




Can the use of zooplankton dormant stages from natural wetlands contribute to restoration of mined wetlands?

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Abstract Wetlands are among the most diverse environments on the planet and are strongly threatened by human activities. Their restoration and/or mitigation of human impacts, therefore, relies on information that can aid to the management of impacted wetlands so that they return to a (semi-) natural state. We investigate in this study the relationship between dormant stages of zooplankton and clay removal in areas subjected to mining. We evaluate whether a gradual increase in topsoil addition from donor natural wetlands to the sediment of mined wetlands influenced the zooplankton community. Eight wetlands were sampled in the Sinos River floodplain, four natural and four mined. In the laboratory, four field sediment

samples were incubated for zooplankton hatching in five treatments comprising sediments from: mined wetlands, natural wetlands, and three treatments containing mined sediments added with low (5%), medium (20%) and high (40%) quantities of sediment from natural wetlands. Hatching consisted of 61 individuals distributed across eight zooplankton taxa. Copepod nauplii were the most abundant (31.1%) followed by *Epiphanes* sp. (29.5%) and *Ovalona glabra* (16.4%). While natural wetlands provided 42.6% of the hatched zooplankton, mined wetlands had just 6.5%. Zooplankton richness and abundance were higher in natural wetland sediments compared with mined and added sediment wetlands. To some degree, the sediment soil donation from natural to mined wetlands was considered viable. As long as prior studies are performed to test the size and quality of the dormant banks present in the sediment of candidate donor wetlands, sediment from donor wetlands may aid in the establishment of a more diverse community in disturbed systems.

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Introduction

Wetlands are among the most diverse habitats on the planet (Batzer and Sharitz 2014; Mitsch and Gosselink 2015), providing valuable ecosystem services to human populations (Bos et al. 2005; Junk et al. 2014; Mitsch and Gosselink 2015; Ramsar Convention on Wetlands 2018). However, these systems are threatened by human activities including urban and industrial expansion and agriculture (Czech and Parsons 2002; Settele et al. 2015; Craft 2016; Hu et al. 2017; Maltchik et al. 2017). Studies estimate a loss of wetland area of around 64% worldwide (Davidson 2014; Ramsar Convention on Wetlands 2018), with considerable spatial variation at loss rates. In Brazil, current rates of wetland loss are unknown (Junk et al. 2014).

Mining is an economically important industrial activity in Brazil (BRL 100 billion per year) (National Mining Agency of Brazil 2018), which is directly and indirectly responsible for significant impacts on biological communities, such as zooplankton, and compromising the integrity of aquatic systems in and around the mined area (Santhosh et al. 2013; Moreira et al. 2016; Goździejewska et al. 2018). Mining in wetlands generates inputs of a range of pollutants, including heavy metals, which reduces water quality (Sodré et al. 2015) and also increases turbidity by the increase in soluble particles through the water column, which limits primary production and zooplankton diversity (Bozelli 1996; Moreira et al. 2016). Mined wetlands are usually disturbed by continuous clay removal, mostly from the topsoil (Dong et al. 2019) that contains the dormant stages of zooplankton (Brendonck and De Meester 2003).

Zooplankton is a key group in the aquatic food web, representing the link between producers and larger consumers (Jenkins and Boulton 2003). Some zooplankton animals have dormancy capability (i.e., long-term resistant stages) that enables them to survive periods of unfavorable conditions (Vargas et al. 2019; Fontaneto 2019). Thus, such dormant stages are crucial for colonization processes and community dynamics in freshwater wetlands, especially temporary ones (Shurin 2000; Badosa et al. 2017; Brendonck et al. 2017). Seemingly isolated temporary wetlands may serve as source of zooplankton propagules via surface water flow, wind, and attached to animals. In floodplains, several ecological processes are

associated with the variation between the aquatic and terrestrial phases (Junk et al. 2006; Tockner et al. 2006). As flood pulse often decreases environmental heterogeneity and bring new colonizers (Thomaz et al. 2007; Bozelli et al. 2015), zooplankton communities are structured by the interaction between dormant stages, stochastic events and the dispersal of propagules from other areas (Shurin et al. 2009).

