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Viability of Phytoplankton, Zooplankton, and Aquatic Macroinvertebrates in Dredged River Sediments of the Upper Mississippi River

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ABSTRACT

Sediment dredging is widely used to manage freshwater ecosystems, including maintaining river navigation corridors. Beneficial uses of the sediments are needed to reduce demands on new and existing stockpile sites. However, using dredged sediments in waterbodies outside the sediment source poses an ecological risk if aquatic organisms remain viable through desiccation-resistant forms. We assessed viability of aquatic organisms in dredged river sediments from four locations along the Upper Mississippi River. We artificially re-inundated sediments that had been dry for < 1, 1–5, and 5–10 years and sampled phytoplankton, zooplankton, and macroinvertebrate for 4 weeks. We hypothesized that richness and density of aquatic organisms would be negatively affected by duration of drying. Although individual samples of the most recently dredged material had the greatest richness and density of phytoplankton, zooplankton, and macroinvertebrates, mean richness and density was not different across sediment drying durations. Overall, richness and density of aquatic organisms was low in relation to in situ sampling especially following the first week of sediment re-inundation. Our results suggest transporting sediments from main channel navigation dredging from the Upper Mississippi River to support beneficial uses in other waterbodies poses some ecological risk given our observations of viable organisms in dredged sediments. We present a multistep risk evaluation procedure that could be applied when considering beneficial uses for dredged sediments.

1 | Introduction

Sediments enter rivers naturally from eroding landscapes creating and maintaining important habitats for freshwater organisms (Meade and Paker 1985; Belmont et al. 2011), but excess sediment is one of the most common pollutant of rivers and streams (U.S. EPA 2023). Excess sedimentation of rivers can occur due to anthropogenic alteration of watersheds (Walling 1999), impairing navigation of rivers and potentially reducing biodiversity (Belmont et al. 2011). To facilitate commercial and recreational navigation of large rivers worldwide, sediments are removed to maintain navigation channels through the process of mechanical or hydraulic dredging (U.S. Army Corps of Engineers 1996; Casper 2008). Dredged river sediments are often stockpiled on islands within the river floodplain or nearby riparian areas until beneficial uses are found. Little is known about organisms that may remain in these sediments following the dredging process. Therefore, the consequences of transporting dredged river sediments to new freshwater habitats remains an important question.

Many freshwater phytoplankton, zooplankton, and macroinvertebrates have desiccation-resistant forms that allow persistence in habitats despite the loss of surface water (Cáceres 1997; Stubbington and Datry 2013; Strachan, Chester, and Robson 2015). Desiccation resistance could enable individuals to survive periods of time when river sediments are moist or dry. However, relatively few studies have explored how prevalent desiccation-resistant organisms are in freshwater ecosystems (Ellegaard and Ribeiro 2018; Stubbington and Datry 2013). Phytoplankton, including diatoms, green algae, and cyanobacteria, are known to have resisting stages that can remain viable for hundreds of years (Radzikowski 2013; Ellegaard and Ribeiro 2018). Zooplankton, including cladocerans and rotifers, may use desiccation-resistant life stages to overcome periods of harsh environmental conditions, such as drying (Radzikowski 2013; Vargas, Santangelo, and Bozelli 2019). Larger freshwater invertebrates (i.e., macroinvertebrates) also have desiccation-resistant forms, including resistant eggs, protective cases, and dormant larvae, that facilitate persistence to harsh environmental conditions (Bogan et al. 2017). Despite evidence that phytoplankton, zooplankton, and macroinvertebrate use desiccation-resistant forms, the presence in sediments and viability of these organisms following reestablishment of favorable conditions has been found to be highly variable and context dependent (Datry et al. 2017; Ellegaard and Ribeiro 2018).

Drying duration, re-inundation period, and historic drying prevalence are important factors to consider in studies of desiccation resistance (Stubbington and Datry 2013; Datry et al. 2017). Survival rates of organisms using desiccation-resistant forms decline in moist or dry sediments during prolonged periods of desiccation (Stubbington and Datry 2013; Vargas, Santangelo, and Bozelli 2019). For example, Stubbington and Datry (2013) reported a steady decrease in taxonomic richness and abundance emerging from dry sediments as dry period duration increased from 0.1 days to 64 days in a meta-analysis of 10 previous studies in intermittent rivers. Studies of phytoplankton resting stages found abundances can vary greatly, from zero to 10,000 individuals per gram of dry sediment. In most cases, viable individuals become active within a few hours to days of rewetting (Bogan et al. 2017), however, relatively little is known about the timing and conditions needed to break dormancy. Therefore, the potential presence of organisms in river sediments with desiccation-resistance forms suggests transporting river sediments from one waterbody to another may pose a risk of introducing novel invertebrate taxa to the receiving waterbody.

The objective of this study is to assess the viability of phytoplankton, zooplankton, and macroinvertebrates in dredged sediments from the Upper Mississippi River. The Upper Mississippi River is similar to many large rivers worldwide, requiring sediment dredging to maintain navigation corridors. We quantified the abundance and taxonomic richness of phytoplankton, zooplankton, and macroinvertebrates in dredged sediments following artificial rewetting. We hypothesized that time since dredging, that is, sediment drying duration, would negatively affect richness and abundance of organisms that emerge from sediments after re-inundation. We assessed risks of transporting biota in sediments to other locations by providing: (1) a taxa list of viable invertebrates from dredged sediments from several dredging locations on the Upper Mississippi River, (2) a comparison of invertebrate viability in relation to sediment time since rewetting and dredging location.

