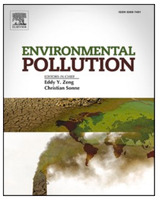
Environmental Pollution 313 (2022) 120127

Contents lists available at ScienceDirect

Environmental Pollution

journal homepage: www.elsevier.com/locate/envpol



Experimental assessment of salinization effects on freshwater zooplankton communities and their trophic interactions under eutrophic conditions[☆]

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ARTICLE INFO

Keywords:
Salinization
Zooplankton
Eutrophication
Mesocosm
Pond

ABSTRACT

Freshwater ecosystems are becoming saltier due to human activities. The effects of increased salinity can lead to cascading trophic interactions, affecting ecosystem functioning and energy transfer, through changes in community and size structure. These effects can be modulated by other environmental factors, such as nutrients. For example, communities developed under eutrophic conditions could be less sensitive to salinization due to cross-tolerance mechanisms. In this study, we used a mesocosm approach to assess the effects of a salinization gradient on the zooplankton community composition and size structure under eutrophic conditions and the cascading effects on algal communities. Our results showed that zooplankton biomass, size diversity and mean body size decreased with increased chloride concentration induced by salt addition. This change in the zooplankton community did not have cascading effects on phytoplankton. The phytoplankton biomass decreased after the chloride concentration threshold of 500 mg L^{-1} was reached, most likely due to direct toxic effects on the osmotic regulation and nutrient uptake processes of certain algae rather than as a response to community turnover or top-down control. Our study can help to put in place mitigation strategies for salinization and eutrophication, which often co-occur in freshwater ecosystems.

1. Introduction

Freshwater ecosystems are becoming saltier globally (Cañedo-Argüelles et al., 2019) due to human activities such as agriculture (Estévez et al., 2019; Thorslund et al., 2021), resource extraction (Ladrera et al., 2016; Vengosh et al., 2014) or the application of salt as a deicing agent for roads (Dugan et al., 2017; Kaushal et al., 2005). Additionally, indirect drivers of salinization include decreased precipitation and increased evaporation exacerbated by global warming, particularly in dry climates (Jeppesen et al., 2020). The increase in the salt concentration of freshwater ecosystems affects the survival of aquatic organisms (Szklarek et al., 2022) such as phytoplankton (Fazio and O’Farrell, 2005; Flöder et al., 2010), biofilms (Vendrell-Puigmitja et al., 2021), zooplankton (Bezirci et al., 2012), insect larvae (Cañedo-Argüelles et al., 2012), amphibians (Karraker et al., 2008) and fish (Baberschke et al., 2019) since they need to maintain an osmotic balance between the salts in the water and the salts within their internal fluids and cells (Bradley, 2008; Hellebusi, 1976).

The effects of freshwater salinization (FS) on biological communities can lead to trophic cascades with implications for ecosystem functioning (Hintz et al., 2022; Hintz and Relyea, 2019; Lin et al., 2017; Vidal et al., 2021). Within this context, zooplankton is a key component of lake food webs, since it facilitates energy transfer by grazing on phytoplankton (Hébert et al., 2017; Jeppesen et al., 2010). Furthermore, the ability of zooplankton to control phytoplankton biomass can determine the ecological state of lake ecosystems (Scheffer and Jeppesen, 2007). Despite its ecological relevance, there are only a few studies on the potential effects of FS on lake food webs (Cunillera-Montcusí et al., 2022). For example, Hintz et al. (2017) found a negative synergistic effect of fish and high salinity on the zooplankton community (i.e.,

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[☆] This paper has been recommended for acceptance by Dr. Sarah Harmon.

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https://doi.org/10.1016/j.envpol.2022.120127
Received 18 February 2022; Received in revised form 2 September 2022; Accepted 4 September 2022

top-down effect) in experimental ponds, which in turn caused an increase in phytoplankton biomass without altering the benthic food web; whereas Astorg et al. (2022) showed alteration of phytoplankton community composition with no change in total biomass along a wide salinity gradient. Moreover, Hébert et al. (2022) showed no compensatory increases in rotifers despite the decreases in crustaceans as a response to increased salinity in an internationally coordinated mesocosm experiment. In another experiment, relatively high chloride concentrations modified wetland community composition and its trophic interactions, but these effects were dependent on the chemistry of allochthonous inputs (Stoler et al., 2017). Finally, Schuler et al. (2017) found that NaCl had little effect on the aquatic communities of experimental ponds, but the organic additives contained in some of the road salts promoted microbial growth thereby increasing algae and zooplankton abundances.