Zooplanktonic species are good indicators of anthropic impacts and useful in understanding processes associated with wetland degradation and restoration (Boix et al. 2008; Ejsmont-Karabin 2012; Brendonck et al. 2017; Marszelewski et al. 2017; Goździejewska et al. 2018). The assessment of the diversity of zooplankton communities through hatching experiments has proven to be a useful and reliable tool for monitoring community changes as dormant banks integrate seasonal and inter-annual variations in environmental conditions (Brendonck and De Meester 2003; García-Roger et al. 2008).

Dormant banks of zooplankton consist mostly of long-term resistant propagules and other similar stress-resistant structures present in wetland sediments (Gaikwad et al. 2008). The hatching of the dormant stages is influenced by one or more environmental and biological factors (Williams 2006), such as, hydroperiod (Brendonck et al. 2017; Stenert et al. 2017), temperature (Gaikwad et al. 2008; Palazzo et al. 2008), photoperiod (Vandekerkhove et al. 2005), dissolved oxygen (Broman et al. 2015), predation (Nielsen et al. 2000; Vendramin et al. 2020) and competition (Shao et al. 2014). In southern Brazil, studies using hatching experiments have been indicating that many zooplanktonic taxa, mainly microcrustaceans, emerge from numerous viable dormant eggs of natural wetland sediments (Freiry et al. 2016, 2020a, b; Stenert et al. 2016; Bandeira et al. 2020), and rice field sediments (Stenert et al. 2010; Ávila et al. 2015).

The management of dormant banks of zooplankton can be used to recuperate wetlands affected by human activities (Gleason et al. 2003, 2004; Jenkins and Boulton 2007; Cui et al. 2018). The employment of topsoil (i.e., the upper sediment layer which comprises the active dormant bank) (Cáceres and Hairston 1998; Brendonck and De Meester 2003) from a donor wetland soil (Burke 1997; Wetland Science Institute 2003) may aid the recovery of biological communities in disturbed wetlands (Brock et al. 2003; Richter and

Stromberg 2005; Moreno-Mateos et al. 2015). Topsoil from donor wetlands may also increase the water retention capacity and, therefore, enhance the establishment of other organisms, important to the nutrient cycling (Burke 1997).

In this sense, we conducted an *ex situ* experiment to investigate the relationship between dormant stages of zooplankton and clay removal in wetlands subject to mining activities. We also assessed whether increasing amount of topsoil addition from natural (non-mined) wetlands in the sediment of mined areas could influence the taxonomic richness, abundance and composition of the zooplankton communities. Based on the evidence that mining activities affect the zooplanktonic communities (Vandysh 2004; Moreira et al. 2016; Goździewska et al. 2018), and that cladoceran ephippial eggs are well-known compared to other taxa, we hypothesized that: (1) the abundance of cladoceran ephippial eggs would be higher in natural wetland sediment when compared with mined ones; (2) richness and abundance of hatchlings would be higher in natural wetland sediments, followed by treatments with high topsoil addition from natural wetlands, and lowest in mined wetland sediments; and (3) mined and natural wetland sediments would have a different taxa composition, but topsoil addition would reduce the community composition dissimilarity between mined and natural wetlands.

Material and methods

Study area

The study was conducted in the Sinos River basin, inserted between the geographic coordinates 50°10' and 51°20' W, and 29°15' and 30°00' S (state of Rio Grande do Sul), in southern Brazil, which encompasses a densely populated area of approximately 3600 km². The water quality of the Sinos River basin varies along its longitudinal gradient, decreasing from the upper reaches towards the lower region of the basin, which is affected by the input of polluting agents from domestic and industrial waste (Bieger et al. 2010; FEPAM 2019).

