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Response of yellow perch to water level fluctuations in oligotrophic, north-temperate inland lakes

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Abstract

Information on yellow perch *Perca flavescens* population dynamics and responses to various abiotic and biotic factors in oligotrophic, north-temperate inland lakes is limited. Water level fluctuations are known to influence available habitat and biological communities within the littoral zones of lakes, yet research is lacking for yellow perch in Wisconsin. The goal of our study was to characterize yellow perch population-level responses to natural water level fluctuations in four northern Wisconsin lakes using a 39-year time series. On average, increasing water level periods correlated with lower mean fyke net and gill net relative abundances (catch-per-unit-effort), though generally not statistically significant. Yellow perch mean relative weight varied among lakes and was significantly greater during increasing water level periods for all lakes except one. The lack of statistically significant findings potentially suggests a buffering mechanism of north-temperate oligotrophic lakes due to their small surface area to volume ratios, relative lack of nutrients, and(or) littoral structural habitat compared to other systems (e.g., shallow eutrophic lakes). Our results suggest that natural water level fluctuations may not be an environmental concern for yellow perch populations in some north-temperate oligotrophic inland lakes.

KEYWORDS
ecology, fish, freshwater, population dynamics

1 | INTRODUCTION

Natural and anthropogenic water level fluctuations in natural lakes and reservoirs have the potential to influence fish population dynamics through changes in available nutrients and habitat. Yellow perch *Perca flavescens* are an ecologically and recreationally important fish species in the Midwestern United States (Beard & Kampa, 1999; Embke et al., 2020). Yet, yellow perch population dynamics and characteristics remain relatively understudied for many inland lakes of Wisconsin (Beard et al., 2003; Feiner et al., 2020; Feucht et al., 2023; Hansen et al., 1998; Mrnak et al., 2021; Rypel et al., 2016). Yellow perch have been frequently studied in the Laurentian Great Lakes (Henderson, 1985; Parker et al., 2009; Stacy-Duffy et al., 2020), Prairie Pothole region of the Northern Great Plains (Dembkowski et al., 2014; Isermann et al., 2007; Lott et al., 1996), Missouri River reservoirs (Martin et al., 1981; Nelson & Walburg, 1977), and the Rainy Lake-Namakan

Reservoir complex in Northern Minnesota and Canada (Kallemeyn, 1987; Larson et al., 2016, 2018). Previous water level studies on fish populations have mainly focused on reservoir systems where water levels fluctuate naturally and anthropogenically, which can influence fishery resources (Groen & Schroeder, 1978; Lantz et al., 1967). Given the ecological and recreational importance of yellow perch and the predicted increase in climate-driven precipitation changes (Carpenter et al., 2011; Kundzewicz et al., 2007), there is a critical need to better understand population characteristics of this species in oligotrophic, north-temperate inland lakes that undergo natural water level fluctuations.

Fluctuations in water level are known to influence fish community structure within littoral habitats of lakes (Logez et al., 2016; Midwood & Chow-Fraser, 2012). Changes in water level have altered the community composition and abundance of macroinvertebrates, macrophytes, and coarse woody habitat within littoral zones of lakes (Carmignani & Roy, 2017; Gaeta et al., 2014; Haxton & Findlay, 2008; Leira & Cantonati, 2008). Water level fluctuations may indirectly affect fish assemblages through shifts in resource availability, structural spawning habitat, and(or) the availability of adequate refuge from predators (Carmignani & Roy, 2017; Gaeta et al., 2014; Wilcox & Meeker, 1992). In north-temperate climates, yellow perch regularly use coarse woody habitat and macrophytes during spring spawning in the littoral zones of small lakes (Becker, 1983; Feucht et al., 2023; Mrnak et al., 2021; Muncy, 1962; Nelson & Walburg, 1977). It has been postulated that higher water levels will submerge more vegetation and coarse woody habitat and positively influence yellow perch populations through increased littoral habitat availability (Nelson & Walburg, 1977; Wilcox & Meeker, 1992). Water level fluctuations have positively influenced recruitment in Missouri River reservoirs and various Laurentian Great Lake bays (Henderson, 1985; Martin et al., 1981; Nelson & Walburg, 1977) and the abundance of yellow perch in several Northern Great Plains systems (e.g., Prairie Pothole glacial lakes and Missouri River reservoirs; Dembkowski et al., 2014; Martin et al., 1981). Other studies on the larger lakes of Minnesota and Canada have suggested that water levels explained very little of the variation observed in yellow perch populations or did not have consistently significant effects (Kallemeyn, 1987; Larson et al., 2016, 2018). For example, age-0 yellow perch abundance estimates were not correlated with fluctuations in water levels (Larson et al., 2016) or increased in only one of several lakes studied (Kallemeyn, 1987; Larson et al., 2018). Other abiotic and biotic factors related to water level such as temperature (Stacy-Duffy et al., 2020), UV radiation (Huff et al., 2004; Williamson et al., 1997), dissolved oxygen (Huff et al., 2004; Zhang et al., 2017), river inflows (Marcek et al., 2021), and wind (Clady, 1976; Nelson & Walburg, 1977; Zhang et al., 2017) have also influenced yellow perch population characteristics across their native range. Despite these studies, a better understanding of habitat characteristics and water level influences on yellow perch in north-temperate oligotrophic lakes is lacking.

