intertidal substrata, including saltmarshes or macroalgae assemblages when present at sediment or rocky surface (Sola, 1996;

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Cardoso et al., 2008). Besides that, *P. ulvae*, can also be a periphyton grazer (Philippart, 1995). So, its strong connection with the sediment can represent a risk of contamination for the higher trophic levels which feed on it and also may represent a threat for the functioning and success of the own species. *P ulvae* is a key-species in several temperate estuaries, due to the great abundances and its role in the trophic web, being the link between primary producers and secondary consumers (Lillebø et al., 1999; Cardoso et al., 2008).

In the literature, there are some studies that assess the metals concentrations in water, sediment and fauna (Mucha et al., 2005; Dauvin, 2008) but very few works have been developed to evaluate the impact of metal contamination on the structure and productivity of coastal fauna (Weis et al., 2004; Rumisha et al., 2012). The scarce bibliography generally debates the effects of metal pollution at the community level and not at the population level. Assuming the relevance of this gastropod species as primary consumer, the main goal of this paper is to evaluate the impact of mercury contamination on the population dynamics of *Peringia ulvae*, using structural indicators such as density and biomass but also using indicators of ecosystem functioning like reproduction and secondary production. In addition, is assessed the potential of the species as vehicle of mercury transfer through the trophic web.

2. Materials and methods

2.1. Study area

The Ria de Aveiro is a shallow costal lagoon, adjacent to the Atlantic Ocean, located in the northwest coast of Portugal (40′38′N, 8′44′W). The system is 45 km long (NNE–SSW direction) with a maximum width of 10 km covering an area of approximately 83 km^2 of wetland in high tide (spring tide) and 66 km^2 in low tide. Ria de Aveiro has a complex topography, with four main channels joining and running to the mouth, several branches, islands and intertidal mud and sandflats (Fig. 1). The Ria de Aveiro has an inner area, called Laranjo Bay, which is considered the most contaminated site in the system. For approximately 5 decades (1950's–1990's), this area has received discharges loaded with mercury from a chlor-alkali plant located upstream at the Estarreja industrial complex. Despite the end of the effluents release, the sediments of the Ria present high mercury concentrations, creating a spatial gradient of contamination (Coelho et al., 2005). Three sampling stations were selected in the Laranjo Bay along a transect defined by the distance from the mercury point source: station 1 was considered to be closest to the mercury point source in the lagoon, and the others stations are progressively further way, respectively 600 m (station 2) and 3000 m (station 3) (Fig. 1).

2.2. Field and sampling procedures

The *P. ulvae* population was monitored monthly, during low tide from September 2010 to September 2011. In each site, 8–10 replicates were collected randomly, with a 141 cm^2 section core to a depth of approximately 20 cm. The biological samples were all washed *in situ* in a 500 μm mesh sieve bag and then placed into plastic bottles. All physicochemical parameters were also measured *in situ* (temperature, dissolved oxygen, pH and salinity) and water from the intertidal water pools for determination of chlorophyll *a*,