The climate in the study area is subtropical (Cfa, Köppen-Geiger classification), with annual mean temperature and precipitation of 20 °C and 1600 mm, respectively (COMITESINOS 2019). The

Sinos River floodplain is scattered with intermittent wetlands, where high precipitation originates occasional flood events that inundate all floodplain habitats throughout various periods of the year (Rio Grande do Sul 2015). Flood pulses can be quite variable within and among the years. While many floods are short in duration (~ 1 week), it is not uncommon the system remain flooded for up to 2 months (Maltchik et al. 2008). During these events, the riparian habitats (between 1000 and 2000 m around the main channel of the river) remains connected through the surface water of the Sinos River. The mean water depth in floodplain wetlands is approximately 60 cm; however, during the flood events, the water depth can reach 200 cm in all floodplain system. The riparian vegetation is represented mainly by native woodland, and wetlands are fed from precipitation and runoff or floodwaters from the Sinos River. Mining activities that occur over the Sinos River basin, include the extraction of clay minerals for the production of ceramics, tiles and construction materials, which is responsible for 21% of the GDP of the state of Rio Grande do Sul (COMITESINOS 2019).

Sampling design

Eight floodplain wetlands located in the lower region of the Sinos River basin were sampled in February 2019, including four natural (donor wetlands) and four mined wetlands, located at least 5 km distant from each other. Sampling was performed in summer when most wetlands decrease in surface water. Some wetlands dry up completely and others may decrease in as much as 60% of their surface area. Sampled natural wetlands showed similar morphological characteristics such as size (~ 1 ha), depth (0.5 m on average) and the composition of aquatic vegetation during flooding, such as herbaceous and emergent plants. Mining consisted of mechanically excavating the wetlands for the use of clay, in which case they would become devoid of aquatic plants.

In order to obtain a representative sediment sample throughout the selected wetlands, 15 subsamples were randomly collected from each of the eight wetlands during a dry period using a core sampler (7.5 cm diameter) inserted to a depth of 5 cm into the substrate (Brendonck and De Meester 2003). The area of each core was 44.15 cm², and the area of the sediment sampled per wetland was about 660 cm². These

subsamples were, then, pooled to represent the entire wetland and stored in black plastic buckets (20 L) for further analysis (Maia-Barbosa et al. 2003; Stenert et al. 2010).

Laboratory procedures

The sediment collected (February 2019) was taken to the laboratory and dehydrated in a dark oven for 96 h at 40 °C. The dry sediment was subsequently stored in dark plastic bags for 60 days and room temperature (23 °C) before the beginning of the experiment (May 2019). Before the experiment, the dehydrated sediment was sieved (1-mm mesh size) for the removal of roots and leaves and homogenized, corresponding to approximately 1.5 kg of dry sediment per wetland. First, the sugar flotation method (Onbé 1978) was used to evaluate the abundance of cladoceran ephippial eggs in natural and mined wetlands. For this, 100 g of dry sediment representing each wetland was added to a 1:1 mix of distilled water and sugar. Then, 15 mL of this solution was centrifuged (at 3000 rpm for 3 min), and the supernatant was washed through a 53- μ m mesh using distilled water.

Additionally, 250 g aliquots from dry topsoil were used to create five incubation treatments (T) with four replicates each: T1 (250 g from mined wetlands, i.e., 0% from donor wetland); T2, low addition of topsoil (12.5 g from natural wetland + 237.5 g from mined sediment, i.e., 5% from donor wetland); T3, medium addition of topsoil (50 g from natural wetland + 200 g from mined sediment, i.e., 20% from donor wetland); T4, high addition of topsoil (100 g from natural wetland + 150 g from mined sediment, i.e., 40% from donor wetland); T5 (250 g of topsoil from natural donor wetlands). The definition of these different proportions of topsoil sediment was based on the fact that low proportions of natural sediments (maximum 40%) would be less costly and more viable in wetland restoration projects developed in situ.

Sediment samples were incubated in trays measuring 30.3 cm in length, 22.1 cm in width and 7.5 cm in depth. The sediment in each one of the twelve trays was kept aerated and submersed under a depth of 2 cm of distilled water. Water level, temperature (23 ± 2 °C), photoperiod (12 h light/12 h dark), and dissolved oxygen (> 6.5 mg/L) were kept constant (Stenert et al. 2010; Ávila et al. 2015).