2 | Methods

2.1 | Study Area

The Upper Mississippi River System, comprising 2000 river kilometers from Minneapolis, MN to near Cape Girardeau, MO, is a large river system managed to allow commercial navigation and other public and private uses (U.S. Army Corps of Engineers 2011, Figure 1). Commercial vessels follow a navigation channel maintained through a system of locks and dams and dredging. The locks and dams create a series of river reaches, referred to as pools and a long-term resource monitoring program exists on several of these pools, including Pool 4 and Pool 8 (Figure 1). The dredged sediment from the main channel of the Mississippi River in Wisconsin is 95%–99% sand (<2 mm; U.S. Army Corps of Engineers 1996). Once removed, the material is stockpiled into temporary or permanent stockpile sites to be made available for beneficial use. Primary upland uses of material stockpiled in the upper reaches of the river are general fill, winter road abrasive, and dairy bedding. Primary terrestrial and aquatic uses of dredged material include habitat restoration and shoreline protection. To date, beneficial use of dredged sediments has occurred within the Upper Mississippi River System exclusively, however, dredged material could service habitat restoration needs within other freshwater systems, including the Great Lakes. A key factor in sediment suitability for those purposes is the risk of introducing phytoplankton, zooplankton, or macroinvertebrates, including known invasive species, into the receiving waters.

Within the study area, wet dredged material is stockpiled at beneficial use sites (Figure 1) and is mechanically mixed through further site excavation and shaping. There is minimal silt and organic matter in the material and consequently minimal moisture retention properties within the well-drained, sandy material (Walczak, Witkowska-Walczak, and Slawinski 2002; U.S. Army Corps of Engineers 1996). Stockpiled dredge sediments can reach surface temperatures of 50°C, limiting the colonization by wind-blown seed or rhizomes. In some cases (Wabasha and Brownsville locations), sediment is moved hydraulically using river water to an upland placement site prior to beneficial use. This has two potential opposing effects on biota; first it provides a potential reintroduction of plankton in carriage water, which is expected to be minor based on the velocities, abrasive particles, and turbulence of the water in the pipeline. Second, the in situ dredge material contains roughly 1%–4% material less than 75-μm in size, which is further sorted through hydraulic offload, reducing the placement site values to roughly 0.5%–1.2% (Coor and Ousley 2019, Strassman unpublished data).

2.2 | Sediment Collection

To assess invertebrate viability in dredged river sediments and account for potential differences in invertebrate viability correlated with increased drying duration, we collected sediment