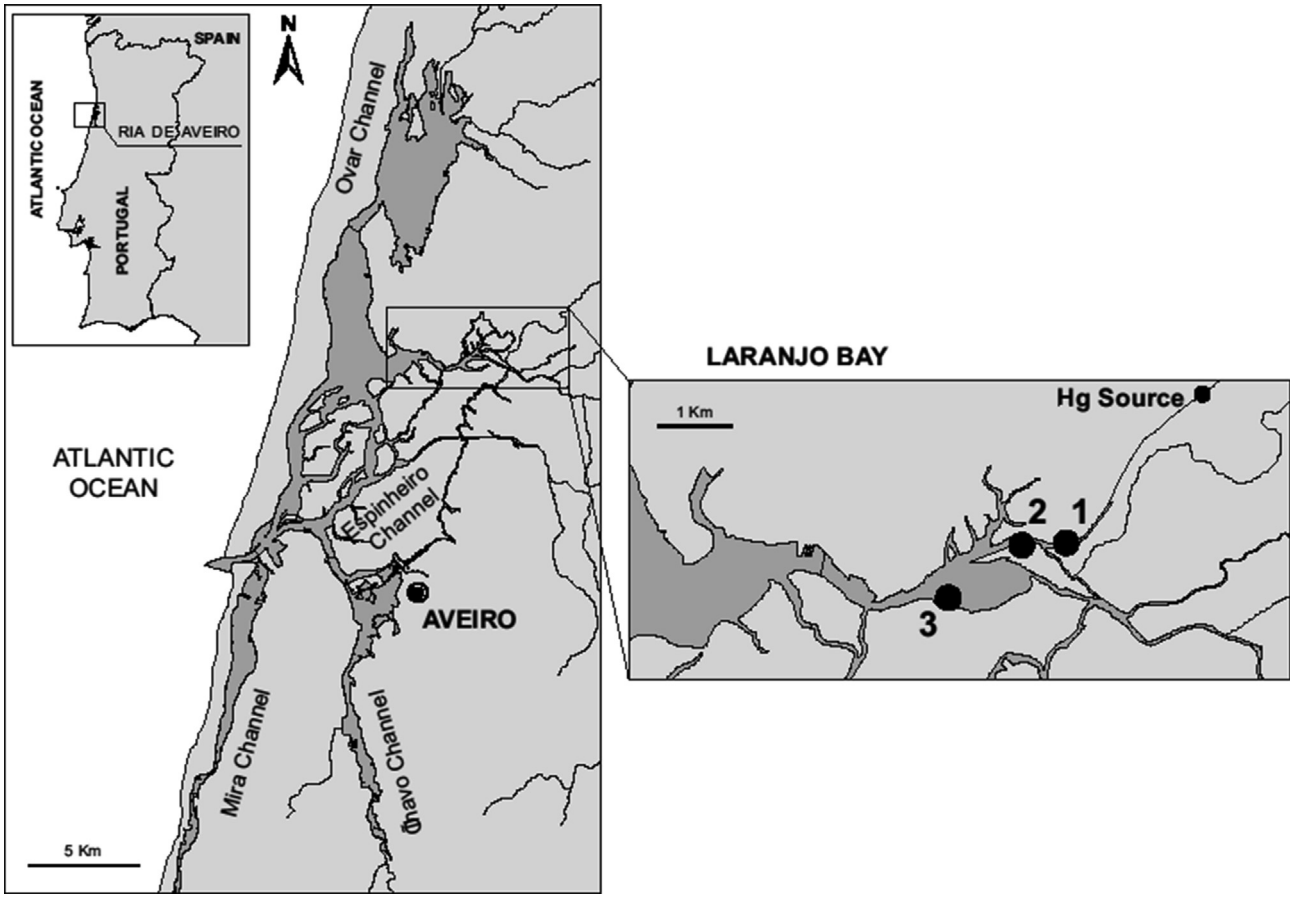

Fig. 1. Location of the sampling areas in the Ria de Aveiro.

total dissolved mercury and total mercury in suspended particulate matter (SPM) was collected.

Sediments from each site (taken at the first 5–7 cm) were also collected for organic matter, total mercury content quantification and granulometry. They were homogenized, a portion was freeze-dried for mercury analysis and the remainder was oven dried at 60 °C and incinerated at 450 °C for 8 h for organic matter quantification. The grain size was analysed and classified according to the following nomenclature: gravel: >2 mm; 2.0 mm > coarse sand >0.5 mm; 0.5 mm > medium sand >0.250 mm; 0.250 mm > fine sand >0.063 mm; 0.063 mm > silt > 0.0039 mm; clay <0.0039 mm (adapted from Gray and Elliott, 2009).

The biological samples were fixed with 4% buffered formalin. Later, the animals were sorted and the *P. ulvae* individuals were identified and kept in 80% alcohol. They were counted and their total shell length (TSL) and maximum width (MW) were measured. A conversion equation was used, based on previous works, (TSL = 2.2289*MW − 0.3886, $n = 339$, $r = 0.97$; Lillebø et al., 1999) and the individuals were classified in different size classes (Cardoso et al., 2005; Grilo et al., 2012). Organisms for mercury determination were collected, separated by age groups, frozen and posteriorly freeze-dried for later analysis.

Length–weight relationships were determined for production estimates. Preliminary ANOVA of length × AFDW relationships indicated no significant seasonal differences, and an overall regression equation was used ($AFDW = 0.0564TSL^{2.2381}$, $n = 191$, $r = 0.98$; Lillebø et al., 1999). Weight-to-weight regression (Dolbeth et al., 2005) was used to determine DW.

Cohorts were tracked using the FAO ICLARM Stock Assessment Tool package (FISAT II software) (Gayanilo and Pauly, 1997) and size frequency distributions were determined over successive sample dates. Following the cohorts since their recruitment until their disappearance (death) it is possible to estimate the life span for each cohort.

2.3. Mercury quantifications

2.3.1. Sediments and organisms

Total mercury content of the sediments and organisms was analysed by atomic absorption spectrometry with gold amalgamation, using a Leco AMA-254 (Advanced Mercury Analyser) (Costley et al., 2000). The analytical quality control of the total mercury determination was performed using Certified Reference Materials (CRMs), the TORT-2 lobster hepatopancreas for the organisms while for the sediments was used the MESS-3 (for least contaminated sediments) and PACS-2 (for high contaminated sediments). The results were corrected according to the daily recovery percentage of the CRM analyses. The values obtained for the whole CRM analysis ranged from 96.8 to 103.3% (at 0.05 significance level) for the sediments and 113% for the organisms. Analyses of CRMs were always performed in triplicate and coefficient of variation was lower than 10%. Additionally, total metal contents in sediments were compared with two sediment quality guidelines (SQG), the ERL (effects range-low) and the ERM (effects range median), which define the concentration ranges that are rarely, occasionally, or frequently linked to adverse biological effects (Long et al., 1995).

2.3.2. Water

Water samples were filtered with 0.45 μm pore size Millipore filters and acidified with concentrated HNO3 "mercury free" to pH < 2 and maintained in a room at 4 °C. Total mercury analysis in water samples was performed by cold vapour atomic fluorescence spectroscopy (CV-AFS), on a PSA cold vapour generator, model 10.003, associated with a Merlin PSA detector, model 10.023, and using SnCl2 as reducing agent. This analytical methodology is highly sensitive, allowing the measurement of 1 ng L^{-1} of mercury (Mucci et al., 1995).