The Northern Highland Lake District of Wisconsin (see Peterson et al., 2003) provides a unique opportunity to study oligotrophic lakes and the availability of littoral habitat in relation to yellow perch populations and natural water level fluctuations. Low nutrient levels of oligotrophic lakes may play an important role in limiting suitable habitat, making any subsequent water level changes potentially more influential. Conversely, these ecosystems may contain different buffering capacities to water level change compared to small, glacial eutrophic lakes (e.g., Prairie Pothole glacial lakes). North-temperate oligotrophic lakes have a smaller surface area to volume ratio than most Prairie Pothole glacial lakes, thus requiring a greater increase in water level for the inundation of shoreline vegetation and coarse woody habitat. Naturally low-nutrient and littoral structural habitat availability may also result in little to no change in habitat availability within littoral zones despite water level fluctuations. Oligotrophic lakes may require a larger increase in water level to observe proportional changes in their surface area to volume ratios than shallower eutrophic systems.

The goal of our study was to test for the influence of water level fluctuations on yellow perch populations across four north-temperate lakes of Wisconsin. Specifically, our objectives were to test for differences in yellow perch condition (i.e., relative weight; *Wr*) and relative abundances (catch-per-unit-effort; using littoral fyke nets and pelagic vertical gillnets) between increasing and decreasing water level periods. Due to the potential for increased littoral habitat, resource availability, and predator refugia, we hypothesized that increasing water level periods would positively influence the condition and abundance of yellow perch populations in our study lakes compared to decreasing water level periods.

2 | MATERIALS AND METHODS

Study area: Four lakes located within the Northern Highland-American Legion State Forest (Vilas County, WI) were selected for our study based on the availability of long-term yellow perch relative abundance (catch-per-unit-effort) and water level data. These lakes included Crystal (46.00470°N, 89.61910°W), Sparkling (46.01580°N, 89.70450°W), Big Muskellunge (46.02730°N, 89.63350°W), and Trout (46.07900°N, 89.70380°W) lakes (Figure 1; Table 1). Since 1981, these lakes have been continuously monitored (e.g., fisheries, limnology, and water chemistry) by the North Temperate Lakes-Long-Term Ecological Research project (NTL-LTER; Magnuson et al., 2006). All lakes can be categorized as north-temperate oligotrophic systems and range in surface area (37–1561 ha), maximum depth (20.0–35.7 m), shoreline length (2.3–25.9 km), volume (3,910,400–228,504,600 m^3), and shoreline development (low to moderate; Table 1).

Water levels: The annual mean water level for each study lake was estimated by averaging monthly (April–August) water surface elevation estimates collected by NTL-LTER (Magnuson et al., 2022a). Water level measurements were taken 507 times in Crystal Lake, 492 in Big Muskellunge Lake, 468 in Sparkling Lake, and 399 in Trout Lake during 1981–2020. Before conducting a time series analysis, we tested for stationarity in the water level data using an Augmented Dickey-Fuller test (Dickey & Fuller, 1979; Mushtaq, 2011). For each lake and all lakes combined, the Augmented Dickey-Fuller test returned a $p < 0.01$, indicating that the water level time series was independent of time. This

