



DELIVERABLE D3.3



Ecosystem threshold response to the main environmental factors



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Integrated water resources and coastal zone management in European lagoons in the context of climate change



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1. General Introduction

The influence of changing environment factors on the support of vital ecosystem processes and services in coastal lagoons

Increases in coastal populations and accompanying shifts in land use (Vitousek and Mooney, 1997), have led to changes that have cumulative impacts on the delivery of nutrients and other pollutants to coastal lagoons. For at least 50 years, researchers have recognised this growing imbalance, especially in estuarine waters and coastal lagoons where nutrient over-enrichment has fuelled accelerated primary production, or eutrophication; and other pollutants have originated severe consequences for the ecosystems and the living forms that inhabit them.

Pollutant sources change, and generally increase, as use of a region intensifies. Wastewater, agricultural discharge, stormwater and industrial sources are among the major contributors. Another change resulting from development of the coastal landscape is an alteration of the transport mechanisms of pollutants to nearby waters. The channelisation of streams and rivers, ditching and the removal of native vegetation all contribute to increased rates and quantities of pollutants transported from the land to adjacent water bodies. Finally, human activities in coastal regions often lead to a significant loss of the natural landscapes that either retain or remove pollutants, such as wetlands.

Due to their location between land and sea, lagoonal ecosystems are subject to an elevated rate of dynamic changes in the natural environment that result in high biological productivity and diversity. In these systems, most of the seafloor lies within the photic zone, which allows benthic primary productivity. As a result, shallow lagoons and bays tend to be dominated by benthic producers such as seagrasses, perennial macroalgae and microphytobenthos, rather than by phytoplankton. Furthermore, as highly productive systems, lagoons usually show marked abundances of macrofaunal species. They also support fish populations, many of great commercial importance, and constitute essential zones for the nest building, migration and hibernation of aquatic birds.

Many authors have demonstrated the importance of benthic primary producers in coastal lagoon nutrient cycles (Fong et al., 1993; Viaroli et al., 1996; McGlathery et al., 1997; Hauxwell et al., 2001; Tyler and McGlathery, 2003; Palomo et al., 2004; Boyer and Fong, 2005), emphasising their role in the resistance of such systems to eutrophication. Benthic plant assimilation from both the water column and sediments reduce nutrient availability for other autotrophs, like phytoplankton or floating macroalgae during the growing season. However, their importance is not only due to nutrient captation; plant metabolism and the decomposition of buried plant tissues also modify biogeochemical processes and nutrient retention in sediments (McGlathery et al., 2007). The role of benthic plants in these shallow ecosystems can be considered as a very effective ‘coastal filter’ of nutrients. The high metabolic rates of benthic primary producers mediate nutrient cycling processes and result in strong benthic-pelagic coupling (e.g. Krause-Jensen et al., 1999; Tyler et al., 2001; Eyre and Ferguson, 2002). For example, because of the importance of benthic-pelagic coupling and the temporary retention of nutrients in longer-lived plant biomass (compared to phytoplankton), coastal bays often have relatively high apparent water quality (low water column nutrient concentrations, low phytoplankton biomass) even when nutrient loading rates are high for much of the year (Valiela et al., 1997; McGlathery et al., 2004; McGlathery et al., 2007).

Lagoonal benthic macrofauna also plays an essential role in the maintenance of the ecosystem’s integrity by mediating exchanges and transformations of energy and materials, including nutrients, between the water column and sediments (Hansen and Kristensen, 1997; Twilley et al., 1999). Furthermore, macrobenthos production provides an important trophic transfer vehicle within the coastal ecosystem (Diaz and Rosenberg, 1995). In this way, benthic macrofauna affect lagoonal nutrient cycles and could modify the functioning of the plant-mediated ‘coastal filter’ via their active grazing on primary producer forms and modulation of nutrient fluxes.

Benthic plants and macroinvertebrates seem to be responsible for certain biotic feedbacks that can ultimately modify an ecosystem’s response to nutrient enrichment. Therefore, stress-induced changes to the benthos need to be understood well enough to apprehend consequential losses of vital ecosystem services.

At first sight, benthic organisms seem extremely vulnerable to the effects of climate change, since such change would have a strong impact on them as regards their survival,

productivity, distribution and function (Nicholls et al., 1999; Short and Neckles, 1999; Simas et al., 2001).

Increasing water temperature will directly affect benthic primary producer metabolism and maintenance of a positive carbon balance established by the relation between photosynthesis and respiration. The direct effects of increased temperature will depend on thermal tolerance and optimum temperatures for photosynthesis, respiration and growth of individual species.

Apart from the rise in temperature, other changes are expected, such as an alteration in rainfall patterns and an increase in the occurrence of extreme climate events, which may lead to a change in the frequency and intensity of storms (Nicholls and Hoozemans, 1996 and references within; Watson et al., 1996). The impacts of these changes on benthic primary producers will be due not only by the direct effect of the erosion caused by the storms and associated wave action, but also by the ‘shading’ caused by suspended sediments resulting from the coastal erosion and the increase of storm water runoffs.

Projections of the sea-level rise for the end of the 21st century range from 15 to 95 cm, with a ‘best estimate’ of 50 cm (Watson et al., 1996; IPCC, 2001; Rahmstorf, 2007). This rise will have direct consequences on the benthic primary producers of coastal environments due to the increase in depth, the reduction of light reaching the bottom, the changes of salinity and the alteration of the hydrodynamism of the areas.

Another consequence of future environmental change is the increased nutrient input into coastal areas, which provide natural resources and suitable space for economic activities and human settlements. It is estimated that 50-70% of the world’s population lives in coastal zones, a figure that shows an increasing tendency. Furthermore, the use of fertilizers for agriculture in the surrounding watersheds is also expected to rise. This increasing pressure may lead to a rise in the discharge of nutrients into coastal systems, which will be aggravated by the expected changes in hydrological regimes and precipitation patterns (Bethoux et al., 1998; Bethoux and Gentili, 1999; Short and Neckles, 1999; Bouraoui et al., 2002; Sumner et al., 2003; Dore, 2005). Eutrophication processes may affect benthic primary producers and consumers of coastal zones through increases in water column light attenuation caused by increased phytoplankton concentrations, the proliferation of floating macroalgae and epiphytes or even more severe phenomena, such as oxygen depletion at the bottom.

The rise in sea water temperature, the reduction of light reaching the bottoms, through the cumulative interaction of the rise in sea-level and increased water column turbidity, and the expected changes in the hydrodynamism and salinities in coastal lagoons are expected to have deleterious effects on the survival of many lagoonal organisms and affect their role in the processes and services they support.

1.1. The Methodological approach

A full description of the methodologies used can be found in the LAGOONS Report D3.1 available at <http://lagoons.web.ua.pt/>. As a summary, and with the final aim of identifying and evaluating key environmental factors that contribute to drastic changes in lagoonal communities and the services they provide, two main methodological approaches are defined:

- i) Community level: Due to their importance in lagoonal environments benthic macrophytes and macrozoobenthos will be considered for this study. For macrophytes, a set of ‘key’ environmental variables can be identified according to the particular characteristics of each species found. Historical data can also be used to identify the main threats to these communities within each case study lagoon and the environmental variables that determined their regression or expansion in the last decades. The study will focus on variables that could affect benthic primary production and/or survival. In this sense, temperature and bottom light regimes constitute key environmental variables for macrophyte production and maintenance of a positive carbon balance. Other variables, such as salinity ranges and shear stress at the bottoms are of enormous importance for macrophyte survival and could help determining which areas are suitable for macrophyte settlement according to the different species optimal ranges for these variables. For macrofaunal communities multivariate methods can be used to analyse the relationship between environmental variables and benthic community structure, which will allow us to infer the relative influence of a set of environmental variables on benthic macrofauna distributions and

elucidate the effect of possible changes of these variables on benthic communities.

- ii) Species level: The response of amphipod species to changes in their environment could help determining the mechanisms responsible for the impact of these changes on the ecosystem. In this sense, the study of survival rates of these indicator species in a changing environment provides useful information for the determination of trends in the biological response to climate change. Furthermore, variations of the toxic response of amphipods to pollutants are to be expected as a consequence of changes on the physical environment (mainly temperature and salinity) and, therefore, this coupled effect on species survival need to be assessed in order to better quantify impacts.

2. Evaluation of the main environmental variables responsible for macrophyte distribution and survival

The Ria de Aveiro lagoon

2.1.1. Rooted macrophyte species in the lagoon and their relationship with environmental variables

The rooted macrophytes species in Ria the Aveiro intertidal area can be divided into two major groups: seagrasses and salt marshes. Presently, the most representative seagrass species in Ria is *Zostera noltii* (Fig. 2.1.1.1). Regarding salt marshes, the low marshes are dominated by *Spartina maritima* (Fig. 2.1.1.2) whilst the high marshes are dominated by *Juncus maritimus* (Fig. 2.1.1.3) (for a more detailed description of seagrasses and salt marshes composition, please see LAGOONS Report D3.2).



Fig. 2.1.1.1A – *Z. noltii* meadows
(Ria de Aveiro, ©AI Lillebø)



Fig. 2.1.1.2A – *S. maritima* marsh
(Ria de Aveiro, ©AI Lillebø)



Fig. 2.1.1.3A – *J. maritimus* marsh
(Ria de Aveiro, ©AI Lillebø)



Fig. 2.1.1.1B – *Z. noltii*
(Ria de Aveiro, ©AI Sousa)



Fig. 2.1.1.2B – *S. maritima*
(Ria de Aveiro, ©AI Lillebø)



Fig. 2.1.1.3B – *J. maritimus*
(Ria de Aveiro, ©AI Lillebø)

The following table summarizes the influence of environmental variables on *Zostera noltii* distribution, growth and photosynthesis (the data for Ria de Aveiro is currently unpublished and result from a close cooperation between LAGOONS project team and the LTER-RAVE project team).

Table 2.1.1.1. Influence of environmental variables on *Zostera noltii* distribution, growth and photosynthesis in Ria de Aveiro Mira Channel

Variable	Range in the lagoon	Optimum	Critical	Source
Water salinity	2-32 ¹	2-41(euryhaline species)	<2 (lower) >41 (upper)	(Fernández-Torquemada et al., 2011) ²
Water temperature	13-30° C ¹	20-25 °C	38 °C (upper)	(Massa et al. 2009) ³
Bottom type	Sandy ¹ Muddy ¹	Muddy	-----	-----
Light	No data available	520 $\mu\text{mol photons m}^{-2} \text{ s}^{-1}$	20 $\mu\text{mol photons m}^{-2} \text{ s}^{-1}$ (lower) 1245 $\mu\text{mol photons m}^{-2} \text{ s}^{-1}$ (upper)	(Peralta et al. 2002) ⁴
		165 $\mu\text{mol photons m}^{-2} \text{ s}^{-1}$	25 $\mu\text{mol photons m}^{-2} \text{ s}^{-1}$ (lower)	(Olivé et al. 2007) ⁵
Nutrients	DIN ⁷ : 2.5 – 195 $\mu\text{mol l}^{-1}$ SRP ⁷ : 0.5-3.6: $\mu\text{mol l}^{-1}$	NH ₄ ⁺ can have a toxic effect on <i>Zostera</i> survival and growth (depending on the internal C balance to sustain N assimilation); PO ₄ ³⁻ addition can alleviate NH ₄ ⁺ toxicity ⁸	Not limited by nutrients in the Ria de Aveiro	(Lopes et al. 2007) ⁶ (Brun et al. 2002) ⁷

¹Unpublished data; ²Laboratory experiment; ³Ria Formosa, Portugal; ⁴Laboratory experiment, Cadiz Bay, Spain; In the original publication the tested light intensities corresponded, respectively to: 0.8 mol photons $\text{m}^{-2} \cdot \text{d}^{-1}$ (2% surface irradiance); 20.6 mol photons $\text{m}^{-2} \cdot \text{d}^{-1}$ (42% surface irradiance); 49.3 mol photons $\text{m}^{-2} \cdot \text{d}^{-1}$ (100% surface irradiance).
⁵Laboratory experiment, Cadiz, Spain; ⁶Max. and min.values considering four seasons, and low and high tide samplings in 2000-2001; ⁷Laboratory experiment, Cadiz Bay, Spain.

The following table summarizes the data for environmental variables on *Spartina maritima* and *Juncus maritimus* distribution in Ria de Aveiro. These data are currently unpublished and result from a close cooperation between LAGOONS project team and the LTER-RAVE project team.

Variable	Range in the lagoon <i>Spartina maritima</i>	Range in the lagoon <i>Juncus maritimus</i>
Sediment salinity	No data available ¹	No data available
Sediment temperature	7-25 °C	7-25 °C
Bottom type	Sandy Muddy	Muddy
Inundation Period	Low marsh ² Tidally inundated (twice a day)	High marsh Tidally inundated (twice a day)

¹The optimum salinity in the sediment for *S. maritima* range between 0 and 35, whilst the critical salinity is > 55 (Adams and Bate, 1995), ²Optimum for: leaf elongation is dry conditions; stem elongation is tidally inundated and submersed; stem production is dry conditions or tidally inundated. Critical for: leaf elongation is tidally inundated or submersed; stem elongation is permanently dry conditions or reduced tidal flushing; stem production is submersed conditions (Adams and Bate, 1995).

2.1.2. The status of rooted macrophytes in the lagoon: Regression events and their causes

Until the 1960's, Ria de Aveiro subtidal areas were colonized by transitional waters' seagrass species (*Ruppia cirrhosa* (Petagna) Grande, 1918, *Stuckenia pectinata* (L.) Börner, 1912, previously known as *Potamogeton pectinatus* and by *Zostera* Linnaeus, 1753) (Silva et al., 2004). Later on, a survey performed in the 1980's at Ovar channel recorded that subtidal areas were vegetated by the seagrasses *Stuckenia pectinata*, *Ruppia cirrhosa*, *Zostera marina* and *Zostera noltii* (also present at intertidal area) (Silva et al., 2004). Throughout the last decades of the 20th century, Ria de Aveiro lagoon suffered a great decline in the seagrass meadows extension mainly due to changes in the system's hydrodynamics as a result of dredging activities (Silva et al., 2004). These changes also altered the tidal prism and increased the water velocity (Picado et al., 2010), resulting in the loss of subtidal seagrass meadows and reducing the intertidal meadows extension and biodiversity (Silva et al., 2004, Silva et al., 2009, Azevedo et al., 2013, Cunha et al., 2013 - for a more detailed description please see LAGOONS Report D3.2). Presently, seagrass meadows of this lagoon are restricted to intertidal areas and *Zostera noltii* is the most representative seagrass species in Ria. Last seagrasses mapping was done one decade ago and an attempt for an updated mapping is now ongoing within the LAGOONS project.

Regarding *Spartina maritima* salt marshes in the Ovar channel close to the Varela bridge there are some evidences of litoral erosion due to the changes in the system hydrodynamics (Figure 2.1.4). In the Baixo Vouga lagunar, characterized by large areas of *Juncus maritimus* marshes, there are evidences of dying plants due to the increase of tidal prism and submersion time (Figure 2.1.5) (for a more detailed description please see LAGOONS Reports D2.1 and D3.2, available at <http://lagoons.web.ua.pt/>).



Fig. 2.1.4 – *Spartina maritima* patches
(Ria de Aveiro, ©AI Lillebø)



Fig. 2.1.5 – *Juncus maritimus* dying
(Ria de Aveiro, ©AI Lillebø)

2.1.3. Rooted macrophyte vulnerability in the context of global climate change

Due to climate change, the frequency of extreme weather events in Ria de Aveiro are predicted to increase, leading also to the increase in the occurrence of flood and drought events (LAGOONS Report D5.1). As a result, seagrasses might be affected by a stronger wave action and consequent erosion, but also by a reduction in light availability (eventually, light limitation by shading effect) due to sediment resuspension (LAGOONS Report D3.3).

Regarding salt marshes, sea level rise predictions for Ria de Aveiro show that salt marshes dynamics and stability will be closely related to/dependent on the hydrodynamic conditions under climate change context, namely regarding sea level rise (Valentim et al., 2013). Thus, nutrients, sediment salinity and moisture, saline stress and relative vegetation patterns, are some of the parameters likely to be affected, compromising salt marshes stability (Valentim et al., 2013).

2.2. The Vistula lagoon

2.2.1. Macrophyte species in the lagoon and their relationship with environmental variables

Research on the effect of environmental factors on the distribution and biology of macrophytes has been particularly conducted on lakes (among others Lehmann et al. 1997, Scheffer et al. 1992, Amano et al. 2011), and more seldom in lagoons (Selig et al. 2007, Li et al. 2008, Prado et al. 2013). Data regarding the scope of abiotic factors tolerated by macrophytes in the waters of the Baltic Sea are very scarce (Kautsky 1988, Dahlgren and Kautsky 2004, Selig et al. 2007).

The present paper concerns the macrophytes of the Polish part of the Vistula Lagoon (Figure 2.2.1.1). It is restricted to species with considerable bottom cover permitting them to develop clearly spatially defined assemblages in the form of clusters or belts of vegetation (Table 2.2.1.1).

The macrophytes were divided into: i. emergent plants – helophytes (*Phragmites australis*, *Typha angustifolia*, *Schenoplectus lacustris*), ii. submerged plants – elodeids (*Potamogeton perfoliatus*, *P. pectinatus*, *Zannichella palustris*, *Ceratophyllum demersum*, *Myriophyllum spicatum*, *Chara sp.*, and iii. plants with floating leaves – nymphaeids (*Nuphar lutea*, *Nymphoides peltata*).

The following were taken into consideration among the factors responsible for macrophyte distribution and survival in the lagoon: Secchi disc visibility and factors affecting visibility (suspended solids, including phytoplankton biomass expressed as chlorophyll-a concentration), salinity, total nitrogen (N_{tot}), and total phosphorus (P_{tot}), as well as potential importance of wind wave action and ice cover.



Figure 2.2.1.1. Distribution of vegetation zones in the Vistula Lagoon

Salinity

The salinity of the Polish part of the Vistula Lagoon changes in a clear gradient – from a mean of 2.17 PSU in the western part of the Lagoon to 3.11-3.48 PSU in its eastern part (Table 2.2.1.2). Therefore, the degree of salinity decreases along the growing distance from the Strait of Baltiysk, and similarly increases along with the growing distance from the mouth of the largest rivers inflowing to the Lagoon on the Polish side: the Elbląg River and the rivers of the delta of the Vistula River (the Szarpawa and Nogat Rivers).

Considerable fluctuations of salinity in time occur, with varied amplitude in particular parts of the Lagoon (Table 2.2.1.2). The highest fluctuations are observed in the western part (0.73-3.82 PSU), and the lowest in the eastern part (1.6-5.2 PSU).

Salinity usually increases in summer and autumn, and decreases in winter and early spring (Żmudziński, Szarejko 1955, Bogdanowicz (2009). This is related to the inflow to the Lagoon of freshwater from the catchment in spring, low water levels in summer, and autumn storms resulting in increased inflow of Baltic Sea water into the Lagoon.

In the most saline western part of the Lagoon, relatively high cover is reached by only three species – a helophyte *Phragmites australis*, and the elodeids: *Potamogeton perfoliatus* and *P. pectinatus* (Table 2.2.1.1).

Table 2.2.1.1. Distribution, frequency, and species richness of macrophytes in the Vistula Lagoon. The names of locations describe shore zones following Pliński et al. (1978), with certain modifications. For the zones distribution see Figure 2.2.1.1.

Location	Phr. aus.	T. ang.	Sch. lac.	Pot. per.	Pot. pec.	Zan. pal.	Cer. dem.	Myr. spic.	Chara sp.	Nuph. lut.	Nym. pel.
Piaski	++										
Braniewo	++		+	++	++						
Krynica	++			+			+	+			
Frombork	++		+	++	+						
Tolkmicko	++		+	++	+						
Suchacz	++		+	++	+		+	+		+	
Łaszki	++	++	++	++	++	+a	+	+	++	+	+a
Frequency [%]	100	14	71	85	71	14	42	42	14	28	14

Phr. aus. - *Phragmites australis*; T. ang. – *Typha angustifolia*; Sch. lac. – *Schenoplectus lacustris*; Pot. per. – *Potamogeton perfoliatus*; Pot. pec. – *P. pectinatus*; Zan. pal. – *Zannichella palustris*; Cer. dem – *Ceratophyllum demersum*; Myr. spic. – *Myriophyllum spicatum*; Nuph. lut. – *Nuphar lutea*; Nym. pel. – *Nymphoides peltata*; + small patches; ++ large patches and extensive belts; a – following Gajewski (2010).

The number of macrophyte species developing clearly distinguishable assemblages gradually increases westwards, reaching the highest species diversity in areas at the largest distance from the inflow of saline Baltic Sea waters – Kąty Bay, and Łaszki, in the vicinity of the mouths of the Vistula Królewiecka and Szkarpa River. Most species also reach the largest individual sizes, cover, and biomass here (Gajewski 2010).

The mean maximum concentrations recorded in the Lagoon, reaching 3.5 PSU, do not exceed the range or optimum values for plants found in the western part of the Lagoon (Table 2.2.1.3). Therefore, for the majority of plants, salinity seems not to be the factor restricting

their occurrence in the Lagoon. Nymphaeids, considered to commonly inhabit fresh or brackish waters, are an exception (Table 2.2.1.3).

Table 2.2.1.2. Physical and chemical water parameters in the Vistula Lagoon (the data obtained from the Voivodship Inspectorate of Environment Protection in Olsztyn). The presented values are means and ranges for the period 2007-2009. The names of locations describe shore sections, with certain modifications following Pliński et al. (1978). TSS - total suspended solids. For shore sections distribution see Figure 2.2.1.1.

Location	Visibility [cm]	TSS [mg/L]	Chl.-a [µg/L]	Salinity [PSU]	Ntot. [mg/L]	Ptot. [mg/L]
Piaski	44	52.54	33.36	3.31	1.65	0.094
	15-80	26-130	14-128	2.6-4.7	1.06-3.00	0.022-0.176
Braniewo	49	42.78	30.93	3.48	1.75	0.092
	30-110	28-70	16.8-86.9	1.6-5.2	1.22-286	0.047-0.0161
Krynica	41	54.48	37.65	3.11	1.87	0.090
	20-130	29.7-82	10.61-108.5	2.1-4.5	0.881-3.659	0.040-0.205
Frombork	39	49.23	32.54	3.29	1.95	0.089
	25-70	21.3-63	8.75-52.38	2.32-4.7	1.56-3.00	0.045-0.162
Tolkmicko	40	52.28	34.66	2.99	2.29	0.097
	30-100	35.8-110	13.1-61.7	1.47-4.61	0.84-4.56	0.052-0.182
Suchacz	39	53.5	46.98	2.39	2.14	0.114
	30-75	18-102	8.42-142.2	1.22-3.82	0.813-4.181	0.048-0.223
Łaszki	50	48.15	36.55	2.17	2.31	0.114
	20-85	9.8-140.0	6.08-145.9	0.73-3.37	0.713-3.839	0.035-0.315

Table 2.2.1.3. Salinity as a variable determining macrophyte distribution

Macrophyte	Range		Optimum	Critical
	Vistula Lagoon	Other sources		
<i>Phragmites australis</i>	2.17-3.48	0-72a, 0-30c	5.84 b, 5f	33.1f
<i>Typha angustifolia</i>	2.17-2.99	0-15k		
<i>T. latifolia</i>	2.99-3.29		<0.5k	
<i>Potamogeton perfoliatus</i>	2.17-3.48	0-8d		
<i>P. pectinatus</i>	2.17-3.48	0-18d	3-12j	
<i>Zannichella palustris</i>	<2.17	0.4-7d		
<i>Ceratophyllum demersum</i>	2.17-3.11	0-9d		
<i>Myriophyllum spicatum</i>	2.17-3.11	0-6d, 0-13.3e	0.17-3.33e	24.5e
			<3.6i	
<i>Nuphar lutea</i>	<2.17		<0.5k	
<i>Nymphoides peltata</i>	<2.17		<0.5l	
<i>Chara</i> sp. sp.	<2.17	0-17d		

a - Achenbach et al. (2013)

b - Gorai et al. (2011)

c - Matoh et al. (1988)

d – Luther (1951)

e - Haller et al. (1974)

f - Lissner and Schierup (1997)

i – Li et al. (2011)

j - Prado et al. (2013)

k - <http://www.wetland.org/downloads/waterfowl%20habitats.pdf>

Nutrients – total nitrogen and total phosphorus

Substantially increased values of nitrogen in water occur in the southern part of the Lagoon, between Frombork and Suchacz, and the highest in the western part (Łaszki) (Table 2.2.1.2). A similar spatial distribution is observed in the case of concentrations of total phosphorus, with the maximum recorded in the western part of the Lagoon – Suchacz and Łaszki. This can be related to the inflow and direction of flow of the main streams of nutrients from the side of the land by underground flow and with rivers). Similar results were obtained by Renk et al. (2001).

The concentrations of nitrogen and phosphorus recorded in the Lagoon were usually respectively 10 and 3-5 times higher than those in bays at the shores of Sweden dominated by macrophytes. The water in the bays showed a similar level of total nitrogen and level of total phosphorus more than twice higher than that of phytoplankton-dominated bays (Dahlgren and Kautsky 2004). This suggests that the concentrations of nutrients in the Vistula Lagoon predestine the ecosystem for remaining in the phytoplankton-dominated state. Despite this, the bottom of the western part of the Lagoon in the estuarial area of rivers is abundant in submerged vegetation, which seems to hinder the development of phytoplankton. In spite of increased concentrations of nutrients, the recorded concentrations of chlorophyll-a remained at a level lower than in the other areas of the Lagoon. Increased concentrations of nutrients in the western part of the Lagoon did not constitute a barrier for the development of underwater meadows of Charales, abundant in this region. Similarly, the dependence of charophytes on low phosphorus levels, as observed in limnic waters, was not verified in the Baltic coastal waters (Selig et al. 2007).

Water transparency, chlorophyll-a, and total suspended solids (TSS)

It is commonly believed that water transparency frequently becomes a factor determining the depth at which plants grow, limiting their development (Chambers, Kalff 1985). Mean values of water transparency, measured by means of Secchi disc, in the entire Lagoon are approximate (39-50 cm; Table 2.2.1.2). Relatively the highest values were recorded in the vicinity of Braniewo (central-southern part of the Lagoon) and in Łaszki (eastern part). In this case, a feedback effect occurs – the submerged vegetation abundant in these areas could be the cause of increased water transparency. On the one hand, more abundant macrophyte assemblages restrict the development of phytoplankton. On the other hand, they decrease the intensity of wave action, reducing the resuspension phenomenon (Carpenter, Lodge 1986). In consequence, both the concentrations of chlorophyll-a and TSS in those two areas were among the lowest (Table 2.2.1.2) in spite of increased concentrations of nutrients.

The mean water transparency values recorded in the years 2007-2009 (43 cm) were somewhat higher than those occurring in the 1950's (40 cm – Łomniewski 1958; 30-40 cm – Ringer 1959), and identical to those determined in the years 1998-1999 (Renk et al. 2001). Chlorophyll-a concentrations were slightly lower (36.1 µg/L) than ten years before (43 µg/L;

Renk et al. 2001). This is at variance with the theses on still maintained trophic status of the Lagoon substantially higher than that in the second half of the previous century (among others Pliński 1995).

This conflict is only apparent, however, considering the fact of the intensive resuspension phenomenon occurring periodically in the Lagoon as a result of strong wave action. Detritus particles swept from the bottom by wave action contribute to the suspension, and then, after the wave action ceases, they fall back to the bottom within approximately 1-2 days (Łomniewski 1958). Therefore, in contrast to phytoplankton (and the remaining fraction of bioseston), they contribute to a decrease in water transparency only periodically, affecting the amplitude of changes. The amplitude, determined for 12 stations in the years 1950-1954, varied from 60 cm to 135 cm, and the maximum values were between 75 cm and 170 cm (Łomniewski 1958). It currently varies from 45 to a maximum of 90 cm, with the maximum values from 75 to 130 cm (Table 2.2.1.2). Assuming that resuspension occurred with similar intensity in both of the compared periods, it becomes obvious that in spite of similar mean water transparency values included in the literature (and presented data), more than fifty years ago the development of phytoplankton affected their values to a much lower degree. The currently strongly developed phytoplankton “hides” the content of suspension resulting from resuspension, and by remaining in the water depths, it reduces the water transparency amplitude.