2.3.3. Suspended particulate matter (SPM)

Filters (from the previous process of water filtration) were oven-dried at 60 °C and weighted again to determine the amount of SPM. Then, they were digested with HNO_3 4 mol L^{-1} for determination of the total mercury concentration in the SPM fraction. After HNO_3 digestion, the particulate total mercury was analysed by CV-AFS (Pato et al., 2008).

2.4. Data analysis

Some data (i.e. physicochemical variables and density/biomass of *P. ulvae*) were presented graphically as moving averages (Oct-10 to Sep-11) to minimize the effects of high-frequency variability and visually simplify the data series. Mercury concentrations were presented each two months from Sep-10 to Sep-11.

One-way ANOVAs were carried out for the environmental parameters in order to detect differences between sampling stations. In addition, statistical differences in mercury concentrations in *P. ulvae* as a function of sampling stations and age class were tested with two-way analysis of variance (ANOVA). All data were previously checked for normality using the Kolmogorov–Smirnov test and for homogeneity of variances using the Levene's test (Zar, 1996). Data not meeting these criteria were transformed appropriately (Zar, 1996) and checked again for normality and homocedasticity. Whenever data did not meet those criteria were applied non-parametric tests.

Chi-square tests were carried out for the *P. ulvae* population distributions in order to detect differences between sampling stations.

2.5. Secondary production

Secondary production (P) was estimated based on cohort recognition, as described in Dauvin (1986). P corresponds to the biomass assimilated in a period of time by a constant number of individuals. The growth production for each cohort was evaluated for a period between two sampling times (t and $t + 1$) using the following expression:

$$P = [(N_t + N_{t+1})/2](\overline{W}_{n+1} - \overline{W}_t) \text{ for } W_{t+1} > W_t$$

where N corresponds to density (ind m^{-2}) of the cohort at each sampling time and $\overline{w}$ is the mean individual biomass of the cohort at each sampling time.

The total productions of each cohort were expressed as:

$$P = \sum_{t=1}^{t=N} \left[\frac{N_t + N_{t+1}}{2}\right] d\overline{w}$$

where $d\overline{w}$ represents the variations of individual mean biomass of the cohort between two sampling times.

Total values of P for the population were expressed as:

$$P = \sum_{n=1}^{N} P_{cn}$$

where P_{cn} is the growth production of cohort n. $\overline{B}$, the annual mean population biomass, is calculated as:

$$\overline{B} = (1/T)\sum_{n=1}^{N} \overline{B}_{nt}$$

T is the period of study, N is the number of successive cohorts in the period T,$\overline{B}_n$ is the mean biomass of cohort n; and t is the duration of the cohort n (Cardoso et al., 2005).

3. Results

3.1. Environmental variables

Regarding the physicochemical parameters, the temperature, salinity and chlorophyll *a* presented a seasonal pattern with lower values during winter and higher values during spring/summer (Fig. 2A–C). No significant differences between the three sampling areas were observed (1-way ANOVA, Temperature: $F_2 = 0.14$, $P > 0.05$; Salinity: $F_2 = 0.18$, $P > 0.05$; chlorophyll *a*: $F_2 = 1.35$, $P > 0.05$).

For the dissolved oxygen (DO) and pH (Fig. 2D–E) an opposite pattern was observed, with higher values during winter and lower values during spring/summer. No significant differences were observed between the three sampling areas (1-way ANOVA, DO: $F_2 = 0.22$, $P > 0.05$; pH: $F_2 = 1.27$, $P > 0.05$). The organic matter (Fig. 2F) presented a quite stable pattern all over the study period with no significant differences between the sampling stations (1-way ANOVA, $F_2 = 2.57$, $P > 0.05$).

Concerning the granulometry, all the stations were dominated by fine sand, ranging from 50% (St. 2) to 77% (St. 3). However, no strong oscillations throughout the year were observed. Regarding most of the sediment fractions (gravel, medium sand and silt) no significant differences were observed between the sampling areas (1-way ANOVA, gravel, $F_2 = 0.18$, $P > 0.05$; medium sand, $F_2 = 0.83$, $P > 0.05$; silt, $F_2 = 0.38$, $P > 0.05$), except for the coarse and fine sand fractions where significant differences were observed between stations 2 and 3 (1-way ANOVA, coarse sand, $F_2 = 6.89$, $P < 0.05$; fine sand, $F_2 = 5.98$, $P < 0.05$).

3.2. Mercury concentrations in sediments, water and SPM

Considering the mercury concentrations in the sediments it was clearly visible a spatial gradient (Fig. 3A). The highest levels were observed at station 1, ranging between 82 and 206 μg Hg g^{-1}. Station 2 presented intermediate values (11–24 μg Hg g^{-1}), followed by station 3 (1.3–2.0 μg Hg g^{-1}) which presented residual values. Significant differences were observed between all the three sampling areas (1 way-ANOVA, $F_2 = 345.18$, $P < 0.05$).

Regarding the total dissolved mercury in the intertidal water pools, the three stations presented a similar pattern all over the study period, showing higher values during summer (Fig. 3B). No significant differences were observed between the three areas (1 way-ANOVA, $F_2 = 0.63$, $P > 0.05$).