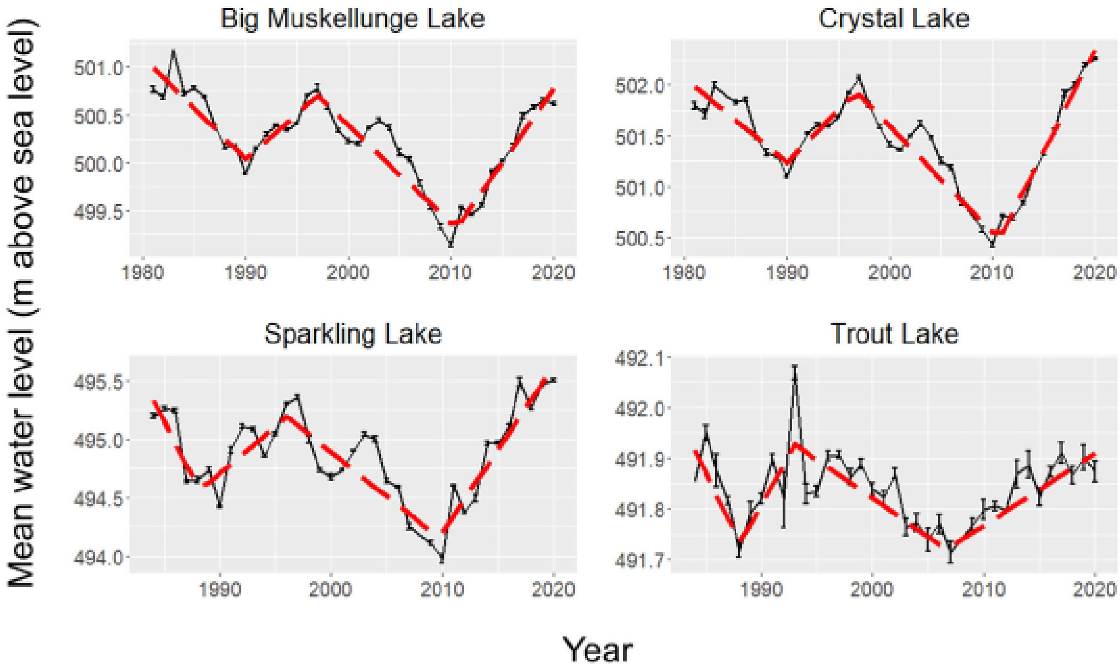

FIGURE 1 Mean annual lake surface elevation during 1981–2020 for Big Muskellunge, Crystal, Sparkling, and Trout lakes, all of which are located in the Northern Highland-American Legion State Forest (Vilas County, WI; black line and points). The dashed red lines correspond to segmented regression analysis statistical break points; Big Muskellunge Lake break points in 1990, 1997, and 2010 (SE = 1.0, 1.0, and 0.6 years, respectively), Crystal Lake break points in 1990, 1996, and 2010 (SE = 0.8, 0.7, and 0.4 years, respectively), Sparkling Lake break points in 1988, 1996, and 2009 (SE = 0.8, 1.0, and 0.7 years, respectively), and Trout Lake break points in 1988, 1993, and 2006 (SE = 0.7, 0.8, and 1.3 years, respectively). Decreasing water level periods correspond to dashed red lines with negative slopes, while increasing water level periods correspond to dashed red lines with positive slopes. Error bars represent one standard error about the mean.

TABLE 1 Surface area (ha), maximum depth, mean depth (m), volume (m^3), shoreline length (km), shoreline development, and percent volume change after ±1 m in average depth for Crystal, Sparkling, Big Muskellunge, and Trout lakes located within the Northern Highland-American Legion State Forest (Vilas County, WI).

Lake	Surface area (ha)	Maximum depth (m)	Mean depth (m)	Volume (m^3)	Shoreline length (km)	Shoreline development	Volumechange ±1 m (%)
Crystal Lake	37.5	20.4	10.4	3,910,400	2.3	Low	9.6
Sparkling Lake	63.7	20.0	10.9	6,943,300	4.3	Moderate	9.2
Big Muskellunge Lake	363.4	21.3	7.5	27,255,000	16.1	Moderate	13.3
Trout Lake	1565.1	35.7	14.6	228,504,600	25.9	Moderate	6.8

allowed us to test for lagged effects of water level on our response variables using a dynamic linear regression (Zeileis, 2019; Zeileis et al., 2005) to identify whether time lags needed to be incorporated into our statistical model (detailed below). We allowed our response variable to depend on the current water level and water level up to 5 years prior. We found no significant lag effect of water level on the yellow perch response variables, *ceteris paribus* (all $p > 0.1$; non-autocorrelated). As such, we used a segmented regression breakpoint analysis on each lake's 39-year water-level time series of data to test for periods of increasing, stable, or decreasing water level (based on breaks and associated slopes; Muggeo, 2017). We found significant break points for the Crystal Lake water level time series in 1990, 1996, and 2010 (SE = 0.8, 0.7, and 0.4 years, respectively; slope significant at $p < 0.05$, $R^2 = 0.89$), for Big Muskellunge Lake in 1990, 1997, and 2010 (SE = 1.0, 1.0, and 0.6 years, respectively; slope significant at $p < 0.05$, $R^2 = 0.84$), for Sparkling Lake in 1988, 1996, and 2009 (SE = 0.8, 1.0, and 0.7 years, respectively; slope significant at $p < 0.05$, $R^2 = 0.77$), and for Trout Lake in 1988, 1993, and 2006 (SE = 0.7, 0.8, and 1.3 years, respectively; slope was significant at $p < 0.05$, $R^2 = 0.57$; Figure 1). Each lake's water level time series consisted of two periods of increasing water level and two periods of decreasing water level. Changes in lake-specific water volumes were estimated using the equation of Taube (2000), given as:

$$\text{Lake volume} = \text{Average lake depth} \times \text{Lake surface area} \times 100,$$

where lake volume is in cubic meters, average lake depth is in meters, and lake surface area is in hectares. Because lake surface area was not directly measured during our 39-year time series, and assumed as a constant as is for most lakes, we estimated water volume change using the Taube (2000) equation by adjusting average lake depth ±1 m and examining the ratio between average lake volume and a ±1 m change in average lake depth. Estimated lake volume deviations from the average lake volume during the time series ranged from 6.8% to 13.3% increase or decrease (Table 1).

Fish sampling: All yellow perch data were accessed from the NTL-LTER website (https://ntl.limnology.wisc.edu/data; Magnuson et al., 2022b). Following standardized methodology, NTL-LTER used a variety of fisheries sampling gears including vertical gillnets, fyke nets, beach seines, trammel nets, and pulsed-DC boat electrofishing to monitor fishes in our study lakes. All fisheries surveys were conducted annually at pre-defined random sample sites (consistent since 1981) between the third week of July and the first week of September. Monofilament gillnets were deployed in the deep portion of each lake (surface to bottom), picked daily, and reset for another consecutive 24-h set (total set time = 48 h), with catch-per-unit effort calculated as fish/net night. Experimental gill nets were 3 × 30 m with stretched mesh sizes of 19, 25, 32, 38, 51, 64, and 89 mm (Mrnak et al., 2021). Three fyke nets (0.8 × 1 m frame, 1 × 12 m lead, 7 mm stretched nylon mesh) were deployed in the littoral zone of each lake. After 24 h, fyke nets were picked and moved to different locations in the littoral zone, reset, and picked the following day after a 24-h soak. Each lake received six net nights of fyke netting effort. Beach seines were 12.2 m × 1.2 m (two 5.5 × 1.2 m wings and a 1.2 × 1.2 × 1.2 m central bag) with wings having 6.4 mm stretched mesh and the central bag having 3.2 mm nylon mesh. At night, seines were pulled perpendicular to shore for 3 × 33 m sections (100 m total) at six sites in each lake. Trammel nets were 30.5 m long by 1.1 m deep with two outer nets (170 mm square 32 kg test mesh multifilament nylon) and an inner panel (51 mm stretch mesh 9 kg test multifilament nylon). Trammel nets were set on the bottom perpendicular to shore at two locations in each lake for 24 h before being pulled. Pulsed-DC electrofishing was conducted in four (before 1997) or three (after 1997) 30-min predetermined transects parallel to shore in 1–2 m of water. All electrofishing runs for a lake were conducted during the same night. Data collected included fish species, total length to the nearest mm, and weight to the nearest 0.1 g. Only data collected from fyke net (yellow perch total length range; 101–277 mm) and gillnet (yellow perch total length range; 101–281 mm) surveys were used in our relative abundance (fish/net night; CPUE) analyses, while biological data collected from all survey types were used in our condition (*Wr*) analysis.

Data analyses: For each lake year, yellow perch relative weight (*Wr*) was calculated for all individuals ≥75 mm across all gears (Willis et al., 1991; Figure 2). Relative weight is a measure of fish condition and was estimated as the ratio of an individual fish weight to a length-specific standard weight determined by a species-specific weight-length regression multiplied by 100 (Wege & Anderson, 1978; Willis et al., 1991). Greater *Wr* values indicate a more robust individual, and values <100 indicate that the fish's weight is below the standard weight for a fish of the same length. For each lake-year-gear (i.e., fyke net or gillnet), fyke net and gillnet relative abundance estimates (fish/net night; *CPUE-f* or *CPUE-g*, respectively) were calculated.