Overall, these studies show that the potential of salinization to generate trophic cascades in lentic ecosystems is rather weak at low and moderate salt concentrations. However, the generation of trophic cascades by salinization might depend on many different factors that deserve to be further investigated. For example, the trophic status of lakes could modulate the effect of FS on aquatic communities and trophic cascades (Cañedo-Argüelles Iglesias, 2020; Schulz and Cañedo-Argüelles, 2019). Large grazers such as cladocerans that are sensitive to salinization tend to be abundant under oligotrophic conditions (Jeppesen et al., 2000; Scheffer et al., 1993). Therefore, salinization can promote phytoplankton growth in oligotrophic lakes through a release of the grazing pressure (Delaune et al., 2021; Moffett et al., 2020; Schuler et al., 2017). However, it is uncertain if this is also true for eutrophic freshwater ecosystems where zooplankton communities are dominated by small-sized zooplankton species (i.e., small copepods and rotifers) with relatively low grazing potential (Beklioğlu et al., 2017); Ersoy et al., 2019) and high tolerance to salinity (Hébert et al., 2022; Schallenberg et al., 2003). Different laboratory and mesocosm experiments assessing salt toxicity under eutrophic conditions showed contradictory results. For example, *Daphnia* in eutrophic conditions with high resource availability had a higher tolerance to salt inputs than *Daphnia* from oligotrophic conditions in a single-species laboratory experiment (Brown and Yan, 2015). However, Greco et al. (2021) did not observe any significant effects of nutrients on the zooplankton community when exposed to increased chloride concentrations. Instead, they showed that the effect of salt addition on phytoplankton and protist abundance and biomass was weak under high nutrient concentrations. Finally, Lind et al. (2018) found additive effects of nutrients and salts on phytoplankton, periphyton, macroalgae, snails, and zooplankton. Thus, the potential interactive effects of salinization and eutrophication on trophic interactions in lentic ecosystems are still unclear and need to be further studied (Cunillera-Montcusí et al., 2022).

In this study, we assess the influence of increased salinity on the plankton community of a eutrophic pond under a trophic food-web perspective, using a mesocosm experiment under semi-controlled conditions. Understanding the consequences of FS on eutrophic ecosystems is important because salt and nutrient pollution often have common drivers and tend to co-occur (Kaushal et al., 2021, 2018). Thus, both stressors need to be jointly managed and regulated. Here, we conducted a mesocosm experiment along a salinity gradient under eutrophic conditions and analyzed changes in zooplankton community and size structure and the potential resulting cascading effects on phytoplankton. We hypothesized that (H1) zooplankton biomass and size structure would not change in response to salt addition, since under eutrophic conditions the zooplankton community would be dominated by small-bodied taxa with high tolerance to salt stress and low grazing potential; (H2) thus, no cascading effects on phytoplankton would be observed along the salinity gradient, and (H3) phytoplankton biomass would not change until reaching a certain threshold of chloride concentration but decrease afterward due to direct toxic effects of salinity on phytoplankton.

2. Material & methods

2.1. Experimental design

The experiment was carried out in 20 outdoor land-based mesocosms (200 L capacity) filled with 200 L of water collected from a freshwater pond located in Tavèrnoles (Barcelona, Spain; 41°57′29.8″N 2°18′43.4″E) for 6 weeks from 8 November to 20 December , 2018. Data are provided in the Supplementary Material. Tavèrnoles pond had eutrophic conditions at the time of the experiment [Total Nitrogen (TN) = 8.11 mg L^{-1}, Total Phosphorus (TP) = 530 μg L^{-1}, chlorophyll-a (Chl-a) = 9.93 μg L^{-1}], it was oligohaline (chloride (Cl^-) = 11.3 mg L^{-1}) and it did not have large cladocerans (the zooplankton community was dominated by copepods and rotifers in terms of relative abundance, see Appendix, Table A1). A gradient design was used by adding pre-weighed sodium chloride (NaCl; > 99.0% purity; PanReac AppliChem, CAS-No: 7647-14-5) to obtain nominal experimental NaCl concentrations from basal levels (no NaCl addition) up to 1500 mg Cl L^{-1} covering ecologically relevant concentrations for salt contamination of lentic ecosystems (Dugan et al., 2017; Hintz et al., 2022). According to Kreyling et al. (2018), gradient designs in experiments are better than replicated designs to detect threshold and non-linear responses across multiple simulations with different environmental drivers. NaCl was added only once at the start of the experiment. Electrical conductivity (EC, μS cm^{-1}), dissolved oxygen (DO, mg L^{-1} and %), water temperature (T, °C) and pH were measured with a YSI Pro Plus multiparametric hand-held probe during the experiment (weeks 0, 2, 4 and 6) (YSI Inc., Yellow Springs, OH, USA).