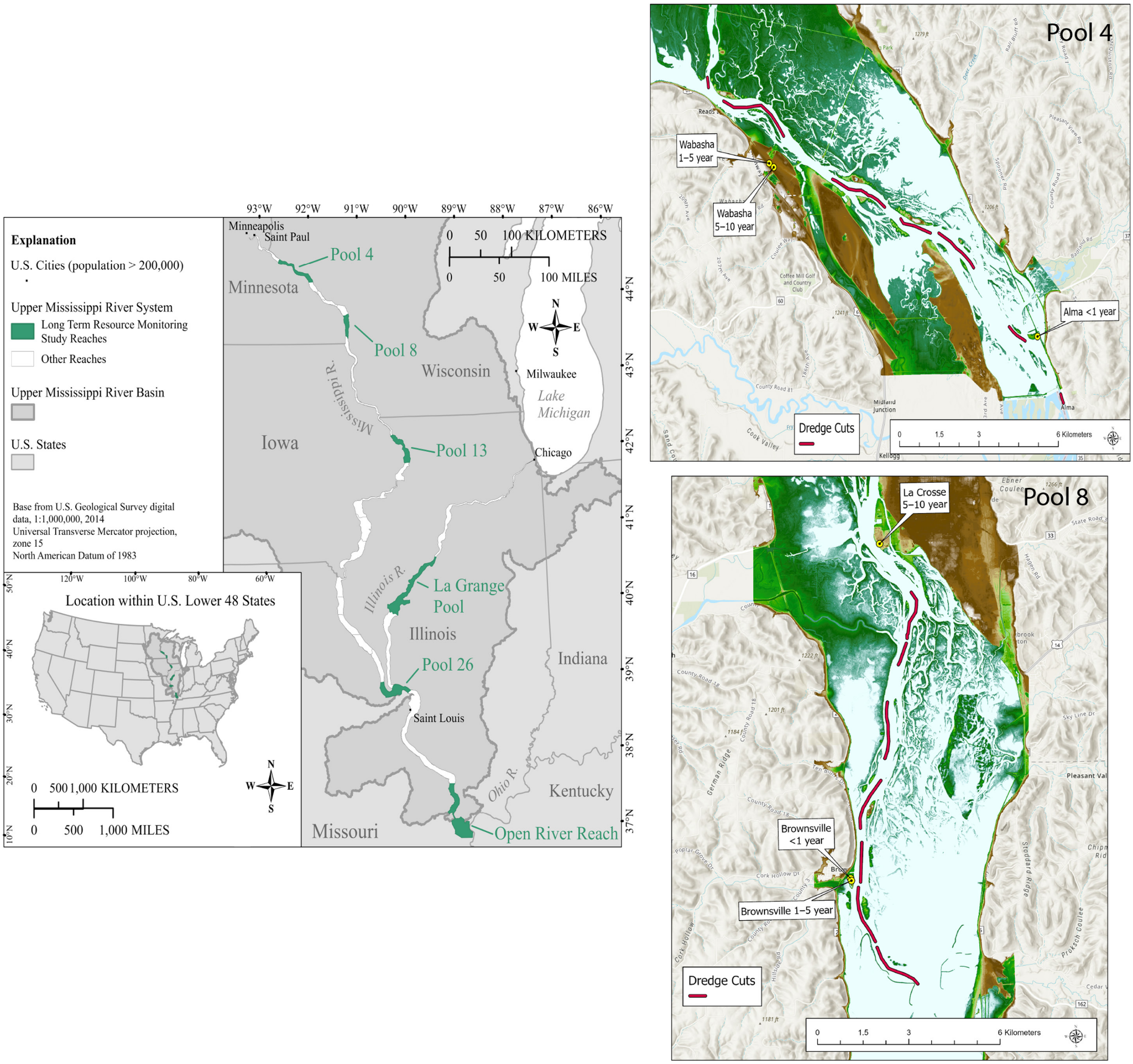

FIGURE 1 | Location of dredged sediment storage sites and dredge cuts (site of dredging) in Pool 4 and Pool 8 of the Upper Mississippi River where sediments were collected for the sediment rewetting experiment. [Color figure can be viewed at wileyonlinelibrary.com]

samples ($n=30$) ranging in time since dredging (age groups: <1, 1–5, 5–10 years) and from four dredge stockpile locations (Wabasha, Alma, Brownsville, and La Crosse). A total of 5 replicate samples from each age class and dredge stockpile combinations were collected. Dredge piles were created from dredge cuts done in nearby areas on Pool 4 and Pool 8 of the Upper Mississippi River (Figure 1). Pool 4 is located immediately downstream from a large lake (Lake Pepin), a known source of phytoplankton blooms. Pool 8 represents riverine habitat with many side channel and backwater areas. Sediments were collected on 14 June 2021 from six unique locations (Table 1).

We removed approximately the first 15-cm of dredge piles before sediment collection to avoid wind-deposited sediments and sediments directly exposed to UV light. Three-liters of dredged river sediments were then collected in food-safe plastic containers (capacity = 18.9 L). Within 24 h of collection, sediments were artificially re-inundated with filtered river water and oxygenated using electric air pumps with air stones. The buckets were stored outside and were exposed to indirect light and ambient air temperatures ranging from 16°C–27°C.

2.3 | Upper Mississippi River Water Collection

River water used to re-inundate sediments was collected along the main channel of the Mississippi River in La Crosse County, WI. Water was run through a 45-μm sieve net attached to PVC piping to remove phytoplankton, zooplankton, and macroinvertebrates. In the lab, water was further filtered using a 1.5-μm glass mesh fiber filter (Whatman, 934-AH) to remove any remaining organisms. We used filtered river water versus using

TABLE 1 | Location and attributes of dredge sites along the Upper Mississippi River where sediments were collected for the sediment rewetting experiment.

Location	Navigation pool	Year class	Dredging year	Latitude	Longitude
Alma	4	< 1	2021	44.339441	−91.928730
Brownsville	8	< 1	2021	43.692184	−91.272708
Brownsville	8	1–5	2016	42.692766	−91.273101
La Crosse	8	1–5	2020	43.790965	−91.257970
Wabasha	4	1–5	2016	44.381960	−91.052864
Wabasha	4	5–10	2011	44.386636	−92.053012

a sterile water source that would not represent natural water chemistry supportive of growth and development of emerging organisms. We processed a blank sample of filtered river water for phytoplankton to ensure that biota had been effectively removed.

2.4 | Phytoplankton, Zooplankton, and Macroinvertebrate Sampling

Phytoplankton samples were taken from a subset of buckets ($n = 5$) from two of our four test sites and collected after 48-h of artificial re-inundation. Only < 1 and 1–5-year age classes were assessed for phytoplankton. For each sample, 200 mL of well-mixed water was poured from each bucket, without sediment agitation, into sample and preserved with 1-mL of Lugol's Solution (a solution of potassium iodide and iodine). Cell numbers of all identifiable phytoplankton taxa were quantified on a per milliliter basis using the Utermöhl method (Lund, Kipling, and Le Cren 1958) in accordance with American Public Health Association Standard Method 10,200 (American Public Health Association 2012). Samples were thoroughly mixed prior to subsampling an aliquot to ensure that the organisms will be evenly distributed. Utermöhl counts were performed on a LEICA DMi1 or DMIL inverted microscope at 800×. Cell biovolumes of all identified phytoplankton taxa were quantified on a per milliliter basis. Biovolumes were estimated using formulae for solid geometric shapes that most closely matched the cell shape (Hillebrand et al. 1999). Biovolume calculations were based on measurements of 10 organisms per taxon for each sample where possible.