Yellow perch mean *Wr*, fyke net *CPUE-f*, and gillnet and *CPUE-g* were calculated for each lake's water level time period identified through the segmented regression break point analysis. We tested for differences in each metric using a one-way analysis of variance with the null hypothesis of no difference in the mean of each response variable between increasing and decreasing water level periods. Statistical significance (i.e., rejection of the null hypothesis) was assessed at $\alpha = 0.05$ for all analyses.

3 | RESULTS

3.1 | Water levels

All lakes followed a similar pattern of decreasing, increasing, decreasing, and increasing water levels during 1981–2020, though the duration of these periods slightly varied (Figure 1). The Crystal Lake water level time series contained 22 years of decreasing water level periods and 17 years of increasing water level periods. Big Muskellunge Lake had 23 years of decreasing and 16 years of increasing water level periods. Sparkling Lake had 17 years of decreasing water levels and 19 years of increasing water levels. The Trout Lake water level time series contained 13 years of decreasing water level and 23 years of increasing water level. Over the time series, the range of water level fluctuation for each lake varied (Figure 1). Big Muskellunge Lake had the largest range in water level (501.18–499.14 m), followed by Crystal Lake (502.25–500.42 m), Sparkling Lake (495.5–493.9 m), and Trout Lake (492.1–491.7 m) lakes.

3.2 | Population characteristics

Condition: Yellow perch condition and relative abundance within each lake differed between water level periods, though differences were generally not statistically significant. Differences in yellow perch *Wr* between water level periods appeared lake-specific and followed a relatively consistent pattern (Figure 2; Table 2). Yellow perch mean *Wr* in Crystal, Trout, and Big Muskellunge lakes were significantly greater during increasing water level periods ($F_{1,1255} = 8.66$, $p < 0.05$, $F_{1,226} = 16.96$, $p < 0.05$, and $F_{1,82} = 8.66$, $p < 0.05$, respectively; Table 2). Yellow perch mean *Wr* did not differ between water level periods in Sparkling Lake ($F_{1,203} = 0.05$, $P = 0.81$; Table 2).

Relative abundance: Yellow perch relative abundance estimates from littoral fyke nets (*CPUE-f*) and pelagic gillnets (*CPUE-g*) did not statistically differ between increasing or decreasing water level periods (Table 2). Nevertheless, there appeared to be a general trend where *CPUE-f* was greatest in Crystal, Sparkling, and Trout lakes during the decreasing water level periods. Mean yellow perch *CPUE-g* was also greatest during decreasing water level periods for Trout, Sparkling, and Big Muskellunge lakes (Table 2).