Concerning the SPM fraction, stations 1 and 2 recorded similar mercury concentrations (st1 – 5–9 μg Hg g^{-1}; st2 – 4.5–11 μg Hg g^{-1}) and higher than station 3 (0.8–3 μg Hg g^{-1}) (Fig. 3C). No significant differences were observed between stations 1 and 2 (Wilcoxon two-sample test, $W = 59$, $P > 0.05$), but significant differences were observed between the stations 1 and 3 (Wilcoxon two-sample test, $W = 77$, $P < 0.05$) and between the stations 2 and 3 (Wilcoxon two-sample test, $W = 77$, $P < 0.05$).

3.3. Peringia ulvae density/biomass

Concerning the density pattern, all the stations presented a similar trend with higher values during winter/spring and lower during summer. Station 2 presented higher values than stations 1 and 3 (Fig. 4A). The biomass pattern was relatively constant all over

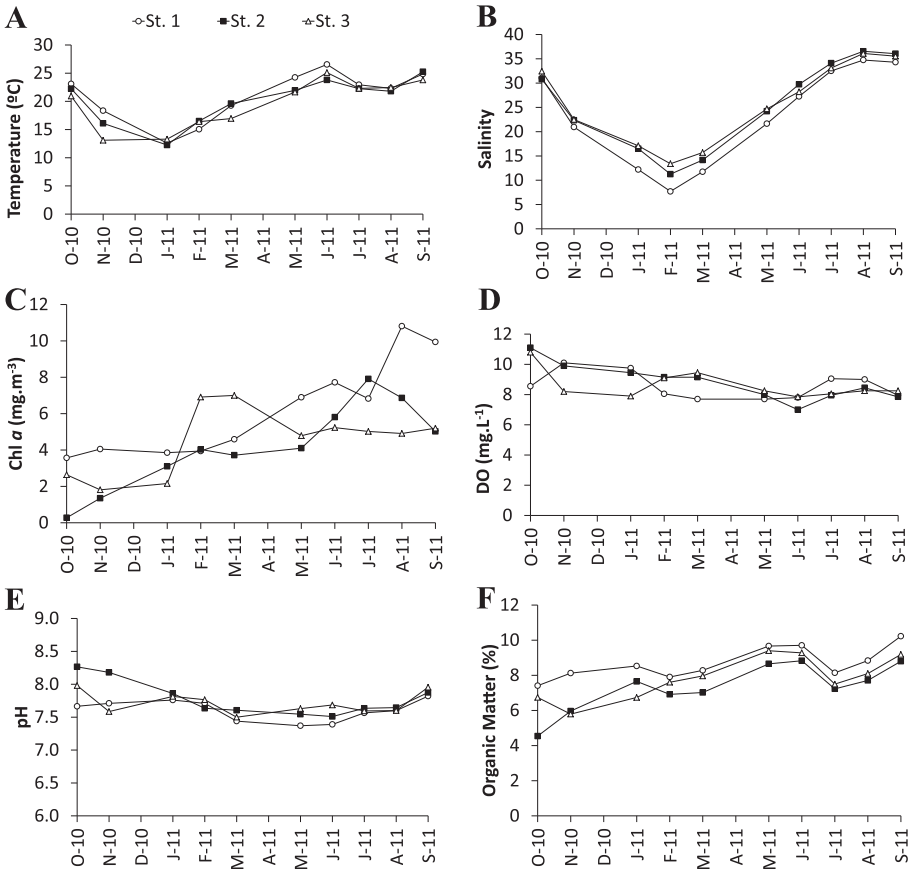

Fig. 2. Monthly variation of the physicochemical parameters in the 3 sampling areas during the study period. A) – Temperature, B) – Salinity, C) – Chlorophyll *a*, D) – Dissolved oxygen, E) – pH, F) – Organic Matter.