2.2. Sampling and processing

The zooplankton community was sampled at the beginning (week 0, only two samples) and the end of the experiment (week 6) from each mesocosm by filtering 30 L water (corresponding to 15% of the total volume) through a 50 μm mesh size, after a gentle mixing of the water. Analyses of the communities from the beginning of the experiment showed no association with the chloride concentration (Hébert et al., 2022), thus they were discarded from further analyses. Subsamples were taken at multiple locations in the mesocosm and combined to account for eventual spatial heterogeneity. All samples were fixed and stored in 70% EtOH solution until further identification. Zooplankton samples were counted until reaching at least 100 individuals of the most abundant taxa and identified to genus level under an inverted microscope using relevant identification keys (Dussart, 1967, 1969; Koste, 1978). At least 20 individuals from each taxon were measured, if possible, to calculate size structure metrics and estimate total biomass. Different developmental stages of copepods (nauplii, copepodites, and adults) were measured separately to account for ontogenetic changes. The counting and measuring methodology were in line with the previous relevant studies (Brucet et al., 2017; Ersoy et al., 2019). Individual biomasses were calculated using the existing allometric relationships from the literature (Bottrell et al., 1976; Ruttner-Kolisko, 1977; García-Chicote, 2015) and the dry weight of rotifers was estimated using a dry/wet weight ratio of 0.25 and carbon units of biomasses for all groups using a 40% ratio (Reiss and Schmid-Araya, 2008).

After gently mixing the water, samples (45 ml) for determining TP and TN concentration were collected directly from each mesocosm (approximately 30 cm depth) with a 50 ml Falcon tube. Water samples were collected at the beginning from a limited number of enclosures (week 0) and end of the experiment (week 6) from each enclosure and immediately stored at −20 °C for determination of TN and TP concentrations. Additionally, 0.5 L of water from each mesocosm was filtered through GF/C Glass microfiber filters (1.2 μm pore size, Whatman ® International) and stored at −20 °C for further determination of phytoplankton biomass measured as chlorophyll-*a* (Chl-*a,* μg L^{-1}) concentration after extraction with 90% acetone for 12 h in the dark at 4 °C.

The samples were sonicated for 2 min to ensure a complete chlorophyll-*a* extraction. Acetone extracts were filtered through 0.7 μm glass fiber filters (GF/F filters, Whatman International), and Chl-*a* concentration was determined spectrophotometrically (NanoPhotometer™ P-360) following the method described by Jeffrey & Humphrey (1975).

2.3. Size diversity

Size diversity index is computed based on the Shannon-Wiener diversity index adjusted for continuous variables and represents size range and regularity in the body size distributions. Size diversity index was calculated from individual zooplankton body mass measurements following Quintana et al. (2008):

$$\mu = -\int_0^\infty p_x(x)\,\log_2 p_x(x)dx$$

where $p_x(x)$ is the probability density function of size x. High size diversity represents both regular distribution of body sizes and wider size range in the community body size distributions.

2.4. Statistical analyses

Generalized additive models (GAMs) with Gaussian distributions were used to assess the response of the zooplankton and phytoplankton community to the chloride gradient, using the *'mgcv'* package (Wood, 2004). GAMs were selected over other modeling approaches because we expect to observe non-linear relationships between chloride, other environmental predictors (i.e., TP) and the planktonic community and size structure metrics (i.e., total biomass, mean body mass, and size diversity). GAMs are developed as an extension of GLMs and they do not assume any a priori relationship. Within this context, GAMs are a very useful tool because they allow testing non-linear responses by fitting non-parametric functions. As a common procedure, during the exploratory data analyses and modeling, we first fitted GLMs and observed the non-linearity in the relationships, which further confirmed the use of GAMs. However, since there weren't non-linear relationships between zooplankton community metrics and chloride concentrations, chloride was included as a parametric term in models with zooplankton community metrics as response variables. The residuals of the models were checked visually for normality and the collinearity of the predictors was assessed using the variance inflation factor (VIF <4; car package; Fox & Weisberg, 2011).