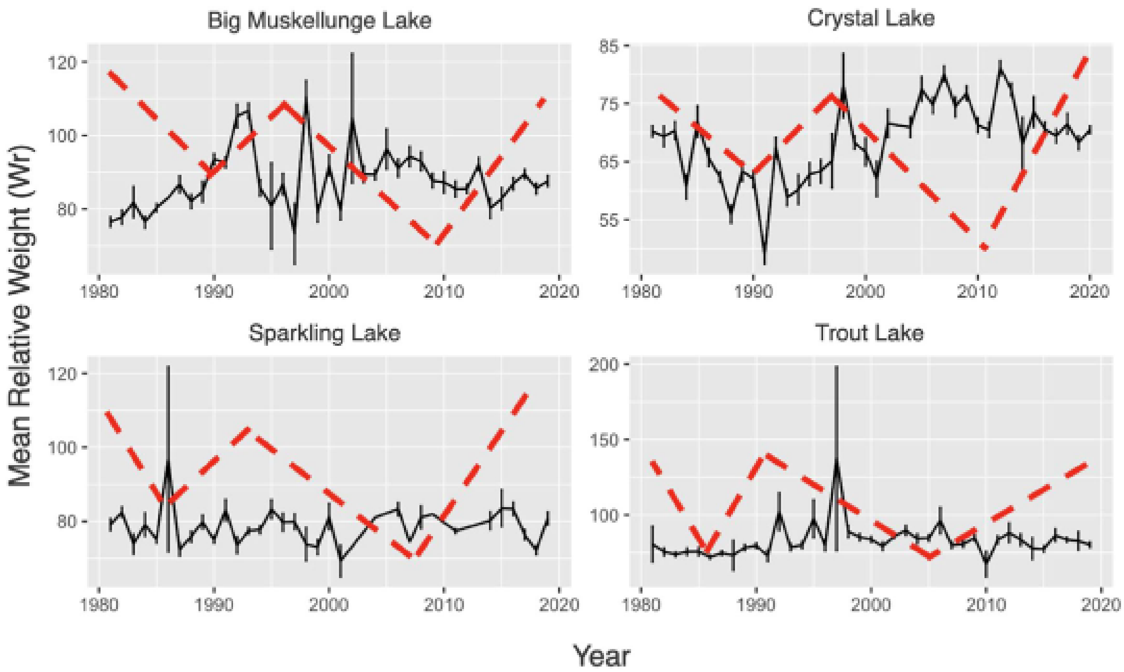

FIGURE 2 Mean annual relative weight (*Wr*) of yellow perch *Perca flavescens* from 1981 to 2020 for Big Muskellunge, Crystal, Sparkling, and Trout lakes, all of which are located in the Northern Highland-American Legion State Forest (Vilas County, WI; black line and points). Error bars represent one standard error about the mean. Decreasing water level periods correspond to dashed red lines with negative slopes, while increasing water level periods correspond to dashed red lines with positive slopes.

TABLE 2 Yellow perch *Perca flavescens* mean (±SE, sample size) relative weight (*Wr*) and catch per unit effort for fyke nets (*CPUE-f*) and gillnets (*CPUE-g*) in four oligotrophic north-temperate inland lakes between decreasing and increasing water level periods.

Variable	Decreasing water level	Increasing water level	F-statistic	d.f.	*p*
Crystal Lake					
Wr	66 (0.5, 357)	68 (0.3, 899)	8.66	1254	<0.05
CPUE-f	30.6 (15.9, 22)	21.7 (5.9, 18)	0.16	38	0.68
CPUE-g	17.9 (6.5, 22)	18.7 (6.3, 18)	0.007	39	0.93
Trout Lake					
Wr	75 (0.7, 145)	81 (1.5, 82)	16.96	225	<0.05
CPUE-f	57.9 (25.2, 14)	44.9 (17.0, 23)	0.19	35	0.66
CPUE-g	0.31 (0.2, 14)	0.08 (0.1, 23)	2.34	36	0.13
Sparkling Lake					
Wr	79 (2.9, 140)	78 (0.5, 64)	0.023	570	0.88
CPUE-f	1.5 (1.1, 17)	0.9 (0.6, 20)	0.26	35	0.61
CPUE-g	2.7 (2.5, 17)	0.3 (0.3, 20)	1.01	36	0.32
Big Muskellunge Lake					
W_r	82 (2.0, 33)	88 (1.0, 60)	6.01	82	<0.05
CPUE-f	89.4 (38.7, 18)	561.9 (309, 22)	2.68	37	0.11
CPUE-g	2.5 (2.4, 18)	1.75 (1.6, 22)	0.05	37	0.81

Note: *p* is the probability of a significant difference based on a one-way analysis of variance ($\alpha = 0.05$). No yellow perch data were collected in 2020 for Trout Lake and Big Muskellunge Lake or in 2010, 2012, 2013, and 2020 for Sparkling Lake. Across all lakes, fyke nets were not deployed in 2020.

4 | DISCUSSION

Natural fluctuations in lake surface elevation leading to extended periods of increasing or decreasing water levels did not appear to have strong effects on yellow perch conditions and relative abundances within the north-temperate, oligotrophic Wisconsin lakes examined here. Among the four study lakes, we identified three statistically significant differences for the three response variables examined between the water level periods. Although most comparisons were not statistically significant, there appeared to be some biologically meaningful differences between increasing or decreasing water level periods. For example, greater relative abundances of yellow perch (across littoral and pelagic gears) were generally observed in the decreasing water level period (excluding Big Muskellunge Lake, but see an associated error in the estimate). In contrast, Dembkowski et al (2014) found mean gill net yellow perch CPUE to be significantly higher in high water years in South Dakota glacial lakes. In our study, yellow perch *Wr* was significantly greater in three of the four lakes during increasing water level periods. This result was somewhat unexpected as yellow perch *Wr* did not change with water level fluctuations in the glacial lakes of South Dakota (Dembkowski et al., 2014) and is potentially related to resource and(or) habitat availability differences between systems. Across all lakes and between water level periods, yellow perch condition estimates were relatively low, particularly in Crystal Lake (Wege & Anderson, 1978; Willis et al., 1991). Low *Wr* values may be indicative of low food availability and(or) poor habitat in these systems ultimately making it challenging to observe statistical or biologically meaningful changes in yellow perch conditions (Garvey & Chipps, 2012; Neumann et al., 2012). Overall, our study found fewer statistically significant differences in the three variables examined within oligotrophic, north-temperate inland lakes between increasing or decreasing water level periods compared to studies on small, eutrophic, prairie pothole lakes, likely resulting from differences in biotic and abiotic conditions of each ecosystem and region.