A higher amplitude of visibility changes in the past, i.e. periodical improvement of light climate conditions, could result in better conditions of development for macrophytes. It is difficult to determine to what degree their deterioration influenced the restriction of the horizontal and vertical range of occurrence of plants in the Lagoon. It seems that this factor is of no crucial importance, at least not affecting directly. Currently, e.g. in the Kały Bay (north-western part of the Lagoon) at the mean water visibility of 50 cm, the range of occurrence of macrophytes (*Chara*) amounts to as much as 160 cm. They do not occur in the central or eastern part of the Lagoon in spite of similar light conditions. Similar conclusions referring to the effect of light climate on the distribution of macrophytes in lakes were derived by Scheffer et al. (1992). According to the authors, “the depth range of the vegetation was hardly related to transparency. An increase in turbidity results in a strong reduction of the vegetated area, but this is due to the disappearance of vegetation not only from the deep, but also from the shallow sites”. The latest literature suggest that periphyton might the main factor restricting

the development of macrophytes. Its abundance on the surface of macrophytes depends on the concentration of nutrients and periphyton grazing invertebrates (Sayer et al. 2010). The mechanism of negative effect of periphyton involves cutting off the access of macrophytes to light, and limiting their access to nutrients. The importance of allelopathy in the mutual relations of both components is also emphasised (Gross et al. 2007).

Bottom sediments

In the littoral zone of the Polish part of the main basin of the Lagoon, sediments are relatively homogenous. They are dominated by sandy fractions, with local admixtures of fine clayey fractions. The current scarcity of more detailed data on the chemical and grain size composition of sediments in relation to the distribution of macrophytes in the Lagoon does not permit a more thorough analysis of these relations. Such research in four brackish lagoons in the southern Baltic Sea was conducted by Selig et al. (2007). According to this study, Charophyte stands favour sediments with smaller mean grain size and higher organic matter and nitrogen contents than other plant communities. *Myriophyllum spicatum*-*Potamogeton pectinatus* communities – indicators of high degradation in the classification system of coastal waters – were present in a broad spectrum of sediment types, i.e. were not more abundant in habitats with nutrient-rich sediments and muddy sediments. The study by Lehmann et al. (1997) suggests a general negative effect of organic matter content on macrophytes (except for *P. pectinatus*), and a positive effect of fine texture and phosphorus. *P. pectinatus* grew better in shallower water. At deeper sites, *P. lucens* appeared to show the best competitive abilities, and *P. perfoliatus* performed best.

Wave action

Wave action is a factor with a considerable importance to the distribution of macrophytes and their species composition in the Lagoon. The shores of the Lagoon, exposed to strong wave action, are primarily inhabited by resistant to waves helophytes, especially *Schoenoplectus lacustris* and elodeids - *Potamogeton perfoliatus* and *P. pectinatus*. Their structure allows them to persist at disturbed sites, where the floating leaves could be broken. Quiet bays and places sheltered with belts of helophytes are colonised by patches of elodeids (*Myriophyllum spicatum*, *Ceratophyllum demersum*) and nymphaeids (*Nuphar lutea*, *Nymphoides candida*).

Ice cover

The mean period of presence of ice cover in the Vistula Lagoon varies from approximately 2.5 to 4 months, and the thickness of ice cover from approximately 30 to 50 cm (Łomniewski 1958). In such conditions, and additionally under a thick snow cover (> 10 cm), the intensity of photosynthesis can be strongly reduced. The thinnest ice cover persisting for the shortest time occurs in the estuarial zones of rivers (Łomniewski 1958), and particularly in the western part of the Lagoon, where the richness of macrophytes is the highest. Therefore, ice cover seems to be a significant factor affecting the distribution of plants.

2.2.2. The status of macrophytes in the lagoon: Regression events and their causes

Research conducted over more than five decades in the Vistula Lagoon (Szarejko 1955, Ringer 1959, Pliński et al. 1978, Pliński 1995) permits the determination of certain trends in changes occurring in macrophyte assemblages (Pliński 1995). During that time, in the majority of the area of the Lagoon, the surface area occupied by helophyte assemblages developed by *Phragmites australis* and *Schoenoplectus lacustris* did not change. The most evident changes occurred in elodeid and nymphaeid assemblages. In the years 1975-1994, the area in the northern part of the Lagoon occupied by them decreased from 277 to 160 ha (Pliński 1995). In the years 1950-1970, certain plants, e.g. *Stratiotes aloides* and *Nymphoides peltata*, disappeared completely. Currently, *Stratiotes aloides* is still not recorded in the Lagoon. *Nymphoides peltata* assemblages are gradually reconstructed. The primary cause of the unfavourable changes occurring in plant assemblages is considered to be the Lagoon's trophic status which increased particularly in the 1980's (Pliński 1995).

2.2.3. Macrophyte vulnerability in the context of climate change

In the future, vegetation changes are likely to occur as a result of increased flooding and salinity associated with global climate change and sea level rise. Macrophytes, typical of fresh/brackish waters (*Nuphar lutea*, *Nymphoides candida*), can be the most prone to such changes. On the other hand, their current distribution is limited to estuarial areas of rivers, providing inflow of freshwater. Considering the strongly elongated shape of the Lagoon and substantial distance of the current sites of occurrence of those plants from the Strait of

Baltiysk, the progressing climate changes do not seem to be able to considerably affect macrophytes through a change in salinity.

The remaining macrophyte species of the Lagoon are distinguished by considerable resistance to changes not only in salinity, but also other abiotic conditions. Moreover, plants can adapt their phenology and morphology to their thermal and salinity response over broad environmental gradients (Santamaria 2002, Santamaria et al. 2003, Achenbach 2013).

An increase in water level can result in the migration of macrophytes to shallower areas, including freshly flooded ones, and their retreat from regions below the compensation point.

Increased water temperature resulting from climate changes does not seem to be a potential limiting factor for the species occurring in the Lagoon. Their geographic distribution already currently shows very wide tolerance to temperature. Molecular-based studies can be helpful in forecasting the response of macrophytes to climate changes, including thermotolerance (e.g. Amano et al. 2011).

A gradual increase in temperature can favour further cyanobacterial blooms, even stronger than those occurring currently, including species producing toxic microcystins (Rybicka 2005). Their increasing concentration can lead to the successive impoverishment of the species composition of macrophytes (Sayer et al. 2010).

2.3. The Mar Menor lagoon

2.3.1. Macrophyte species in the lagoon and their relationship with environmental variables

At the present time, two main macrophyte species are present in the Mar Menor lagoon. The macroalga *Caulerpa prolifera* covers approximately 90% of the lagoon bottoms as a dense monospecific bed on muddy sediments. *Cymodocea nodosa*, the traditional phanerogam that once dominated in the lagoon is now restricted to small patches in sandy shallow areas.

Other macrophytes can also be found in the lagoon, such as, photophyllic algae growing on shallow rocky bottoms, mainly around the islands within the lagoon, and *Ruppia cirrhosa*, a phanerogam that is present in areas influenced by freshwater inputs.

This study will only focus on the two main macrophyte species in the lagoon, *Caulerpa prolifera* and *Cymodocea nodosa* (Figure 2.3.1.1).

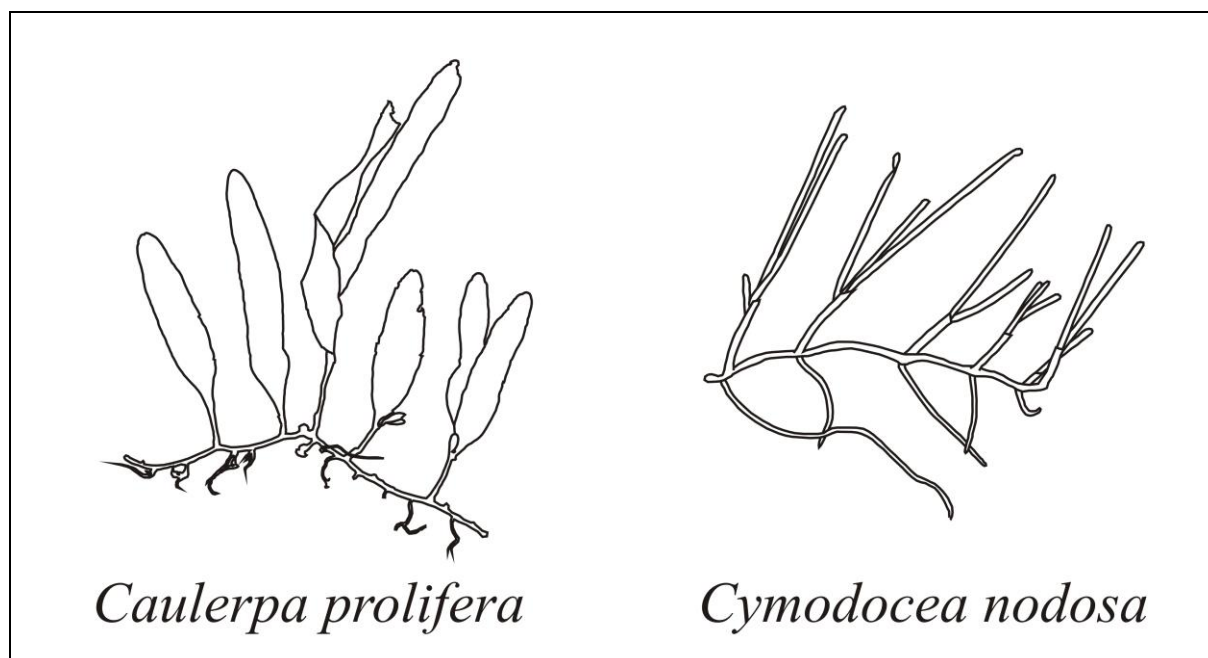


Figure 2.3.1.1. A schematic representation of the talli of the main macrophyte species *Caulerpa prolifera* (Forsskal) Lamouroux and *Cymodocea nodosa* (Ucria) Ascherson inhabiting the bottoms of the Mar Menor lagoon.

Salinity

Due to the restricted water exchange with the adjacent Mediterranean Sea and the high evaporation rates salinities in the Mar Menor lagoon are high and ranges from 41 to 47. Both species of macrophytes seem to be adapted to this range of salinities.

Terrados (1991) carried out some experiments to test the salinity limits for *C. prolifera* in the Mar Menor lagoon. *C. prolifera* showed negative growth at salities above 50. At this salinity value, frond degradation increases and at salities of 60 fronds start dieing rapidly. In addition salinities below 30 cause apical degradation of fronds.

Temperature

Water temperatures in the lagoon oscillate between 10° C in the winter and 31° C during the summer. At this temperature range both species of macrophytes are able to grow and maintain their biomass in the lagoon.

Experiments carried out by Terrados (1991) showed that *C. prolifera* is quite sensitive to both extremes of this temperature range. At temperatures below 10° C *C. prolifera* is unable to maintain a positive carbon balance and starts dieing. The same way, at temperatures above 30° C, *C. prolifera* respiration exceeds photosynthesis and at temperatures of 35° C the plant dies.

C. nodosa does not seem to be limited at these temperatures and still maintains a positive carbon balance, leaf and rizhome groth at 10° C and at 35° C (Terrados, 1991).

Bottom type

Although both species of macrophytes can be found growing on sand and mud in the Mar Menor lagoon, *C. nodosa* displays a certain preference for sandy shallow bottoms and *C. prolifera* occupies the deeper muddy bottoms of the lagoon.

Light

According to Terrados (1991) and Lloret et al. (2008), light seems to be one of the most important limiting factors for the growth and photosynthesis of *C. prolifera* and *C. nodosa*.

C. prolifera is adapted to lower irradiances, and reaches its maximum photosynthetic rates at irradiances around 20-50 $\mu\text{mol photon m}^{-2} \text{s}^{-1}$ PAR, allowing its growth in the deepest parts of the lagoon. In contrast, *C. nodosa* requires irradiances of about 70-380 $\mu\text{mol photon m}^{-2} \text{s}^{-1}$ PAR, a fact that, under current light conditions in the area, limits its growth to the shallow bottoms of the lagoon. Furthermore, *C. prolifera* displays photoinhibition at high irradiances, a fact that explains its limited growth in those shallow areas of the lagoon where *C. nodosa* is present.

Nutrients

Experiments carried out by Terrados (1991) showed that *C. prolifera* does not seem to be limited by nutrient availability in the Mar Menor. Experiments showed that additions of nitrogen slightly stimulated *C. prolifera* growth.

C. nodosa seems to be limited by nutrient availability in the Mar Menor lagoon. Nitrogen and phosphorus additions stimulated leaf and rhizome growth and increased the ratio photosynthesis/biomass.

The following tables summarize the above information about the influence of the different environmental variables on macrophytes in the Mar Menor:

Table 2.3.1.1. Influence of environmental variables on *C. prolifera* distribution, growth and photosynthesis

Variable	Range in the lagoon	Optimum	Critical	Source
Salinity	41-47	37-47	30 (lower) 50 (upper)	Terrados (1991)
Temperature	10-31° C	20-25° C	10° C (lower) 30° C (upper)	Terrados (1991) Lloret et al. (2008)
Bottom type	Sandy Muddy Rocky	Muddy	Not applicable	Terrados (1991)
Light	Not applicable	20-50 $\mu\text{mol m}^{-2} \text{s}^{-1}$	10 $\mu\text{mol m}^{-2} \text{s}^{-1}$ (lower) 300 $\mu\text{mol m}^{-2} \text{s}^{-1}$ (upper)	Terrados (1991) Lloret et al. (2008)
Nutrients	DIN: 1.51-8.91 $\mu\text{mol l}^{-1}$ SRP: 0.00-0.06 $\mu\text{mol l}^{-1}$	No data	Not limited by nutrients in the Mar Menor	Terrados (1991) Lloret et al. (2008)

Table 2.3.1.2. Influence of environmental variables on *C. nodosa* distribution, growth and photosynthesis

Variable	Range in the lagoon	Optimum	Critical	Source
Salinity	41-47	37-47	No data	
Temperature	10-31° C	15-30° C	5° C (lower) 35° C (upper)	Terrados (1991) Lloret et al. (2008)
Bottom type	Sandy Muddy Rocky	Sandy	Not applicable	Terrados (1991)
Light	Not applicable	70-380 $\mu\text{mol m}^{-2} \text{s}^{-1}$	50 $\mu\text{mol m}^{-2} \text{s}^{-1}$ (lower)	Terrados (1991) Lloret et al. (2008)
Nutrients	DIN: 1.51-8.91 $\mu\text{mol l}^{-1}$ SRP: 0.00-0.06 $\mu\text{mol l}^{-1}$	No data	Limited by nutrients in the Mar Menor	Terrados (1991) Lloret et al. (2008)

2.3.2. The status of macrophytes in the lagoon. Regression events and their causes

Historically, the principal benthic macrophyte in the Mar Menor lagoon was the phanerogam *Cymodocea nodosa* (Ucria) Ascherson. In the early 1970s, one of the channels that communicate the lagoon with the adjacent Mediterranean Sea (El Estacio) was dredged and widened to make it navigable. Since then, it has become the lagoon's main connection with the sea. The enlargement of El Estacio channel led to a substantial increase of water renewal rates from the Mediterranean, as well as subsequent changes in water temperatures and salinities. These changes favoured the colonisation of the lagoon by numerous marine species as lagoonal temperatures and salinities reached less extreme values (Pérez-Ruzafa et al., 1991).

One of the main events in relation to this 'Mediterraneanisation' process was probably the colonisation of the lagoon by the invasive alga *Caulerpa prolifera* (Forsskal) Lamouroux. During the early 1980s, only a few years after the enlargement of El Estacio channel, the bottoms were covered by a mixed meadow of *C. prolifera* and *C. nodosa*.

Around the same time, water derived from the Tajo-Segura river diversion, generated a profound transformation of the agricultural practises in the adjacent agricultural area, Campo de Cartagena, that changed from extensive dry crop farming of cereals, olives, almonds and carob beans to intensively irrigated crops. Due to increased agricultural water usage and decreased groundwater exploitation, phreatic levels have risen. As a result, some watercourses, such as El Albuñón wadi, now maintain a regular flux that is fed by ground water with high nitrate levels. As a consequence of increased inputs, the waters of the Mar Menor have experienced rising nutrient levels that have led to planktonic changes in the lagoon (Gilabert, 2001, Pérez-Ruzafa et al., 2005). These changes have also favoured the proliferation of the jellyfish species *Cotylorhiza tuberculata* and *Rhizostoma pulmo*, with severe consequences for touristic activities in the area (Pérez-Ruzafa et al., 2002). Furthermore, modified light conditions of the lagoon waters might have favoured the expansion of *C. prolifera* on the bottoms of the lagoon and the confinement of the traditional phanerogam *C. nodosa* to shallow areas.

At the present time, a dense monospecific bed of *C. prolifera* covers most of the bottom of the lagoon, and the distribution of *C. nodosa* is restricted to very small patches in the shallowest areas (Figure 2.3.2.1).

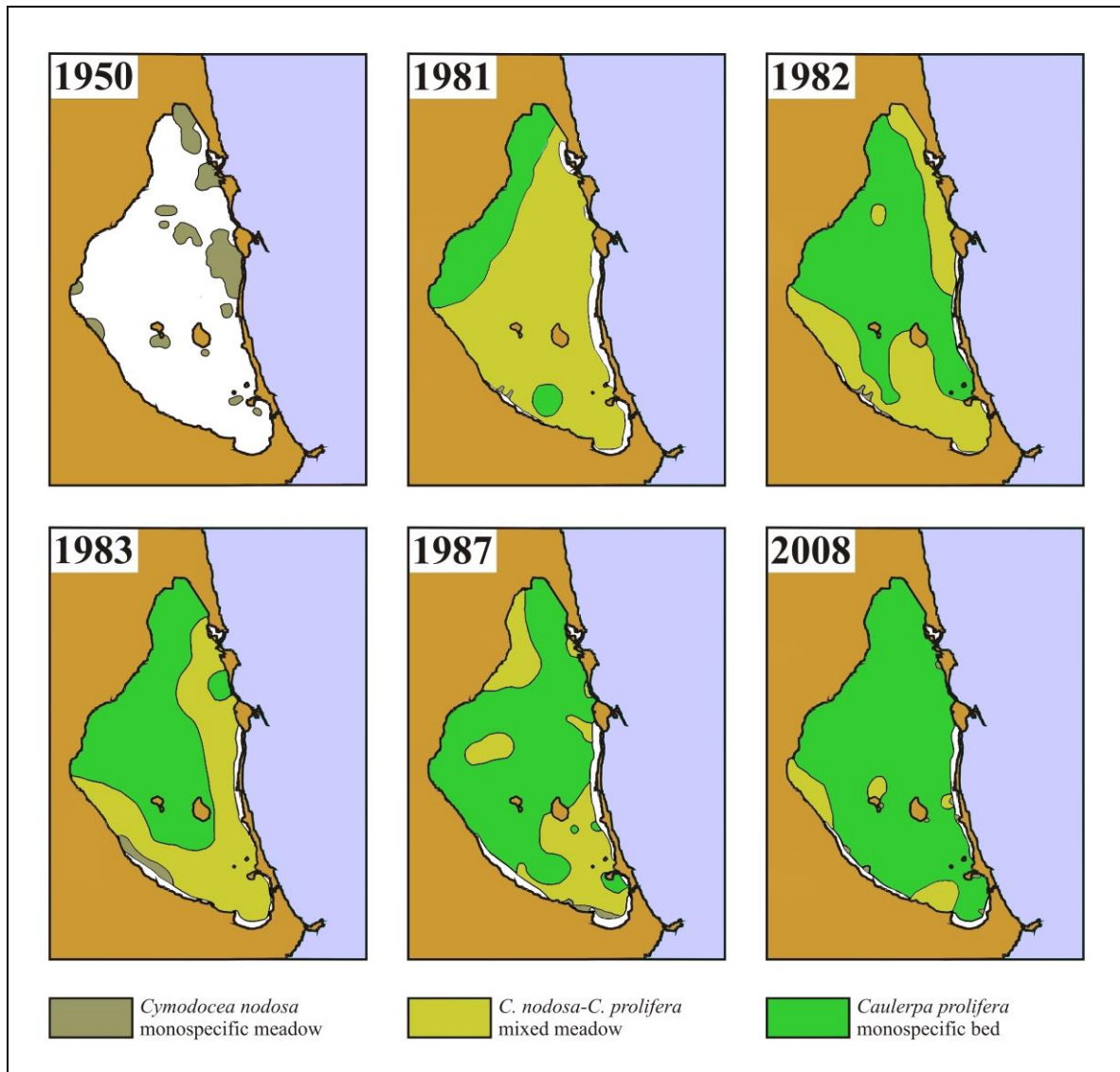


Figure 2.3.2.1. Evolution of the coverage of the bottom by the main macrophytes in the Mar Menor lagoon (Redrawn and adapted from Lozano, 1954 and Pérez-Ruzafa et al., 2009).

2.3.3. Macrophyte vulnerability in the context of climate change

In the Mar Menor lagoon *C. prolifera* maintains a positive carbon balance throughout the year. Current environmental conditions allow algal growth and photosynthesis in a wide depth range of the lagoon (Figure 2.3.3.1) and its development and distribution is probably only limited by photoinhibition, emersion stress and extreme temperatures in the shallowest areas where *C. nodosa* is able to grow (Terrados and Ros, 1995; Lloret et al., 2005).

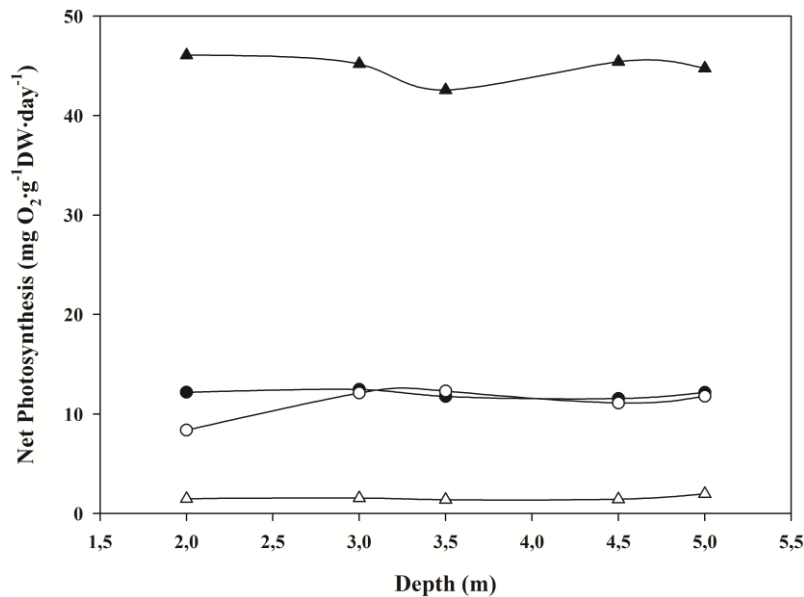


Figure 2.3.3.1. Net Photosynthesis of *C. prolifera* at different depths in spring (black circles) summer (black triangles), autumn (white circles) and winter (white triangles).

As in other coastal zones, a decrease in the amount of light reaching the bottom of the Mar Menor lagoon is to be expected as a consequence of global climate change. This decrease exists as a result of the combined effect of the rise in the sea-level and the decrease in water column transparency caused by the increase in nutrient inputs and suspended solids runoff (Nicholls and Hoozemans, 1996; Short and Neckles, 1999).

Lloret et al. (2008) pointed to a certain degree of vulnerability of the benthic communities of the Mar Menor lagoon to eutrophication processes in the next decades as a consequence of the expected effects of global climate change. The critical values obtained for water column light attenuation have already been reached in other Mediterranean lagoons with similar nutrient inputs (De Casabianca et al., 1997; Sfriso et al., 2003). Furthermore, nutrient concentrations have been increasing in the last decades in the Mar Menor lagoon and

are expected to continue increasing (Pérez-Ruzafa et al., 1991, 2002, 2005; Lloret et al., 2005; Velasco et al., 2006). The relative oligotrophic conditions of the lagoon are expected to change and eutrophication processes may appear. The proliferation of jellyfish observed in recent years in the lagoon constitutes an example of the consequences of alteration in the trophic status of the system and could be an indicator of the instability of the lagoon parameters (Pérez-Ruzafa et al., 2002). The loss of benthic macrophytes and the appearance of eutrophication processes could drive to a significant decrease of habitat quality in the lagoon with unexpected consequences to the biological diversity of its communities.

The highest densities of phytoplankton in the Mar Menor lagoon are found in summer, and these cause an increase in light attenuation. Summer photosynthesis seems to be the most fragile and vulnerable to the expected climate change effects. The coupled rise of water column light attenuation and temperature could have a deleterious impact on *C. prolifera*. Critical values of water temperature could easily be reached if predictions become true. A rise in air temperature is rapidly reflected in a rise of water temperatures in semi-enclosed water bodies. In the Mar Menor lagoon, there is a marked linear correlation between both variables, and the slope of the fit (close to 1) indicates that the expected increase of 3-5 °C in atmospheric temperature in the warmest months will probably cause a similar increase in the water temperature due to the high surface/volume ratio of this water body and the low water exchange with the adjacent Mediterranean Sea. A rise in sea water temperature to above 30 °C causes a decrease in *C. prolifera* photosynthesis and, therefore, critical values are expected to be reached more frequently (Figure 2.3.3.2). As Terrados and Ros (1991) previously hypothesized, summer seems to be a critical time of year for the functioning of the lagoon. The highest phytoplankton populations are found in this season, therefore, light extinction in the water column increases. Any increase in temperatures as a consequence of rising mean atmospheric temperatures with climate change in this season may aggravate the situation since, if critical values are reached, *C. prolifera* photosynthesis will be affected and, if this situation is maintained, extensive masses of the algae could die.

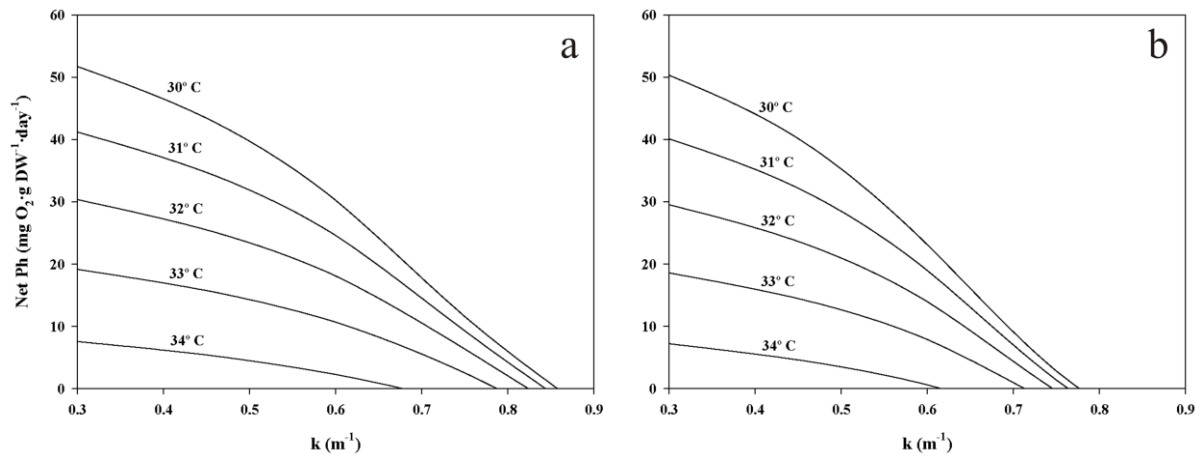


Figure 2.3.3.2. Influence of the coupled effect of water temperature and light extinction coefficient on *C. prolifera* net photosynthesis at 5 m depth in summer at (a) current sea-level and (b) future sea-level rise of 0.5 m.

Decreased *C. prolifera* photosynthesis (or its death) in summer and the consequent decrease in its nutrient uptake will probably originate phytoplankton proliferation in the lagoon since higher nutrient concentrations would become available. As a result, critical values of light attenuation could easily be reached the following season and *C. prolifera* would be unable to recover after the impact. Any increase in light attenuation does not need to be high to affect *C. prolifera* in the lagoon. In addition, most of the macroalgae biomass is found below 5-m depth, where even slight increases in light attenuation cause high decreases of light reaching the bottom. The loss of these deeper masses of *C. prolifera* represents approximately 54% of the total estimated biomass and, therefore, the nutrient uptake of *C. prolifera* in the lagoon would be profoundly affected. A planktonic control of the system could then easily appear since a rise in phytoplankton densities in the water column would block the development of benthic macrophytes in the lagoon.

Furthermore, an increase in light attenuation due to the loss of *C. prolifera* will be aggravated by the release of nutrients from the organically enriched sediments and the resuspension of solids of unvegetated bottom, which would now be more exposed to erosion. In addition, dead masses of the algae would increase the availability of nutrients in the system as the biomass decomposes and releases its nitrogen and phosphorus compounds.

If climate change predictions become true, the current status of the Mar Menor lagoon is advocated to collapse, resulting in a profound deterioration of the whole ecosystem through the appearance of eutrophication processes with higher nutrient concentrations, the proliferation of phytoplankton and floating macroalgae or even more serious phenomena, such as hypoxia, affecting not only the biological communities but also the main local economic activities, fisheries and tourism.

2.4. The Tyligulskyi lagoon

2.4.1. Macrophyte species in the lagoon and their relationship with environmental variables

Among the environmental factors that exert significant impact on the macrophyte species composition in various parts of the Tyligulskyi Liman lagoon it is the water level in the lagoon, water salinity and temperature and oxygen regime, which is closely related to wind activity, that could be highlighted.