Zooplankton and macroinvertebrate samples were taken at 7, 14, 21, and 28 days after re-inundation on: June 23, June 30, July 8, and July 14, 2021. Samples were collected following thorough mixing of the water column and upper 5-cm of the sediment surface by hand. The water column was then poured through a 500-μm sieve to collect macroinvertebrates and then through a 60-μm sieve to collect zooplankton. Invertebrates were washed into glass jars and stored with 70% ethanol. Zooplankton were washed into glass jars and stored in a 10% Formalin solution (4% formaldehyde). In the lab, Rose Bengal solution was added to aid in counting small organisms, mainly zooplankton. Zooplankton and macroinvertebrates were identified to the lowest practicable taxonomic level (Haney et al. 2024, Merritt, Cummins, and Berg 2019).

2.5 | Statistical Analysis

We used one-way analysis of variance tests (ANOVA) to compare the effects of dredging year (< 1, 1–5, 5–10 years) and location (Pool 4, Pool 8) on taxonomic richness and abundance or biovolume (dependent variable) of phytoplankton, zooplankton, and macroinvertebrates. Separate tests were run for each combination of independent and dependent variables across the different taxonomic groups. Plotted residual variances and Levene's test were used to validate ANOVA model assumptions. Subsequently, $\log_{10}(x)$-transformations were used to improve normality and homogeneity of variance.

Indicator species analysis was used to explore relationships between phytoplankton and macroinvertebrate taxa and dredging years, using the R package "indicspecies" (De Cáceres and Legendre 2009). This analysis calculates indicator values, an association between the taxa and site group, for each taxa and tests the statistical significance of these associations using permutation tests ($n = 999$). We were unable to perform indicator species analysis for zooplankton due to low taxa richness. All statistical analyses and figures were created using R (R version 4.1.1, R Core Team 2022).

3 | Results

3.1 | Phytoplankton Richness and Biovolume

A total of 10 phytoplankton taxa within three functional classifications were found after rewetting sediments that had been previously dry for < 1 and 1–5 years (Table 2). There was no difference in mean phytoplankton richness between the < 1-year (3 ± 2.2 taxa, mean ± SD) and 1–5-years (1.6 ± 0.9 taxa) age groups (one-way ANOVA, Table 3). Mean total biovolume of phytoplankton cells across all subsamples was 7.73E + 8 μm^3/L, with individual values ranging from 0 to 6.6E+9 μm^3/L. There was no difference in mean phytoplankton biovolume between the < 1 and 1–5-year age groups (one-way ANOVA, Table 3, Figure 2). There was no phytoplankton found in the control consisting of filtered river water.

3.2 | Zooplankton Richness and Density

A total of 2 zooplankton taxa (Rotifera and Copepoda) were found after rewetting sediments that had been previously dry for <1, 1–5, and 5–10 years. Across all samples ($n=120$), there were between 0 and 18 zooplankton individuals found per sample (1 ± 2.8 mean ± SD). There was no difference in mean zooplankton abundance between the <1-year (0.5 ± 0.9 taxa), 1–5-year (0.9 ± 3.0 taxa), and 5–10-year (2.2 ± 4.2 taxa) groups (one-way ANOVA, Table 3, Figure 3). Zooplankton abundance was similar in samples collected from Pool 4 (1.0 ± 2.6) and Pool 8 (0.9 ± 3.0; one-way ANOVA, Table 3). Across all sample types, zooplankton emergence occurred immediately following rewetting, within the first week of rewetting. No organisms emerged in weeks 2–4.

3.3 | Macroinvertebrate Richness and Density

A total of 18 macroinvertebrate taxa were found after rewetting sediments that had been previously dry for <1, 1–5, and 5–10 years (Table 4). Across all samples ($n=120$), there were between 0 and 102 macroinvertebrate individuals found per sample (7 ± 12.2; mean ± SD). Macroinvertebrate richness in <1 and 5–10-year group were significantly higher than in 1–5-year samples (one-way ANOVA, Table 3, Figure 4). Macroinvertebrate abundance in <1 and 5–10 year samples were significantly higher than in 1–5-year samples (one-way ANOVA, Table 3). Samples collected from Pool 8 had significantly higher macroinvertebrate richness (one-way ANOVA, Table 3) and abundance (one-way ANOVA, Table 3) compared to Pool 4.

TABLE 2 | Phytoplankton taxa found after rewetting dredged sediments collected from Pool 4 and Pool 8 of the Upper Mississippi River that had been previously dry for <1 year and 1–5 years.

Taxonomic name	Taxonomic group
Cyclotella sp.	Diatom
Diatoma sp.	Diatom
Encyonema sp.	Diatom
Gomphonema sp.	Diatom
Navicula sp.	Diatom
Chlorella spp.	Green algae
Microspora sp.	Green algae
Monoraphidium griffithii	Green algae
Trachelomonas sp.	Euglenoid

3.4 | Indicator Species Analysis

We found no phytoplankton taxa indicative of specific drying durations or sampling locations. The macroinvertebrate taxa Ephemeroptera larvae and Chironomidae larvae were associated with sediments dried for less than 1 year (indicator value (ind.val.) = 0.63, $p=0.0003$ and ind.val. = 0.55, $p=0.0343$, respectively). Nematoda was indicative of sediments dried for 5–10 years (ind.val. = 0.75, $p<0.001$). Ephemeroptera larvae were only found at the Brownsville sampling location (ind.val. = 0.62, $p<0.0001$).