After checking the correlation coefficients of the other environmental variables, a maximum of two predictors were selected given the size of our dataset ($n = 20$) (Appendix, Figure A3). The direct effect of chloride concentration was tested on zooplankton community and size structure metrics, phytoplankton biomass (using Chl-*a* as a proxy) and total phosphorus (TP) concentration (as an indicator of resource availability). Additionally, phytoplankton biomass and zooplankton community metrics were included as potential bottom-up and top-down controls in the zooplankton and phytoplankton models. For example, in models for zooplankton community and size structure metrics, phytoplankton biomass was added as an indicator for a bottom-up control. In different models for phytoplankton biomass, zooplankton total biomass, mean body mass and size diversity were included separately as proxies for top-down control by zooplankton.

Separately, to investigate the tolerance of different taxa to salt addition, linear models were conducted for multivariate abundance data, using the '*manyany*' function from the '*mvabund*' package (Wang et al., 2012) with a negative-binomial distribution, followed by univariate tests for all zooplankton taxa with *p-value* adjustment. Statistical analyses were performed using R version 4.0 (R Core Team, 2020). All graphs were produced with the *'ggplot2'* package (Wickham, 2009).

3. Results

There was strong evidence that zooplankton total biomass, mean body mass, and size diversity decreased with the increase in chloride concentration (Table 1, Fig. 1, $p < 0.01$). However, there was no evidence that phytoplankton biomass (measured as chlorophyll-*a* concentration) had any effect on the zooplankton community (i.e., bottom-up control), except the zooplankton mean body mass. We only found weak evidence for a negative relationship between Chl-*a* and zooplankton mean body mass (Table 1, Fig. 2, $p = 0.06$). The analysis of multivariate abundance data (Appendix, Figures A1& A2) revealed strong evidence for the negative effect of salt addition on zooplankton abundance ($p < 0.01$) along the chloride gradient, with adult copepod abundance showing a significant decrease with increased chloride concentration ($p_{adj} = 0.04$). Phytoplankton biomass (i.e., Chl-*a*) showed a threshold response to salt addition. We found moderate evidence that Chl-*a* decreased after a certain chloride threshold (500 mg L^{-1}) was reached and zooplankton total biomass had a weak negative effect on phytoplankton biomass (Table 1, Fig. 2, $p = 0.05$ for both). Finally, there was strong evidence that total phosphorus (TP) concentration at the end of the experiment increased along the chloride gradient (Table 1, Fig. 2, $p < 0.01$).

4. Discussion

Freshwater salinization is a great threat to the functioning and health of aquatic ecosystems, and it often co-occurs with eutrophication (Kaushal et al., 2018; Lind et al., 2018). Thus, understanding how salinization shapes trophic interactions under eutrophic conditions is crucial in the context of multiple and interacting stressors (Velasco et al., 2019; Cunillera-Montcusí et al., 2022). We found a direct negative effect of increasing chloride concentrations on the zooplankton community and size structure as well as on the phytoplankton biomass under eutrophic conditions. However, salt addition did not affect trophic interactions except for a weak negative effect of zooplankton total biomass on phytoplankton biomass.

Contrary to our expectations (H1), total zooplankton biomass decreased significantly with increased chloride concentration as shown by previous studies (Hintz et al., 2017; Searle et al., 2016). Likewise, a recent mesocosm experiment (Greco et al., 2021) showed that salt addition caused a reduction in copepod and rotifer biomass regardless of the nutrient level. However, we found no strong relationship between zooplankton and phytoplankton biomass, suggesting that changes in the zooplankton community were exclusively related to direct chloride

Table 1
Results of Generalized Additive Models (GAMs) showing the effects of chloride concentration and potential bottom-up and top-down controls on zooplankton and phytoplankton communities, respectively. Results showing only strong evidences are highlighted in bold. F and t-value are shown for smooth and parametric variables, respectively. s: smoothing parameter.