In view of a deficiency of freshwater balance, being negative on account of intense evaporation from the water surface of the lagoon in the summer period, it is under the absence of water exchange with the sea through the connecting canal that the water level in the lagoon may decrease 0,6-1,0 m throughout a number of years, as it was, e.g. in 2006 - 2009. This leads to draining of the traditional macrophyte habitats on the stony and the sandy-and-slimy ground in the coastal zone in relatively narrow shallow water coastal belt of the deep southern and central parts of the lagoon.

Salinity of the lagoonal waters has the strongest influence on the macrophyte species composition. The southern, deeper half of the lagoon, where the sea water with the salinity of 11-16 ‰ inflows during operation of the connecting canal, is characterized by the most stable water salinity ranging 15 - 22 ‰ and it is here that the greatest species diversity of the macrophytes is observed. In the northern half of the lagoon, depending on the availability and intensity of the Tiligul river runoff, the water salinity during the annual cycle can vary from a few ‰ (during the spring flood or the high water caused by intense spring and summer rains) to 24 ‰ in August and September (under the lack of the Tiligul river runoff due to intensive evaporation). As a result, macrophyte species variety in the northern part of the lagoon is almost 2 times narrower than in the southern one. Salt water and brackish-salt water species

of macrophytobenthos (100% and 82.8%, respectively) are prevalent in the southern part of the lagoon, while in the northern part the role of fresh-brackish water species increases. In the years when the surface runoff of fresh waters into the lagoon increases (for example, under an intense spring flood) the proportion of brackish water and salt water species, as well as the overall species variety, decreases (see LAGOONS Report D3.2.3, Kovtun, 2012). Classification of macrophytes observed in the Tyligulskyi Liman lagoon, according to their attitude towards salinity is shown in the Table 2.4.1.1. According to Mitvalli (1970), in the case of *Cystoseirabarbata*, the brown algae, which is extinct in the north-western part of the Black Sea, but still grows in the southern part of the Tyligulskyi Liman lagoon, the salinity range of 15-25 ‰ is optimum for the growth and the lower salinity limit comprises 5 ‰.

In the summer period, the water in the shallow northern part, as well as in the shallow areas of the coastal zone in the central part of the lagoon are heated to the temperatures exceeding the optimum values for growth of the macrophytes (28 °C, according to Fu-Liu Xu et al., 1999). The maximum values of water temperature (30-34 °C) were registered in the late July - early August. The high water temperature in the shallow water zone contributes to intensive biochemical oxidation of the dead remains of algae organic matter and in the periods of long calms, which often occur in the summer, this leads to emergence of anoxic zones and death of hydrobionts in the shallow coastal zone, as, e.g., in the northern part of the lagoon in June 2013.

In a cold winter the lagoon may be covered with ice for a period of 1 - 2 months. The thickness of the ice cover may be of up to 0.5 m. This causes winterkilling of the macrophyte associations in the shallow coastal zone and change in the dominant species within the association in the subsequent spring and summer period (2011). The minimum water temperature, under which growth of the macrophytes secured in winter, is 5 °C (Fu-Liu Xu et al., 1999).

Wind conditions affect the species composition and spatial distribution of the macrophytes in the lagoon in three ways: 1) provide vertical oxygen ventilation of the lagoonal waters; 2) cause resuspension of sediments during storms leading to reduction in the water transparency and creating favorable conditions for assimilation of nutrients by the macrophytes thus contributing to water mixing in the dense thickets of aquatic plants reaching the water surface; 3) initiate intense wind-wave alongshore currents under the northerly and

the southerly winds prevailing in the area, which may cause detachment of algae from the substrate and prevent their re-settlement. Various species of algae resist the hydrodynamic factor to varying degrees. For example, *S. barbata* is well adapted to sustaining conditions of strong wind and wave impact. The combined influence of the type of bottom sediment and hydrodynamic conditions cause high-level mosaic in the spatial distribution of algae and availability of sites, which are almost completely devoid of attached algae, in many places of the shallow water zone of the lagoon on the silty-and-sandy grounds and numerous spits (Kovtun, 2012).

Table 2.4.1.1. Ecological-and-geographical analysis of the macrophyte algae and higher aquatic vegetation in the Tyligulskyi Liman lagoon (Kovtun, 2012)

№	Taxon	Towards		Occurrence			
		Salinity	Contamination	In the study period (seasonal)			
				Autumn	Winter	Spring	Summer
CHLOROPHYTA							
ULVOPHYCEAE							
ULOTRICHALESBohl.							
1	Enteromorphaclathrata (Roth) Grev.	B-S	m	+	+	+	+
2	E.compressa (L.) Grev	B-S	m	-	-	+	-
3	E. flexuosa (Wulf.)J.Ag.	B-S	m	+	-	-	+
4	E. intestinalis (L.) Link	B	p	+	+	+	+
5	E. maeotica Pr. –Lavr.	B-S	-	+	-	+	+
6	EntocladiaviridisReinke	S	o	+	-	+	+
7	Gomontia polyrrhiza (Lagerh.) Born. et Flah	B-S	m	+	-	-	+
8	Percursaria percura (C. Ag.) Bory	S	-	-	-	-	+
9	Pringsheimiella scutata (Reinke) Schmidt	S	-	+	-	-	+
10	Stigeoclonium tenue (Ag.) Kutz.	S	-	-	-	+	-
11	Ulothrix flacca (Dillw.)Thur.	B-S	m	-	-	+	+
12	U. implexa Kutz	B-S	p	+	+	-	-

13	<i>U. limnetica</i> Lemm.	F-B	-	+	+	-	-
14	<i>U. pseudoflacca</i> Wille	B-S	m	-	-	+	-
15	<i>U. tenerrima</i> Kutz.	F-B	m	+	-	+	+
16	<i>U. tenuissima</i> Kutz.	F-B	m	+	-	+	+
17	<i>Ulvarigida</i> Ag.	B-S	m	-	-	+	+
CLADOPHORALES Fritsch							
18	<i>Chaetomorpha</i> aerea (Dillw.) Kutz	B-S	o	+	+	+	+
19	<i>Ch. chlorotica</i> (Mont.) Kutz.	B-S	m	+	-	+	+
20	<i>Ch. linum</i> (O. Mull.) Kutz	S	o	+	-	+	+
21	<i>Cladophora</i> albidula (Huds.) Kutz.	B-S	m	+	-	+	+
22	<i>C. glomerata</i> (L.) Kutz.	F-B	m	+	-	+	+
23	<i>C. siwaschensis</i> C. Meyer	B-S	m	-	-	+	+
24	<i>C. sericea</i> (Huds.) Kutz.	B-S	m	+	-	+	+
25	<i>C. vadorum</i> (Aresch.) Kutz.	S	-	+	-	+	+
26	<i>C. vagabunda</i> (L.) Van Hoek (= <i>C. fracta</i> f. <i>marina</i> Hauck.)	B-S	p	+	+	+	+
27	<i>Rhizoclonium implexum</i> (Dillw.) Kutz.	B	p	+	-	+	+
	<i>R. tortuosum</i> (Dillw.)	S	-	-	-	+	+
28	SIPHONALES (Endl.) Blackm. et Tansi						
29	<i>Bryopsis</i> hypnoides Lamour.	B-S	m	+	-	+	+
30	<i>B. plumose</i> (Huds.) Ag.	B-S	m	+	+	+	+
	ZYGNEMATOPHYCEAE ZIGNEMATALES Krieg.						
31	<i>Spirogyra</i> decimina (Mull.) Kutz.	B-S	-	-	-	+	+
32	<i>S. hassalli</i> (Jenner) Petit	B-S	-	-	-	+	+
33	<i>S. subsalina</i> Kutz.	F-B	m	-	-	+	-
	CHAROPHYCEAE CHARALES Dumor.						
34	<i>Chara</i> canescens Desv. et Lois. (= <i>C. crinita</i> Wallr.)	B	-	-	-	-	+
XANTHOPHYTA							
	XANTHOPHYCEAE VAUCHERIALES Bohl.						
35	<i>Vaucheria</i> dichotoma (L.) C. Ag. - f. <i>submarina</i> Lyngb	F-B	m	+	+	+	+
36	<i>V. geminata</i> (Vauch.) DC (= <i>V. hamata</i> Walz)	B-S	m	+	-	+	+

37	V. litorea Hofm. et C. Ag. in C. Ag.	S	m	+	+	+	-
RHODOPHYTA							
BANGIOPHYCEAE CONIOTRYCHALES Skuja							
38	Goniotrichum elegans (Chauv.) Zanard.	S	m	+	-	-	+
39	Chroodactylon ramosum (Thwait.) Hansg. (=Asterocystis ramosa (Thwait) Gobi.)	B-S	p	+	-	-	+
40	Ch. wolleanum Hansg. (=Asterocystis wolleana (Hansg.) Lagerh.)	F-B	m	+	-	-	+
BANGIALES Schmitz							
41	Bangia atropurpurea (Roth) C. Ag. (=B. fuscopurpurea (Dillw.) Lyngb.)	F-B	p	-	+	-	-
42	Erythrocladia subintegra Rosenv.	S	o	+	-	-	+
43	Eritrotrichia carnea (Dillw.) J. Ag.	S	m	+	-	+	+
FLORIDEOPHYCEAE ACROCHAETIALES Garb.							
44	Colaconemasavianum (Menegh.) Perest. (=Acrochaetium thuretii (Born.) Coll. et Herv.)	S	m	+	+	-	+
45	Kyliniasecundata (Lyngb.) Papenf.	B-S	m	+	-	-	-
46	K. virgatula (Harv.) Papenf.	B-S		+	-	-	+
47	Rhodochorton purpureum (Lightf.) Rosenv.	S	o	+	-	+	+
CORRALLINALES Silva et Johansen							
48	Fosliella farinosa (Lamour.) Howe. (=Mel Obesia farinosa Lamour.)	S	o	+	-	-	-
CERAMIALES Olm.							
49	Callithamnion corymbosum (J. E. Smitt.) Lyngb.	S	p	+	-	+	+
50	C. granulatum (Ducl.) C. Ag.	S	o	+	-	-	+
51	Ceramium arborescens J. Ag.	S	o	+	+	+	-
52	C. diaphanum (Lightf.) Roth	S	p	+	+	+	+
53	C. elegans Ducl.	B-S	m	+	-	+	+
54	C. pedicellatum (Duby) J. Ag.	S	p	+	-	+	+

55	<i>C. rubrum</i> (Huds) J. Ag.	S	p	+	-	+	+
56	<i>C. strictum</i> Grev. et Harv.	B-S	m	+	-	+	+
57	<i>C. tenuissimum</i> (Lyngb.) J. Ag.	S	m	+	-	-	-
58	<i>Chondriatenuissima</i> (Good. et Wood.) C. Ag.	S	o	+	-	+	+
59	<i>Lophosiphonia obscura</i> (C. Ag.) Falkenb.	B-S	m	-	-	+	+
60	<i>Polysiphonia denudata</i> (Dillw.) Kutz. – f. <i>denudata</i>	F-B	m	+	+	+	+
61	- f. <i>fragilis</i> (Sperk) Woronich.	B	-	-	+	+	-
62	<i>P. elondata</i> (Huds.) Harv.	S	o	+	-	+	+
63	<i>P. opaca</i> (C. Ag.) Zanard.	S	m	+	+	+	-
64	<i>P. pulvinata</i> Kutz.	S	m	+	-	-	+
65	<i>P. subulifera</i> (C. Ag.) Harv.	S	o	+	+	+	-
PHAEOPHYTA							
PHAEOSPOROPHYCEAE							
ECTO CARPALES Oltn.							
66	<i>Ectocarpus confervoides</i> (Roth) Le Jolis	B-S	m	-	-	+	+
67	<i>E. siliculosus</i> (Dillw.) Lyngb.	B-S	m	-	-	+	+
68	<i>Pylaiella littoralis</i> (L.) Kjellm.	S	-	+	-	-	+
CHORDARIALES Setch. et Gardn							
69	<i>Cladosiphon contortus</i> (Thur.) Kylin	S	o	+	-	-	+
70	<i>Corynophlaea umbellata</i> (C. Ag.) Kutz.	S	o	-	-	-	+
71	<i>Leathesiadifformis</i> (L.) Aresch.	S	o	-	-	-	+
SCYTOSIPHONALES Feldm.							
72	<i>Scytosiphon lomentaria</i> (Lyngb.) J. Ag.	B-S	m	-	-	-	+
PUNCTARIALES Kylin							
73	<i>Desmotrichum undulatum</i> (J. Ag.) Reinke (= <i>Punctaria tenuissima</i> Kutz.)	S	m	-	+	-	-
74	<i>Punctaria latifolia</i> Grev.	S	o	-	-	+	+
75	<i>Striaria attenuata</i> (C. Ag.) Grev.	S	o	-	-	-	+
76	<i>Stictyosiphon adriaticus</i> Kutz	S	-	+	-	-	-
CYCLOSPOROPHYCEAE							
FUCALES Kylin							
77	<i>Cystoseira barbata</i> (Good. et Wood.) C. Ag.	S	o	-	-	+	+
MAGNOLIOHYTA							

78	<i>Ceratophyllum demersum</i> L.	-	o	+	-	-	+
79	<i>Myriophyllum spicatum</i> (L.)	-	o	+	-	+	+
80	<i>Phragmites australis</i> (Cav.) Trin ex Steud.	-	o	+	+	+	+
81	<i>Potamogeton pectinatus</i> L.	-	m	+	-	+	+
82	<i>Typha angustifolia</i> L.	-	o	+	-	-	+
83	<i>Ruppia spiralis</i> L.	-	-	+	+	+	+
84	<i>R. cirrhosa</i> (Petagna) Grande	-	-	+	+	+	+
85	<i>Zannichellia major</i> Boenn. ex Reichenb.	-	o	+	+	+	+
86	<i>Zosteranoltii</i> Hornem.	-	o	+	+	+	+
87	<i>Zostera marina</i> L.	-	-	+	+	+	+

Notation conventions: **HalobityCategory:** S – salt water; B – brackish water, B-S – brackish-salt water, F-B – fresh-brackish water species. **Saprobity Category:** o-oligosaprobites; m - mesosaprobites, p - polysaprobites.

Floristic composition of the macrophytobenthos is distributed between the saprobity groups as follows: 42% of all the known species in the lagoon are mesosaprobites, 22.1% - oligosaprobites, 13.0% - polysaprobites (Table 2.4.1.2). Comparison with the data on the Black Sea (Kalugina-Gutnik, 1975) (polysaprobites - 7.9%, mesosaprobites - 30.8%, oligosaprobites - 61.3%) indicates that the Tyligulskyi Liman lagoon as a whole has a high level of organic pollution, since the share of the polysaprobic species in it is almost 2 times higher, and the oligosaprobic – almost three times lower than in the Black Sea (Kovtun, 2012).

Classification of the macrophytobenthos species observed in the Tyligulskyi Liman lagoon by their attitude to water saprobity is given in the Table 1.

Table 2.4.1.2. Phytosaprobity composition of the macrophytobenthos in the Tyligulskyi Liman (Kovtun, 2012)

Groups	Divisions			Total
	<i>CHLOROPHYTA</i>	<i>RHODOPHYTA</i>	<i>PHAEOPHYTA</i>	
polysaprobity	4 (11,8 %)	6 (21,4 %)	- (0 %)	10(13,0%)
mesosaprobity	17 (50,0)	12 (42,9)	4 (33,3)	33 (42,9)
oligosaprobity	3 (8,8)	8 (28,6)	6 (50,0)	17 (22,1)
an unidentified	10 (29,4)	2 (7,2)	2 (16,7)	17 (22,1)

Total:	34 (100)	28 (100)	12 (100)	77 (100)
<i>Note:</i> in parentheses -% of the total number of taxones				

To further analyze the relationship between macroalgal community structure and environmental variables two sampling stations were selected from available benthic sampling in the area (Stations 2 and 5 in Figure 2.4.1.1). These sampling stations were selected since both environmental and macroalgal species matrices contained information for the period 2002-2006. Samples were taken at 0.5 m depth in June-July.



Figure 2.4.1.1. Location of sampling stations 2 and 5 selected to link environmental and macroalgal data.

MDS ordination of samples, based on square root transformed biomass and Bray-Curtis similarities, was carried out (Figure 2.4.1.2). A greater degree of variability in community structure of the station 2 was found. Another significant feature is the strong difference in the community composition between the years 2002 (station 3) and 2003 (station 5). The community structure is similar in 2006 for both stations.

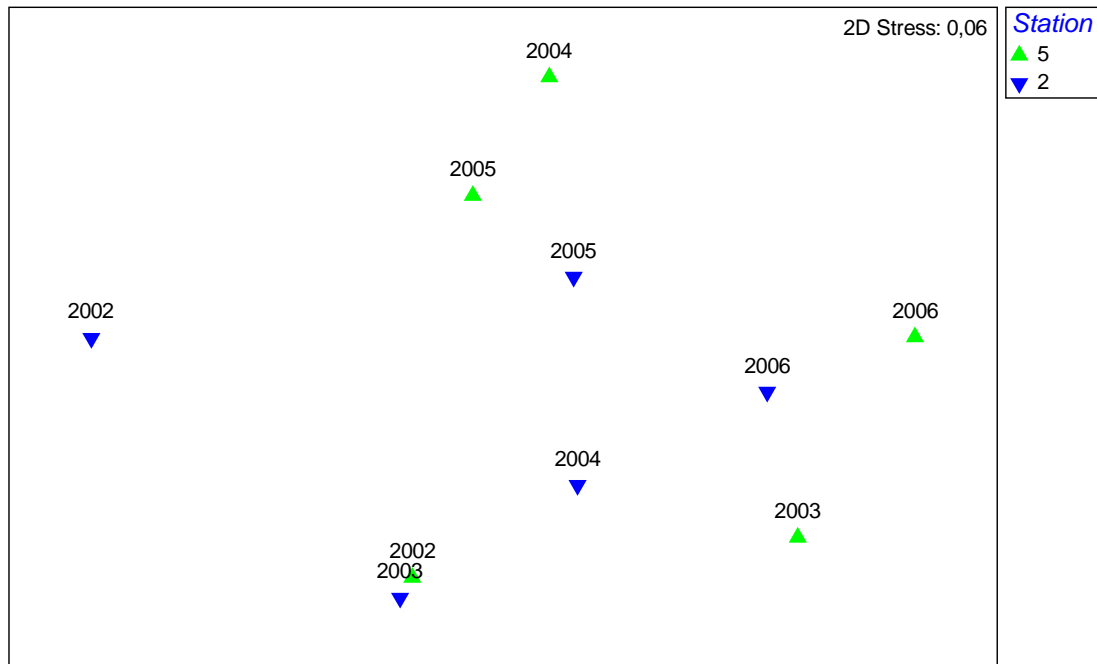


Figure 2.4.1.2. MDS ordination of sampling stations 2 and 5 based on fourth root of macroalgal biomass and Bray-Curtis similarities (stress=0.06).

To evaluate the effect of depth on algal community structure, samples were collected at different depths (0-0.5 m, 0.5-1 m, 1-2 m and 2-3 m) in the sampling station 5 (July 2011). PCO ordination was performed to the square root transformed biomass of macroalgae. The PCO ordination shows an environmental change of algae with the depth (Figure 2.4.1.3). The percentage of total variation inherent in the resemblance matrix that is explained by the first two axes is 98.8 %. PCO1 axis discriminated the gradual change of algal structure with the depth, while PCO2 axis separated deeper station (2-3 m).

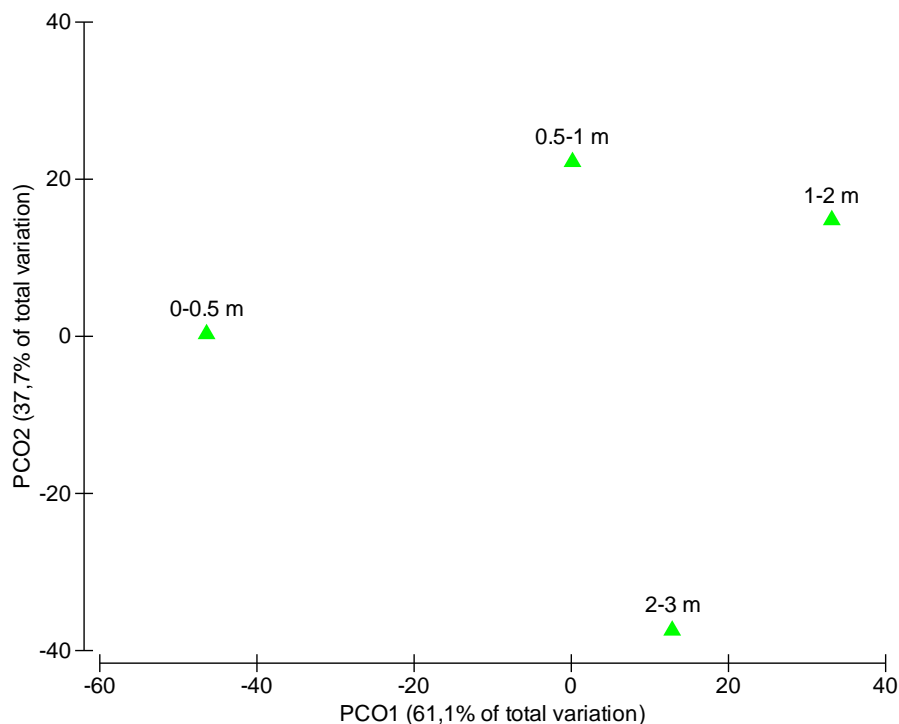


Figure 2.4.1.3. PCO ordination of macroalgal community at different depths (station 5).

PCA (Principal Components Analysis) was applied for the ordination of environmental data. The environmental variables which showed a great deal of skew were transformed with a log transformation (TOC, Pval, NO₂, NO₃, Nmin, Norg, Nval and SiO₃) or square root transformation (Salinity, Oxygen, NH₃, Porg and DOM).

PCA analysis showed that many of the environmental variables are strongly correlated with the first PCO axis, showing a gradual change in the sampling stations from 2002 to 2006. The year 2005 in the station 2 is clearly distinguished from the other years. There is lesser gradual change of the environmental pattern in the station 5. The PCA1, PCA2 and PCA3 explain 75.6 % of the total variability. The environmental variables with higher eigenvector values (>300) were LogToC, Sqrt POrg, Log Pval, Log NO₂ and Log Nmin for PCA1, and Sqrt Oxygen, PO₄, Sqrt NH₃ and Log NO₃ for PCA2 (Figure 2.4.1.4).

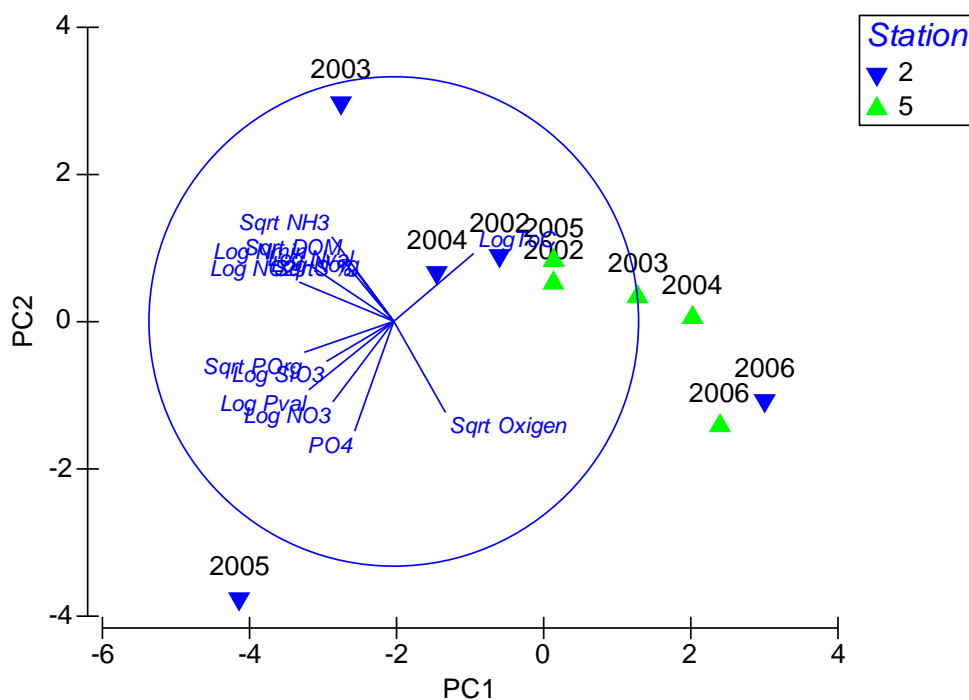


Figure 2.4.1.4. PCA ordination of the environmental variables on the basis of a Euclidean matrix.

DISTLM analysis was used to identify potential models for macroalgae in Tyligulskyi lagoon in response to several measured environmental variables. In the sequential test of DISTLM analysis, only the variables SqrtS‰ and SqrtDOM had the higher significant relationship with the macroalgae multivariate matrix. When the “Best” selection procedure was used, the best model solutions contained 8 variables (LogToC, SqrtS ‰, SqrtOxygen, Sqrt POrg, LogPval, SqrtNH3, LogNmin and SqrtDOM). The R^2 of the 8 variables explained 97.5% of variation for the model. However, in the marginal test only 3 variables were statistically significant (SqrtS ‰, SqrtOxygen and SqrtDOM) (Figure 2.4.1.5).

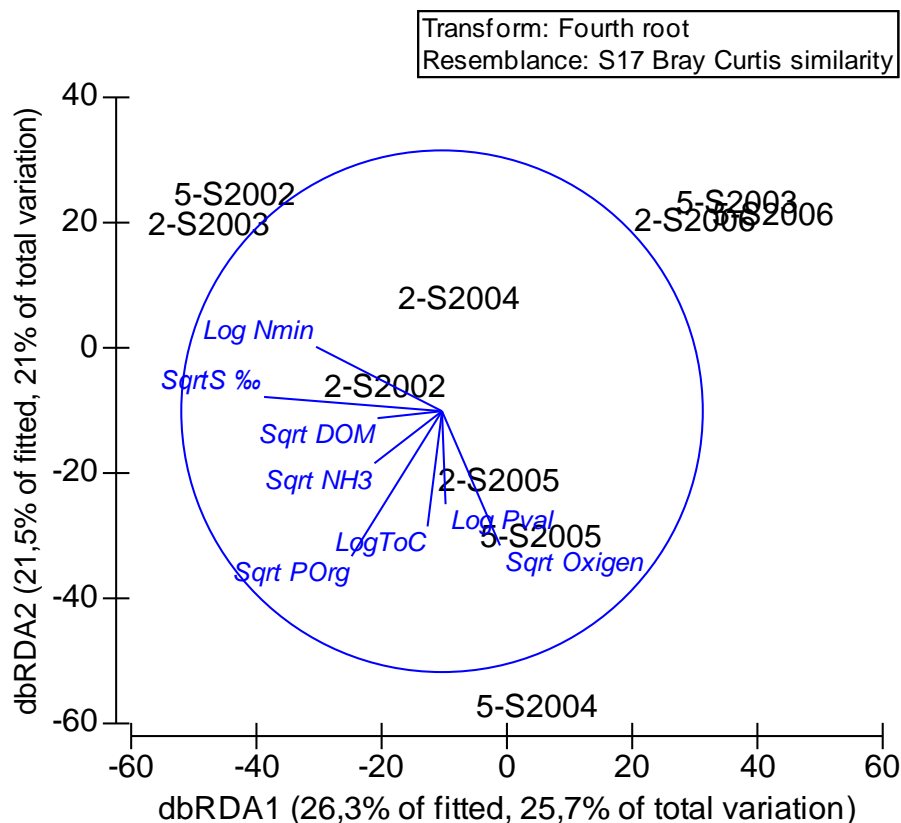


Figure 2.4.1.5. dbRDA plot of macroalgal data with the 8 variables that explained 97.5% of variation for the model.

CAP (Canonical analysis of principal coordinates) analysis showed a similar ordination of macroalgal species obtained with MDS analysis (Figure 2.4.1.6). The first canonical axis separated macroalgal assemblages sampled in 2005 and in 2004 at station 2 from those sampled in other years, while the second canonical axis separated macroalgal assemblages sampled in 2002 and 2003 at station 2 from the other period sampling. For exploratory purposes we projected the Spearman rank correlations of individual macroalgal species with the CAP axes. *Cystoseira barbata*, *Ceramium rubrum* and *Ceramium diaphanum* showed a high correlation with the CAP1, while some algal species such as *Enteromorpha intestinalis* or *Ectocarpus confervoides* were correlated with CAP2 axis (Table 2.4.1.3).