In north-temperate oligotrophic systems, increases in water level generally increase littoral structural habitat availability following the inundation of vegetation and coarse woody habitat. In turn, this generally leads to greater production at lower trophic levels (i.e., microbes and macroinvertebrates; Sass et al., 2019, 2022). The absence of coarse woody habitat within littoral regions of north-temperate glacial lakes has been shown to coincide with reduced lake levels and yellow perch CPUE, potentially resulting from a loss of spawning habitat and refuge and increased predator-prey encounter rates (Gaeta et al., 2014). Thus, it is not surprising that yellow perch may be selected for littoral habitat over pelagic zones at higher lake water levels. Yet, we did not observe a marked increase in littoral relative abundance during increasing water level periods. This could be a response to lower water levels reducing the total volume of the lake, thus resulting in higher densities and CPUE of yellow perch. However, this may also be due to our experimental design where all years were forced into a binary category of increasing or decreasing water level. For example, our break point analysis indicated that Crystal Lake during 2011 (mean water elevation of 500.7 m) was in an increasing water level period, while Crystal Lake in 1983 (mean water elevation of 501.9 m) was in a decreasing period. Therefore, our experimental design may not have been able to capture yellow perch population characteristics or dynamics around extreme water levels (i.e., high or low). However, the availability of coarse woody habitat has been suggested to be important as a refuge for fish and invertebrate prey (Roth et al., 2007; Sass et al., 2006b; Vander Zanden & Vadeboncoeur, 2002) and for influencing predator-prey interactions (Anderson, 1984; Sass et al., 2006a; Savino & Stein, 1982). With rising water levels and subsequent inundation of shoreline habitat, we would expect to observe greater increases in littoral habitat use of yellow perch in oligotrophic, north-temperate inland lake ecosystems.

We did not test for differences in yellow perch population size structure (i.e., index of the proportion of available length classes within a population; Gabelhouse, 1984; Guy et al., 2007) between increasing or decreasing water level periods in our study lakes due to sample size limitations. However, significant differences in yellow perch population size structure found in glacial lakes in South Dakota could be attributed to density-dependent growth patterns, with more, but smaller, yellow perch during high water years increasing intra-specific competition and thus reducing growth rates of larger yellow perch (Dembkowski et al., 2014). Additionally, water levels may influence age-dependent effects that favor smaller individuals over larger individuals during increased water levels due to increased availability of prey refugia or an overall greater habitat volume to avoid predators. Because north-temperate oligotrophic systems are generally much larger in volume than shallow, eutrophic glacial lakes, we hypothesized that density dependence may not have as much of an influence on individual growth rates. However, productivity differences between these types of systems may overcome the differences in available physical habitat and lead to density-dependent effects (i.e., reduced growth rates) with support for our hypothesis deriving from the observed low yellow perch *Wr* estimates for each lake. Regardless, increased availability of niche space in oligotrophic, north-temperate inland lakes potentially helps to buffer any resource demands following an increase in the recruitment of smaller fishes (Mrnak et al., 2023). A greater yellow perch population size structure during increasing water levels could potentially result from increased prey availability and feeding efficiency; however, additional research is needed to test this question and plausible mechanisms.

The paucity of statistically significant findings in our research could be attributed to a lack of power to detect effects due to low sample sizes. However, we hypothesize that the lack of statistically significant results could also be potentially due to enhanced buffering capabilities of north-temperate oligotrophic lakes to water level change. All lakes used in our study are clear, deep, cold-water systems with limited nutrients and macrophytes, particularly when compared to other systems such as the shallow eutrophic Prairie Pothole glacial lakes (e.g., Dembkowski et al., 2014). Because our study lakes have a smaller surface area to volume ratio than South Dakota glacial lakes, it may require a more significant increase in water level for sufficient inundation of shoreline vegetation and coarse woody habitat to elicit a stronger response from the yellow perch populations. Indeed, similar changes in water level will have a greater influence on systems with a gradual slope (glacial lakes) than steep-sided systems (north-temperate oligotrophic lakes; this study). Additionally, natural low-nutrient and littoral structural habitat availability in north-temperate lakes may result in little to no change in habitat availability within littoral zones despite water level fluctuations.