Response variable	Predictors	F/t-value	*p-value*
TP	s(Chloride)	6.04	**0.006**
log(total zooplankton biomass)	s(Chl-*a*)	2.39	0.09
	Chloride	−3.69	**0.002**
Zooplankton size diversity	s(Chl-*a*)	2.24	0.11
	Chloride	−4.40	**0.005**
log(zooplankton mean body mass)	s(Chl-*a*)	2.74	0.06
	Chloride	−3.45	**0.004**
Chl-*a*	s(Chloride)	5.43	**0.01**
	Total zooplankton biomass	−2.13	0.05
Chl-*a*	s(Chloride)	4.30	**0.03**
	Zooplankton size diversity	−1.62	0.12
Chl-*a*	s(Chloride)	3.51	0.05
	Zooplankton mean body mass	−0.94	0.36

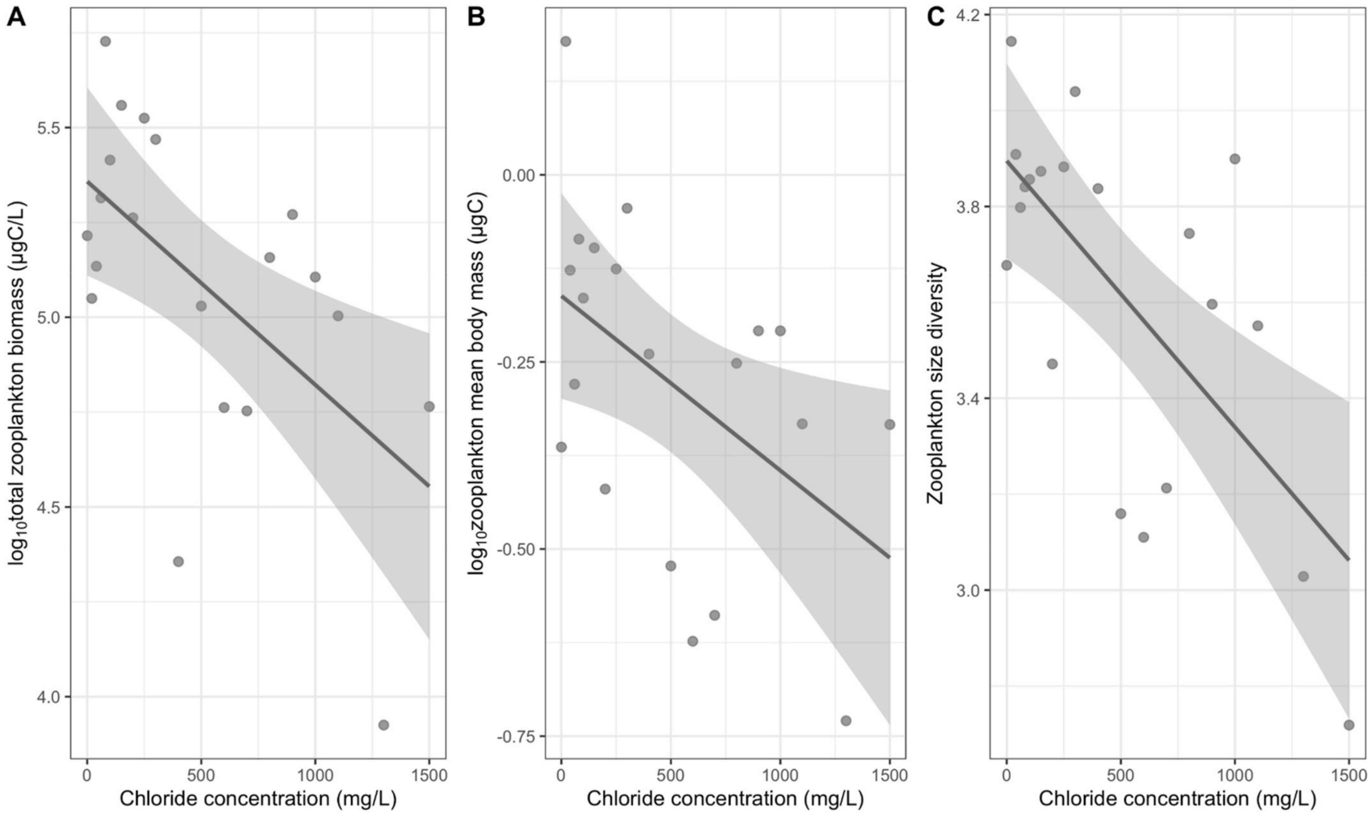

Fig. 1. Effect of chloride concentration on zooplankton community A) total zooplankton biomass, B) zooplankton mean body mass and C) zooplankton size diversity. Data are fitted with linear models with a 95% confidence interval ($n = 20$). Lines show the fitted models and shaded regions represent the confidence intervals.