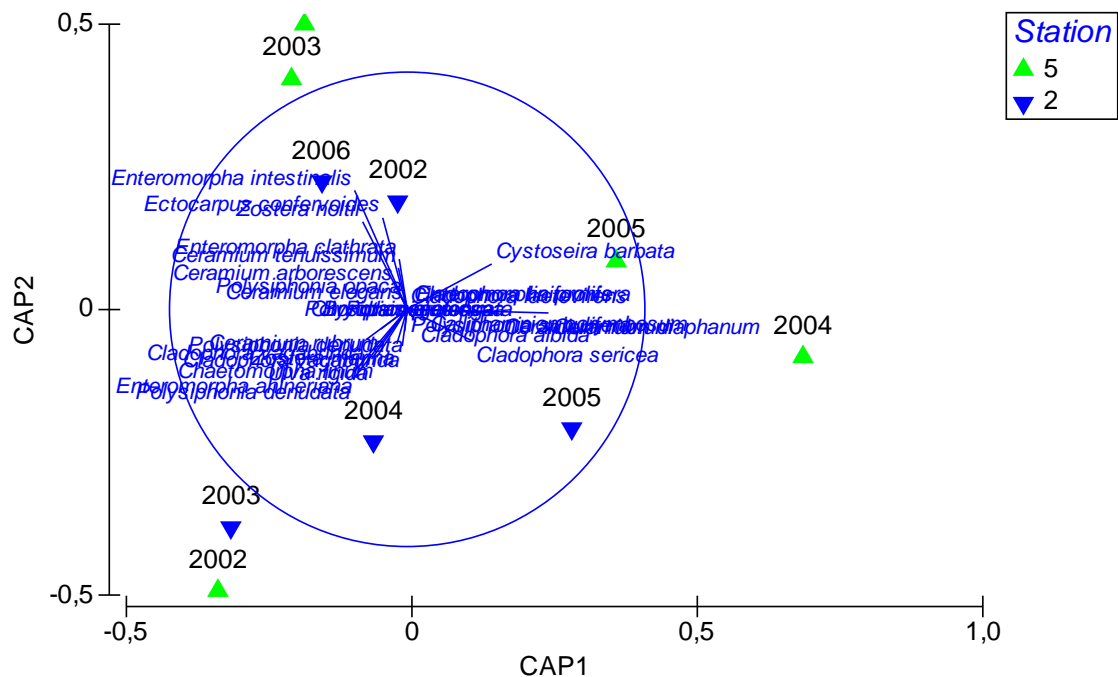


Figure 2.4.1.6. CAP ordination of macroalgal species from sampling station 2 and 5 during the years 2002 to 2006. Vector overlay of Spearman rank correlations of individual seaweed species with the CAP axes.

Table 2.4.1.3. Canonical eigenvectors in the space of X (Coefficients for linear combinations of X's to form axes that have maximum correlation with canonical coordinates)

Variable	CAP1	CAP2	CAP3
<i>Cystoseira barbata</i>	0,356	0,191	-0,270
<i>Cladophora vagabunda</i>	-0,021	-0,155	0,109
<i>Cladophora liniformis</i>	0,013	0,007	-0,010
<i>Cladophora albida</i>	0,036	-0,054	0,033
<i>Cladophora laetevirens</i>	0,000	0,000	0,000
<i>Cladophora vagabunda</i>	-0,172	-0,127	-0,240
<i>Cladophora sericea</i>	0,276	-0,134	0,154
<i>Ceramium rubrum</i>	0,388	-0,017	-0,023
<i>Ceramium diaphanum</i>	0,594	-0,015	-0,049
<i>Ceramium elegans</i>	-0,007	0,015	0,004
<i>Ceramium tenuissimum</i>	-0,037	0,175	-0,012
<i>Ceramium arborescens</i>	-0,047	0,097	0,051
<i>Ceramium rubrum</i>	-0,111	-0,084	-0,154
<i>Chondria tenuisima</i>	0,000	0,000	0,000
<i>Chaetomorpha linum</i>	-0,129	-0,203	-0,112
<i>Enteromorpha intestinalis</i>	-0,221	0,501	0,189
<i>Enteromorpha ahlneriana</i>	-0,225	-0,264	-0,147
<i>Enteromorpha prolifera</i>	0,017	0,009	-0,013
<i>Enteromorpha clathrata</i>	-0,033	0,211	-0,840
<i>Ulva rigida</i>	-0,150	-0,210	-0,016
<i>Polysiphonia denudata</i>	-0,227	-0,293	-0,070
<i>Polysiphonia opaca</i>	-0,016	0,046	0,007

<i>Polysiphonia subulifera</i>	0,000	-0,007	0,005
<i>Polysiphonia denudata</i>	-0,011	-0,093	0,063
<i>Polysiphonia elongata</i>	0,000	0,000	0,000
<i>Callithamnion corymbosum</i>	0,080	-0,005	-0,005
<i>Ectocarpus confervoides</i>	-0,103	0,386	-0,012
<i>Bryopsis plumosa</i>	0,000	0,000	0,000

Results confirmed the importance of salinity on determining macrophyte community structure in the Tyligulskyi lagoon. Furthermore, other variables such as oxygen and organic matter concentrations seem to be also important for macrophyte distribution. The clear north-south salinity gradient, as well as, water depth and its influence on light, primary production, organic matter accumulation and oxygen contents, are among the most important factors for macrophyte composition, distribution and survival in the lagoon.

2.4.2. The status of macrophytes in the lagoon: Regression events and their causes

The main hydrological factor which conditions a long-term tendency in variability of species composition of the bottom flora and fauna is its water salinity. Since launching of the connecting canal to the sea in 1959 salinity of the lagoonal waters has been rising gradually due to accumulation of salts inflowing with the sea waters. In the 1960s the average values of water salinity in the northern part of the lagoon made up 8.7 ‰, in the central part - 11.4 ‰, and in the southern part - 13–14 ‰ (Rozengurt, 1974). Under the present-day conditions the water salinity in both the southern and the northern parts of the lagoon may increase up to 19–23 ‰ by the late summer – early autumn. The average annual salinity of the lagoonal waters is above 18 ‰. In the years with intensive spring flood water salinity in the northern part of the lagoon may decrease to 3–5 ‰.

Within the latest 50 years, under the influence of the increased water salinity in the lagoon, the distribution of macrophytobenthos into environmental groups has changed towards an increase in salt water species, from 18,1 to 45,6 %, brackish – salt water species – from 22,3 to 37,7 % and a decrease in fresh-brackish water species – from 21,4 to 11,7 %. Currently the freshwater species have almost completely lost their value and persist only in the mouth of the intermittent Tyligul river.

The main causes of these changes are irregular inflow of sea water into the lagoon through the artificial connecting canal to the sea, reduction and instability of fresh water inflow from the lagoonal catchment area due to both anthropogenic and climatic factors.

Relative isolation of the lagoon from the sea facilitated persistence of the isolated population of *Cystoseira barbata*, which had become extinct in the North-Western part of the Black Sea in the 1980s, in the Tyligulskyi Liman lagoon. Among the macrophytes inhabiting the lagoon, 1 species (*Chara canescens*) is in the Red Book of Ukraine and 2 species of aquatic flowering plants (*Zostera noltii*, *Z. marina*) are in the Red Book of the Black Sea. The red alga of *Rhodochorton purpureum* (Lighth.) Rosenv., which was found on *Cystoseira barbata*, is rare in Ukraine. *Vaucheria litorea* Hofm. et Ag. in C. Ag., still being a mass species in the lagoon, is among the ones with decreasing number of specimens in Ukraine.

2.4.3. Macrophyte vulnerability in the context of climate change

For the Tyligulskyi Liman lagoon vulnerability of the macrophyte associations related to the climate change will be determined by the volume of freshwater runoff from the drainage area of the lagoon and the operating conditions for the connecting canal with the sea.

The forecasted increase in the air temperature in the 21st century will condition an increase in the evaporation rate and, consequently, a decrease in the inflow of fresh water into the lagoon. On the stipulation that a periodic water exchange with the sea through the connecting canal is not provided, the rate of annual decline in water level in the lagoon, according to preliminary water balance estimates, will comprise 0,26 m per year in the period of 2011-2040 and 0,4 m per year in the period of 2071-2098. Consequently, rapid shoaling of the lagoon will occur and the macrophytes will lose their traditional habitats in the coastal zone, on the stony, the sandy-and-silty bottom in the relatively narrow shallow coastal zone of the deep southern and central parts in the lagoon. Probability of their development on fine-dispersed silty soils, which prevail in the deeper parts of the lagoon, is very limited. Moreover, drainage of large water areas in the shallow northern part of the lagoon will take place. Shallowing of the lagoon will be accompanied by an increase in the salinity of its waters and in the eutrophication level of the water body due to a reduction in the water volume as a result of intensive evaporation.

When the connecting canal operates systematically only in the spring months (when the salinity of the inflowing sea water is minimal) in order to stabilize the water level in the

lagoon, accumulation of salts in the lagoon will be going on, and in the second half of the 21st century it will turn into a hyper saline water body with the water salinity of above 40 ‰, which will lead to a significant impoverishment of the macrophyte species composition.

When operation of the connecting canal is regulated throughout a year, it will be possible to stabilize the lagoonal water salinity on the whole. At the same time, in the southern part, it will be close to the seawater salinity (16-19 ‰), and in the northern part it will depend on the volume and duration of the Tyligul river runoff.

Under all of the three considered scenarios for the changes in the lagoonal water salinity related to the climate change the dominant role of marine macrophyte species in functioning of the aquatic ecosystem of the Tyligulskyi Liman lagoon is to become more significant.

Higher water temperatures in all seasons will entail an increase in production of organic matter by the macrophytes in the winter and the spring periods and, as a consequence, in the summer will lead to an increase in dissolved oxygen consumption for biochemical oxidation of dead remains of the macrophytes. Therefore, an increase in frequency and expansion area of hypoxic and anoxic phenomena in the shallow parts of the lagoon are to be expected during periods of prolonged calm.

3. Evaluation of the main environmental variables responsible for benthic macrofauna distribution

3.1. The Ria de Aveiro lagoon

3.1.1. Data sources and previous studies

Comparatively to the other case study lagoons there is a lack of information regarding the ecology of benthic macrofauna in Ria de Aveiro, meaning that there are no time series or even seasonal data regarding the benthic communities. The most complete assessment to study the benthic macrofauna distribution in Ria de Aveiro was done by Rodrigues et al. (2011). These authors sampled the benthic communities covering the full salinity gradient and the whole channel system of Ria de Aveiro in Spring-Summer 1999. The study covered 248 sampling sites and resulted in the identification of a total of 120 taxa in more than 76,350 specimens. According to the authors, “the most abundant and frequent taxa were all annelids, namely *Alkmaria romijni*, *Streblospio shrubsolii*, *Tharyx* sp., *Tubificoides benedii*, *Nereis diversicolor*, *Capitella* sp., *Pygospio elegans*, *Polydora ligni* and an unidentified oligochaete” (For a complete list of the macroinvertebrates and mobile megafauna species occurring in Ria de Aveiro lagoon please see LAGOONS D 3.2). In order to explore the relationship between environmental and biological data the BIOENV procedure from the Primer v6 software analysis was adopted by the referred authors. Rodrigues et al. (2011) found that the main environmental variables responsible for benthic macrofauna distribution in Ria de Aveiro lagoon were: the sediment shear stress and Eh, the current velocity and flux, and the salinity. These authors also found that benthic macrofauna “increased from euhaline to limnetic areas but species richness and diversity were higher at intermediate regions”.

3.1.2. Interpretation of results in the context of climate change

As previously stated by Rodrigues et al. (2011), macrofaunal distributions in the Ria de Aveiro lagoon clearly correspond to the main environmental gradients observed in the study area, mainly that of salinity. Current velocities and their effect on sediment properties seem to have also a clear effect on macrofaunal composition and distributions in the lagoon.

The expected consequences of global climate change in the area are likely to alter both variables in the Ria de Aveiro lagoon. Future salinity and current velocity changes in the

lagoon may cause a shift on benthic community composition and distribution. Macrofaunal assemblages in the study area respond to the strong gradient from those sites located close to the main inlet, characterised by more oceanic conditions, to sites clearly influenced by freshwater inputs. Future variations of tidal regimes, sea-level and/or freshwater discharges will have a clear impact on the salinity gradient, as well as on many other environmental variables. Inferring the consequences of these changes on benthic macrofaunal distributions is a difficult task but an 'upstream-migration' of benthic communities, as well as the colonisation of the lagoon by marine species is to be expected.

3.2. The Vistula lagoon

3.2.1. Study area and sampling procedure

The study was carried out in the western, i.e. Polish part of the Vistula Lagoon. The Vistula Lagoon is situated in the southeastern part of the Baltic Sea and it spreads along the distance of 91 km covering the Polish as well as Russian coast of the Gulf of Gdansk. A sand bar formed on the way of natural processes and called Mierzeja Wiślana separates the Lagoon from the sea. The Pilawska Straits forms the only connection between the Lagoon and the sea and it makes it possible water exchange between these water reservoirs. Muddy sediments occupy a predominating part of the bottom in the Lagoon, and only a narrow strip stretching in the coastal zone, as well as shoals, both not exceeding the depth of about 1.5 -2,0 m, are covered with sandy or muddy-sandy sediments. Fine-grained and medium-grained sand predominate. Due to shallowness of the Lagoon, the region is exposed to significant wind induced mixing and consequently the thermal stratification of water column is non-existent, (the difference in temperature at the surface and near bottom does not exceed 0.5-1.0°C). The seasonal variability of water temperature remains in a range from -0.2°C to over 25 °C. A very fast warming of water in spring is a characteristic feature of the region. In winter, ice cover may appear, and it may occupy the region on average for 75 days; during severe winters, the presence of ice cover may extend up to 120 days. The maximal thickness of ice cover may reach 60 cm. Salinity in the Polish zone in the range about 0.5-3,6 PSU (Łomniewski, 1958; Czubarenko and Margoński 2008).

Starting from the 80's of the XXth century, macrozoobenthos in the Polish zone of the Vistula Lagoon was dominated by freshwater species (Chironomidae and Oligochaeta larvae). Over the last decades, an invasive species of polychaete *Marenzelleria* sp. has been predominating with respect to biomass (Cywińska and Róžańska 1978, Róžańska and Cywińska 1983, Żmudziński, 1996).

3.2.2. Multivariate statistics

In order to define the connections between the found species/taxa and the physical spatial environmental variations the canonical correspondence analysis was adopted (CCA-package "Vegan") (Palmer, 1993; Gockzin and Zettler, 2008; Rousi et al., 2011).

The data collected in 2009-2010 constituted an input database for the analysis of impact of physical parameters on distribution of macrozoobenthos. Over the summer period (July-August), samples of the sediment and fauna were collected at 56 stations covering the entire area of Polish zone, and sampling procedures were accompanied by measurements of physical parameters sediment sampling for analyses on granulometry and organic matter content. Samples were taken with the help of hand operated corer type HAPS (penetration depth – 30 cm; sampling area – 168 cm²). All the stations were located beyond the zone with coastal vegetation, starting from the depth of ca. 0.5 m.

Species abundances were used in the statistical analysis. Number of species was determined for each sample separately and then the average values for three samples were calculated on each station, then the obtained numbers were recalculated per m².

In total 20 taxa were found (Table 3.2.2.1) and nearly entirely they belong to infauna.. All the taxa found on less than 6% of all the stations were discarded from the analysis (Glockzin and Zettler, 2008; Rousi et al. 2011). After rejection of all the stations at which measurements and analyses of all the abiotic parameters were not made, and after a selection of species provided for analyses, there remained 40 stations and 9 taxa (Table 3.2.2.1).

Table 3.2.2.1. Taxa used for CCA analyses, number of stations where particular species was found mean abundance and species/taxa used for CCA analyses.

taxa	number of CCA stations where found	mean abundance ind. x m ⁻²	used in the CCA analyses +/-
<i>Marenzelleria</i> spp.	39	1812,4	+

Oligochaeta nd.	39	773,04	+
Chironomus semireductus	33	820,17	+
<i>Prostoma obscurum</i>	6	5,17	+
Glyptotendipes gr. gripekoveni	5	31,58	+
Procladius sp.	5	11,78	+
<i>Polypedilum gr. nubeculosum</i>	4	2,03	+
Orthocladinae nd	4	2,55	+
Hydrobiidae nd.	3	2,18	+
<i>Dreissena polymorpha</i>	2	1,44	-
Cryptochironomus defectus	2	2,2	-
<i>Criodrilus lacuum</i>	2	4,88	-
<i>Boccardiella ligerica</i>	1	6,61	-
<i>Gammarus sp.</i>	1	0,58	-
<i>Gammarus tigrinus</i>	1	0,58	-
<i>Dikerogammarus sp.</i>	1	0,78	-
<i>Rhitropanopeus harrisii</i>	1	0,58	-
Microchironomus sp.	1	0,3	-
<i>Polypedilum gr. scalaenum</i>	1	0,58	-
<i>Paratanytarsus sp.</i>	1	4,88	-

Prior to analysis, the abundances were subjected to $\sqrt{\sqrt{\cdot}}$ transformation. Four abiotic environmental variables i.e. depth, salinity, sediment granulation and content of organic matter (LoI) in sediments were used as physical features of environment in CCA analyses. In interpretation of the obtained results some additional measurements carried out in 2012-2013 i.e. oxygen content in near-bottom water and redox potential, were also utilized.

In order to exclude the environmental variables that can mask impact of other factors (Glockzin and Zettler, 2008; Rousi et al., 2011), four CCA analyses were carried out (Figures 3.2.2.1-4) and in these analyses (i) four environmental variable were used (Figure 3.2.2.1), (ii) salinity was excluded (Figure 3.2.2.2) (iii) depth was excluded (Figure 3.2.2.3)), (iv) only depth was used (Figure 3.2.2.4). All four analyses point to sediment features (granulation and LoI) as variables having impact on distribution of bottom invertebrates. However, they explain distribution variability only in relatively little degree. Among the dominants, *Marenzelleria sp.* prefers sandy bottom, whereas *Chironomus fl semireductus* is associated with muddy sediments. In each analysis, oligochaeta can be found in the center of Biplot, which indicates its adaptation to various types of sediments. That points to the fact that within this group there do not occur specialized species and determination of all Oligochaeta species would not significantly change the results of the analysis. Generally, salinity variability does not play an essential role, with an exception of few Chironomidae taxa (*Glyptotendipes gr gripekoveni*, *Orthocladinae nd.* *Polypedilum sp.*, which were found on stations located nearby

the outlets of small rivers. The depth (Figure 3.2.2.4) is only related to distribution of *Chironomus semireductus* but it seems to be autocorrelation with sediment parameters.

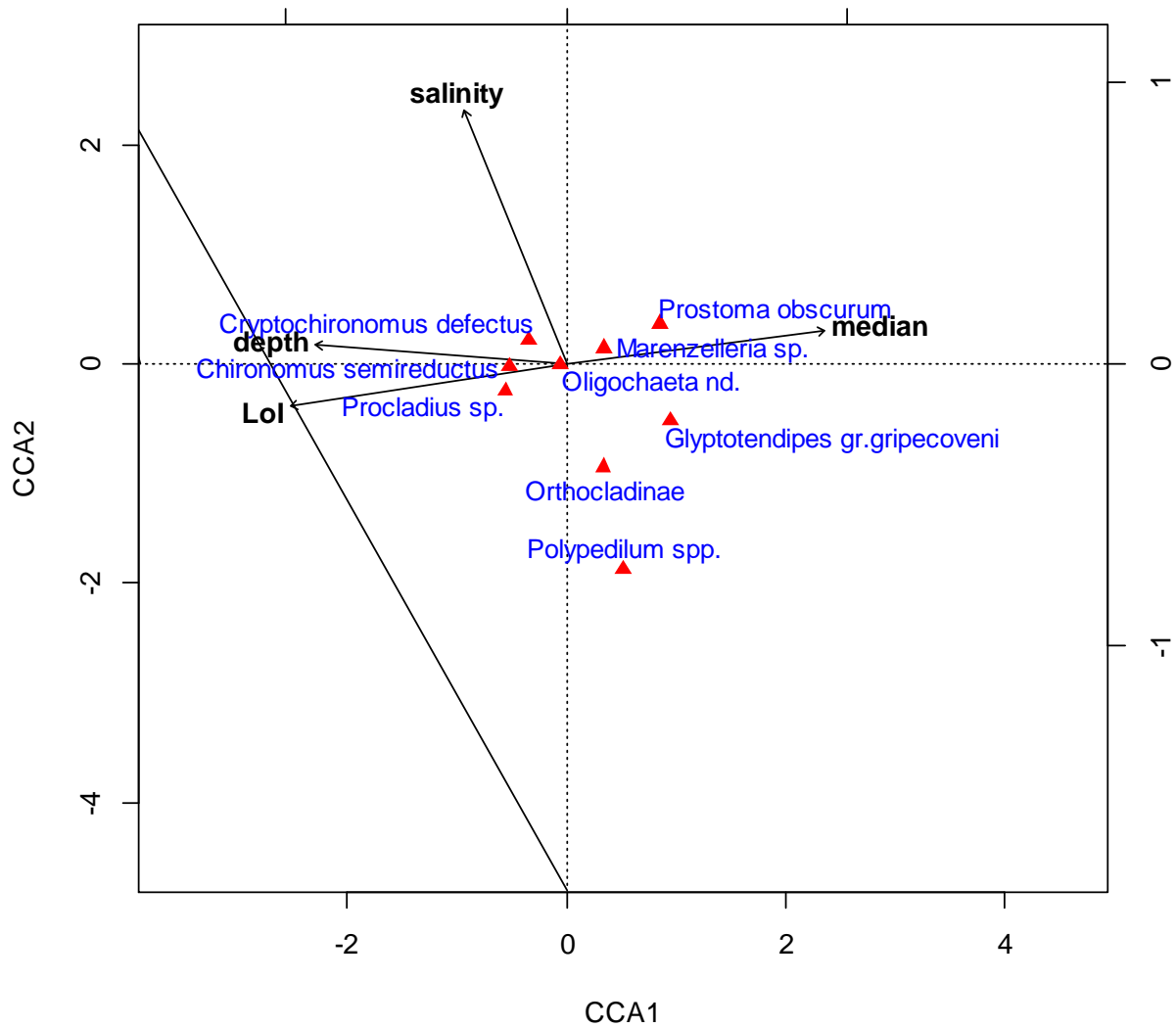


Figure 3.2.2.1. Results of the CCA analysis calculated for transformed $\sqrt{\sqrt{}}$ abundances of macrofauna and depth, median grain size, organic matter (LoI) and salinity.

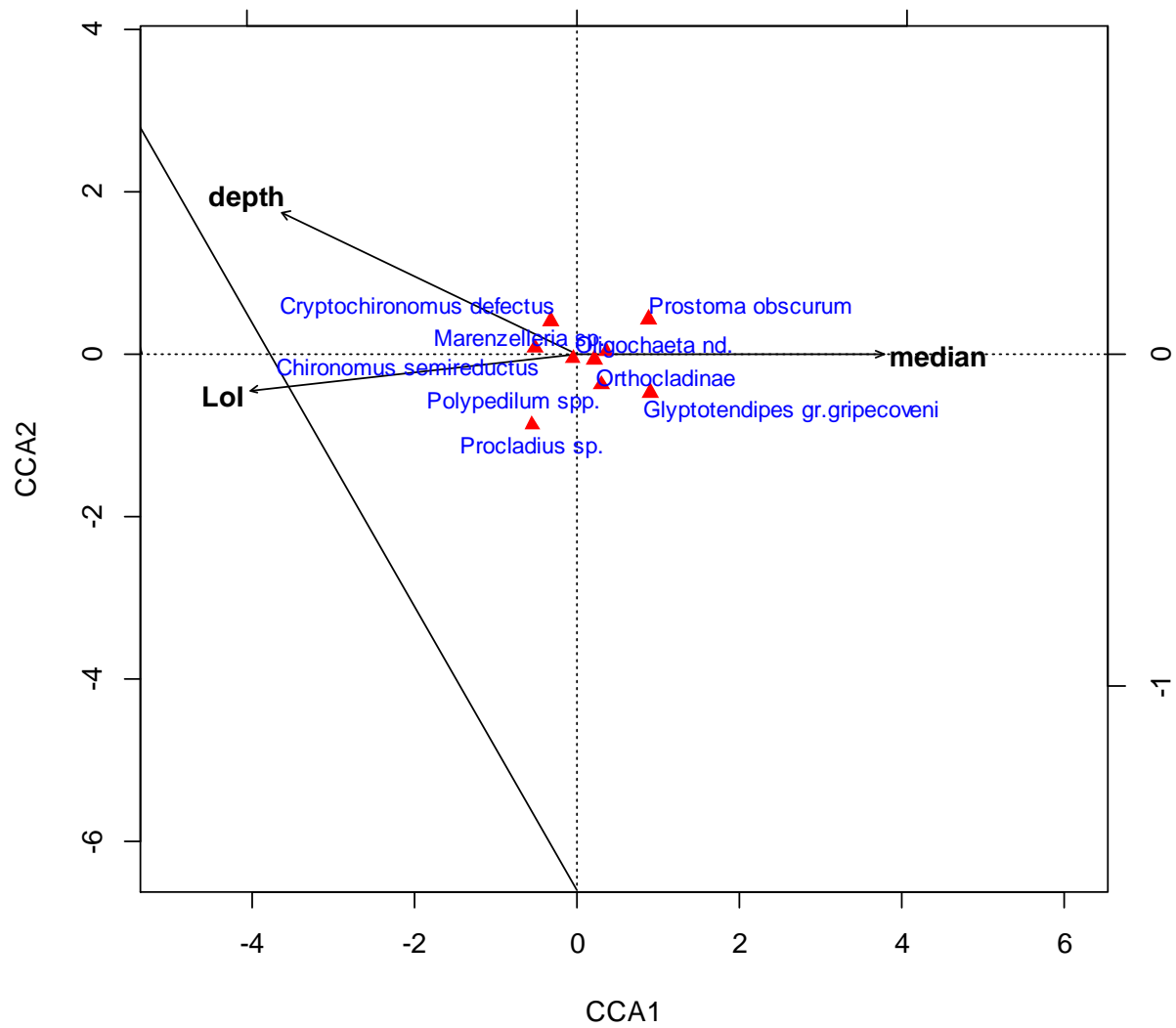


Figure 3.2.2.2. Results of the CCA analysis calculated for transformed $\sqrt{\sqrt{\cdot}}$ abundances of macrofauna and abiotic parameters excluding salinity.

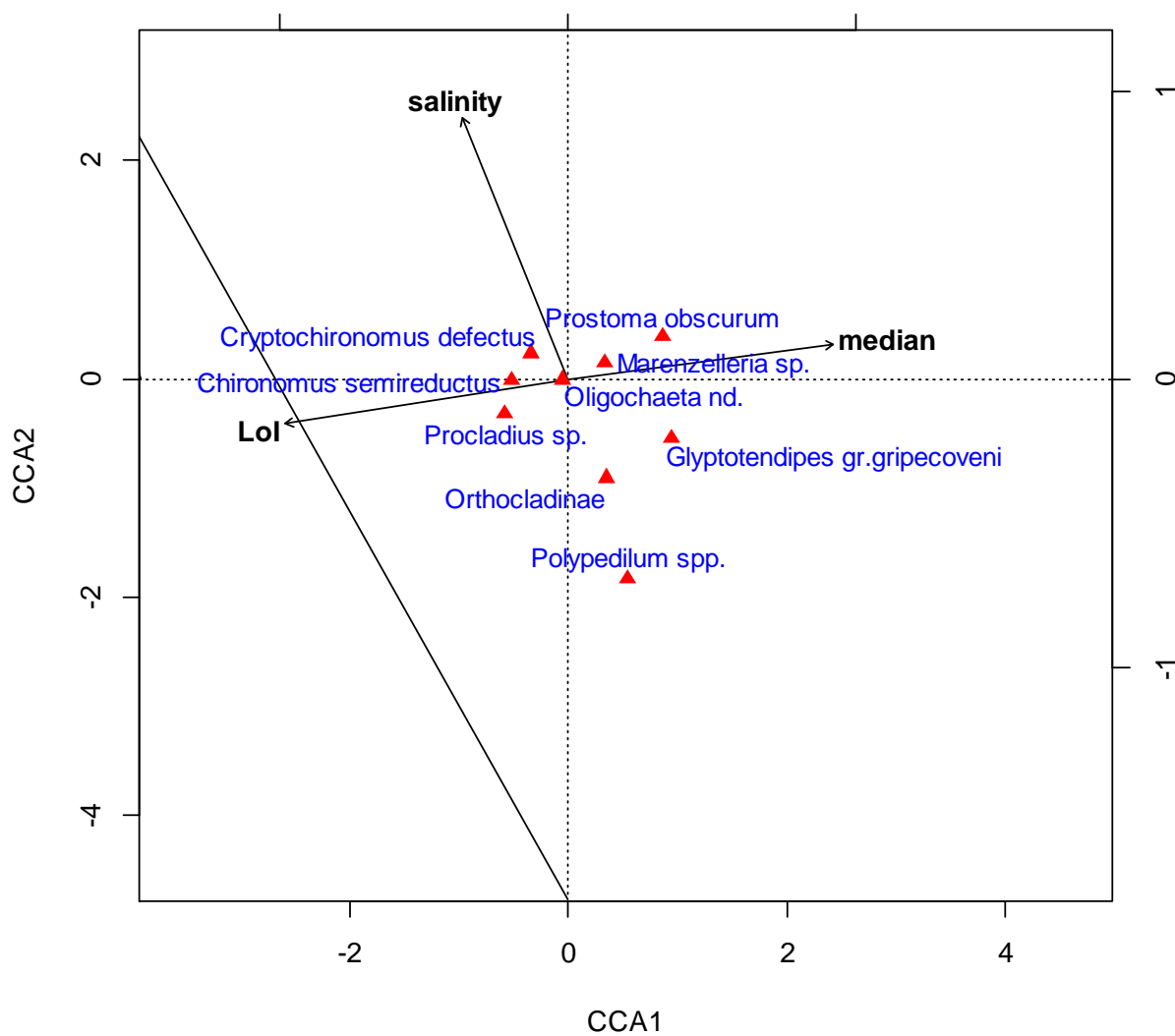


Figure 3.2.2.3. Results of the CCA analysis calculated for transformed $\sqrt{\sqrt{}}$ abundances of macrofauna and abiotic parameters excluding depth.