The experiment was maintained in the laboratory for 4 weeks, and hatchlings were collected three times per week to avoid reproduction. Several studies suggest that sampling intervals of 2–3 days ensure that the individuals collected are from the dormant bank and not from reproduction (Brock et al. 2005; Nielsen et al. 2013). The duration of the experiment was based on previous work from our research group (Freiry et al. 2020b; Vendramin et al. 2020) and others (Brock et al. 2003). The samples were collected by stirring up the sediment and then sweeping above the overlying water (three times at each tray) using a 22.1 cm wide net (53 μ m mesh). The content of the sample was then transferred to a Bogorov chamber for sorting of hatchlings using a stereomicroscopic (Zeiss Stemi 2000-C). After the sampling of hatchlings, the ephippial eggs found in the sample were returned to their respective trays. The hatchlings collected were transferred to 1.5-mL polypropylene tubes with 80% ethanol. Identification was performed to the species level whenever possible using specialized literature (Koste 1978; Elmoor-Loureiro 1997, 2000; Gazulha 2012) and additional help from taxonomists. Individuals or their parts (antennae, antennule, post-abdomen) were prepared with a drop of glycerine and examined under a microscope. The Copepod nauplii were only counted. Samples of the identified organisms are kept at the Laboratory of Ecology and Conservation of Aquatic Ecosystems of UNISINOS.

Data analyses

To compare richness and abundance of zooplankton across topsoil sediment treatments, we used a generalized linear model (GLM). Response variables were taxa number and number of hatchlings, and the model was adjusted for the Poisson distribution for richness and negative binomial distribution for abundance (due to major overdispersion of residuals). Statistical significances were assessed using the ANOVA function from car package which performs Wald Chi-Square Test (Fox and Weisberg 2019).

A permutational variance analysis (PERMANOVA) was used to assess differences in zooplankton composition among the five treatments, based on Bray–Curtis dissimilarity. Because this analysis is sensitive to data dispersion and may therefore confound between within-group variation and among-group variation, we performed an analysis of

multivariate homogeneity (PERMDISP), as recommended by Anderson and Walsh (2013). We used a Bray–Curtis dissimilarity matrix for PERMDISP and a permutation test (999 runs) to assess significance. Similarity percentages (SIMPER) analysis was used to investigate the sources of variation (taxa) responsible for composition dissimilarity identified in the PERMANOVA. A compound graph (Dambros 2014) was used to characterize species distribution across topsoil treatments. All analyses were performed using the vegan (Oksanen et al. 2019) and stats packages for R (R Development Core Team 2020).

Results

In the flotation experiment, 23 cladoceran ephippial eggs were found, distributed into two genera and two morphotypes (*Simocephalus*—6, *Ilyocryptus*—1, morphotype 1—9, and morphotype 2—7). All of them were observed in the natural wetland sediments (4, 2, 1 and 16 eggs in the four natural wetlands). The density of the ephippial eggs varied from 0.01 to 0.16 eggs/g of sediment.

A total of 61 hatchlings distributed across eight zooplanktonic taxa were recorded in the five treatments. Copepod nauplii were the most abundant (31.1%), followed by *Epiphanes* sp. (29.5%), *Ovalona glabra* (16.4%) and the bdelloid rotifers (9.8%) (Table 1). In mined wetland sediments (T1), just copepod nauplii and rotifers (*Cephalodella* sp. and Bdelloidea) hatched, while all the eight taxa were

found in natural wetland sediments (T5). While natural wetland sediments (T5) contained 42.6% of hatched zooplankton, treatments with added topsoil (T2, T3, and T4) contained 13.0, 16.4 and 21.3% of hatched zooplankton, respectively. Mined wetland sediments (T1) resulted in just 6.5%.

Zooplankton richness and abundance differed between treatments (Richness: $X^2 = 11.55$, $d.f. = 2$, $P = 0.021$; Abundance: $X^2 = 12.389$, $d.f. = 2$, $P = 0.015$), but only sediments from natural wetlands had positive relationship with both variables (Table 2; Fig. 1). Richness and abundance were higher in natural wetland sediments than in the other treatments. However, the richness and abundance of hatchlings did not change between mined wetland sediments (T1) and the different topsoil addition treatments from donor wetlands (T2, T3 and T4).