toxicity and not driven by bottom-up processes (i.e., phytoplankton) (confirming H2), although there was slight but not statistically strong evidence of the increase in phytoplankton biomass at the lower salinity levels, which could be due to salt tolerance of zooplankton community at lower salinity levels. These results do not align with previous studies conducted in oligotrophic lakes where the reduction in zooplankton biomass at high salinity led to compensatory increases in phytoplankton due to a release of grazing pressure (Delaune et al., 2021; Moffett et al., 2020). This is most likely related to the differences in zooplankton community structure between oligotrophic and eutrophic lakes, with the latter being dominated by smaller taxa (Jeppesen et al., 2020; Scheffer et al., 1993) that tend to have a lower phytoplankton grazing potential (Dupuis and Hann, 2009; Vanni and Temte, 1990).

Our results showed strong evidence of a reduction in zooplankton mean body mass and size diversity along the ecologically relevant chloride gradient, indicating that the community size range was lower and dominated by small-sized individuals at the highest end of the gradient. Supporting this decline in the zooplankton size diversity and mean body mass, we found a significant decrease in the abundance of adult copepods as observed in a recent study using a similar experimental set-up (Greco et al., 2021). This decrease in zooplankton biomass and size metrics could be explained by the direct toxic effects of salt, causing decreased reproduction and increased mortality (Arnott et al., 2020) and a shift in community composition along the salinity gradient (Brucet et al., 2010; Gutierrez et al., 2018; Jeppesen et al., 2007a). Additionally, the low diversity in our zooplankton community and low grazing potential (i.e., absence of big cladocerans), most likely related to the eutrophic conditions of the local pond (Badosa et al., 2007; Duré et al., 2021), could have contributed to the decrease in size diversity, through decreased niche partitioning (Brucet et al., 2017; Ersoy et al., 2017). Moreover, we cannot exclude the effect of seasonal dynamics on the zooplankton assemblages, since big cladocerans become dominant during early spring in permanent ponds whereas copepods occur widely during winter when our experiment took place (Brucet et al., 2005; Vad et al., 2012). This decline in zooplankton body size metrics as a response to the chloride gradient may impair the energy flow in the natural aquatic food webs, by reducing the trophic energy transfer from the lower (i.e., primary producers) to the upper trophic levels (i.e., predators) (Ye et al., 2013; García-Comas et al., 2016).

In line with our third hypothesis (H3), no substantial change in phytoplankton biomass was observed until a certain threshold. Chlorophyll-*a* concentrations started to decrease after reaching a 500 mg L^{-1} chloride concentration, despite the potentially lower grazing pressure caused by lower zooplankton biomass and zooplankton body size. This finding agrees with experiments conducted in stream (Cañedo-Argüelles et al., 2014, 2017) and pond mesocosms (Astorg et al., 2022), where increased salinity did not affect algal growth because the algae communities were able to tolerate wide ranges of salinity. However, as previous studies on both benthic and planktonic primary producers suggested (Cochero et al., 2017; Flöder et al., 2010; Vendrell-Puigmitja et al., 2021), a decrease in primary producers' biomass can be observed at high salinities due to the direct toxic effect of increased salinity on osmotic regulation in certain phytoplankton groups (i.e., diatoms), thereby changing their cell morphology and decreasing their nutrient uptake efficiencies. Further, the linear relationship between TP and Chl-*a* can also become decoupled with chemical characteristics of the freshwaters (Quinlan et al., 2021), for example as a result of salinization. Accordingly, inefficient nutrient uptake rates by phytoplankton could clarify the absence of nutrient effect on phytoplankton growth, despite the increased nutrient concentrations at higher salinity levels. These available nutrients might have not been sequestered by phytoplankton due to their low biomass and low nutrient uptake capacity. Since we didn't measure nutrient uptake rates, we cannot discard the possibility of these effects. These impaired energy and nutrient transfer mechanisms in the food web are likely to have an impact on ecosystem functioning (Jeppesen et al., 2015).