4 | Discussion

In this study, we assessed the viability of phytoplankton, zooplankton, and macroinvertebrates in sediments artificially rewetted following dredging from two navigation pools in the Upper Mississippi River. We found a total of 10 phytoplankton, 2 zooplankton, and 18 macroinvertebrate taxa after artificially rewetting sediments for up to 4 weeks. Despite taxa remaining viable in dredged sediments, taxa richness and abundance of phytoplankton, zooplankton, and macroinvertebrates was low compared to in situ samples from past studies. We found no support for our hypothesis that duration of sediment drying (<1, 1-5, 5–10-year groups) would negatively affect richness and abundance of all three taxonomic groups.

TABLE 3 | The effect of drying duration (Year), rewetting duration (Week), and site location (Pool) on dependent variables related to phytoplankton, zooplankton, and macroinvertebrate abundance and richness, determined using ANOVA tests.

Taxonomic group	Dependent variable	Factor	d.f.	Mean Square	*F*	*p*
Phytoplankton						
	Biovolume (log10(x))	Year	8	3.76	3.755	0.771
	Richness	Year	8	4.90	1.690	0.230
Zooplankton	Abundance (log10(x))	Year	117	0.81	1.780	0.173
		Pool	118	0.05	0.108	0.744
Macroinvertebrate	Abundance (log10(x))	Year	117	1.64	4.707	**0.011**
		Year × Week	114	2.25	7.261	**0.001**
		Pool	118	2.93	8.425	**0.004**
	Richness (log10(x))	Year	117	0.36	4.424	**0.014**
		Year × Week	114	0.08	0.976	0.380
		Pool	118	0.37	4.528	**0.035**

Note: Biovolume and abundance for all taxonomic groups and macroinvertebrate richness were log10(x)-transformed. Statistical significance is indicated in bold.

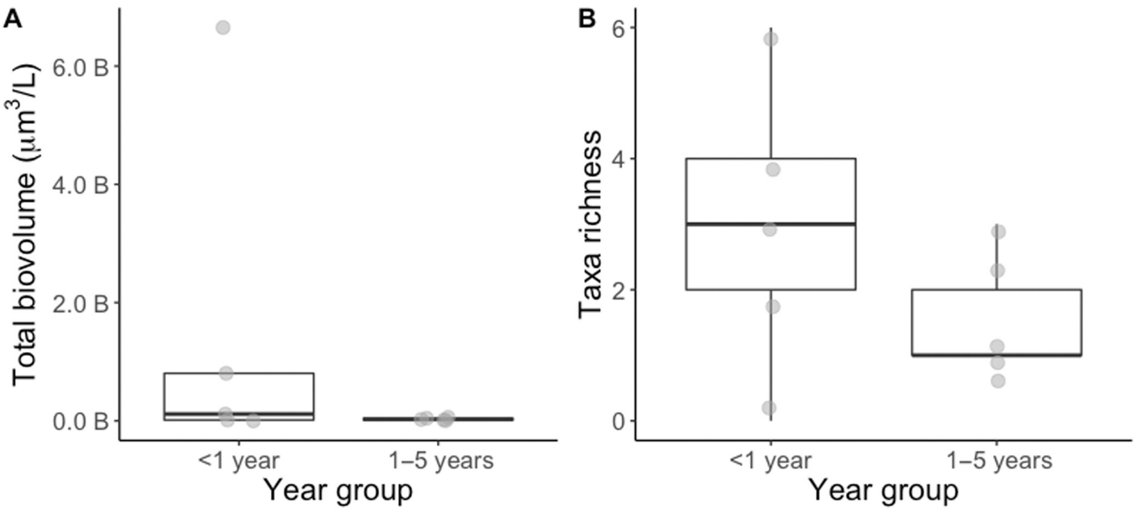

FIGURE 2 | Phytoplankton biovolume and taxonomic richness collected from re-inundated river sediments following dredging in Pool 4 and Pool 8 of the Upper Mississippi River from four dredge stockpile locations (Wabasha, MN, Alma, WI, Brownsville, WI, and La Crosse, WI, USA) with different duration of drying since dredging.