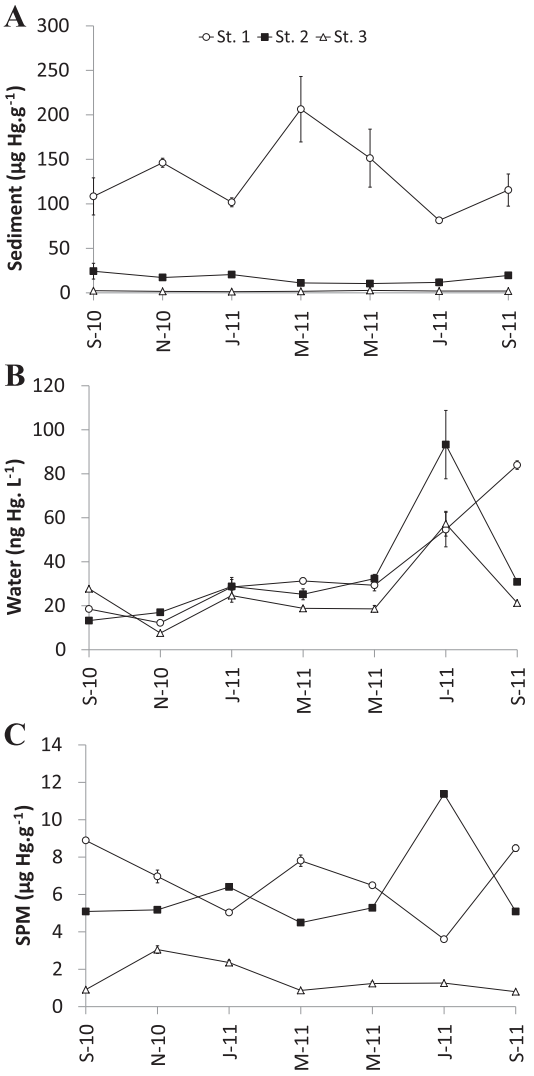

Fig. 3. Mercury concentrations in A) Sediment, B) Water and C) SPM. Error bars correspond to standard deviations.

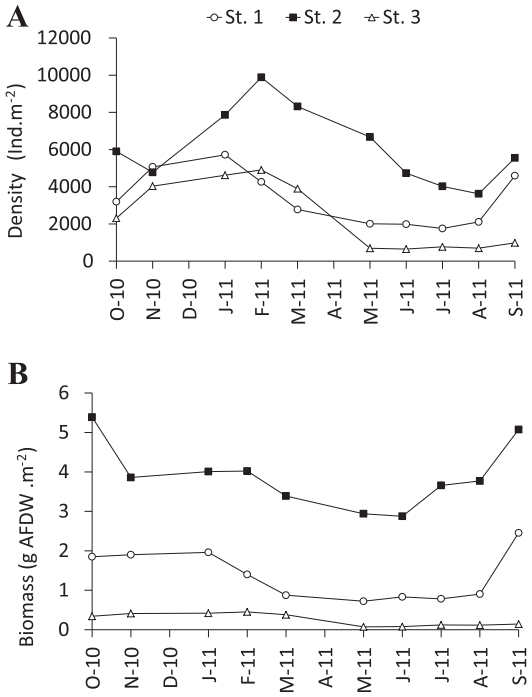

Fig. 4. Density (A) and Biomass (B) variations of *Peringia ulvae* in the 3 stations all over the study period.

the time for the three areas. Station 2 recorded the highest values (3–5 g AFDW m^{-2}) followed by station 1 (1–3 g AFDW m^{-2}) and finally station 3 with the lowest values (0.1–0.6 g AFDW m^{-2}) (Fig. 4B).

3.4. Population structure and growth

Marked differences in the population structure were observed at the three sites. At station 2 several modes were apparent in the population representing individuals of all age classes (juveniles <2 mm TSL; young individuals 2–3 mm TSL; adults >3 mm TSL (Cardoso et al., 2005). In contrast, stations 1 and 3, mainly station 1 were dominated by small individuals being significantly different from station 2 (χ^2 test, $P < 0.05$) (Fig. 5). Also, population structure at station 1 was significantly different from the one at station 3 (χ^2 test, $P < 0.05$).

Regarding the percentage of the different age groups, station 2 was dominated by adults (38%), contrarily to stations 1 and 3 which were dominated by young individuals and juveniles (St. 1 – young ind: 47%, juveniles: 36%; St.3 – young ind: 42%, juveniles: 35%).

3.5. Life span and productivity

Analyzing Fig. 6, it is apparent that *P. ulvae* presents 4 cohorts per year (1 spring cohort – February/March, 2 summer cohorts - June and July and 1 autumn cohort – September/October) at all stations. Generally, the autumn and spring cohorts lived longer than the summer ones. At station 2 the individuals lived longer (mean: 13 months) than at station 3 (mean: 10 months) and lastly station 1 (mean: 8 months) (Table 1).

Growth production (P) and mean population biomass ($\overline{B}$) were considerably higher at station 2 (185 g DW m^{-2} y^{-1}) than at station 1 (68.96 g DW m^{-2} y^{-1}) and finally station 3 (5.55 g DW m^{-2} y^{-1}). The $P/\overline{B}$ ratios were slightly higher at stations 1 and 2 than at station 3 (Table 2).

3.6. Mercury accumulation in biota

Total mercury levels in *P. ulvae* were generally higher in the most contaminated area (St1) (ranging from 0.050 to 0.250 µg Hg g^{-1}) declining along the Hg gradient (St2 – 0.050–0.070 µg Hg g^{-1}, St3 – 0.030–0.050 µg Hg g^{-1}) (Fig. 7). Significant differences between individuals of station 1 and 2 and station 1 and 3 were observed (2-way ANOVA, $F_2 = 34.4$, $P < 0.05$) however, no significant differences were observed between individuals of stations 2 and 3 (2-way ANOVA, $P > 0.05$). On the other hand, significant differences between juveniles and adults and between young individuals and adults were observed (2-way ANOVA, $F_2 = 13.83$, $P < 0.05$). Despite these differences between age classes, no metal accumulation through life was observed (Fig. 7). At station 1, a great variation in mercury concentration between age classes was observed.