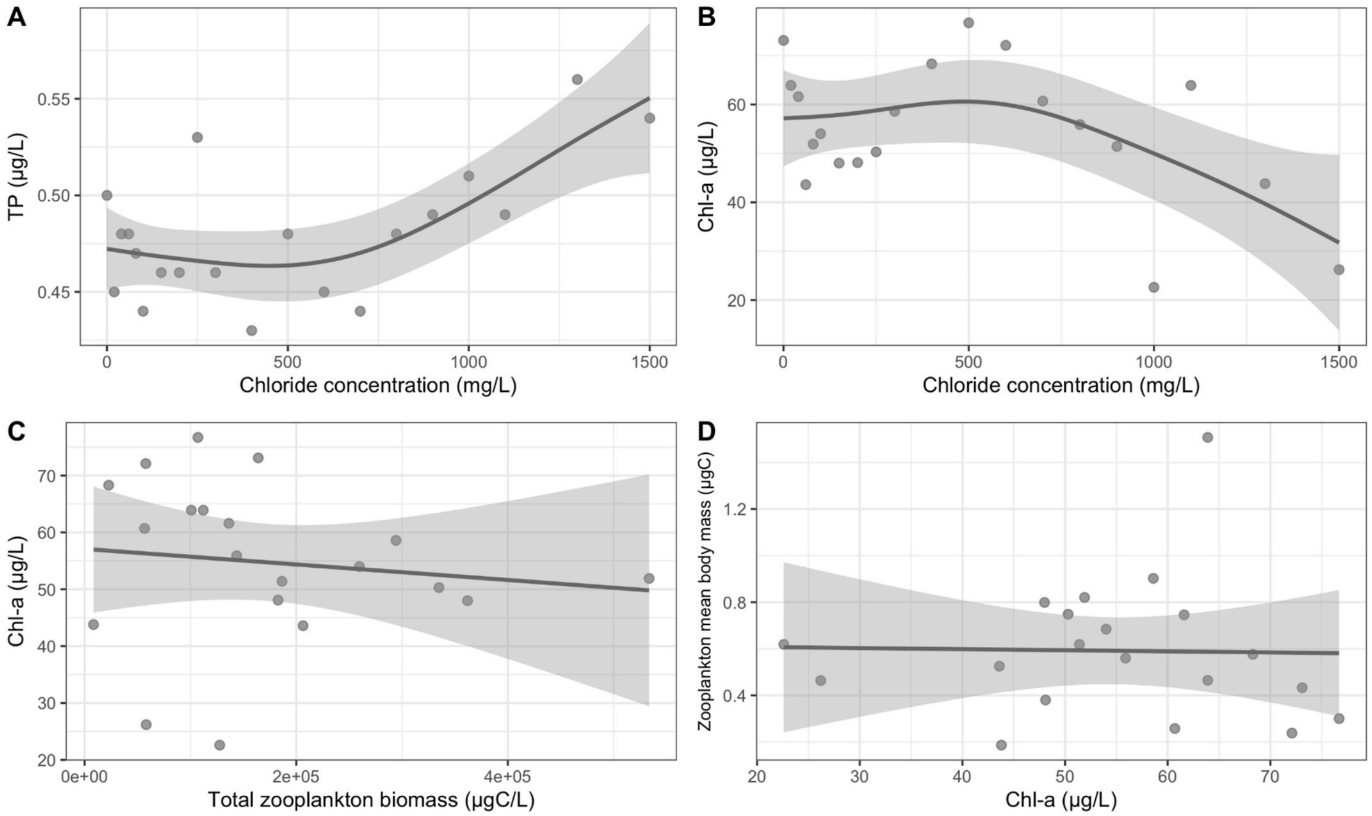

Fig. 2. Effect of chloride concentration on A) total phosphorus (TP) concentration; B) phytoplankton biomass (Chl-*a*); C) effect of total zooplankton biomass on Chl-*a*; D) effect of Chl-*a* on zooplankton mean body mass. Data are fitted with GAM and LM with a 95% confidence interval (n = 20). Lines show the fitted models and shaded regions represent the confidence intervals.