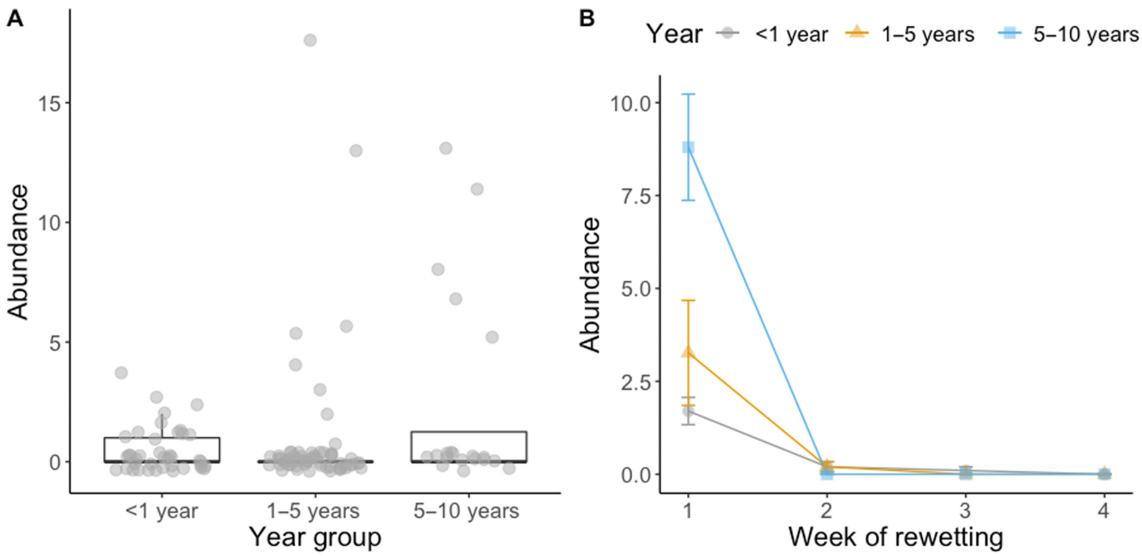

FIGURE 3 | Zooplankton abundance collected from re-inundated river sediments following dredging in Pool 4 and Pool 8 of the Upper Mississippi River from four dredge stockpile locations (Wabasha, MN, Alma, WI, Brownsville, WI, and La Crosse, WI, USA) with different duration of drying since dredging (A) and timing of emergence following re-inundation (B). [Color figure can be viewed at wileyonlinelibrary.com]

Previous literature suggests the duration of sediment drying plays an important role in the viability of aquatic organisms in dry sediments (Stubbington and Datry 2013; Ellegaard and Ribeiro 2018; Vargas, Santangelo, and Bozelli 2019). Our results only partially support that there is a negative relationship between duration of sediment drying and the abundance and richness of aquatic organisms emerging from rewetted sediments. Our highest levels of phytoplankton and macroinvertebrate richness and density were found in sediments dredged <1 year before rewetting. However, we found few significant correlations between richness or density and duration of sediment drying. This is likely due, in part, to the high amount of variability in richness and density found within each year group. This indicates viability of organisms may differ within sediments of the same drying duration. For example, one site (Brownsville) sampled in the <1-year group comprised sediments that were actively being deposited on shore from a dredge cut, whereas a replicate site (Alma) from the same year group comprised sediments collected approximately 2 weeks after dredging. Richness and density of macroinvertebrates in sediments collected from the Brownsville site were 1.7× and 5.4×, greater than those collected at Alma, respectively. Future efforts aiming to quantify the effect duration of sediment drying has on viability of aquatic organisms could attempt to select sediments along a continuous gradient of duration rather than categories as done in this study.

The presence of taxa with desiccation-resistant forms in a biotic community may be context or habitat dependent (sensu Datry et al. 2017, Lair 2006) affecting the viability of aquatic

TABLE 4 | List of macroinvertebrate taxa found in rewetted sediments from Pool 4 and Pool 8 of the Upper Mississippi River that had been previously dry for < 1 year and 1–5 years.

Order	Family	Life stage	Habitat	% of total abundance	Week of peak emergence (trend)
Araneae	Arachnida	Adult	Terrestrial	< 1%	Weeks 2–3
Coleoptera	Unknown	Adult	Terrestrial	12%	Weeks 1–4, steady
Diptera	Ceratopogonidae	Larvae, adult	Aquatic	< 1%	Weeks 1–4, steady
Diptera	Chironomidae	Adult	Aquatic	5%	Weeks 1–4, decline weekly
Diptera	Sciaridae	Adult	Terrestrial	< 1%	Weeks 1–4, steady
Diptera	Simuliidae	Lavae, adult	Aquatic	< 1%	Weeks 1–2
Entomobryomorpha	Entomobryidae	Adult	Terrestrial	26%	Week 4
Ephemeroptera	Leptohyphidae	Larvae	Aquatic	< 1%	Week 1,2
Ephemeroptera	Unknown	Larvae	Aquatic	31%	Week 1,2
Hymenoptera	Unknown	Adult	Terrestrial	< 1%	Week 4
Nematoda[a]	Unknown	Adult	Aquatic	17%	Weeks 1–4, decline weekly
Oligochaeta[b]	Unknown	Adult	Aquatic	< 1%	Week 1
Symphypleona	Sminthuridae	Adult	Semi-aquatic	< 1%	Week 2
Trichoptera	Leptoceridae	Adult	Aquatic	< 1%	Week 2
Trichoptera	Hydroptilidae	Adult	Aquatic	< 1%	Week 4
Trombidiformes	Unknown	Adult	Terrestrial	< 1%	Week 2, 4

[a]Phylum.
[b]Subclass.

organisms in sediments at differ across locations. When sample sizes allowed for comparison, we found sampling location had no effect on the richness and density of zooplankton and macroinvertebrates. This suggests that many Upper Mississippi River main channel sediment stockpiles may have similar desiccation-resistant biota profiles based on the similarities of the hydraulic and habitat conditions near the main channel dredge cut locations that likely drive community composition. Dredged sediments in this study had low percentages of fine particles (fewer than 2% particles less than 75-μm in size), reducing moisture content, and could have been exposed to high temperatures (> 50°C) and sunlight. Therefore, the physical composition of dredged sediments and environmental exposure may reduce the likelihood of viable taxa following rewetting considering moisture, light, and temperature are key factors that influence re-establishment following dormancy (Ellegaard and Ribeiro 2018; Stubbington and Datry 2013).