Based on the estimated annual growth production values it is possible to determine the Hg associated to the species, annually, and consequently to assess the role of *P. ulvae* on the Hg transfer to the estuarine food web (Table 2). Production was considerably higher in the most contaminated areas (St2 – 185 g DW m^{-2} yr^{-1},

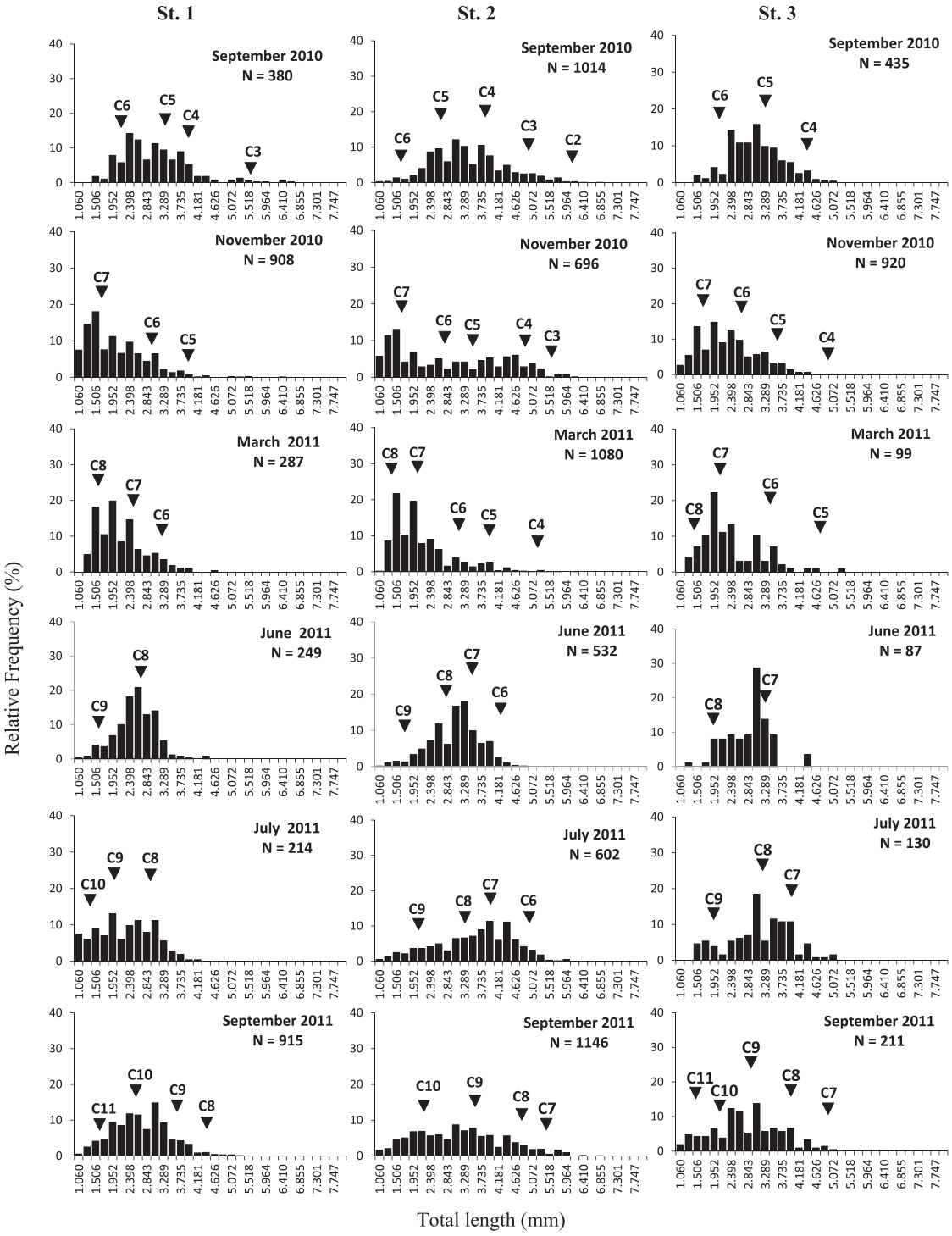

Fig. 5-. Size–frequency distribution of *Peringia ulvae* in the 3 stations. The cohorts (C) and the number of individuals are shown.

St1 – 68.96 g DW m^{-2} yr^{-1}) than in the least contaminated one (St3 – 42.72 g DW m^{-2} yr^{-1}). Considering the annual mercury bioaccumulation, this gastropod was responsible for the bulk Hg removal from the sediment, incorporating almost 31.45 μg Hg m^{-2} yr^{-1} in the intermediate contaminated area and 25.5 μg Hg m^{-2} yr^{-1} in the most contaminated area (Table 2). In the least contaminated area (St3) removal of Hg declined approximately 80%. In fact, extrapolating the results from the three sampling stations to the approximate 0.6 km^2 (St1–St3) of the intertidal flats existent in the inner Laranjo Bay, it is possible to conclude that this species is responsible for the mean annual incorporation of 12.5 g of Hg from the sediments.