Hatchling assemblages showed differences in taxa composition associated with substrate type that explained 29% of variability observed (PERMANOVA, Pseudo $F_{4,15} = 1.563$, $P = 0.043$). The PERMANOVA results were not affected by multivariate dispersion ($F_{4,15} = 2.2815$, $P = 0.105$). SIMPER analysis showed that sediment from mining activities were particularly restricted to one cladoceran species (*O. glabra*) ($P = 0.004$) and rotifers (*Lecane* sp.) ($P = 0.03$) when compared to hatchlings from natural wetlands (Fig. 2). Alone *O. glabra* contributed to 21.6% of the overall dissimilarity. Abundance of copepod nauplii (Bray–Curtis dissimilarity = 0.52; $P = 0.014$) and rotifers from the subclass Bdelloidea (Bray–Curtis dissimilarity = 0.24; $P = 0.03$) were the

Table 1 Abundance of zooplankton hatchlings in southern Brazil wetlands

	T1	T2	T3	T4	T5	Abundance
COPEPODA						
Nauplii	2	5	2	1	9	19
CLADOCERA						
<i>Ovalona glabra</i>			1		9	10
<i>Simocephalus serrulatus</i>					1	1
<i>Simocephalus</i> sp.			1			1
ROTIFERA						
<i>Epiphanes</i> sp.		1	4	11	2	18
<i>Lecane</i> sp.					1	1
<i>Cephalodella</i> sp.	1	2		1	1	5
Bdelloidea	1		2		3	6
Richness	3	3	5	3	7	8
Abundance	4	8	10	13	26	61

T1 refers to sediment from mined wetlands; T5 refers to sediment from natural wetlands; mined sediments added with donor sediment from natural wetlands is T2 (5%), T3 (20%) and T4 (40%)

Table 2 Results of the GLM performed to test the effect of sediment type on zooplankton hatchlings in southern Brazil wetlands

	Treatment	Coefficients	SE	P
Richness	T1	-0.287	0.577	0.618
	T2	0.511	0.703	0.484
	T3	0.693	0.707	0.327
	T4	≈ 0	0.816	0.99
	T5	1.54	0.632	0.002*
Abundance	T1	≈ 0	0.654	0.99
	T2	0.693	0.855	0.417
	T3	0.916	0.840	0.275
	T4	1.179	0.826	0.153
	T5	2.327	0.793	0.003*

Significant differences were denoted with*. SE, standard error; T1 refers to sediment from mined wetlands; T5 refers to sediment from natural wetlands; mined sediments added with donor sediment from natural wetlands is T2 (5%), T3 (20%) and T4 (40%)

main taxa that accounted for variability in community composition among mined areas (T1) and low (T2) and medium topsoil addition (T3) treatments, respectively.

Discussion

It has been shown that mining and its associated activities have important environmental impacts not only at the mined area but at its surroundings, due to vegetation suppression and disruption of the surface layers of the soil (Santhosh et al. 2013; Dong et al. 2019). In the present study, the flotation experiment showed an absence of cladoceran ephippial eggs throughout mined wetlands. Also, the density of eggs in natural wetlands was low when compared with wetlands in other areas of southern Brazil (Vendramin et al. 2020), as well as the rest of the country (Santangelo et al. 2011) and the world (Brendonck and De Meester 2003). Several studies show that the density of invertebrate resting stages in the sediment throughout the world can be quite variable. According to Hairston (1996), the top sediment layer in natural wetlands can reach from 10^3 to 10^6 dormant stages of zooplankton species per square meter. In Australia, this is estimated as 1.2×10^7 (per square meter) (Shiel