Different fish community assemblages within each lake (and across system types) may also play a role in influencing yellow perch populations, particularly regarding the portfolio effect and invasive species influences (McCann 2000; Mrnak et al., 2023; Rooney et al., 2006). The portfolio effect suggests that systems with high spatial community diversity experience greater heterogeneity in species abundances through time, resulting in greater ecosystem stability (McCann, 2000; Mrnak et al., 2023; Wilcox et al., 2017). Predators are ecologically important for regulating community structure down to the base of food webs (Mrnak et al., 2023; Pace et al., 1999; Terborgh & Estes, 2010). The fish community composition and predator density greatly varied in our study lakes (and compared to other studies; e.g., Dembkowski et al., 2014; Larson et al., 2018; Wilcox & Meeker, 1992), with Trout Lake containing the greatest diversity and density of predators and Crystal Lake containing the least community diversity and no predators (Martin et al., 2023; Mrnak et al., 2023). Predation may dictate abundances of yellow perch within predator-dominated systems acting as another potential buffering mechanism. Conversely, yellow perch in predator-deficient systems may exhibit more pronounced responses to abiotic drivers due to the lack of predator-mediated community regulation (McCann, 2000; Mrnak et al., 2023). It is important to note the presence of invasive Rainbow Smelt *Osmerus mordax* in Sparkling and Crystal lakes. Rainbow Smelt negatively interact with yellow perch and drive major reductions in their populations through competitive interactions and direct predation (Evans & Loftus, 1987; Hrabik et al., 1998, 2001; Krueger & Hrabik, 2005; Mercado-Silva et al., 2007; Mrnak et al., 2023). Due to different fish community assemblages and inter-specific interactions within each lake, it may be difficult to observe long-term trends following changes in non-stationary abiotic variables (e.g., water level, temperature, and nutrient loading) when other biotic interactions may have a greater influence on yellow perch population dynamics. Therefore, examining water level trends (increasing or decreasing) may be more informative and useful to management than an analysis of the overall water level.

4.1 | Management implications

Outside of relative abundance indices (i.e., Beard et al., 2003; Feiner et al., 2020; Hansen et al., 1998; Rypel et al., 2016), little information on yellow perch population dynamics exists for north-temperate, oligotrophic inland lakes (Feucht et al., 2023; Mrnak et al., 2021). Furthermore, yellow perch population status and trends are a "black box" for some state agencies such as the Wisconsin Department of Natural Resources due to unstandardized sampling protocols and limited resources (Mrnak et al., 2021). Yet, yellow perch are a popular sportfish and a critically important prey species (Embke et al., 2020), that when in decline, may negatively affect the recruitment and production of upper trophic level fishes (Beard et al., 2003; Gaeta et al., 2014; Sass et al., 2006b). For example, young and adult walleye *Sander vitreus* are known to selectively consume age-0 yellow perch and influence their recruitment, serving as an important prey source to their populations (Forney, 1974; Haas & Schaeffer, 1992; Herbst et al., 2016; Lyons & Magnuson, 1987). Additionally, age-0 and juvenile yellow perch potentially buffer predation effects on age-0 walleye by other species, thus supporting important fisheries within the Midwest. Natural water level fluctuations did not appear to be an environmental concern for yellow perch populations in our oligotrophic, north-temperate inland study lakes, although water level influences on littoral structural habitat were not considered here (see Gaeta et al., 2014). Thus, Wisconsin fisheries management efforts could be more focused on other environmental concerns that pose a greater threat to yellow perch populations in these systems, such as habitat availability (Gaeta et al., 2014; Sass et al., 2006b), invasive species expansions (Hrabik et al., 1998; Mrnak et al., 2023), and(or) climate change (Brandt et al., 2022; Feiner et al., 2022). Despite our findings, future climate shifts may lead to more variable fluctuations in water level (Carpenter et al., 2011; Kundzewicz et al., 2007) and periods of droughts (Gaeta et al., 2014; Lake, 2011; Romm, 2011). These climatic events may have a more significant influence on yellow perch in the future. We recommend continual monitoring and assessment in relation to environmental factors not discussed within our study (e.g., temperature, shoreline development, and fish community changes) to test for additional drivers of yellow perch population dynamics in north-temperate, oligotrophic lakes.

AUTHOR CONTRIBUTIONS

Gabrielle Shay: Conceptualization; formal analysis; investigation; visualization; writing—original draft; writing—review and editing. **Gregory Sass**: Conceptualization; funding acquisition; supervision; writing—review and editing. **Joseph Mrnak**: Conceptualization; formal analysis; funding acquisition; investigation; supervision; visualization; writing—original draft; writing—review and editing.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

DATA AVAILABILITY STATEMENT

Data are available at https://ntl.limnology.wisc.edu/data.

ETHICS STATEMENT

All animals were handled following IACUC protocols.

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