In comparison to previous in situ studies, richness and abundance of phytoplankton, zooplankton, and macroinvertebrates collected from rewetting dredged sediments were low. For example, Manier et al. (2021) collected 47 phytoplankton genera from Pools 8 and 13 and cyanobacteria were present in 96% of samples. Appel et al. (2020) found 40 unique zooplankton taxa with abundances ≥ 4000 ind./L. At least 144 macroinvertebrate taxa inhabit benthic substrates in the Upper Mississippi River (Angradi et al. 2009). Therefore, we expect that a low proportion of taxa inhabiting the large river can remain viable following the dredging process and subsequent desiccation.

From an evolutionary standpoint, taxonomic richness and abundance of riverine communities is influenced by their associated environmental conditions (Lytle and Poff 2004). Therefore, there may be higher prevalence of taxa that use desiccation-resistant forms in habitats that have historically experienced drying conditions. For example, Datry et al. (2017) found differences across sites in the proportion of macroinvertebrate taxa with desiccation-resistant forms in communities were positively correlated with increasing regional drying prevalence. In the context of our study, aquatic organisms collected from the main river channel have likely experienced similar environmental conditions and little desiccation risk which could explain our lack of differences in viability across sites. Future studies could compare main channel sediments to those in floodplain habitats, where drying occurs, to further explore how localized site and habitat conditions may drive community composition, as supported by other studies (Lair 2006; Casper and Thorpe 2007).

5 | Applicability of Upper Mississippi River Dredged River Sediments in Beneficial Uses

One potential beneficial use of dredged sediments from the Upper Mississippi River is transporting them to other waterbodies,

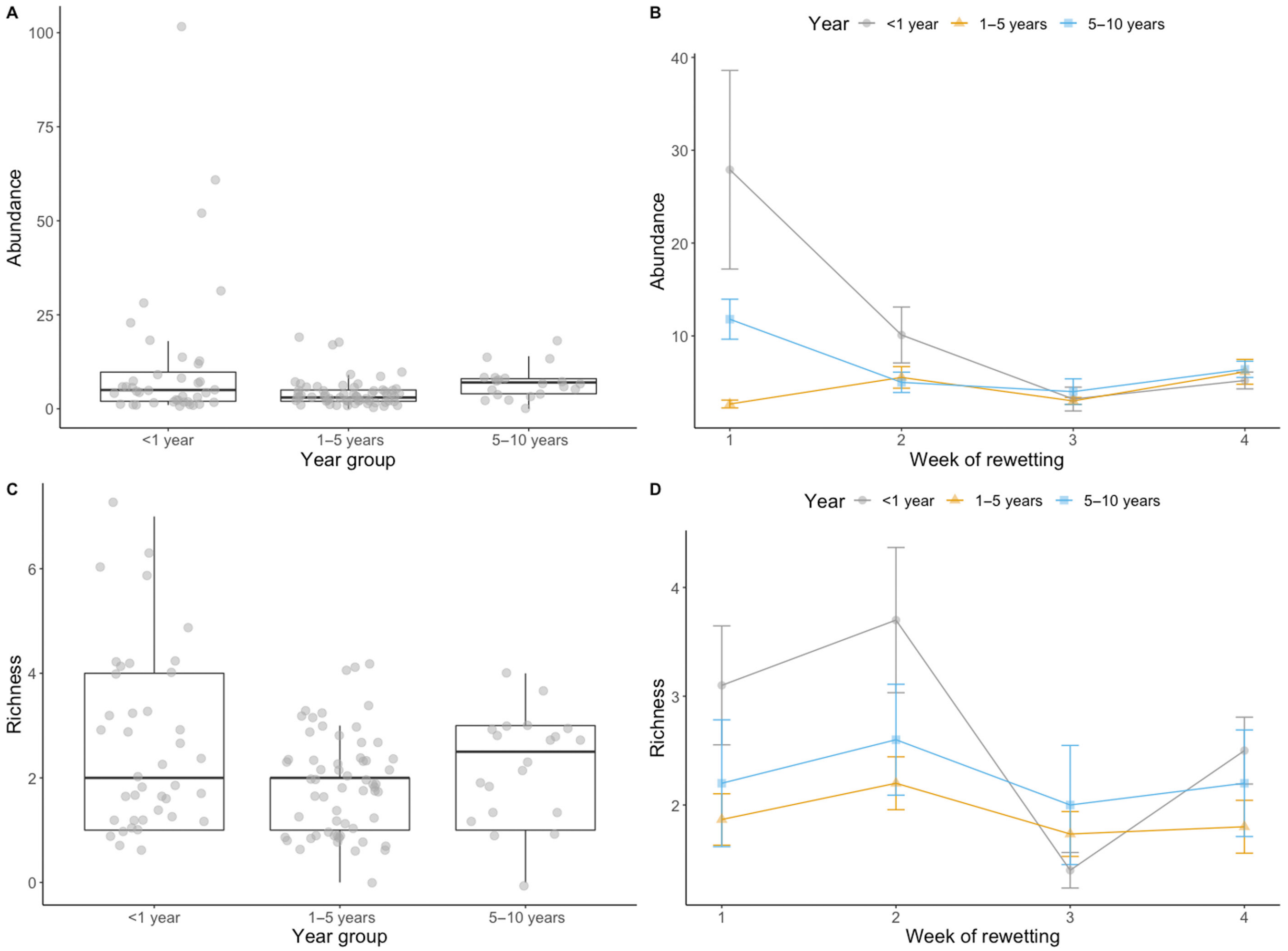

FIGURE 4 | Macroinvertebrate abundance and taxonomic richness collected from re-inundated river sediments following dredging in Pool 4 and Pool 8 of the Upper Mississippi River from four dredge stockpile locations (Wabasha, MN, Alma, WI, Brownsville, WI, and La Crosse, WI, USA) with different duration of drying since dredging (A, C) and after 1–4 weeks of re-inundation (B, D). [Color figure can be viewed at wileyonlinelibrary.com]

including the Great Lakes, to create or maintain sandy shorelines. We explored the risk of transporting biota, including invasive species, into other waterbodies and developed a risk evaluation procedure that could be broadly applied to other waterbodies.