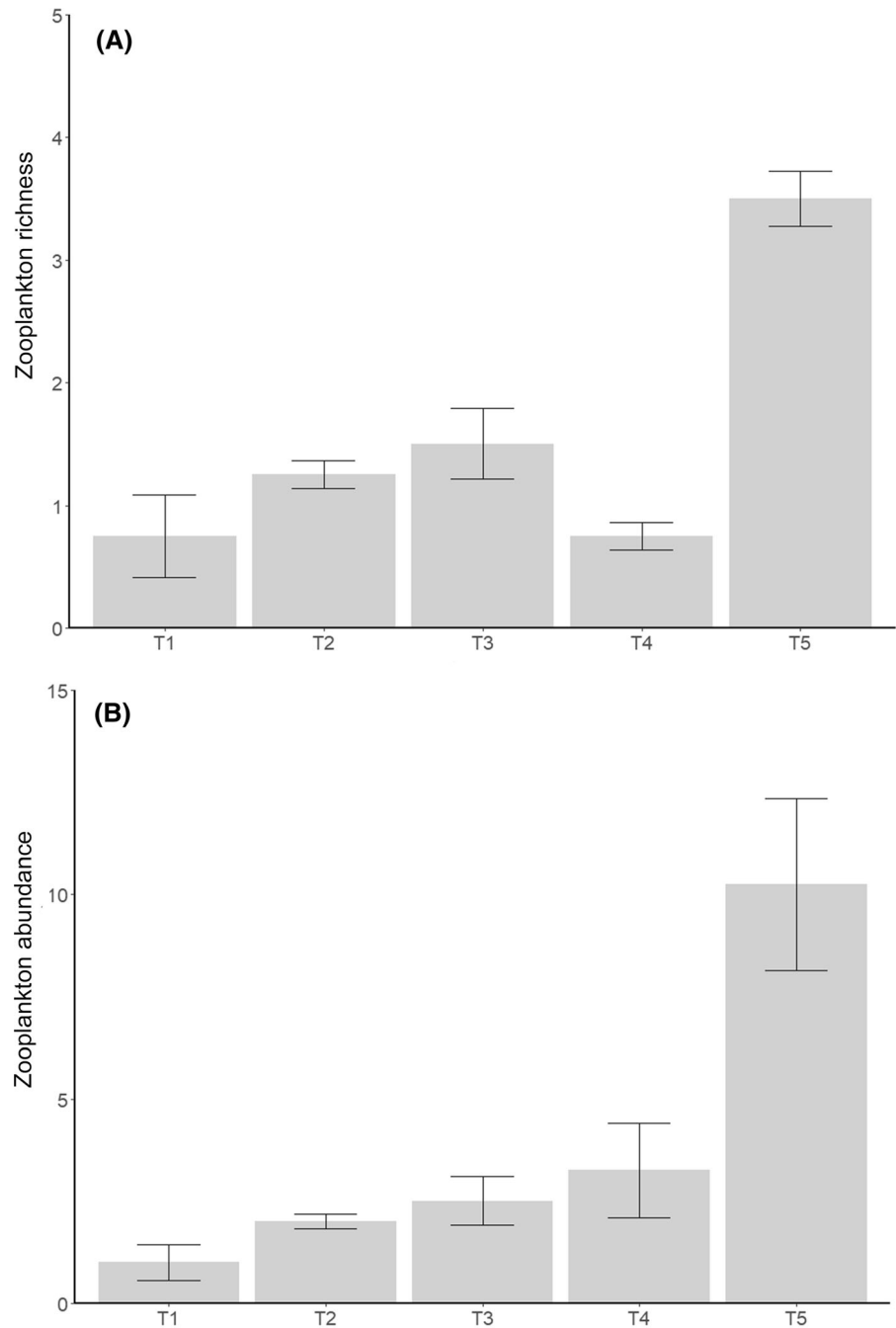
et al. 2001). On the other hand, García-Roger et al. (2008) have reported as low as eight eggs per gram of sediment in brackish ponds and a saline lake from Eastern Spain, and Bailey et al. (2004) only 1.25 eggs per gram of sediment from the Great Lakes in Canada.

The low observed richness and abundance of hatchlings in the experiment can therefore have been the result of a low density of dormant stages found in the sediment of the study wetlands. A range of studies show that the production and hatching of dormant stages of zooplankton are linked to specific biological cues triggered by water conditions (Vandekerckhove et al. 2005; Gaikwad et al. 2008; Broman et al. 2015; Brendonck et al. 2017). Our results differ from a range of studies performed in southern Brazil, where a greater richness and abundance of hatchlings from zooplanktonic dormant bank was found in rehydrated sediments under the same experimental conditions (Stenert et al. 2010, 2016, 2017; Ávila et al. 2015; Freiry et al. 2016, 2020a,b; Vendramin et al. 2020).