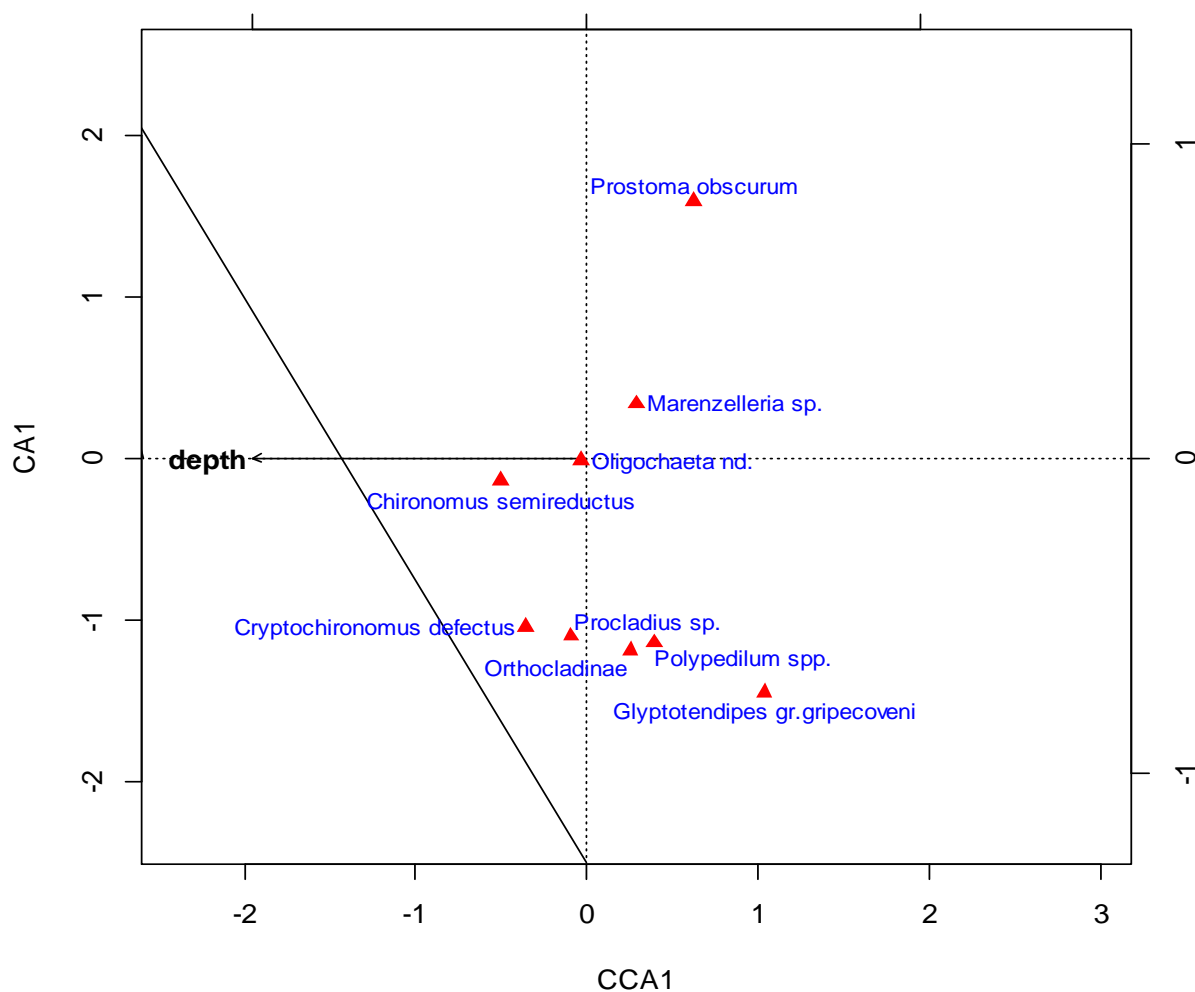


Figure 3.2.2.4. Results of the CCA analysis calculated for transformed $\sqrt{\sqrt{\text{abundances}}}$ showing the relationships between macrofauna and depth.

Distribution of species/taxa in relation to environmental variables reflects also feeding preferences of animals. The muddy sediment is predominantly occupied by detritofag *Chironomus* f.l. *semireductus*, whereas sandy sediment is occupied by facultative suspension feeder *Marenzelleria* sp, which also occurs, though less abundantly, in muddy sediments where it can feed as a detritofag. *Orthocladinae* and *Procladius* sp. belong to species, which do not show connection with sediments in the analysis.

3.2.3. Interpretation of the results from CCA analysis

The results obtained with CCA analysis and concerning the impact of sediment characteristics on zoobenthos, probably would be different if also eastern (Russian) part of Lagoon is included. As to salinity gradient at the cross section of the entire Lagoon, it is much stronger than in the Polish part alone. It particularly concerns the region in vicinity of Pilawska Straits situated in the eastern (Russian) part of the Lagoon. In that part of the Lagoon, salinity reaches the values close to those found in coastal waters of the southwestern Baltic Sea and this region is inhabited by species typical for the Gulf of Gdansk. Willer (1927) defined the eastern part of the Lagoon as polyhaline, the western part of Polish zone – as oligohaline; between these two zones there is a mesohaline zone. The same author demonstrated impact of salinity on horizontal distribution of zoobenthos. It is worth pointing that salinity in the western part of the Lagoon depends on water inflows from the open Baltic; therefore, it significantly changes, even within hours (Figures 3.2.3.1-2).

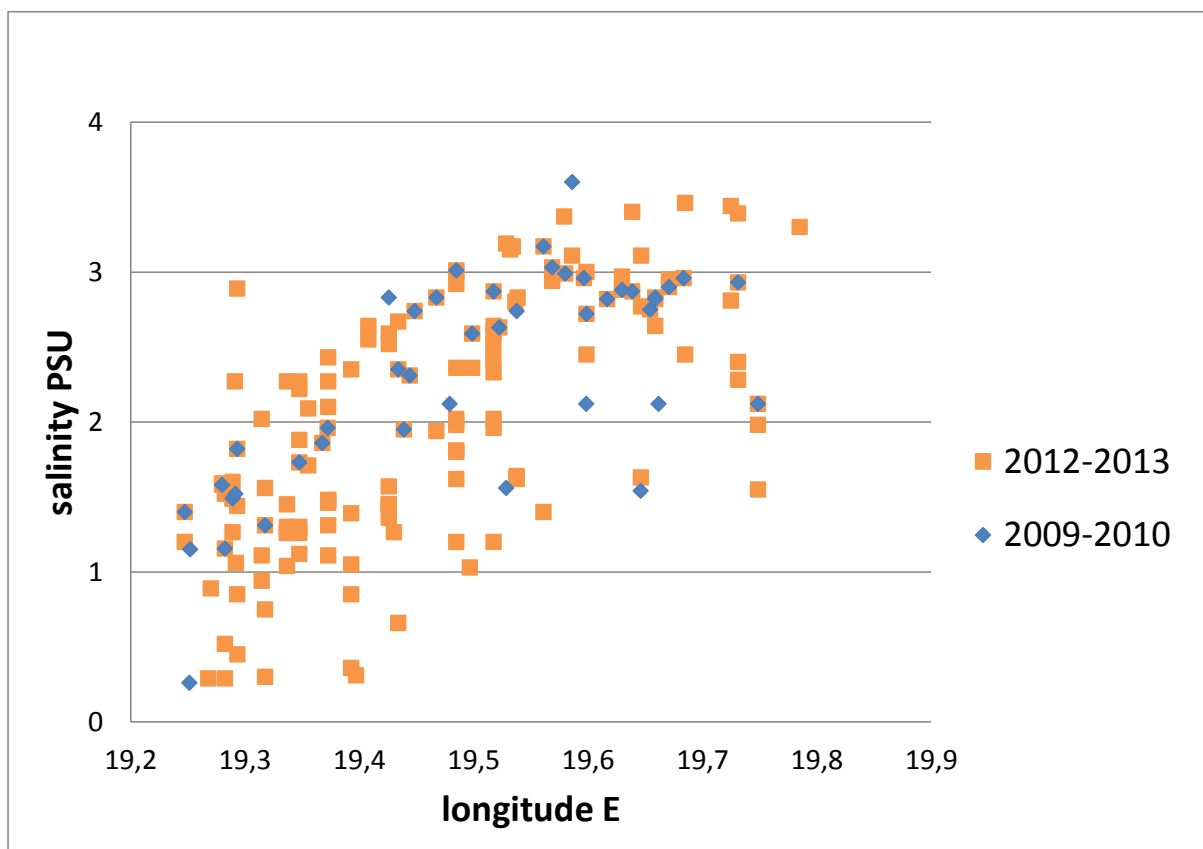


Figure 3.2.3.1. The variability of salinity in 2009-2013 along longitude with salinity values from 2009-2010 used for CCA analyses.

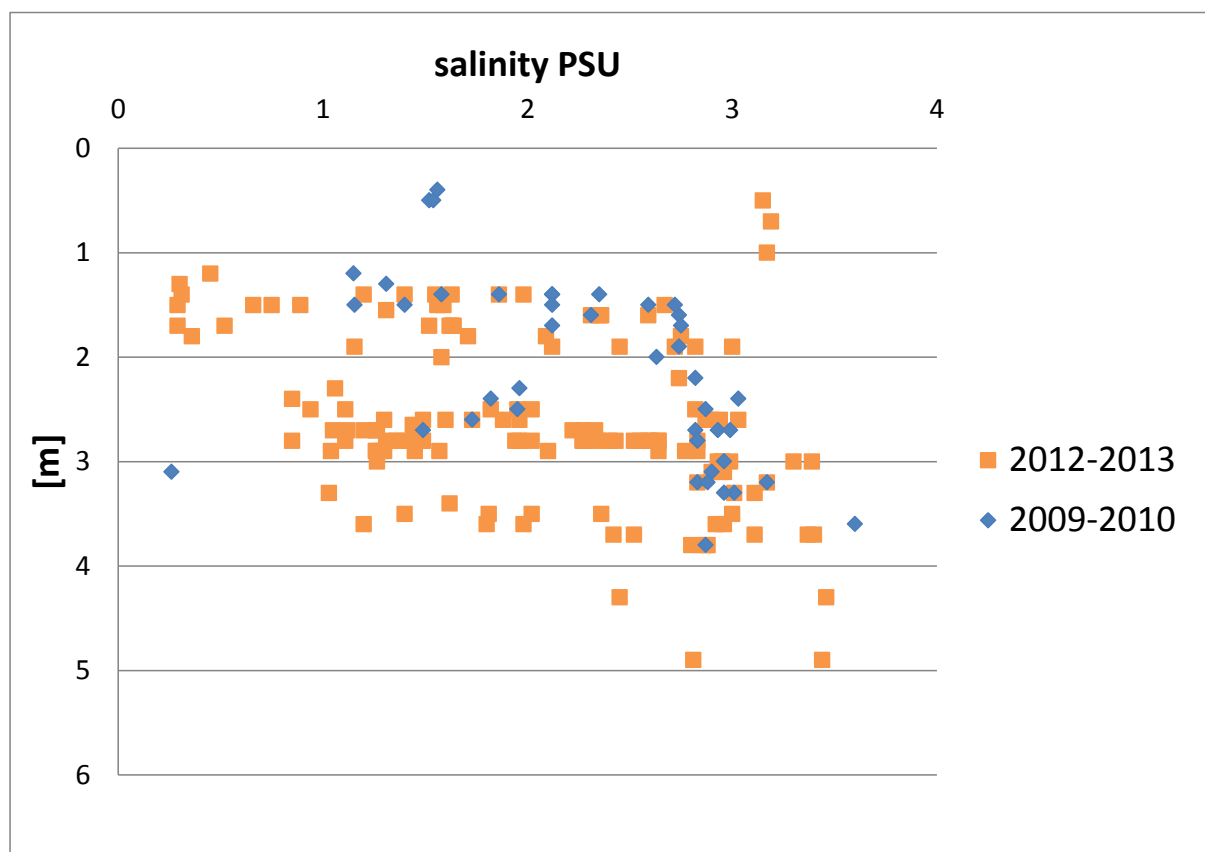


Figure 3.2.3.2. Salinity from 2009-2013 vs. depth.

However, salinity variability in the range occurring in the western part of the Lagoon is not a significant factor impacting the occurrence of the bottom fauna, because the latter is exclusively represented by euryhaline marine species (*Marenzelleria* sp., *Prostoma obscurum*, *Ritropanopeus harrisi*) or by freshwater species (Oligochaeta, Chironomidae). Practically, there do not occur specialized freshwater species. The only exception is *Gammarus zaddachi*, sporadically seen in the plant zone.

The other environmental factor that may also mask the results of CCA analysis is water oxygenation, which is highly variable (Figures 3.2.3.3-4).

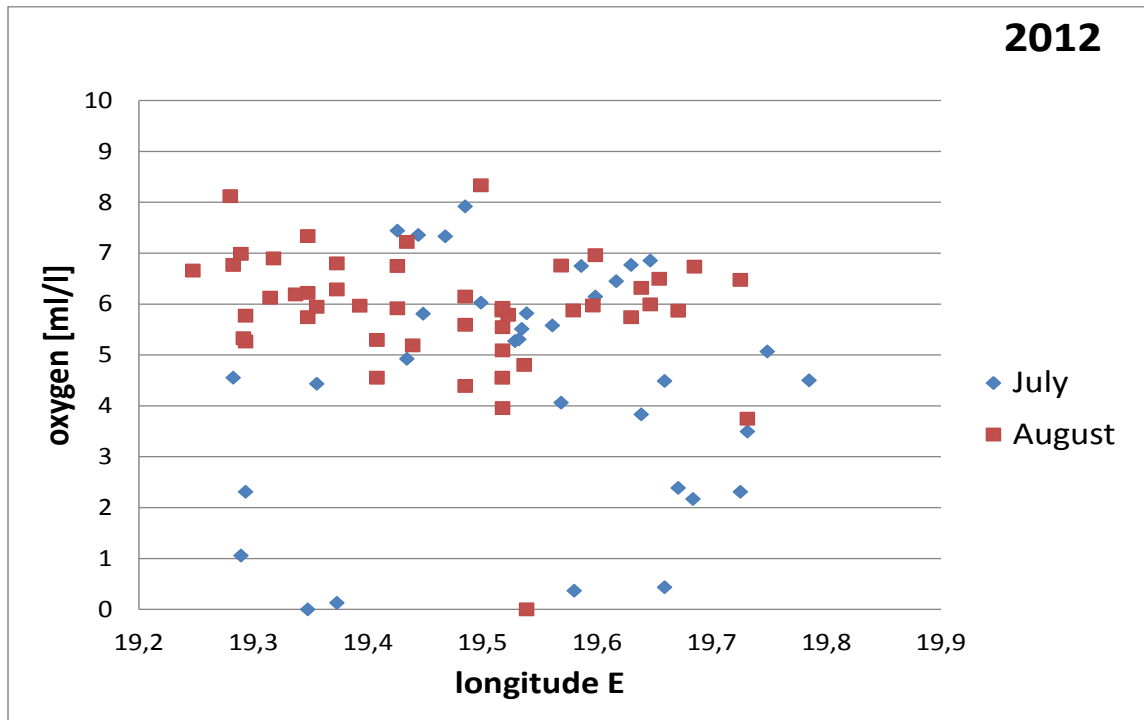
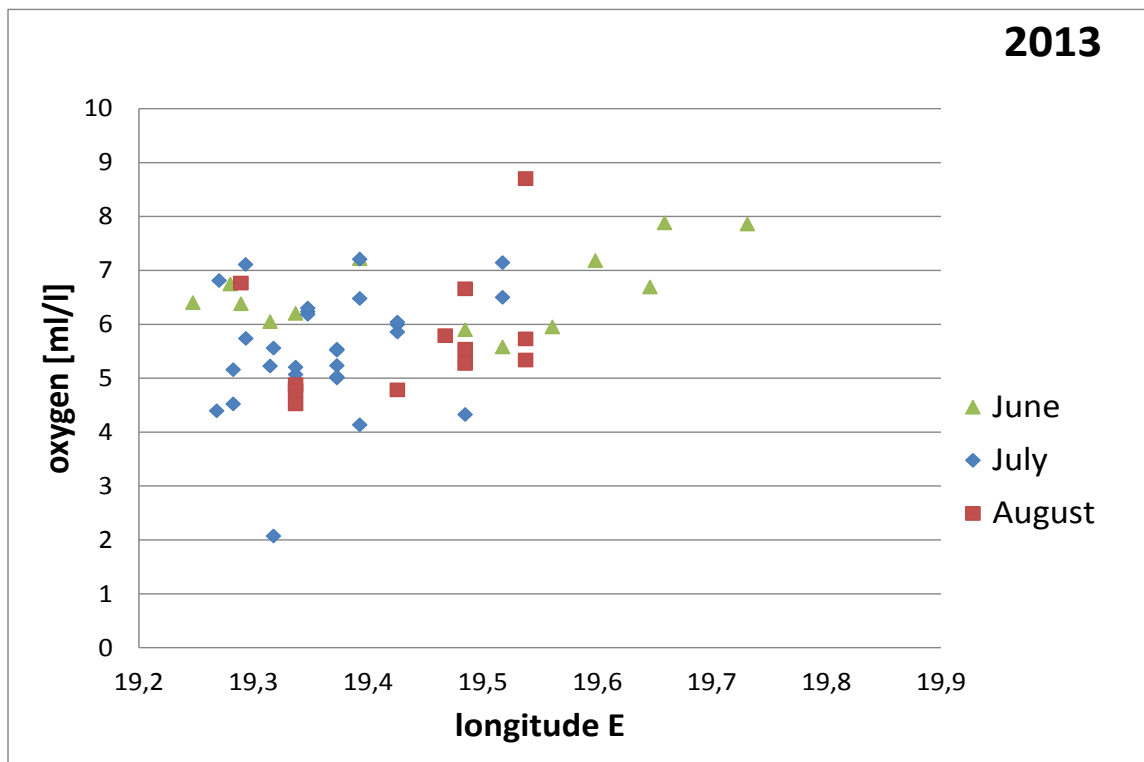
**A****B**

Figure 3.2.3.3. Oxygen content (ml O₂/l) in the near bottom water layer in summer seasons of 2012 (A) and 2013 (B).

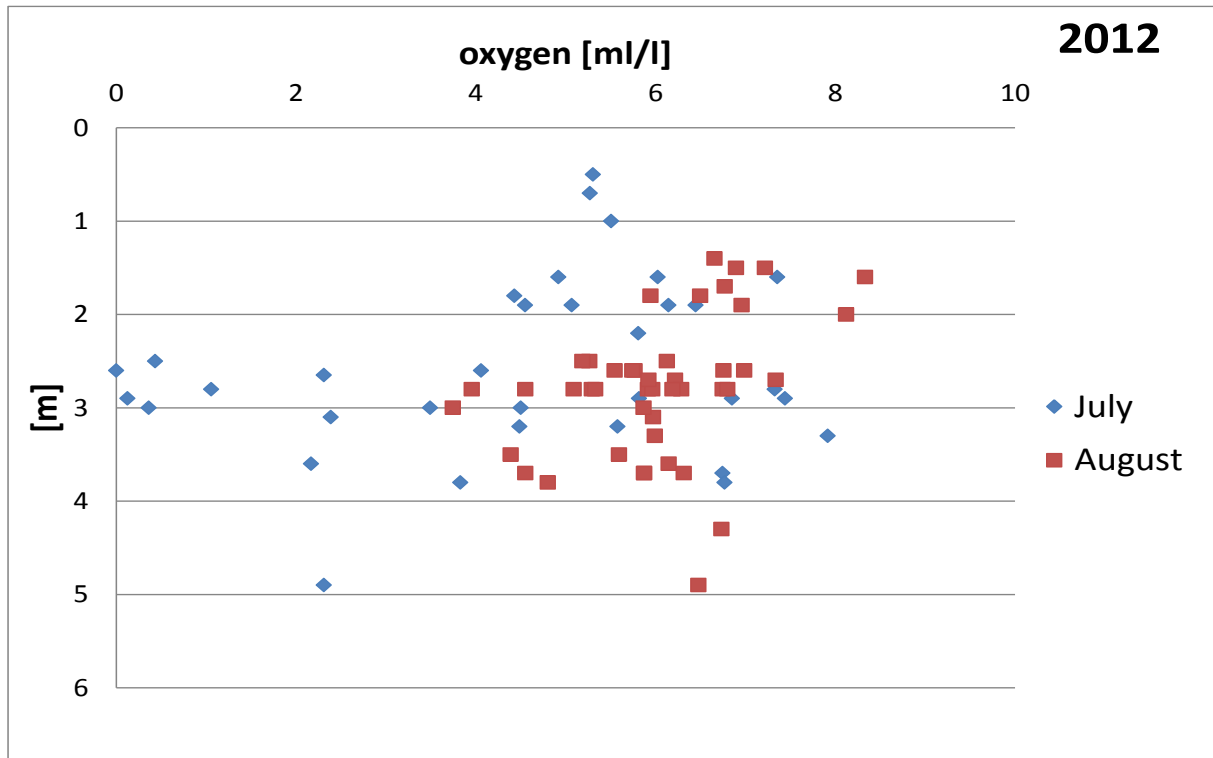
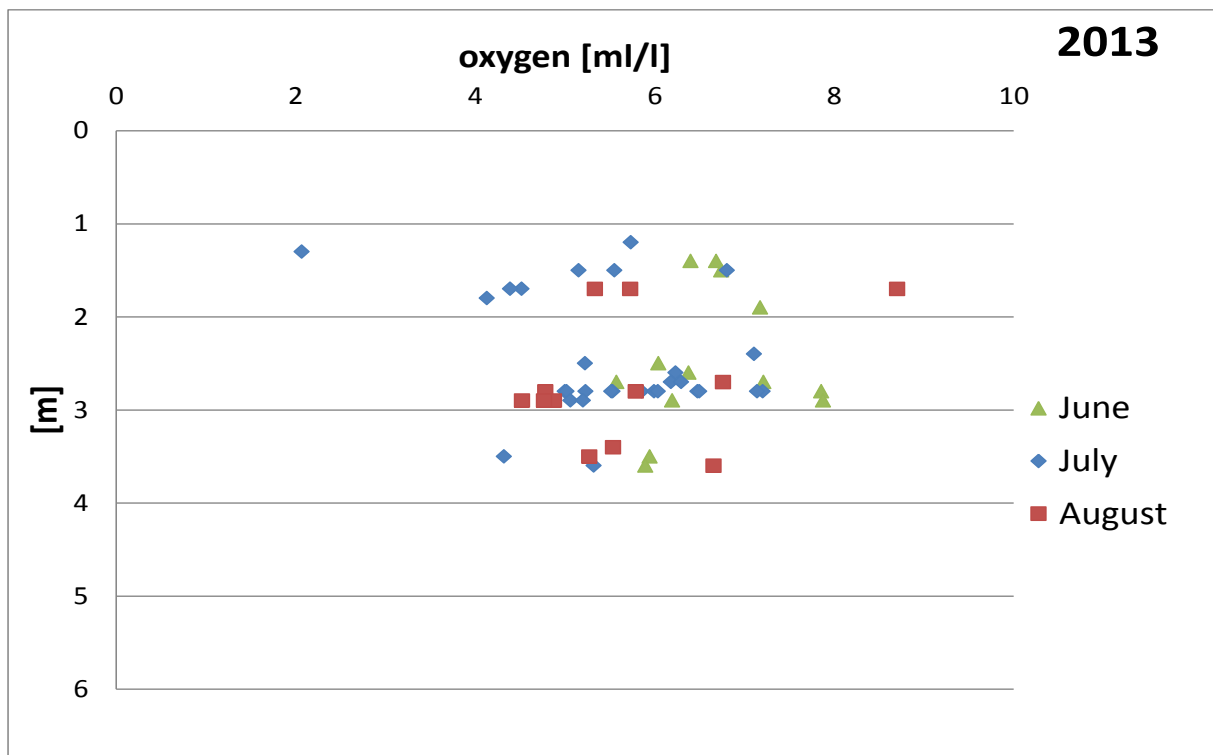
**A****B**

Figure 3.2.3.4. Oxygen content in the near bottom water layer, in summer seasons of 2012 (A) and 2013 (B) vs. depth.

High variability of this parameter made us omit it in the analysis. The significance of this factor has, so far, been omitted in studies and the explanation lies in the fact that the region is shallow and exposed to huge water dynamics and therefore, it should be characterized by good water oxygenation. In 2010-2013, we conducted measurements of water oxygenation (with optical sound) in the coastal zone (10 cm above the bottom) at different wind stress, in different hours of the day and at different salinity, either indicating inflows of saline waters or a stagnation period. The obtained results (Figures 3.2.3.3-4) confirmed that usually there occurs good oxygenation, but significant declines in water oxygenation or even oxygen deficits may occasionally occur (Figure 3.2.3.3A). Oxygen deficit was observed only in eastern, the most sheltered part of the Lagoon at windless conditions and at high water temperature (ca. 25 °C). Such situations lasted only some hours; inflows of saline waters, indicated by increasing salinity, or wind induced water mixing cause an increase in water oxygenation. Such, short lasting deficits of oxygen most probably are not dangerous for dominating taxa, which on the other hand are characterized by exceptional resistance to oxygen deficiency (Chironomidae, Oligochaeta, *Marenzelleria* sp.). However, such unfavorable conditions may limit development of juveniles. Analyses of redox potential and black color of sandy sediments may suggest occurrence of long-lasting oxygen deficits and presence of hydrogen sulfide. Measurements of redox potential in sandy sediments indicated that only an upper 2-5 cm layer was oxygenated; beneath it, reductive processes predominated. In muddy sediments even in the surface sediment layer, negative values of redox potential were detected. These results, together with drastic reduction of macrozoobenthos which followed a long iciness in winter 2012/2013, suggest that wintertime may be critical. Confirmation of these suppositions calls for measurements of water oxygenation and detection of hydrogen sulfide, also during periods with ice cover. Ice has always covered Lagoon in winter, therefore there should occur fauna, which is adapted to such conditions. It may be supposed that worsening of oxygen conditions under ice cover appeared together with process of eutrophication, which began in the 60s. of the XXth century (Wiktor, 1978; Róžańska and Cywińska, 1974; Renk, 2001) and caused increase in organic matter in water column and on the bottom. This period coincides with the observed extinction of “Baltic species” (e.g. *Corophium volutator*, *Gammarus zaddachi*) which were earlier observed in the western part of the Lagoon.

As a summary:

- In the western part of the Vistula Lagoon there are no strong horizontal and bathymetric gradients in salinity, temperature and oxygen content. The bottom is also rather homogenous and consists of two main types; sandy and muddy only.
- The characteristic feature of the Lagoon is strong short term variability in salinity and oxygen content in the water.
- Among macrobenthic invertebrates, living in the western part of the Vistula Lagoon, distribution of *Marenzelleria* sp. and *Chironomus* f.l. *semireductus* reflects the sediment types. Oligochaeta did not showed any habitat preferences.
- Oxygen deficiencies, related also to eutrophication, are proposed as an important environmental factor for the composition and distribution of macrobenthic fauna.

3.3. The Mar Menor lagoon

3.3.1. Study area, sampling procedure and methodologies

The study was focussed on *C. prolifera* bottoms, a habitat that represents more than 90% of the lagoon's bottoms. A transect of five sampling stations was selected following the main environmental gradients observed in the lagoon, from shallow areas influenced by terrigenous inputs of nutrients and particulate materials (LG1), to the central deeper areas far from the influence of inputs (LG5). The study was carried out in 2003 (Lloret and Marin, 2011).

Macrobenthic fauna samples were collected at sites and sorted using a stereo microscope, then classified at species level and counted. At the same time main environmental variables were measured.