4. Discussion

Understanding the impact of pollutants, like mercury on the structure and dynamics of benthic invertebrates is an important issue due to their impact on the food web structure and

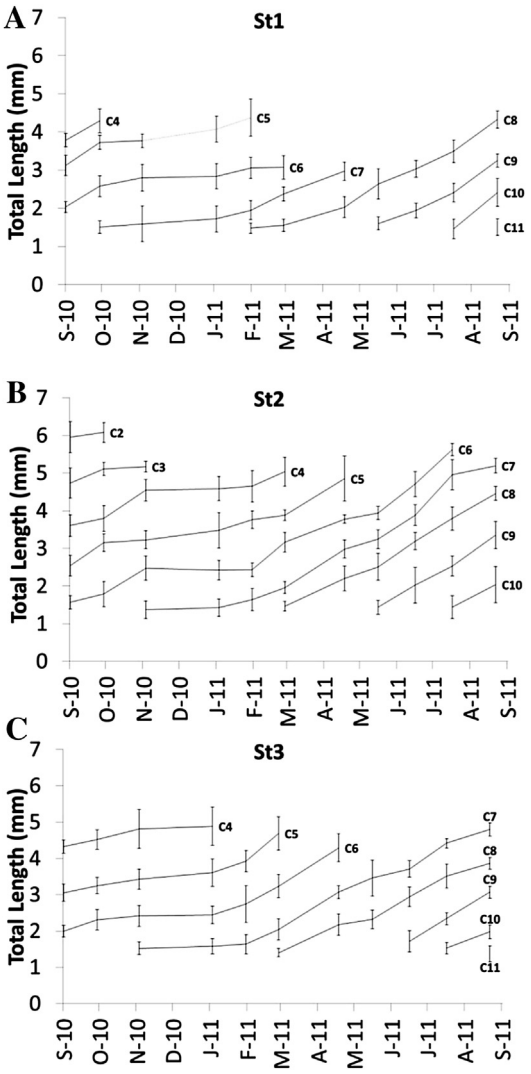

Fig. 6. Estimated growth of *Peringia ulvae* cohorts in the 3 stations. A) St.1, B) St.2 and C) St.3.

functioning. Benthic macroinvertebrate are usually considered good indicators of environmental health, since they are sedentary organisms and respond to the environmental impacts (Calabretta and Oviatt, 2008; Dauvin, 2008).

The present work revealed clear differences in the dynamics and structure of *P. ulvae* population along the mercury contamination gradient. This gastropod presented higher abundances and biomasses, as well as secondary production values at the intermediate contaminated area (St. 2) than at the most contaminated (St. 1) and least contaminated (St. 3) areas. This response seems to be related to the hormesis phenomenon. Hormesis represents a phenomenon in which a chemical that causes harm at greater doses paradoxically

Table 1
Estimated life span for the different cohorts (spring, summer and autumn) in the 3 stations. – Unavailable data.

	Spring Cohort (months)	1° Summer Cohort (months)	2° Summer Cohort (months)	Autumn Cohort (months)
St1	7–8	7–8	7–8	7–8
St2	12–13	10–11	12–13	12–14
St3	10–11	8–9	9–10	–

Table 2
Growth production estimations and associated mercury available for trophic transfer in *P. ulvae* in the 3 stations.

	P (g DW m^{-2} y^{-1})	$\overline{B}$ (g AFDW m^{-2})	$P/\overline{B}$	Associated Hg (μg m^{-2} y^{-1})
St1	68.96	1.23	3.54	25.52
St2	185	3.58	3.23	31.45
St3	5.55	1.17	2.34	5.55

results in beneficial effects at a low dose (Calabrese, 2008). Hormesis has been reported to be widespread with metals, including mercury (Calabrese and Blain, 2004) and despite most references are related to experimental work, this phenomenon could be applied to the present study considering St.3 as a control, St.2 as a low dose and St.1 as a high dose. A typical hormesis response was observed in a study in which were injected mallard (*Anas platyrhynchos*) eggs with methylmercury chloride. In this study a case of hormesis seemed to occur because hatching success of eggs injected with the lowest dose was significantly greater (93.3%) than that of controls (72.6%), whereas hatching success decreased at progressively greater doses of mercury (Heinz et al., 2012). There are multiple hormetic responses, however the maximum stimulatory response (in general is 130–160% greater than the control) has become the most distinguishing characteristic of the dose–response relationship (hormesis) (Calabrese, 2008).

On the other hand, *Peringia ulvae* population presents a paradox, given its considerable abundance and biomass values in an area with very high levels of sediment contamination (St. 1). This is in accordance with the results obtained in the Seine estuary for the subtidal macrobenthic *Abra alba-Pectinaria koreni* community (Dauvin, 2008). This result also corroborates the findings of McLusky et al. (1986) which pointed out that in general, mollusks are one of the taxonomic groups least sensitive to metallic pollution compared to annelids and crustaceans. However, this fact is contrary to most of the macrobenthic species which usually present lower abundances and biomasses in the most contaminated areas (Mucha et al., 2005; Calabretta and Oviatt, 2008; Amin et al., 2009). Also, the present work is contrary to the findings of Araújo et al. (2012), which found that *Peringia ulvae* when exposed to sediments with different degrees of contamination (including several contaminants) avoided the most contaminated sediments by escaping to the least contaminated ones. Indeed, the greater the percentage of sediments' contamination the greater the percentage of avoidance of snails and even the organisms presented in 100% contaminated sediments suffered a reduction in their activities (Araújo et al., 2012).