With that in mind, the density of dormant stages in the topsoil sediment of the non-mined study sites indicates that some wetlands have naturally low density of dormant stages, thus, not being suitable as donors of sediment in the context of restoring damaged wetlands. The prediction that the addition of topsoil to mined wetlands would reduce the community composition dissimilarity between mined and natural wetlands could not be asserted, given the low overall density of eggs in the present study natural wetlands. It cannot be implied, though, that the addition of topsoil to mined wetlands could not be an approach to their restoration. The same experimental methodology was performed in similar studies in southern Brazil, but none of them had wetland restoration as ultimate goal. These studies found high numbers of viable dormant eggs of zooplankton in natural wetlands (Freiry et al. 2016, 2020a, b; Stenert et al. 2016, 2017; Vendramin et al. 2020) and rice fields (Stenert et al. 2010; Ávila et al. 2015).

Therefore, the low number of dormant stages observed in the present study may be related to the portion of the Sinos River basin sampled. The lower region of the Sinos River corresponds to a highly populated and industrialized area in the south of Brazil which can negatively impact water quality, oxygen levels and the overall quality of natural habitats (Bieger et al. 2010; FEPAM 2019). During the flooding season, the studied natural floodplain

Fig. 1 Mean richness (a) and mean abundance (b) of zooplankton hatchlings in southern Brazil wetlands. Bars represent standard error. Richness corresponds to taxa number and abundance to number of hatchlings. T1 refers to sediment from mined wetlands; T5 refers to sediment from natural wetlands; mined sediments added with donor sediment from natural wetlands are T2 (5%), T3 (20%) and T4 (40%)



wetlands can connect to the main river channel, which may have affected the establishment of a healthy and viable zooplanktonic dormant bank. Many eutrophicated wetlands have anoxic sediments where the dormant stages of zooplankton are deposited (Broman et al. 2015). Sediments with oxygen deficiency are rich in toxic compounds, which potentially destroy buried

zooplankton resting stages (Bagarinao 1992). Besides, the increase in organic matter in the sediment originated from the main channel and from human activities can lead to eutrophication, which in turn increases microbial activity and other biological factors capable of destroying dormant eggs or deposit

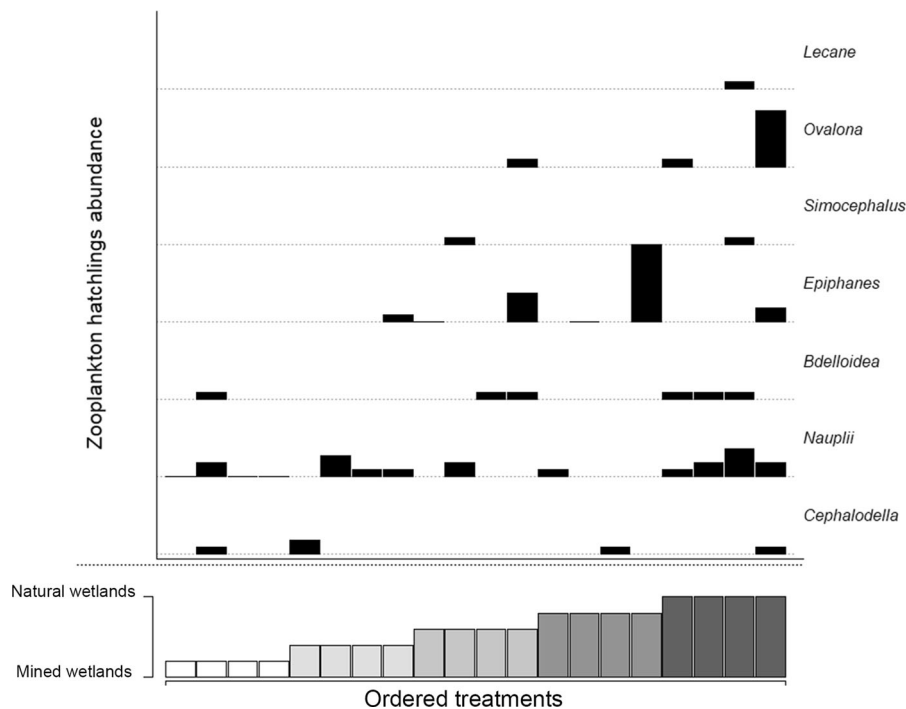


Fig. 2 Distribution of zooplankton hatchlings in different classes of topsoil. Bar size is proportional to total number of individuals. White to dark grey bars represent different

treatments (mining areas → lowly enriched → medium enriched → highly enriched → natural wetland)

waste and toxic products around the eggs turning them non-viable (Brendonk and De Meester 2003).