Principal Component Analysis, PCA, (correlation-based transformed and normalised matrix) was performed using a set of environmental variables including: dissolved inorganic nitrogen (DIN), POM and chlorophyll concentrations, water column temperature, SOM and fine fraction (FF) contents, and *C. prolifera* biomass. The following transformations were necessary to reduce the skewness in the variable values: an arcsine of the square-root transformation of SOM and FF percentages and a $\log(x + 1)$ transformation for the rest of the variables.

Spatial and temporal differences in the species/abundance matrix were analysed using permutational analysis of variance (PERMANOVA, Anderson, 2001). The design consisted of two factors: location (Lo: five levels-stations) and season (Se: four levels). The analyses were based on Bray-Curtis dissimilarities of $\log(x + 1)$ transformed data and each term was tested using 9999 permutations of the appropriate units.

To test how much of the variability was explained by the set of environmental variables, a non-parametric multiple regression analysis was used, under the DISTLM.exe routine (McArdle and Anderson, 2001). Each environmental variable was initially analysed separately (excluding other variables) in the marginal test, to test the potential relationship with the multivariate community (species/abundance) data. Variables were then subject to a forward selection procedure (sequential test, R² selection criterion), in which the amount of variability explained by each variable added to the model was conditional of the variables already in the model. P-values for the marginal tests were obtained by using 9999 permutations of the non-normalised and transformed data. Then, sequential tests were done using 9999 permutations of residuals under the reduced model (Anderson, 2001). All tests were based on Bray-Curtis dissimilarities, calculated among species/abundance observations.

3.3.2. PCA analysis of environmental variables in the sampling sites

PCA analysis of environmental data showed that, for all stations and seasons considered, the joined input of the first two axes explained 58.47% of total variability. PC1 accounted for 34.24% of the variability and was essentially a combination of SOM, *C. prolifera* biomass and POM. PC2 accounted for 24.23% of the variability, and was mainly a combination temperature, DIN and chlorophyll a concentrations. The 2-D PCA showed the separation of environmental responses within groups of stations and seasons.

Spatial variability was mainly recorded on PC1 axis, with LG1 station located in the direction of increasing DIN, POM and chlorophyll a concentrations and decreasing SOM and *C. prolifera* biomass. The remaining stations were located along the gradient of increasing SOM and *C. prolifera* biomass and decreasing POM.

Seasonal variability was mainly recorded in PC2 axis with summer samples located in the direction of increasing temperature and chlorophyll a concentrations and winter samples in the direction of increasing DIN (Figure 3.3.2.1).

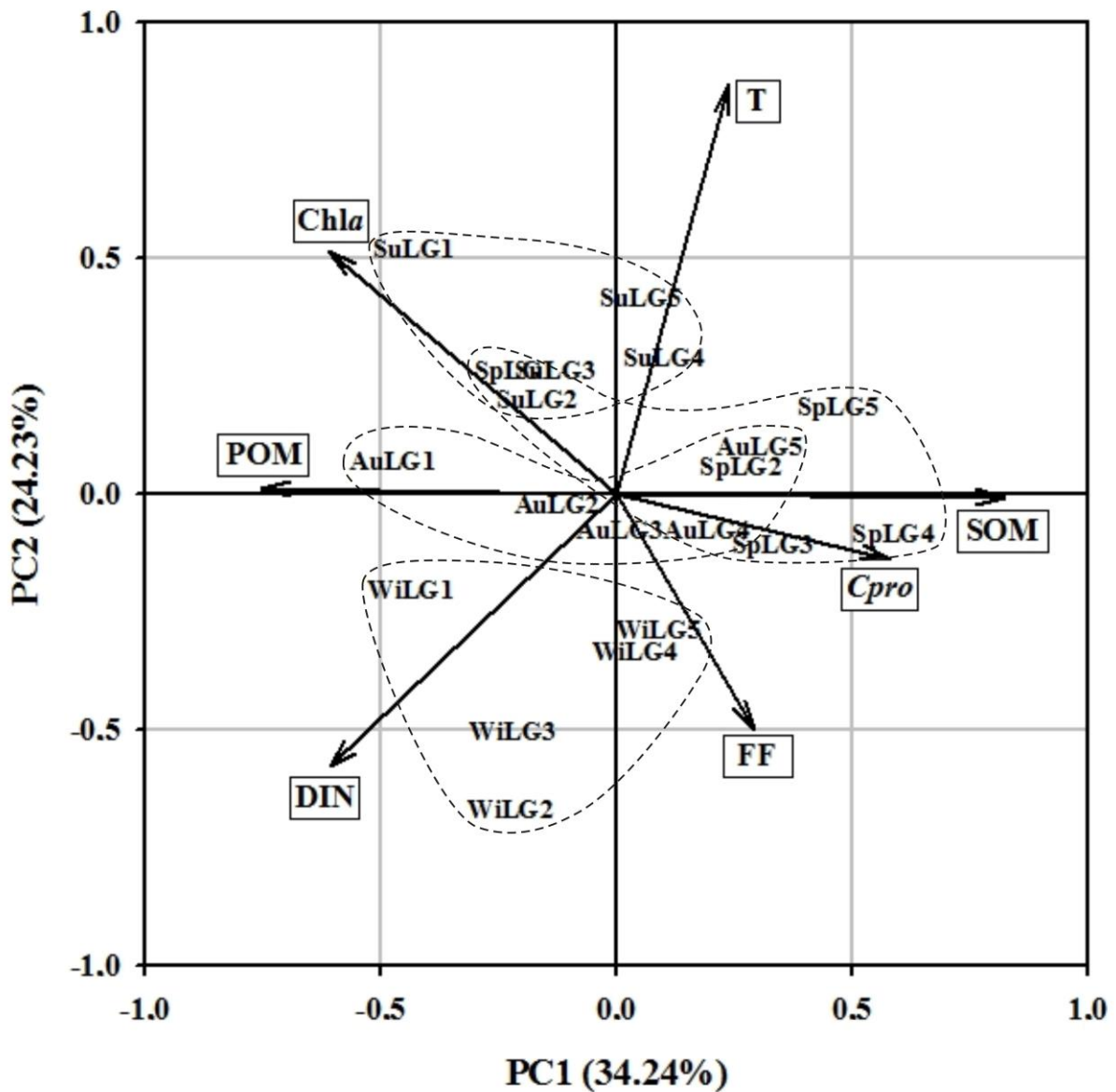


Figure 3.3.2.1. 2D-PCA plot. Due to the high number of samples, only the centroids were represented. Dashed lines represent the different seasons (Sp: Spring, Su: Summer, Au: Autumn, Wi: Winter).

3.3.3. Macrofaunal species/abundance distributions

A total of 53 benthic macrofauna species were identified in the study area. Abundances showed a clear seasonal pattern, with higher abundances in spring and lower abundances in summer and autumn (Table 3.3.3.1). MDS ordination of the samples showed a clear separation of stations in spring and winter, while those of summer and autumn displayed a more disperse ordination (Figure 3.3.3.1).

Table 3.3.3.1. Abundances (indiv m⁻²) of the main ecological groups of benthic macrofauna found in the Mar Menor lagoon during the study period (Number of species is shown in parenthesis).

Season	Station	Herbivores	Deposit-feeders	Suspension-feeders	Carnivores	Epifauna	Infauna
Spring	LG1	159 (2)	4460 (11)	386 (6)	369 (8)	721 (13)	4653 (14)
	LG2	244 (4)	6074 (12)	415 (6)	250 (4)	2835 (11)	4148 (15)
	LG3	91 (4)	6420 (12)	375 (7)	472 (6)	2216 (14)	5142 (15)
	LG4	119 (3)	2568 (6)	2352 (6)	182 (2)	4420 (8)	801 (9)
	LG5	688 (2)	2466 (12)	983 (7)	699 (7)	1983 (14)	2853 (14)
Summer	LG1	28 (3)	886 (9)	119 (3)	148 (5)	306 (10)	875 (10)
	LG2	182 (3)	744 (7)	216 (5)	57 (5)	262 (10)	937 (10)
	LG3	23 (2)	1699 (8)	119 (7)	57 (5)	177 (10)	1721 (12)
	LG4	17 (2)	659 (5)	420 (5)	63 (2)	460 (9)	699 (5)
	LG5	6 (1)	182 (4)	153 (4)	23 (1)	171 (5)	193 (5)
Autumn	LG1	17 (2)	307 (7)	244 (6)	108 (3)	199 (8)	477 (10)
	LG2	17 (2)	659 (5)	131 (2)	97 (3)	154 (6)	750 (6)
	LG3	6 (1)	1108 (6)	1307 (4)	57 (6)	233 (9)	2245 (8)
	LG4	28 (1)	1091 (6)	790 (6)	91 (2)	1216 (8)	784 (7)
	LG5	11 (1)	273 (3)	335 (6)	6 (1)	273 (6)	352 (5)
Winter	LG1	0 (0)	949 (11)	159 (6)	80 (5)	188 (8)	1000 (14)
	LG2	466 (5)	6409 (13)	239 (6)	233 (6)	2852 (13)	4495 (17)
	LG3	63 (2)	3097 (11)	256 (4)	205 (4)	620 (8)	3001 (13)
	LG4	34 (2)	727 (9)	875 (4)	102 (3)	1454 (10)	284 (8)
	LG5	477 (1)	1392 (9)	545 (5)	199 (4)	1255 (9)	1358 (10)

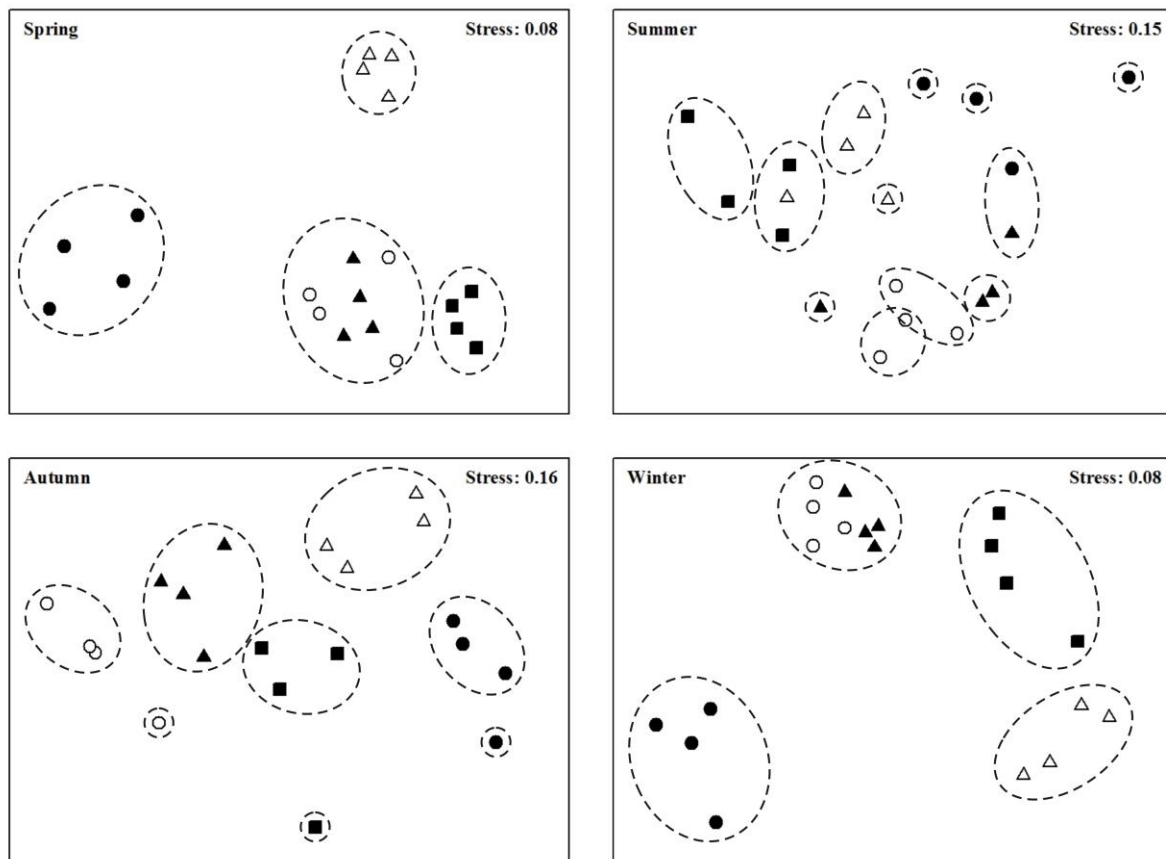


Figure 3.3.3.1. MDS ordination of macrofauna samples in the Mar Menor lagoon during the study period (LG1: black circles, LG2: white circles, LG3: black triangles, LG4: white triangles, LG5: black squares). Dashed lines represent clusters based on Bray-Curtis similarities of at least 60%.

PERMANOVA analysis showed significant differences among the samples for both factors considered, location (station) and season and the combination of both.

DISTLM marginal tests identified SOM as the main source of variability for the species/abundance matrix, explaining 11.91% of the variability, followed by *C. prolifera* biomass (10.64%) and chlorophyll a concentrations (7.25%). The inclusion of all the variables considered in the model only explained a total of 40.44% of the total variability (Table 3.3.3.2).

Table 3.3.3.2. Results of the DistLM analysis of the samples.

Marginal tests				Sequential tests				
Variable	Pseudo-F	P	% Var	Variable	Pseudo-F	P	% Var	% Cum
SOM	10.5460	0.0001	11.91	SOM	10.546	0.0001	11.91	11.91
<i>C. prolifera</i>	9.2947	0.0001	10.65	Chl <i>a</i>	8.0963	0.0001	8.38	20.29
Chl <i>a</i>	6.1002	0.0001	7.25	<i>C. prolifera</i>	5.3087	0.0001	5.20	25.50
FF	4.9202	0.0001	5.93	POM	4.5116	0.0001	4.23	29.72
POM	4.3345	0.0001	5.26	T	3.7405	0.0003	3.38	33.10
T	2.4543	0.0122	3.05	DIN	5.1675	0.0001	4.42	37.53
DIN	2.3726	0.0140	2.95	FF	3.5199	0.0005	2.91	40.44

3.3.4. Interpretation of results in the context of climate change

The analysis of recorded environmental variables confirmed the existence of strong environmental gradients in the study area. These gradients seem to have a small influence on macrobenthic species distributions that are mostly defined by the organic matter content of the sediments and the presence of *C. prolifera*. The existence of a well developed bed of the macroalga is allowing the settlement of this benthic community and its survival is determined by the presence of *C. prolifera* in the lagoon.

Temperature also has a very strong influence on the community structure. The extremely high temperatures of the summer have a clear effect on benthic species abundances, causing an impact on benthic community structure. It is likely that future increases of water temperatures or longer warmer seasons in the area as a consequence of climate changes may originate a severe impact on these communities and the processes and services they support.

3.4. The Tyligulskyi lagoon

3.4.1. Study area and sampling procedure

The macrozoobenthos in the Tyligulskyi Liman lagoon in 2001 - 2011 was characterized on the basis of observations conducted at 59 stations during 12 missions of the Odessa branch of the Institute of Biology of the Southern Seas of the National Academy of Sciences of Ukraine. Samples were predominantly collected in the eastern part of the lagoon. The benthic macrofauna in the shallow coastal areas with the depths of <1.3 m at 45 stations (76.3 % of the total number) was the most comprehensively studied. Location of the sampling

stations and their zoning, depending on the environmental factors of influence are shown in Figure 3.4.2.1.

The hydroecological conditions in the district C are formed under the influence of water exchange with the sea through the connecting canal, which usually operates from late April or early May till late July or mid-August. In April and June the quality of sea water inflowing into the lagoon is influenced by the river runoff of the Dnieper and the Southern Bug (during the spring flood). In this regard, toxic pollutants, e.g., oil products, dissolved and emulsified in the sea water, may get into the southern part of the lagoon. Moreover, there is an international automobile highway with heavy traffic which runs along the southern coast of the lagoon and can be a source of water and sediment pollution with heavy metals contained in depleted fuel products in the lagoonal district C. Deep-water bottom sediments in this district of the lagoon are characterized by the magnesium-sodium fine-dispersed silt muds which have an enhanced capacity to absorb pollutants from the near-bottom waters.

The lagoonal districts A and B are separated from the district C by relatively shallow bridge in the area of Chilova spit with the depths of 1.5 - 4.5 m, which prevents penetration of sea water towards the north. In spring and during the periods of intense rainfall in summer, the hydroecological conditions in the district A, especially in the shallow northern part of it, are affected by the water quality of the Tyligul river runoff. The sediments in the districts A and B at the depths of > 1.3 m are represented by silt with coquina.

The coastal areas in the three districts are characterized by a high mosaicity of sediments dominated by silt with sand and coquina, silt with sand and muddy sand. The waters of the shallow coastal zone are mixed up to the bottom, as a result of wind action, and have a homogeneous hydrological and hydrochemical vertical structure.

In the summer, depending on wind conditions and the vertical thermohaline water stratification, conditions for development of hypoxia and anoxia at the depths of > 5 m (the upper quasihomogeneous water layer) in the lagoon may occur, which creates unfavorable conditions for life of the benthic organisms. Under strong winds with the component being transverse to the lagoonal axis, hypoxic bottom water at the deeper areas of the lagoon may lens out into the shallow waters of the coastal zone. Moreover, hypoxic conditions may develop in the shallow waters in the northern part of the lagoon during long periods of calms, as a result of intensive biochemical oxidation of organic matter from dead macrophytes.

3.4.2. Results of the MDS ordination of samples

Faunal samples were taken in three areas of the Tyliguskyi lagoon (station A, station B and station C) in shallow bottoms ($< 1,3$ m) and deeper bottoms ($> 1,3$ m).

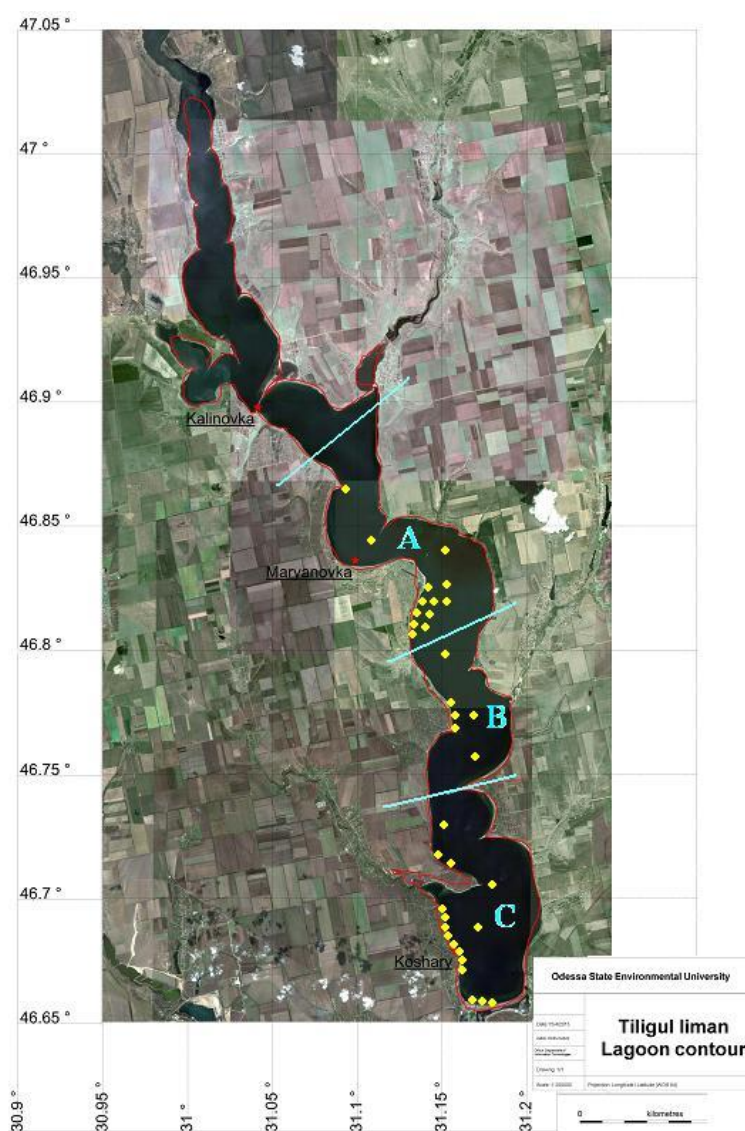


Figure 3.4.2.1. Sampling stations (A, B and C) for macrozoobenthos sampling in the Tyligulskyi lagoon. Yellow points represent subsamples into the sampling station.

One main feature is clear from the MDS analysis is the ordination according with the depth. There is a greater degree of variability in faunal community structure of the deeper samples ($> 1,3$ m). The Cluster analysis indicates a high similarity between the shallow sampling stations and a high dissimilarity of the deeper station C.

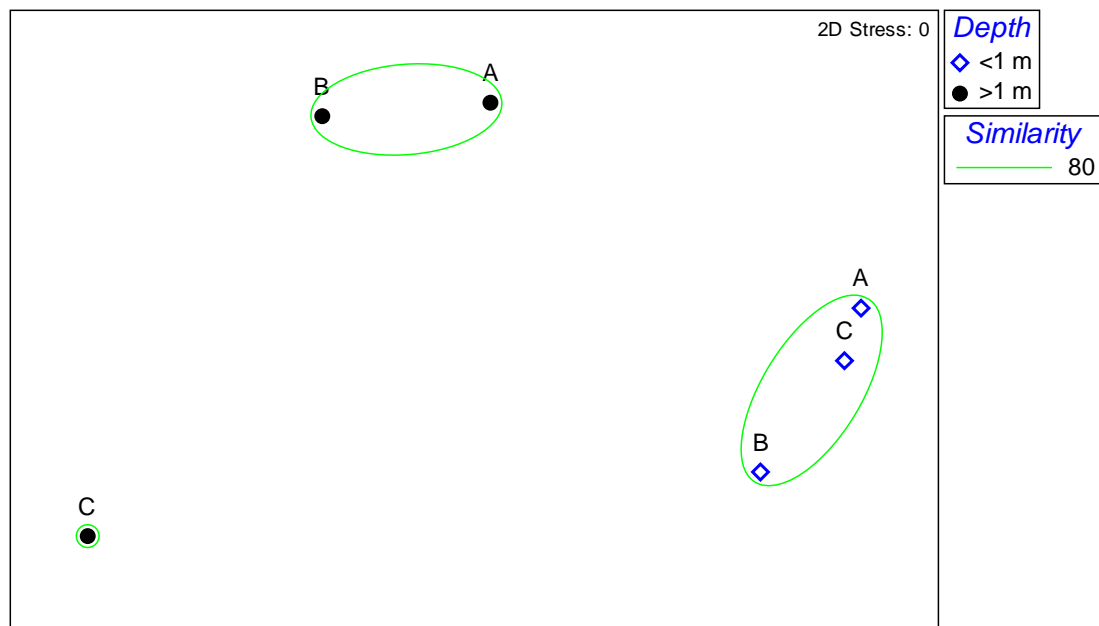


Figure 3.4.2.2. MDS ordination of macroinvertebrates assemblages from north (A), center (B) and south (C) of Tyligulskyi lagoon with superimposed cluster analysis at similarity levels of 80% (green line). The data was Log(x+1) transformed before the calculation of Bray-Curtis similarities (stress<0.01).

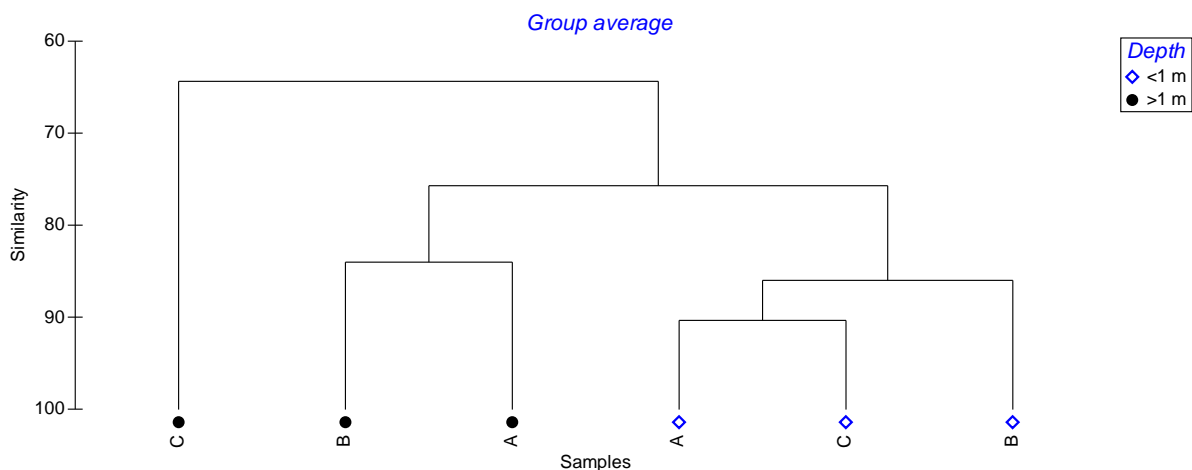


Figure 3.4.2.3. Cluster analysis of macroinvertebrate assemblages.

3.4.3. Variability on benthic macrofauna species/abundance distributions

The detailed information on the macrozoobenthos species composition, number and biomass of various species at the depths of less and more than 1,3 m in the three selected areas is provided in LAGOONS Report D3.2.3.

In 1980-1983, according to (Polischuk, Zambriborsch, Timchenko et al., 1990), 64 species of bottom-dwelling invertebrates were observed in the Tyligulskyi Liman lagoon, 76,2 % of which were marine species, and 16 % - both brackish water and freshwater ones. By the number of specimens and the biomass Molluscs were the dominant group, which in various years comprised 21.9 to 83,2 % of the total number and 93,0-95,6 % of the total biomass.

According to the data obtained in 2001 – 2011, 95,5 % of the number of specimens and 99,6 % of the biomass of the bottom macrofauna in the lagoon were formed by 29 species (83 % of the total number) of euryhaline marine complex. The brackish water fauna was represented by 6 species (17%). Molluscs prevailed among the basic taxonomical groups. The contribution of them in the coastal zone at the depth of < 1,3 m comprised 67,7% of the total number and 79.7% of the total biomass, and in the deeper areas – 79,4% and 98,4%, respectively.

In the coastal zone (at the depths of <1,3 m) in all of the three districts of the lagoon there observed the greatest species diversity of the macrozoobenthos. Of the 35 species inhabiting, the species of *Harmothoeimbricata* (L.) is not found in the districts A and B, as well as *Microdeutopusgryllotalpa* A. Costa– in the district B.

The macrozoobenthos species diversity at the depths of > 1,3 m in all of the three districts is narrower than that of the coastal zone (A – 27 species, B – 28, C – 7). The difference is especially notable for the district C, where only 7 species out of the 35 observed in the coastal zone were registered at the depth of more than 1,3m.

In the districts A and B the total number of macrozoobenthic organisms in the shallow coastal zone is 1,8 and 1,2 times higher than the relevant indices for the depths of > 1,3 m, respectively, and the total biomass, on the contrary, 4,6 times and 5,6 lower. In the district C values of the number of macrozoobenthic organisms in the coastal zone and farther off the coast are close in value, and the biomass farther off the coast is 13,5 times greater than at the coastline on account of *Mytilasterlineatus*.

3.4.4. Interpretation of results in the context of climate change

Changes in climatic conditions in the north-western part of the Black Sea and the adjacent Tyligulskyi lagoon become apparent in a long-term trend of rise in air and water

temperatures, increased evaporation rates, reduced wind speeds (Ilyin, Yu.P., et al, 2012) and freshwater runoff from the drainage area of the lagoon (LAGOONS Report D2.1d). The average annual rate of the Tyligul river decreased from 1,06 m³/s for the period of 1961-1989 to 0,67 m³/s for the period of 1991-2007.

The long-term trend of increase in water salinity in the Tyligulskyi lagoon, caused by the increase in the air temperature, evaporation, decrease in the fresh water inflow from the drainage area and accumulation of salts inflowing into the lagoon with the sea water, leads to an increase in the proportion of euryhaline marine species of the macrozoobenthos, on the account of a decline in brackish water species and gradual extinction of freshwater species.

More intense heating of the surface layer of the lagoon waters in the spring and the summer, and the resulting intensification of production-and-destruction processes in the ecosystem, paired with a decrease in wind activity, lead to an increased frequency of hypoxic and anoxic conditions both in the bottom layer of the deeper areas of the lagoon and in the shallow water in the period of prolonged calm.

As a result of the combined action of the above mentioned climatic factors, the macrozoobenthos species diversity has decreased from 64 in the early 1980s to 35 species at present.