We compared the taxa list generated through the replicated re-wetting experiment to known invasive species within the Upper Mississippi River. Of the 30 species currently considered as exotic or native transplants in the Upper Mississippi River drainage (Table S1), none were detected in rewetted sediments during this study. However, the coarse taxonomic resolution of zooplankton in this study does not allow us to confidently rule out the possibility of invasive zooplankton in dredged sediments. For three of the most likely invasive zooplankton known in the region, *Daphnia lumholtzi*, *Cercopagis pengoi*, and *Bythotrephes longimanus*, we did not detect any individuals in our samples based on their distinguishing characteristics. For other potential invasive zooplankton (e.g., copeopods, rotifers) known to the region, we could not rely on distinguishing characteristics although most records of these taxa are from the Great Lakes and nearby tributaries (U.S. Geological Survey 2024).

One possible explanation for not finding invasive species in rewetted sediments despite their presence in the Upper Mississippi River is that many of the presently known invasive species do not have desiccation-resistance forms. We could not find desiccation thresholds for all invasive taxa in the literature highlighting a knowledge gap that could be addressed in future studies. Most freshwater bivalves have limited ability to resist desiccation, including invasive zebra mussels (*Dreissena polymorpha*) which have a LT_{99} of 42 h in desiccation experiments (Collas et al. 2014). However, taxa known to possess desiccation resistant forms, such as *D. lumholtzi* (Panov and Caceres 2007; Strachan, Chester, and Robson 2015; Bogan et al. 2017) were also not detected in this study despite presence in lentic habitats immediately upstream from our Pool 4 study site (Burdis and Hirsch 2005). For invasive species known to use desiccation-resistant forms, one may predict the absence of these taxa is due to lack of life history or environmental cues that trigger these use of these forms (Strachan, Chester, and Robson 2015; Ellegaard and Ribeiro 2018). However, Smith, Acharya, and Jack (2009) found *D. lumholtzi* produced desiccation-resistant resting eggs despite the absence of drying suggesting factors other than desiccation influence the presence of desiccation-resistant taxa. Given we could not precisely measure desiccation duration and environmental exposure of the dredged sediments studied, it is difficult to assess the effect of all potential factors. However, we expect that the mobile sands comprising dredge cuts in the main channel of the Upper Mississippi River are less likely to host invertebrate taxa with desiccation-resistance forms (Chester and Robson 2011) than those from habitats with greater organic matter content (Stubbington and Datry 2013) or slower water velocities (Lair 2006; Shiozawa 1991; Richardson 1992).

Comparing habitat and environmental characteristics (e.g., water quality) of the source and receiving waterbodies is needed to evaluate the likelihood of invasive species establishment. For example, habitats of similar substrate, water velocity, and nutrient levels between waterbodies could facilitate the establishment of transported species. The risk of establishment could be mitigated by placing beneficial use sites in habitats outside of the environmental tolerances of the viable taxa within dredged sediment. Furthermore, placement of sediments could be done in locations or time periods that place abiotic and biotic limitations on desiccation-resistant forms, such as light, temperature, oxygen, and predation (Ellegaard and Ribeiro 2018). Lastly, it may be possible to reduce presence of viable biota through treatment processes (UV exposure) or active harvesting, such as rewetting sediments in a controlled setting and exchanging the overlying water after emergence has occurred.

From a management perspective, the following steps could be used as part of a risk evaluation procedure to organize information and inform decision-making of sediment beneficial-use projects. (1) Collate known and likely invasive species within the dredging waterbody and ensure the duration of sediment drying exceeds any known lethal thresholds for those species. (2) Conduct repeated and controlled artificial rewetting experiments using the method presented here to quantify emergence. (3) Compare habitat and environmental conditions between source and receiving waterbodies to evaluate the likelihood of invasive species establishment, assuming adequate data exists. Finally, an evaluation of practical dredged material treatment (e.g., UV exposure, rewetting) steps may offer implementation guidelines that reduce uncertainty around potential introductions.

6 | Conclusions

Our results provide an initial assessment of the viability of aquatic organisms in dredged river sediments of the Upper Mississippi River. Although invasive species were not present after artificially rewetting the sand-dominated, main channel sediments, proposed transfers of dredged sediments between waterbodies could use the risk evaluation procedure developed here as a precautionary measure. Our results did not support the hypothesis that the presence of desiccation-resistant forms in a community is positively correlated with drying prevalence (sensu Datry et al. 2017), based on the lack of statistical differences among year groups. To further test this hypothesis within the context of large rivers, future studies could compare viability of aquatic organisms in sediments from floodplain habitats subjected to wet/dry cycles to those found in the main river channel where dredging occurs.

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Data Availability Statement

The authors confirm that the data supporting the findings of this study are available within the article [and/or] its Supporting Information.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section.