Overall, our results are in line with the idea that environmental disturbances mediated by humans can shape community structure by filtering species sharing some traits vulnerable to the new environmental conditions (Mol and Ouboter 2004; Palmer et al. 2010; Hitt and Chambers 2015). Hence, clay removal has a clear negative effect on the dormant propagule banks in wetlands, with only copepod nauplii and rotifers hatching in mined wetlands. Although we found no influence of the relocation of topsoil on richness and abundance of dormant zooplankton, our results showed that even a small addition may affect species composition, favoring the emergency of more generalist taxa.

Our expectations for topsoil addition were not fully supported, although the different outcomes suggest that species introduced together with the topsoil can become prey food for consumers thus aiding recovery of the biota in wetlands affected by human activities. It is important to recognize here that, although some species tolerate mining disturbances, they are less

abundant than in sediments from natural wetlands, since in mined wetland sediments just copepod nauplii and rotifers hatched. Rotifers are opportunistic organisms (Maia-Barbosa et al. 2014) well adapted to disturbance both natural (Melo and Medeiros 2013) and from anthropic origin (Ricci 2001; Cabral et al. 2019). Goździewicz et al. (2018) showed for reservoirs located in mined areas that the richness of zooplankton was represented mostly by rotifer species. Also, particulate organic matter and minerals in the water have been shown to be an important substrate for bacterial growth allowing for a greater abundance of rotifers (Mayer et al. 1997; Ejsmont-Karabin 2012).

While it might seem reasonable to assume that clay removal per se was responsible for the results presented, we should be careful when extrapolating which factor is driving zooplankton assemblage structure. Studies have shown that acidification, salinization and pollution by mining activities lead to the reduction in aquatic invertebrate (and other taxa) richness and to lower abundance on natural ecosystems (Bielańska-Grajner and Gladysz 2010;

Rönicke et al. 2010; Moser and Weisse 2011; Bielańska-Grajner and Cudak 2014; Ferrari et al. 2015; Sienkiewicz and Gąsiorowski 2016; Pocięcha et al. 2017). So, there are several factors related with clay mining influencing zooplankton assemblages. Exactly what is the most important is hard to identify and is beyond the scope of our experiment.

In conclusion, our results showed that some natural wetlands may not have a bank of zooplankton dormant stages large enough for them to be considered as donors of topsoil sediment to aid in the recovery of wetlands subjected to mining activities. These results show some degree of unpredictability in the strategy of sediment soil donation between natural wetlands and those subject to mining activities, which can be reduced as long as prior studies are performed to test the size and quality of the dormant banks present in the sediment of candidate donor wetlands. That is actually an important result from our study, since we draw attention to the fact that the quality of donor habitats (their topsoil, water and habitat structure), must also be taken into account when using the sediment transfers scheme in order to manage disturbed systems. This should be considered not only to wetlands but to other systems as well. Taking such conditions into account, sediment from donor wetlands can therefore aid in the establishment of a more diverse community in disturbed systems. Further studies on the use of in situ techniques for such remediation and species-traits that favor colonization will help to develop additional procedures for the proper management and restoration of wetlands. Such methodologies would enable not only the restoration of disturbed wetlands but also to help the conciliation between human activities and a lower impact of such activities to the natural environment.

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Authors' contribution LM and CS conceived the idea, sampling design, and led the writing of the manuscript. DV organized the field and laboratory work, carried out the statistical analyses, and drafted the manuscript. AGP, AEBS, LM, and PHOH participated in the laboratory work. ESFM and LFBM revised the statistical analyses and the manuscript. All authors read and approved the final version of the manuscript.

Data availability statement Data will be made available on reasonable request.

Declarations

Conflict of interest The authors declare no conflict of interest.

Ethical approval We declare that data collection complied with current Brazilian laws (SISBIO - 53781-4).

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