4. Evaluation of the response of key lagoonal species to changes in main environmental variables

4.1. Introduction

Temperature has long been known to modify the chemistry of a number of chemical pollutants resulting in significant alterations of their toxicities, e.g. for fish. It is also generally accepted that higher temperatures increase the rate of uptake of pollutants via changes in ventilation rate, in response to an increased metabolic rate and decreased oxygen solubility (Kennedy and Walsh, 1997). An underlying mechanism of this interactive toxicity is that temperature alters the toxicokinetics of chemical pollutants in exposed biota (Buchwalter et al., 2003; Lydy et al., 1999; Maruya et al., 2005). Another mechanism probably influencing this enhanced toxicity is the fact that increasing temperatures can alter homeostasis and other key physiological mechanisms, thereby exacerbating the adverse effects of contaminants (Anderson and Peterson, 1969; Broomhall, 2002, 2004; Gordon, 2003; Heath et al., 1994; Patra et al., 2007). Some populations, particularly those living at the edge of their homeostatic or physiological tolerance range, may be more vulnerable to the dual stresses of climate change and contaminant exposures (Anderson and Peterson, 1969; Gordon, 2003; Heath et al., 1994; Patra et al., 2007).

The analysis carried out in this study compared the current situation in each coastal lagoon and future conditions due to climate change effects. The effects of actual temperature average have been compared with the future temperature amphipods will stand and also the effect of toxic sediments from their natural environment (Dorgelo, 1976, 1977; Kinne, 1983; Steele and Steele, 1991; Charmantier, 2000; Cacabelos et al., 2010).

Table 4.4.1.1. Ranges of environmental variables influencing studied amphipod species survival and distribution.

Species	Variables	Range in the lagoon	Optimum	Critical	Source
<i>G. aequicauda</i>	T ^a (°C)	10 - 30	18-20	35	(Gilabert, 2001; Prato et al., 2008)
	Salinity (‰)	39 - 47	15-36	0-3	(Gilabert, 2001; Prato et al., 2008)
<i>G. chevreuxi</i>	T ^a (°C)	12- 23	8-25	26	(Subida et al., 2005)
	Salinity (‰)	0 - 36	0-15	No data	(Vaz and Dias, 2008).
<i>Dikerogammarus sp.</i>	T ^a (°C)	(- 0) - 26	No data	27.8	(Wijnhoven et al., 2003)
	Salinity (‰)	0 - 6.5	No data	25	(Bruijs et al., 2001)

4.2. The Ria de Aveiro lagoon

4.2.1. Acute toxicity tests

Following standard protocols for acute toxicity test, the present study revealed that *G. chevreuxi* mortality increased with temperature. Figure 4.2.1.2 shows *G. chevreuxi* mortality at three temperatures and sediments from different stations ordered in a gradual range of toxicity on a 10 days assay. These data evidenced that the mortality rate of animals increase dramatically with the temperature (ANOVA, $p < 0.05$). We can appreciate that control station shows mortality rate below the accepted 20% control mortality for routine toxicity testing under the three temperatures tested. We also run a 26°C test; in this case we obtained a 0% survive rate in all stations. On the other hand, in the other three stations the mortality statistically increases with the temperature (ANOVA $p < 0.05$; Post-hoc test, Tukey test). Even under low temperature (15°C) the *G. chevreuxi* mortality rate increases in Bico, Chegado, Estarreja. It may be due to toxic sediments. This results are coincident with the ones obtained in Mar Menor test. Besides, the station with more metals concentration (Estarreja) is the one which presents the highest mortality rate, in 20°C and 23°C confirming that the mortality increased as temperature increased in toxic sediments.



Figure 4.2.1.1. Pictures showing the sampling site and the experimental set up of laboratory acute assays with *Gammarus chevreuxi*.

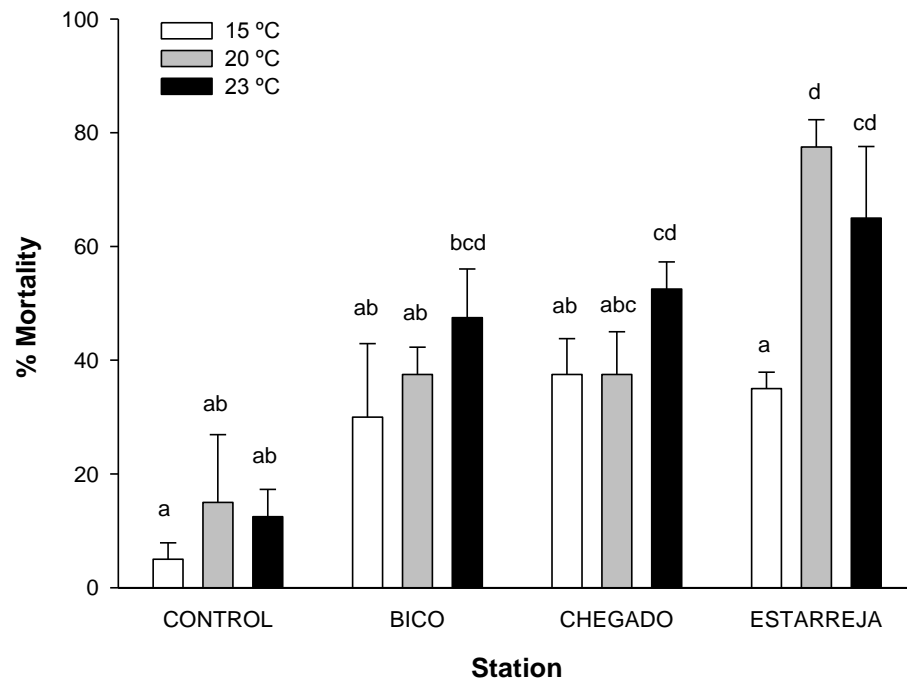


Figure 4.2.1.2. *Gammarus chevreuxi* mortality at three temperatures and sediments from different stations ordered in a gradual range of toxicity on a 10 days assay. The letters show the stations with significant difference, Post-hoc Tukey test ($p < 0.05$). Vertical bars indicate the mean \pm SE.

4.2.2. Behavioural tests

Regarding to behavioural test (Fig. 4.2.2.1), in this case there is not previous information about the daily pattern of *G. chevreuxi*. Previous tests revealed that this amphipod showed an increase of its activity during the light period. This is the opposite behaviour as *G. aequicauda*. Nevertheless, after the 48h test with two temperatures and same sediments than in acute toxicity test the results revealed that *G. chevreuxi* experiments a reduction of the activity when the control station was compared with the other three stations, for both temperatures, as expected. The study showed also significant differences in swimming activity of Control and Bico stations between the two temperatures exposed. *G. chevreuxi* has more vertical swimming at 23°C in this two stations, one without contamination and the other one with low levels of metals. On the other hand, the amphipod shows a decrease of

swimming activity in both temperatures when is exposed in stations with higher metals concentration.

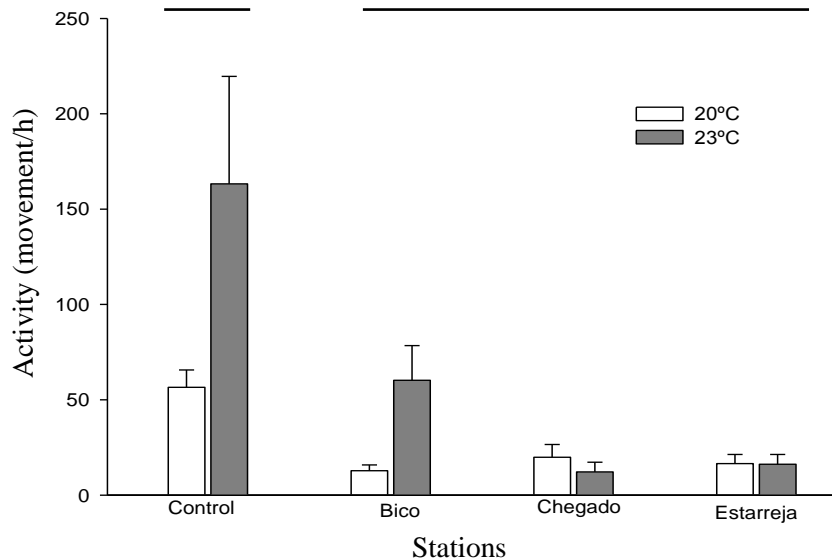


Figure 4.2.2.1. Activity recorded for *G. chevreuxi* during light period. The horizontal bars show the stations with significant differences with post hoc Tukey test HSD ($p < 0.05$). Vertical bars indicate the mean \pm SE. Asterisks indicate significant differences at the same station at different temperatures (Student's t test, $p > 0.05$).

4.2.3. Interpretation of results in the context of climate change

The Aveiro region is industrially, agricultural and commercially important and, historically, this has caused anthropogenic pressure on the lagoon, namely through industrial effluent loadings in the past. Some of these are still of particular concern in specific areas of the lagoon as a result of this historical contamination by metals, e.g. mercury contamination in the 2 km² inner Laranjo basin (Pereira et al., 2009). The sediments in this basin area near the industrial complex of Estarreja are still contaminated, and exceed statutory standards for mercury (Pereira et al., 2009). The sampling stations were chosen along a gradient of metals concentration. Estarreja is the station with highest contamination, metals content decrease in Chegado and Bico respectively. We analyzed the likely relation between the increase of water temperature due to the climate change and its effects in toxic sediments for *G. chevreuxi* as

amphipod representative of this area. The results showed an increase of mortality rate with an increase of temperature, it also happens to *Corophium insidiosum* (Prato et al., 2008b). As both amphipods are present in Ria de Aveiro, it is suggested that the obtained results are due to a bioenergetic disadvantage. Regarding the reduction of locomotor activity of *G. chevreuxi* in sublethal experiment, similar results have been found in *Talitrus saltator* (Ugolini et al., 2012) showing higher sensitivity to sediments contaminated with different Hg concentrations. These results showed the possible negative effects that an increase of water temperature could have in *G. chevreuxi*, since disturbance of the role of key species could cause cascade effects at higher levels (population and community). Therefore, focusing on the behavioural responses of all species with a key role in ecosystems could markedly improve the environmental risk assessments.

4.3. The Mar Menor and Tyligulskyi lagoons

4.3.1. Selection of the test species

The Mar Menor and Tyligulskyi lagoons present many similarities regarding temperature and salinity. Both systems have big temperature gradient between Winter and Summer periods. The high temperatures occur also during the driest period. In addition, both lagoons are characterised by scarce freshwater inputs. Mar Menor presents ephemeral watercourses (called wadis), which are generally inactive, while Tylihulskyi lagoon receives freshwater from Tyligul River, which nowadays decreased its runoff considerably. Due to current hydrological situation both lagoons are already unable to compensate the water losses through evaporation. This phenomena cause an increase of water salinity, hence, causing a shift of microphytobenthos distribution towards an increase in salt water species.

Gammarus aequicauda is one of the most common and abundant amphipods from lagoonal and brakish environments of the Mediterranean and Black Sea and a common species present in both studied lagoons. This amphipod tolerates a highly variable salinity, including hypo-osmotic conditions (Prato and Biandolino, 1990; Petrescu, 1994; Delgado et al., 2009). Due to the similarities between both lagoons in key environmental variables, as temperature and salinity, and taking in account that *G. aequicauda* is present in Mar Menor and Tylihulskyi we had run acute and sublethal toxicity test with metals in order to analyze the possible response to a climate changes scenario by estuarine organisms.

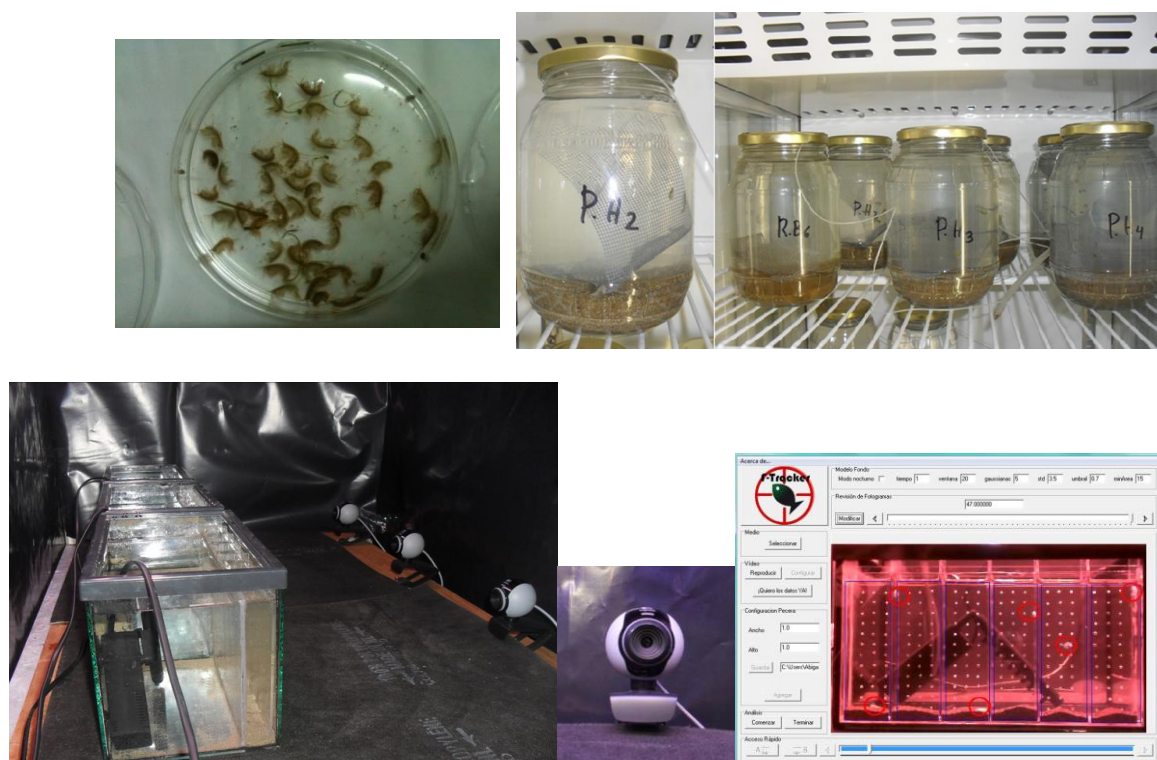


Figure 4.3.1.1. Pictures showing the experimental set up of laboratory acute and behavioural assays with *Gammarus aequicauda*.

4.3.2. Acute toxicity tests

Regarding acute toxicity test, the present research revealed that mortality of *G. aequicauda* increased with temperature. Figure 4.3.2.1 shows *G. aequicauda* mortality, after being exposed, for 10 days, to sediments from different stations and at three temperatures in a complete cross design experiment. These data evidenced that the mortality rate of animals increase dramatically with the temperature (ANOVA, $p < 0.05$). We can appreciate that the control station shows a mortality statistically similar to Station 1 and Station 2, both with toxic sediments. As well, presented similarities with the mortality results of 25°C of Station 3, which present the highest metal concentration. Besides, mortality rates were significantly affected by temperature in each station (Post-hoc, Tukey test), they showed an increase in mortality rate at 25°C and 30°C. We run also experiments at 35°C and the results showed a 100% of mortality including in the Control station. These data evidenced that the mortality increased with temperature. Although, all treatments exhibited low control mortality rates

(below the accepted 20% control mortality for routine toxicity testing), mortality rates were significantly affected by temperature and sediments. These results are supported also by Prato (Prato, et al., 2009) for the amphipod *G. aequicauda* and by the works done by Khan et al. (2007) in crayfish. The temperature-related increase in toxicity of metals has been reported in other crustaceans (Vernberg and Vernberg, 1972; O'Hara, 1973) and invertebrates (Bryan, 1971; Manley, 1983; Depledge, 1987; Khan et al., 2000).

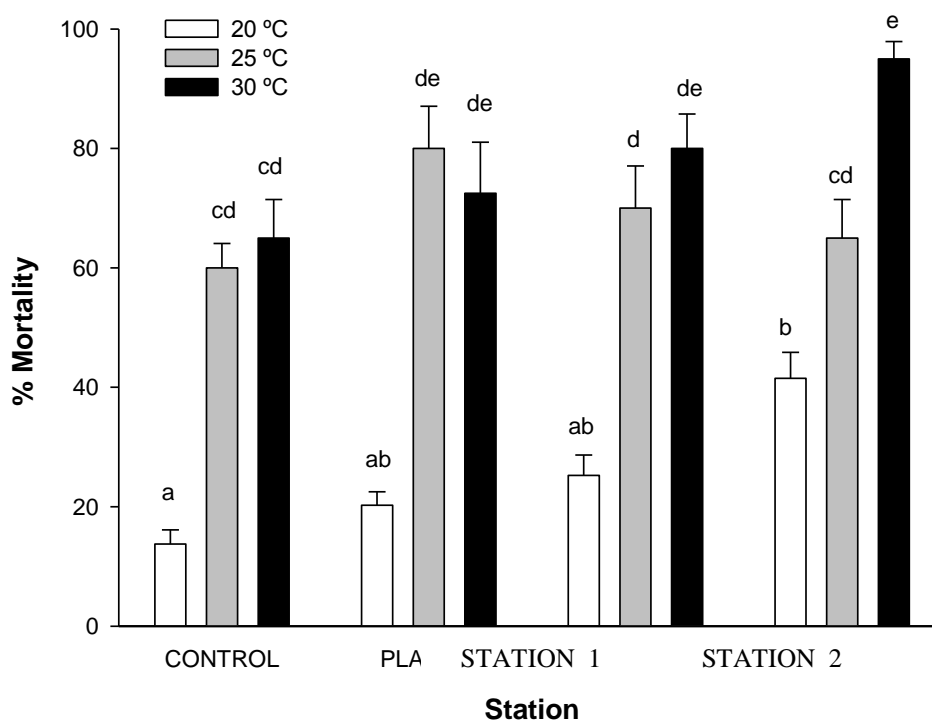


Figure 4.3.2.1. *Gammarus aequicauda* mortality at three temperatures and sediments from different stations ordered in a gradual range of toxicity on a 10 days assay. The letters show the stations with significant difference, Post-hoc, Tukey test ($p < 0.05$). Vertical bars indicate the mean \pm SE.

4.3.3. Behavioural tests

Previous studies from Morillo-Velarde (2010) revealed that *G. aequicauda* displays a nocturnal activity pattern, with increased activity during the dark period. Their study also showed that the most noticeable effects of a metal (Cd) was observed in the total movements recorded during the dark period. On this basis we study the activity of *G. aequicauda* during

48h with several temperatures and using same sediments as in the toxic test. The results (Fig. 4.3.3.1) showed again this daily pattern with higher activity during dark period. We observed that at 30°C the activity showed a significant increase in the control station, which may be due to an escape response to an unfavorable condition (a primary reaction of animals to stressful conditions) (Gerhardt, 1999). This behaviour is also observed in the amphipods tested in Station 1 sediment. On the other hand in Station 2 we observed a statistical decrease of the activity in both temperatures. It might be due at sediments toxicity is high enough to affect their locomotion escape response. (Two way ANOVA, $p < 0.005$ and Post-hoc test, Tukey test). These results agree with previous studies (Wallace and Estephan, 2004) that found vertical swimming activity more sensitive to Cd exposure because of the presumed greater energetic costs associated with producing enough thrust to attain the lift required to make vertical ascent into the water.

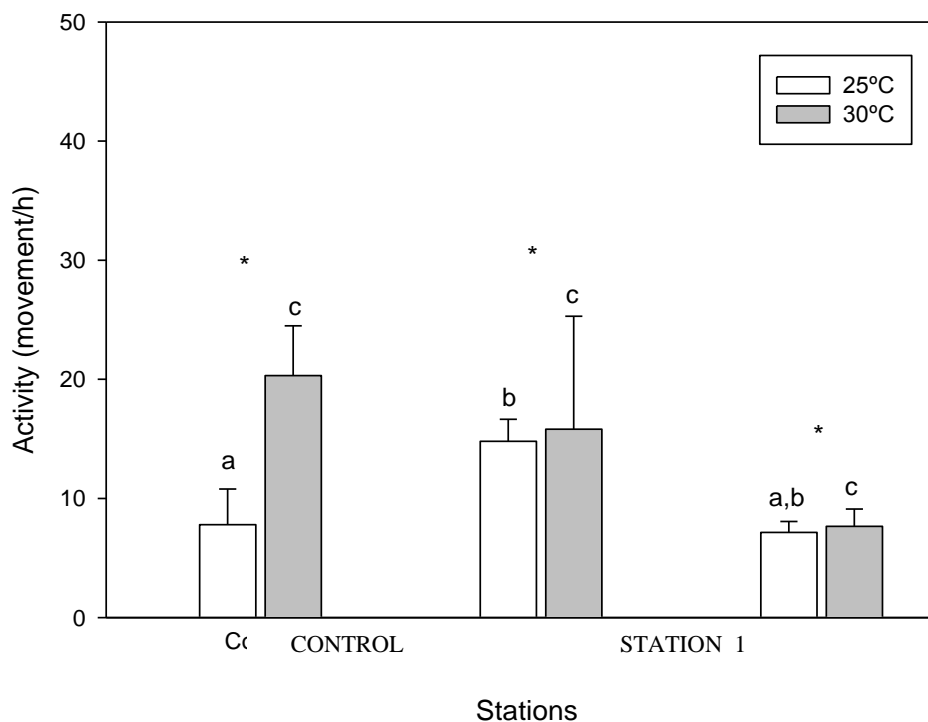


Figure 4.3.3.1. Activity recorded for *G. aequicauda* during dark period. The letters show the stations with significant difference post hoc test of Tukey ($p < 0.05$). Vertical bars indicate the mean \pm SE. Asterisks indicate significant differences at the same station at different temperatures (Student t test, $p > 0.05$).

4.3.4. Interpretation of results in the context of climate change

The results of the present trials have provided information about a free-living epibenthic amphipod, as a potential test species in the sediments toxicity assessment. *Gammarus aequicauda* was chosen for many reasons. It is readily available and easy to handle. Also it shows a high tolerance to different salinities, temperatures and is sensitivity to the reference toxicant.

Sensitivity to the heavy metals has been reported by Prato (Prato and Biandolino, 2005; Morillo-Velarde et al., 2011), to salinity by Delgado (L. Delgado et al., 2011) and for temperature by Prato (E. Prato, 2011). Our study considers the correlation between temperature and heavy metals as a likely situation due to climate change. The results showed that *G. aequicauda* present a low survival in sediments with high heavy metal concentration and also there is an influence once the exposure happened at high temperature. On the other hand behavioural test allowed us to study the effects of sublethal exposure in the amphipods. Even if the heavy metal concentrations have not lethal levels the impact in the organism, their locomotion can be affected which is required by gammarids to find food, avoid predation, and mate, like other biologic functions. The results of our behavioural test showed a clear effect in the vertical swimming activity due to a high temperature exposure in the control station while in stations with heavy metals the reaction is again a decrease of the activity but likely due to the effects of the toxic sediments.

4.4. **The Vistula lagoon**

4.4.1. Preliminary results of behavioural tests

The Ponto-Caspian amphipod, *Dikerogammarus sp.* is a recent and successful invader of the River Rhine. As it has an extensive invasion history in Europe, it is thought to have potential to reach and invade the Great Lakes in America. It has dispersed over large distances in a short time (Bruijs et al., 2001). By these capacities for adaptation *Dikerogammarus* may be able to survive ballast water exchange and subsequently be dispersed over large distances by means of ballast water and to develop large populations in temperate areas on a global scale.



Figure 4.4.1.1. Pictures showing the sampling site and the experimental set up of laboratory acute assays with *Dikerogammarus sp.*

Behavioural responses are linked to complex biochemical and physiologic changes and may act as sensitive indicators of the sublethal effects of pollutants. This trial investigates for the first time changes in the locomotors activity rhythms of the amphipod *Dikerogammarus* exposed to tree different temperatures as a model to study the effect of likely Climate changes on an ecologically important species. In the light of these preliminary results, the *Dikerogammarus* showed a nocturnal behaviour. We exposed *Dikerogammarus* to three different temperatures in order to studied his natural behaviour (18°C) and the reaction of the amphipod to high temperature, 26°C temperature is still inside the current rate of the lagoon. On the other side we run a third temperature, 29 °C as extreme one expected due to climate change events. The results showed (fig. x) amphipod presents higher activity during the dark part of the day (18°C). Therefore when it is exposed a high temperature its behaviour change increasing the diurnal activity. It may be due a escape response to an unfavorable

condition (Gerhardt 1999). Hence the preliminary results showed that the effects of the temperature during the period with higher activity, which correspond with dark phase, showed that the vertical activity of *Dikerogammarus* present an tendency to decrease, but we this preliminary results were not support for an robust statistic.

Therefore when it is exposed to an extreme temperature (29°C) the day activity decreases again (fig 4.4.1.2-3). Probably, ventilatory activity increased to a maximum as water temperature increased, because of a higher metabolic rate on one hand and decrease in dissolved oxygen concentration on the other. This energy expenditure constitutes a detriment in the swimming capacity. But to assure this results further research would be required in order to optimize the methodology and obtain statistical difference.

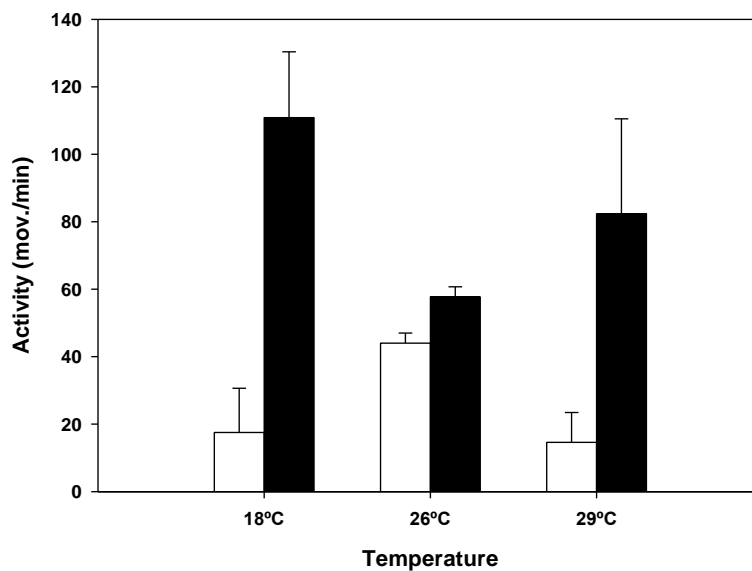


Figure 4.4.1.2. Movements per minute of *Dikerogammarus* during a day. White bars represents the activity (mov./min) during the light phase of the day. Dark bars showed the activity during the dark phase of the day. Data is expressed as mean \pm S.D.

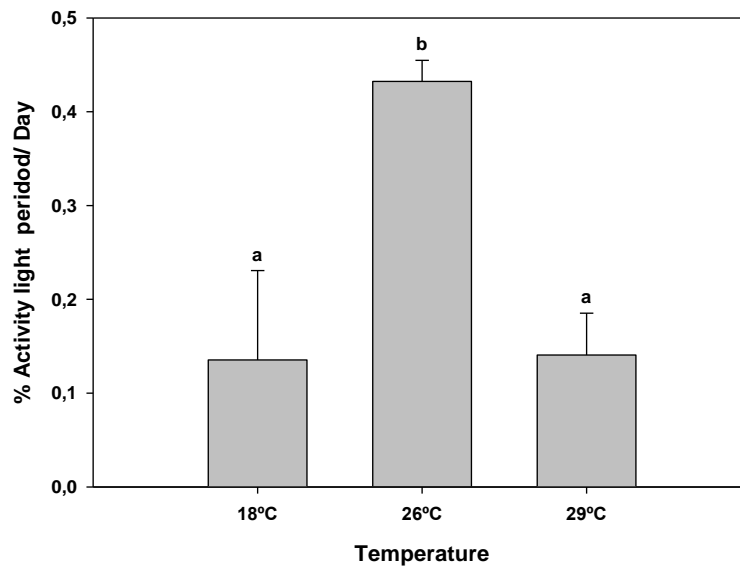


Fig 4.4.1.3. Percentage of activity during the light phase of the day. Data is expressed as mean \pm S.D. Different letters indicated statistically differences from each other (t-Student test. $p < 0.05$)

4.4.2. Interpretation of results in the context of climate change

Results of the preliminary behavioural tests showed a clear influence of temperature over amphipod's activity patterns. The variations observed in the number of movements per minute in the range between 18 and 26 °C corresponded to a 'typical' or expected result, considering the effects of temperature on amphipod metabolism and movement rate and the range of temperatures observed in the Vistula lagoon. However, temperatures of 29 °C, only 3 °C above the maximum temperature observed in the lagoon in summer, are enough to originate a detrimental effect over amphipod activity patterns and caused a decrease in the number of movements during the day. This fact highlights the existence of a certain degree of amphipod population vulnerability to the expected temperature increases due to climate change in the study area.