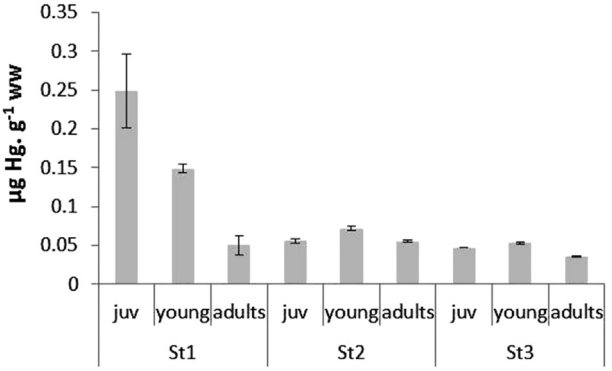

Fig. 7. Mercury accumulation in *P. ulvae* age classes in the 3 stations. Error bars correspond to standard errors. ww – wet weight.

In terms of population structure and growth, *P. ulvae* was clearly negatively affected by the intense mercury contamination since the most contaminated area was characterized by the dominance of juveniles and young individuals. Station 1 presented the lowest percentage of adults of the three areas. It was also evident that the life span of the individuals living in the most contaminated area was strongly reduced comparatively to the other areas. Individuals from St.2 were the ones with higher life span, followed by the ones of St3 and finally those of St.1. These results may also be related to the phenomenon of hormesis, previously explained.

According to SQG (Sediment Quality Guidelines), values much higher than ERM (effects range median) for mercury were observed at station 1, which may be responsible for the adverse biological effects observed in the *P. ulvae* population. According to Long et al. (1995), in sediments with mercury concentrations higher than 0.71 ppm (ERM) the incidence of biological effects corresponds to approximately 42%. Considering that at station 1 the Hg concentrations in the sediments ranges between 80 and 200 ppm, the incidence of biological impacts, surely would be greater. Also, according with OSPAR 2009, the EQS (Ecological Quality Standards) adopted for the Hg in sediments is 0.5 ppm.

In terms of growth production, the species was not greatly affected by the strong mercury concentrations since higher values were recorded at station 1 than at station 3. This is in accordance with the higher biomasses observed in that area. So, mercury contamination seems to negatively affect the species until a certain extent, since only the growth and life span were the most affected parameters in the presence of high Hg concentrations. Otherwise, comparing the most contaminated area (St.1) with the least contaminated one (St. 3), the latter presented always lower abundance, biomass and productivity. In fact, *P. ulvae* was revealed to be a tolerant species to mercury. In contaminated sites, the replacement of sensitive species by pollution-tolerant species may occur (Weis et al., 2004 and references therein). For example, this gastropod revealed an opposite response compared to the bivalve *Scrobicularia plana*. The latter was strongly affected in terms of abundance, biomass and productivity by Hg contamination (personal communication). So, different groups of mollusks may respond differently to the same contaminant. This can be related with the living position and feeding guilds of the species. While *P. ulvae* is an epibenthic species, living in the surface of the sediment, where the metal concentrations are lower than in deeper layers, the bivalve *S plana* is an endofaunal species which lives in direct contact with higher metal concentrations. Rumisha et al. (2012) also found that different species of mollusks responded differently to the same contaminants.

In terms of mercury bioaccumulation in this gastropod species it was observed that at station 1 (most contaminated area) a greater variability in mercury concentrations between age groups was observed compared with others stations. However, as far as possible from the discharge point, the variations in mercury concentrations due to size were minimized. The same conclusions were observed by Elliott and Griffiths (1986) for the bivalve *Mytilus edulis*. On the other hand, juveniles and young individuals presented always higher mercury concentrations than adults, which is contrary to some invertebrate species (e.g. *Scrobicularia plana* – Coelho et al., 2006) that showed mercury bioaccumulation through lifespan. However, some explanations for this fact may be related with the efficiency of assimilation rates, which can be higher in younger individuals while the excretion rates may be higher in adults. Another explanation could be based on the growth dilution phenomenon. Growth dilution happens when the organism's growth is faster than its rate of metal absorption (Tavares et al., 2011). This phenomenon is typical of species with a fast growth and was also observed in some fish species, like *Liza aurata* (Tavares et al., 2011) and the Atlantic salmon *Salmo salar* (Ward et al., 2010).

Our findings are important in an environmental point of view since, *P. ulvae* is a very abundant species in intertidal areas and a common food resource for several species of predators, like waders (Cabral et al., 1999) and fishes, some of them with economic importance like, *Platichthys flesus* (Aarnio and Mattila, 2000), contributing to the mercury transfer from the sediments to the estuarine trophic web. In fact, *P. ulvae* is the responsible for the mean annual incorporation of approximately 15 g of mercury in an area of 0.6 Km^2. This value may seem negligible when comparing with the large amount of mercury accumulated in the Laranjo Bay sediments ($\approx$30 tons; Pereira et al., 2009). However, if we consider the other species that are directly associated to this gastropod, this value can have a strong relevance in terms of mercury biomagnification. Studies like the present one are important to assess the effects of contaminants on the structure and dynamics of macrobenthic species playing a key role in the functioning of the higher trophic levels.

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