We acknowledge the limitations of our experimental design, as mesocosm experiments commonly fail to capture the complexity of real freshwater ecosystems (Stewart et al., 2013). For example, since our mesocosms were isolated, recolonization via dispersal was not possible (Hintz and Relyea, 2019; Sinclair and Arnott, 2018) as it was in other studies conducted in natural ecosystems along salinity gradients (Gutierrez et al., 2018; Jeppesen et al., 2007b; Vidal et al., 2021). We also did not account for the adaptation responses to increased salinity due to the short duration of the experiment (6 weeks), although adaptation was possible after 2.5 months (Coldsnow et al., 2017). Longer experiments are needed at the community level to understand the evolutionary adaptation dynamics. Moreover, we did not incorporate habitat heterogeneity in our mesocosms such as aquatic plants which could potentially have stabilizing effects for the salt toxicity in natural pond ecosystems (Coldsnow and Relyea, 2021; Hintz and Relyea, 2019), nor accounted for the effect of temperature change during the experiment, that may influence algae growth. Despite the temperature decrease from the beginning to the end of the experiment, phytoplankton biomass increased in all enclosures, probably due to high nutrient availability. Finally, we did not incorporate fish in our mesocosms. Fish can be very sensitive to salinization (Hintz and Relyea, 2017; Leite et al., 2022) and they have a great potential to modulate planktonic food webs in lentic ecosystems (Carpenter et al., 2010). Nevertheless, when compared to the studies performed in real ecosystems along natural salinity gradients, controlled field-based mesocosm experiments can be used to establish cause-effect relationships that help to understand the underlying mechanisms of complex ecosystems, including the effects of salinization on freshwater ecosystems (Hébert et al., 2022; Hintz et al., 2022).

5. Conclusions

We found significant negative effects of salt addition on the zooplankton community and size structure. However, no cascading effect on phytoplankton was observed, due to the low grazing potential of the zooplankton community with low functional diversity under eutrophic conditions. Furthermore, our results suggest that the direct toxic effects of salts on the phytoplankton community may interfere with the release of the grazing pressure on the phytoplankton from zooplankton, through reduced osmotic regulation and nutrient uptake processes of phytoplankton. However, the exact mechanisms remain unclear. Taken together, our results suggest that the potential of salinization to cause trophic cascades in lake ecosystems might be buffered to some extent in eutrophic ecosystems, especially if they contain salt-tolerant phytoplankton taxa that could compensate for the loss of sensitive taxa and sustain primary production. However, these findings also suggest that the restoration of degraded lake ecosystems simply through the reduction in nutrient loads might not be effective, since salinization effects might become stronger once the lake returns to oligotrophic conditions. Given that salts and nutrients often co-occur (e.g., fertilizer pollution and salinization are common drivers of freshwater ecosystem degradation in agricultural landscapes), we argue that they should be managed together. For this purpose, we need a better understanding of how their interaction affects biodiversity, trophic cascades, and ecosystem functioning.

Credit author statement

Zeynep Ersoy: Conceptualization, Formal analysis, Data curation, Visualization, Writing – original draft preparation; Meritxell Abril: Conceptualization, Methodology, Writing- Reviewing and Editing, Resources, Funding acquisition; Miguel Cañedo-Argüelles: Methodology, Supervision, Project administration, Resources, Writing- Reviewing and Editing, Funding acquisition; Carmen Espinosa : Methodology, Investigation, Writing- Reviewing and Editing; Lidia Vendrell-Puigmitja: Methodology, Investigation, Writing- Reviewing and Editing; Lorenzo Proia: Conceptualization, Methodology, Investigation, Supervision, Project administration, Resources, Writing- Reviewing and Editing, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data is provided in the Supplementary Material.

Acknowledgments

This study is part of the Global Lakes Ecological Observatory Network (GLEON), Global Salt Initiative. We thank FEHM and CT-BETA for allowing us to use their research facilities. LP has received funding from the Beatriu de Pinós Postdoctoral Fellowships Programme, funded by the Secretary of Universities and Research (Government of Catalonia) and by the Horizon 2020 Programme of Research and Innovation of the European Union under the Marie Skłodowska-Curie Grant Agreement No. 801370 and by a Ramón y Cajal contract funded by the Spanish Ministry of Science and Innovation (RYC 2020-029829-I). MCA was supported by the Serra-Hunter programme funded by the Generalitat of Catalunya and by a Ramón y Cajal contract funded by the Spanish Ministry of Science and Innovation (RYC 2020-029829-I). We also thank three anonymous reviewers for their constructive comments and suggestions.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2022.120127.

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