5. References

- Achenbach L., Eller F., Nguyen L.X., Brix H. 2013. Differences in salinity tolerance of genetically distinct *Phragmites australis* clones. AoB PLANTS.
- Amano M., Iida S., Kosuge K. 2011. Comparative studies of thermotolerance: different modes of heat acclimation between tolerant and intolerant aquatic plants of the genus *Potamogeton*. Annals of Botany 109: 443-452.
- Anderson J.M., Peterson M.R. 1969. DDT-sublethal effects on brook trout nervous system. Science, 164(3878): 440-1.
- Anderson M.J. 2001. Permutation tests for univariate or multivariate analysis of variance and regression. Canadian Journal of Fisheries and Aquatic Sciences 58, 626-639.
- Azevedo A., Sousa A.I., Lencart e Silva J.D., Dias J.M., Lillebø A.I. 2013. Application of the generic DPSIR framework to seagrass communities of Ria de Aveiro: a better understanding of this coastal lagoon. Journal of Coastal Research. SI 65, 19-24.
- Bethoux J.P., Morin P., Chaumery C., Connan O., Gentili B., Ruiz-Pino B. 1998. Nutrients in the Mediterranean Sea, mass balance and statistical analysis of concentrations with respect to environmental change. Marine Chemistry 63, 155-169.
- Bethoux J.P., Gentili B. 1999. Functioning of the Mediterranean Sea: past and present changes related to freshwater input and climate changes. Journal of Marine Systems 20, 33-47.
- Bouraoui F., Galbiati L., Bidoglio G. 2002. Climate change impacts on nutrient loads in the Yorkshire Ouse catchment (UK). Hydrology and Earth System Sciences 6, 197-209.
- Boyer K.E., Fong P. 2005. Macroalgal-mediated transfers of water column nitrogen to intertidal sediments and salt marsh plants. Journal of Experimental Marine Biology and Ecology 321, 59-69.
- Buijs M.C.M., Kelleher B., van der Velde G., de Vaate A.B. 2001. Oxygen consumption, temperature and salinity tolerance of the invasive amphipod *Dikerogammarus villosus*: indicators of further dispersal via ballast water transport. Archiv Fur Hydrobiologie 152: 633–646.

- Brun F.G., Hernández I., Vergara J.J., Peralta G., Pérez-Lloréns J.L. 2002. Assessing the toxicity of ammonium pulses to the survival and growth of *Zostera noltii*. Marine Ecology Progress Series 225: 177–187.
- Bryan G.W. 1971. The effect of heavy metals (other than mercury) on marine and estuarine organisms. Proc R Soc Lond B 177:389–410.
- Buchwalter D.B., Jenkins J.J., Curtis L.R. 2003. Temperature influences on water permeability and chlorpyrifos uptake in aquatic insects with differing respiratory strategies. Environmental Toxicology and Chemistry 22(11): 2806–12.
- Casagrande C., Dridi M.S., Boudouresque C.F. 2006. Abundance, population structure and production of macroinvertebrate shredders in a Mediterranean brackish lagoon, Lake Ichkeul, Tunisia. Estuarine, Coastal and Shelf Science, 66: 437–446.
- Cascabelos E., Lourido A., Troncoso J.S. 2010. Composition and distribution of subtidal and intertidal crustacean assemblages in soft-bottoms of the Ria de Vigo (NW Spain). Scientia Marina 74, 455–464.
- Carpenter S.R., Lodge D.M. 1986. Effects of submersed macrophytes on ecosystem processes. Aquatic Botany 26: 341–379.
- Charmantier G. 2000. Ontogeny of osmoregulation in crustaceans: the embryonic phase. American Zoologist 40, 971–971.
- Chambers P.A., Kalff J. 1985. Depth distribution and biomass of submersed macrophyte communities in relation to Secchi depth. Canadian Journal of Fisheries and Aquatic Sciences 42:701–709.
- Chubarenko B., Margoński P. 2008. The Vistula Lagoon. In: Schiewer U. (Ed.) Ecology of Baltic Coastal Waters. Ecological Studies 197: 167–195.
- Cunha A.H., Assis J.F., Serrão E.A. 2013. Seagrasses in Portugal: A most endangered marine habitat. Aquatic Botany 104, 193–203.
- Cywińska A., Róžańska Z. 1978. Zoobentos of the Vistula Lagoon (In Polish). Stud. Mater. Oceanolog., Biologia Morza 4: 145–160.

- De Casabianca M.L., Laugier T., Marinho-Soriano E., 1997. Seasonal changes of nutrients in water and sediment in a Mediterranean lagoon with shellfish farming activity (Thau Lagoon, France). *ICES Journal of Marine Science* 54, 905-916.
- Delgado L. Guerao G., Ribera C. 2009. The Gammaridea (Amphipoda) fauna in a Mediterranean coastal lagoon: considerations on population structure and reproductive biology. *Crustaceana*, vol. 82, no. 2, pp. 191–218.
- Depledge H.M. 1987. Enhanced copper toxicity resulting from environmental stress factors synergies. *Comp Biochem Physiol C* 87:15–19.
- Diaz R.J., Rosenberg R. 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. *Oceanography and Marine Biology Annual Review* 33, 245-303.
- Dore M.H.I., 2005. Climate change and changes in global precipitation patterns: What do we know? *Environment International* 31, 1167-1181.
- Dorgelo J. 1976. Salt Tolerance in Crustacea and Influence of Temperature Upon It. *Biological Reviews of the Cambridge Philosophical Society* 51, 255–290.
- Dorgelo J. 1977. Comparative Ecophysiology of Gammarids (crustacea-Amphipoda) from Marine, Brackish-Water and Freshwater Habitats Exposed to Influence of Salinity Temperature Combinations .4. Blood Sodium Regulation. *Netherlands Journal of Sea Research*, 11, 184–199.
- Eyre B.D., Ferguson A.J.P. 2002. Comparison of carbon production and decomposition, benthic nutrient fluxes and denitrification in seagrass, phytoplankton, benthic microalgae and macroalgae-dominated warm-temperate Australian lagoons. *Marine Ecology Progress Series* 229, 43-59.
- Ezowa E., Żmudziński L., Maciejewska K. 2005. Long-term trends in the macrozoobenthos of the Vistula Lagoon, southeastern Baltic Sea. Species composition and biomass distribution. *Bull. Sea Fisheries Inst.* 1(164): 55-73.
- Fernández-Torquemada Y., Sánchez-Lizaso J.L. 2011. Responses of two Mediterranean seagrasses to experimental changes in salinity. *Hydrobiologia* 669:21–33.

- Fong P., Zedler J.B., Donohoe R.M. 1993. Nitrogen vs. phosphorus limitation of algal biomass in shallow coastal lagoons. *Limnology and Oceanography* 38, 906-923.
- Gajewski L. (Ed.) 2010. Bottom investigations of the Polish part of the Vistula Lagoon including Elbląg Bay. Instytut Morski, Gdańsk. Typescript. [In Polish]
- Gerhardt A. 1999. Recent trends in online biomonitoring for water quality control. In: Gerhardt A (Ed.) *Biomonitoring of polluted water. Reviews on actual topics*, vol 9. Environmental Research Forum, TTP, Switzerland, pp 95–118
- Gilabert J. 2001. Seasonal plankton dynamics in a Mediterranean hypersaline coastal lagoon: The Mar Menor. *Journal of Plankton Research*, 23, 207–217.
- Glockzin M., Zettler M. 2008. Spatial macrozoobenthic distribution patterns in relation to major environmental factors- A case study from the Pomeranian Bay (southern Baltic Sea). *Journal of Sea Research*, 59: 144-161.
- Gordon C.J. 2003. Role of environmental stress in the physiological response to chemical toxicants. *Environ Res*; 92(1):1–7.
- Gorai M., Ennajeh M., Khemira H., Neffati M. 2011. Influence of NaCl-salinity on growth, photosynthesis, water relations and solute accumulation in *Phragmites australis*. *Acta Physiologiae Plantarum*, 33:963-971.
- Gross E.M., Hilt S., Lombardo P., Mulderij G. 2007. Searching for allelopathy interaction—State of the art and open questions. *Hydrobiologia* 584:77–88.
- Hansen K., Kristensen E. 1997. Impact of macrofaunal recolonization on benthic metabolism and nutrient fluxes in a shallow marine sediment previously overgrown with macroalgal mats. *Estuarine Coastal and Shelf Science* 45, 613-628.
- Hauxwell J., Cebrian J., Valiela I. 2001. Macroalgal canopies contribute to eelgrass (*Zostera marina*) decline in temperate estuarine ecosystems. *Ecology* 82, 1007-1022.
- Heath S., Bennett W.A., Kennedy J., Beitinger T.L. 1994. Heat and cold tolerance of the fathead minnow, *Pimephales promelas*, exposed to the synthetic pyrethroid cyfluthrin. *Can J Fish Aquat Sci*; 51(2):437–40.
- Ilyin Y.P., et al, 2012. *Hydrometeorological Conditions in the Ukrainian Seas. Volume 2. Black Sea, Sevastopol*, 421 p.

- Intergovernmental Panel on Climate Change (IPCC), 2001. Third Assessment Report - WG2. Technical Summary Climate Change 2001: Impacts, Adaptation and Vulnerability, 20-73.
- Jones J.I., Sayer C.D. 2003. Does the fish-invertebrate-periphyton cascade precipitate plant loss in shallow lakes? *Ecology* 84: 2155-2167.
- Kalugina-Gutnik A.A. 1975. Phytobenthos in the Black Sea. Kiev, 246 pp.(in Russian)
- Kautsky L. 1988. Life strategies of aquatic soft bottom macrophytes. *Oikos* 53: 126-135.
- Kennedy C.J., Walsh P.J. 1997. Effects of temperature on xenobiotic metabolism. In: Wood C.M., McDonald D.G. (Eds.), *Global Warming – Implications for Freshwater and Marine Fish*. Cambridge University Press, pp. 303–324.
- Khan M. a. Q., Ahmed S.A., Salazar A., Gurumendi J., Khan A., Vargas M., von Catalin B. 2007. Effect of temperature on heavy metal toxicity to earthworm *Lumbricus terrestris* (Annelida: Oligochaeta). *Environmental Toxicology*, 22, 487–494.
- Kinne O. 1983. Citation Classic - the Effects of Temperature and Salinity on Marine and Brackish Water Animals .2. Salinity and Temperature-Salinity Combinations. *Current Contents/Agriculture Biology and Environmental Sciences*, 22–22.
- Kovtun O.A. 2012. Phytobenthos in the Tyligulskyi Liman Lagoon (Black Sea, Ukraine). Ecological and Biological, Morphological and Taxonomic Features. LAMBERT Academic Publishing, 353 p.
- Krause-Jensen D., Christensen P.B., Rysgaard S. 1999. Oxygen and nutrient dynamics within mats of the filamentous macroalga *Chaetomorpha linum*. *Estuaries* 22, 31-38.
- LAGOONS. 2012. Description of common methodology. LAGOONS Report D3.1. pp. 40.
- LAGOONS. 2012. The Ria de Aveiro Lagoon – Current knowledge base and knowledge gaps. LAGOONS Report D2.1b. pp. 52.
- LAGOONS. 2012. The Tyligulskyi Lagoon – Current knowledge base and knowledge gaps. LAGOONS Report D2.1d. pp. 54.
- LAGOONS. 2013. Results of the problem base science analysis. LAGOONS Report D3.2. pp. 191.
- LAGOONS. 2013. Results of climate impact assessment. LAGOONS Report D5.1. pp. 107.

- Lehmann A., Castella E., Lachavanne J.B. 1997. Morphological traits and spatial heterogeneity of aquatic plants along sediment and depth gradients, Lake Geneva, Switzerland. *Aquatic Botany* 55: 281–299.
- Li F., Xie Y., Chen X., Chen X., Pan Y., Deng Z., Li X. 2011. Plant distribution can be reflected by physiological responses to salinity of three submerged macrophytes. *Fundamental and Applied Limnology*, 179: 159-167.
- Lissner J., Schierup H. 1997. Effects of salinity on the growth of *Phragmites australis*. *Aquatic Botany* 55: 247-260.
- Lloret J., Marín A., Marín-Guirao L., Velasco J., 2005. Changes in macrophytes distribution in a hypersaline coastal lagoon associated with the development of intensively irrigated agriculture. 2005. *Ocean and Coastal Management* 48, 828-842.
- Lloret J., Marín A., Marín-Guirao L. 2008. Is coastal lagoon eutrophication likely to be aggravated by global climate change? *Estuarine, Coastal and Shelf Science*, 78: 403-412.
- Lloret J., Marín A. 2011. The contribution of benthic macrofauna to the nutrient filter in coastal lagoons. *Marine Pollution Bulletin*, 62: 2732-2740.
- Łomniewski K. 1958. Vistula Lagoon (Zalew Wiślany). Państwowe Wydawnictwo Naukowe, *Prace Geograficzne*, 15: 106 pp.
- Łomniewski K. 1958. The firth of Vistula. PWN, Warsaw. [In Polish]
- Lozano-Cabo, F., 1954. Una campaña de prospección pesquera en el Mar Menor (Murcia). *Boletín del Instituto Español de Oceanografía* 66, 1-40.
- Luther H. 1951. Verbreitung und Ökologie der höheren Wasserpflanzen im Brackwasser der Ekenäs-Gegend in Süd-Finnland. II. Spezieller Teil. *Acta Bot. Fennica* 55: 1-61.
- Lydy M.J., Belden J.B., Ternes M.A. 1999. Effects of temperature on the toxicity of M-parathion, chlorpyrifos, and pentachlorobenzene to *Chironomus tentans*. *Archives of Environmental Contamination and Toxicology*, 37, 542–547.
- Manley R.A. 1983. The effects of copper on the behavior, respiration, filtration and ventilation activity of *Mytilus edulis*. *J Mar Biol Assoc UK* 63:205–222

- Maruya K.A., Smalling K.L., Vetter W. 2005. Temperature and congener structure affect the enantio selectivity of toxaphene elimination by fish. *Environ Sci Technol* 39(11):3999–4004.
- Massa S.I., Arnaud-Haond S., Pearson G.A., Serrão E.A. 2009. Temperature tolerance and survival of intertidal populations of the seagrass *Zostera noltii* (Hornemann) in Southern Europe (Ria Formosa, Portugal). *Hydrobiologia* 619:195–201.
- Matoh T., Matsushita N., Takahashi E. 1988. Salt tolerance of the reed plant *Phragmites communis*. *Physiologia Plantarum* 72: 8-14.
- McArdle B.H., Anderson M.J. 2001. Fitting multivariate models to community data: a comment on distance-based redundancy analysis. *Ecology* 82, 290-297.
- McGlathery K.J., Krause-Jensen D., Rysgaard S., Christensen P.B. 1997. Patterns of ammonium uptake within dense mats of the filamentous macroalga *Chaetomorpha linum*. *Aquatic Botany* 59, 99-115.
- McGlathery K.J., Sundbäck K., Anderson I.C. 2004. The importance of primary producers for benthic nitrogen and phosphorus cycling. In: Nielsen, S.L., Banta, G.T., Pedersen, M.F. (eds.). *Estuarine nutrient cycling: the influence of primary producers*. Kluwer Academic, Dordrecht, pp. 231-262.
- McGlathery K.J., Sundbäck K., Anderson I.C. 2007. Eutrophication in shallow coastal bays and lagoons: the role of plants in the coastal filter. *Marine Ecology Progress Series* 348, 1-18.
- Menendez M., Comín F. A. 1990. Consumption of macrophytes by invertebrates in Tancada lagoon (NE Spain). *Scientia Marina*, vol. 54, pp. 139–144.
- Mitvalli, A.H. 1970. Influence of Water Salinity on the Biological, Physiological and Biochemical Peculiarities of the Brown Alga in the Odessa Bay of the Black Sea. Abstract of thesis of the Candidate of Science (Biology), Odessa, 19 pp. (in Russian)
- Morillo-Velarde P.S., Lloret J., Marin A., Sanchez-Vazquez F.J. 2011. Effects of Cadmium on Locomotor Activity Rhythms of the Amphipod *Gammarus aequicauda*. *Archives of Environmental Contamination and Toxicology*, 60, 444–451.

- Nicholls R.J., Hoozemans F.M.J. 1996. The Mediterranean: vulnerability to coastal implications of climate change. *Ocean and Coastal Management* 31, 105-132.
- Nicholls R.J., Hoozemans F.M.J., Marchand M. 1999. Increasing flood risk and wetland losses due to global sea-level rise: regional and global analyses. *Global Environmental Change* 9, S69-S87.
- Palmer M.W. 1993. Putting things In even Belter order: the advantages of canonical correspondence analysis. *Ecology* 74: 2215-2230.
- Palomo L., Clavero V., Izquierdo J.J., Avilés A., Becerra J., Niell F.X., 2004. Influence of macrophytes on sediment phosphorus accumulation in a eutrophic estuary (Palmones River, Southern Spain). *Aquatic Botany* 80, 103-113.
- Patra R.W., Chapman J.C., Lim R.P., Gehrke P.C. 2007. The effects of three organic chemicals on the upper thermal tolerances of four freshwater fishes. *Environmental Toxicology and Chemistry* 26(7):1454–1459.
- Pearson T.H., Stanley S.O. 1979. Comparative measurement of the redox potential of marine sediments as a rapid means of assessing the effect of organic pollution. *Marine Biology*, 53: 371-379.
- Pereira M.E., Lillebø A.I., Pato P., Válega M., Coelho J.P., Lopes C., Rodrigues S., Cachada A., Otero M., Pardal M.A., Duarte A.C. 2009. Mercury pollution in Ria de Aveiro (Portugal): a review of the system assessment. *Environmental Monitoring and Assessment* . 155, 39-49.
- Pérez-Ruzafa A., Marcos C., Ros J.D. 1991. Environmental and biological changes related to recent human activities in the Mar Menor (SE of Spain). *Marine Pollution Bulletin* 23, 747-751.
- Pérez-Ruzafa A., Gilabert J., Gutiérrez J.M., Fernández A.I., Marcos C., Sabah S., 2002. Evidence of a planktonic food web response to changes in nutrient input dynamics in the Mar Menor coastal lagoon, Spain. *Hydrobiologia*, 475/476, 359-369.
- Pérez-Ruzafa A., Fernández A.I., Marcos C., Gilabert J., Quispe J.I., García-Charton J.A. 2005. Spatial and temporal variations of hydrological conditions, nutrients and chlorophyll *a* in a Mediterranean coastal lagoon (Mar Menor, Spain). *Hydrobiologia* 550, 11-27.

- Pérez-Ruzafa A., Marcos C., Pérez-Ruzafa I.M. 2009. 30 años de estudio en la laguna costera del Mar Menor: de la descripción del ecosistema a la comprensión de los procesos y la solución de los problemas ambientales. In: Cabezas F., Martínez-Nieto A. (Eds.). El Mar Menor. Estado actual del conocimiento científico. Instituto Euromediterráneo del agua, Murcia, pp. 17-46.
- Petrescu I. 1994. Contribution to the knowledge of amphipods (Crustacea) from Romania. II. *Gammarus aequicauda* (Martynov), *G. balcanicus* (Chaferna) and *Orchestia cavimana* (Heller), Travaux du muséum d'histoire naturelle "Grigore Antipa", vol. 34, pp. 303–324.
- Picado A., Dias J.M., Fortunato A.B. 2010. Tidal changes in estuarine systems induced by local geomorphologic modifications. Continental Shelf Research 30, 1854-1864.
- Pliński M., Kreńska B., Wnorowski T. 1978. Floristic relations and biomass of vascular plants in the Vistula lagoon. Biologia Morza 4: 161-196. [In Polish]
- Pliński M. 1995. Vascular plants of the northern part of the Vistula Lagoon. Bull. Mar. Inst. 22: 81–87.
- Prado P., Caiola N., Ibáñez C. 2013. Spatio-temporal patterns of submerged macrophytes in three hydrologically altered Mediterranean coastal lagoons. Estuaries and Coasts 36: 414-429.
- Prato E., Biandolino F. 2003. Seasonal changes in population of the Amphipod *Gammarus aequicauda* (Martynov, 1931), Mediterranean Marine Science, vol. 4, no. 1, pp. 49–56.
- Prato E., Biandolino F. 2005. *Gammarus aequicauda* (Crustacea: Amphipoda): A potential test species in marine sediment toxicity assessment. *Aquatic Ecosystem Health and Management*, 8, 475–482.
- Prato E., Biandolino F., Scardicchio C. 2008. Implications for toxicity tests with amphipod *Gammarus aequicauda*: Effects of temperature and salinity on life cycle. *Environmental Technology*, 29, 1349–1356.
- Pogrebniak I.I. 1965. Benthic Vegetation in Lagoons in the North-Western Black Sea Region and the Adjacent Black Sea Water Areas. Abstract of Doctoral Thesis in Biology. Odessa: Odessa State University, 31 p.

- Rahmstorf S., 2007. A semi-empirical approach to projecting future sea-level rise. *Science* 315, 368-370.
- Rao D.G.V.P., Khan M.A.Q. 2000. Zebra mussels: Temperature potentiation of copper toxicity. *Water Environ Res* 72:175–178.
- Renk H., Ochocki S., Zalewski M., Chmielowski H. 2000. Environmental factors controlling primary production in the Polish part of the Vistula. *Bull. Sea Fish. Inst.Gdynia* 152: 77-95.
- Rodrigues A.M., Quintino V., Sampaio L., Freitas R., Neves R. 2011. Benthic Biodiversity patterns in Ria de Aveiro, Western Portugal: environmental biological relationships. *Estuarine Coastal and Shelf Science* 95,338-348.
- Rousi H., Peltonen H., Mattila J., Bäck S., Bonsdorff E. 2011. Impacts of physical environmental characteristics on the distribution of benthic fauna in the northern Baltic Sea. *Boreal Env. Res.*, 16: 521-523.
- Rozengurt M.S. 1974. Hydrology and Prospects for Natural Resources Rearrangement in the Lagoons of the Odessa Oblast. Kiev, 'Naukovadumka', 224 pp. (in Russian).
- Ringer Z. 1959. An attempt to estimate the biomass of the littoral flora of the Vistula Lagoon based on the studies carried out in 1955. *Prace MIR w Gdyni* 10: 193-214 [in Polish].
- Rodrigues A.M., Quintino V., Sampaio L., Freitas R., Neves R. 2011. Benthic Biodiversity patterns in Ria de Aveiro, Western Portugal: environmental biological relationships. *Estuarine Coastal and Shelf Science* 95, 338-348.
- Róžańska Z., Cywińska A. 1983. Characteristics of abundance and biomass of the bottom fauna of the Vistula Lagoon (In Polish). *Oceanol.* 14:188-200.
- Rybicka, D. 2005. Potentially toxic blue-green algae (Cyanoprocarvota) in the Vistula Lagoon, *Oceanol. Hydrobiol. Stud.*: 161–176.
- Sayer C.D., Burgess A., Kari K., Peglar S., Davidson T.A., Yang H., Rose N. 2010. Long-term dynamics of submerged macrophytes and algae in a small and shallow, eutrophic lake: implications for the stability of macrophyte-dominance. *Freshwater Biology* 55: 565–583.

- Scheffer M., De Redelijkheid M.R., Noppert F. 1992. Distribution and dynamics of submerged vegetation in a chain of shallow eutrophic lakes. *Aquatic Botany* 42: 199-216.
- Selig U., Schubert M., Eggert A., Steinhardt T., Sagert S., Schubert H. 2007. The influence of sediments on soft bottom vegetation in inner coastal waters of Mecklenburg-Vorpommern (Germany). *Estuarine, Coastal and Shelf Science* 71: 241-249.
- Sfriso A., Birkemeyer T., Ghetti P.F. 2001. Benthic macrofauna changes in areas of Venice lagoon populated by seagrasses or seaweeds. *Marine Environmental Research*, vol. 52, no. 4, pp. 323–349.
- Sfriso A., Facca C., Ghetti P.F. 2003. Temporal and spatial changes of macroalgae and phytoplankton in a Mediterranean coastal area: the Venice lagoon as a case study. *Marine Environmental Research* 56, 617-636.
- Short F.T., Neckles H.A. 1999. The effects of climate change on seagrasses. 1999. *Aquatic Botany* 63, 169-196.
- Silva J.F., Duck R.W., Catarino J.B. 2004. Seagrasses and sediment response to changing physical forcing in a coastal lagoon. *Hydrology and Earth System Sciences* 8, 151-159.
- Silva J.F., Duck R.W., Catarino J.B. 2009. Nutrient retention in the sediments and the submerged aquatic vegetation of the coastal lagoon of the Ria de Aveiro, Portugal. *Journal of Sea Research* 62, 276-285.
- Simas T., Nunes J.P., Ferreira J.G., 2001. Effects of global climate change on coastal salt marshes. *Ecological Modelling* 139, 1-15.
- Steele D., Steele V. 1991. Effects of Salinity on the Survival, Growth-Rate, and Reproductive Output of *Gammarus-Lawrencianus* (crustacea, Amphipoda). *Marine Ecology Progress Series*, 78, 49–56.
- Subida M.D., Cunha M.R., Moreira M.H. 2005. Life history, reproduction, and production of *Gammarus chevreuxi* (Amphipoda: Gammaridae) in the Ria de Aveiro, northwestern Portugal. *Journal of the North American Benthological Society*, 24, 82–100.
- Sumner G.N., Romero R., Homar V., Ramis C., Alonso S., Zorita E. 2003. An estimate of the effects of climate change on the rainfall of Mediterranean Spain by the late twenty first century. *Climate Dynamics* 20, 789-805.

- Szarejko D. 1955. Vegetation of the Vistula Lagoon. Prace MIR w Gdyni 8: 235-254. [In Polish]
- Terrados J. 1991. Crecimiento y produccion de las praderas de macrofitos del Mar Menor, Murcia. Ph.D. thesis, University of Murcia.
- Terrados J., Ros J., 1995. Temporal variation of the biomass and structure of *Caulerpa prolifera* (Forsskal) Lamouroux meadows in the Mar Menor lagoon (SE Spain). Scientia Marina 59, 49-56.
- Twilley R.R., Cowan J., Miller-Way T., Montagna P.A., Mortaavi B. 1999. Benthic nutrient fluxes in selected estuaries in the Gulf of Mexico. In: Bianchi, T.S., Pennock, J.R., Twilley, R.R. (eds.), Biogeochemistry of Gulf of Mexico Estuaries. John Willey and Sons, Inc. New York, pp. 1863-209.
- Tyler A.C., McGlathery K.J., Anderson I.C. 2001. Macroalgal mediation of dissolved organic nitrogen fluxes in a temperate coastal lagoon. Estuarine, Coastal and Shelf Science 53, 155-168.
- Tyler A.C., McGlathery K.J. 2003. Benthic algae control sediment–water column fluxes of organic and inorganic nitrogen compounds in a temperate lagoon. Limnology and Oceanography 48, 2125-2137.
- Ugolini A., Pasquali V., Baroni D., Ungherese G. 2012. Behavioural responses of the supralittoral amphipod *Talitrus saltator* (Montagu) to trace metals contamination. *Ecotoxicology (London, England)*, 21, 139–147.
- Valentim J.M., Vaz N., Silva H., Duarte B., Caçador I., Dias J.M. 2013. Tagus estuary and Ria de Aveiro salt marsh dynamics and the impact of sea level rise. Estuarine, Coastal and Shelf Science, 1-14.
- Valiela I., McClelland J., Hauxwell J., Behr P.J., Hersh D., Foreman K. 1997. Macroalgal blooms in shallow estuaries: Controls and ecophysiological and ecosystem consequences. Limnology and Oceanography 42, 1105-1118.
- Velasco J., Lloret J., Millán A., Marín A., Barahona J., Abellán P., Sánchez-Fernández D. 2006. Nutrient and particulate inputs into the Mar Menor lagoon (SE Spain) from an intensive agricultural watershed. Water, Air and Soil Pollution 176, 37-56.

- Viaroli P., Bartoli M., Bondavalli C., Christian R.R., Giordan G., Naldi M. 1996. Macrophyte communities and their impact benthic fluxes of oxygen, sulphide and nutrients in shallow eutrophic environments. *Hidrobiologia* 329, 105-119.
- Vitousek P.M., Mooney H.A. 1997. Estimates of coastal populations. *Science* 278, 1211-1212.
- Wallace W., Estephan A. 2004. Differential susceptibility of horizontal and vertical swimming activity to cadmium exposure in a gammaridean amphipod (*Gammarus lawrencianus*). *Aquatic Toxicology*, 69(3), 289–297. doi:10.1016/j.aquatox.2004.05.010
- Watson R.T., Zinyowera M.C., Moss R.H. 1996. Climate change 1995 - Impacts, adaptations, and mitigation of climate change: Scientific-technical analysis. Contribution of working group II to the Second Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, NY, pp. 878.
- Wijnhoven S., van Riel M.C., van der Velde G. 2003. Exotic and indigenous freshwater gammarid species: physiological tolerance to water temperature in relation to ionic content of the water. *Aquatic Ecology*, 37, 151–158.
- Willer A. 1925. Studien über das Frische Haff. *Z. Fisch.* 23:317-349.
- Xu F.L., Jorgensen S.E., Tao S., Li B.G. 1999. Modeling the effects of ecological engineering on ecosystem health of a shallow eutrophic Chinese lake (Lake Chao). *Ecological modeling* 117, p. 239-260
- Żmudzinski L. 1996. The effect of the introduction of the american species *Marenzelleria viridis* (Polychaeta: Spionidae) on the benthic ecosystem of Vistula Lagoon. *Publ. St. Napoli J, Marine Ecology*, Blackwell Wissenschafts Verlag, Berlin, 17, No.1-3: 221-226.