Brien P. Rose[1] and **Matthew G. Mesa**, U.S. Geological Survey, Western Fisheries Research Center, Columbia River Research Laboratory, 5501 Cook-Underwood Road, Cook, Washington 98605

Effects of Summer Drawdown on the Fishes and Larval Chironomids in Beulah Reservoir, Oregon

Abstract

Summer drawdown of Beulah Reservoir, Oregon, could adversely affect fish and invertebrate production, limit sport fishing opportunities, and hinder the recovery of threatened species. To assess the impacts of drawdown, we sampled fish and Chironomidae larvae in Beulah Reservoir in the springs of 2006 to 2008. The reservoir was reduced to 68% of full pool in 2006 and to run-of-river level in 2007. From spring 2006 to spring 2007, the catch per unit effort (CPUE) of fyke nets decreased significantly for dace [*Rhinichthys* spp.] and northern pikeminnow [*Ptychocheilus oregonensis*], increased significantly for suckers [*Catastomus* spp.] and white crappies [*Pomoxis nigromaculatus*], and was similar for redside shiners [*Richardsonius balteatus*]. CPUE of gillnets either increased significantly or remained similar depending on genera, and the size structure of redside shiners, suckers, and white crappies changed appreciably. From 2007 to 2008, the CPUE of northern pikeminnow, redside shiners, suckers, and white crappies decreased significantly depending on gear and the size structure of most fishes changed. Springtime densities of chironomid larvae in the water column were significantly higher in 2006 than in 2008, but other comparisons were similar. The densities of benthic chironomids were significantly lower in substrates that were frequently dewatered compared to areas that were partially or usually not dewatered. Individuals from frequently dewatered areas were significantly smaller than those from other areas and the densities of benthic chironomids in 2008 were significantly lower than other years. Summer drawdown can reduce the catch and alter the size structure of fishes and chironomid larvae in Beulah Reservoir.

Keywords: reservoir, drawdown, bull trout, fish, chironomids

Introduction

Water storage reservoirs are common throughout the U.S. and characterized by seasonal drawdowns that can restrict the establishment of defined littoral zones. Drastic or ill-timed drawdowns can lead to dewatering of littoral zone habitat, poor spawning success, lower fish abundances, and reduced benthic insect production (Benson and Hudson 1975, Martin et al. 1981, Gaboury and Patalas 1984, Furey et al. 2006). On the other hand, species richness and fish abundance in Par Pond, South Carolina, recovered within 9 months of a 50% drawdown (Paller 1997) and Heman et al. (1969) reported improved growth of largemouth bass [*Micropterus salmoides*] in Little Dixie Lake, Missouri, following a 58% reduction in water volume in the summer. Such contrasting results indicate that the effects of reservoir drawdown are varied and may depend upon the magnitude and timing and species assemblage, among other factors. A more complete understanding of the effects of reservoir drawdown would result from more studies in different locales, focusing on a variety of fish and insect species and water management strategies.

In the Pacific Northwest, Beulah Reservoir in northeastern Oregon provides irrigation water to nearby farms and ranches and supports a population of bull trout [*Salvelinus confluentus*] that overwinter in the reservoir from November through early May (Gonzalez 1998, Schwabe and Tiley 1999). Water level management of this reservoir is controlled by the U.S. Bureau of Reclamation (BOR) and has been driven largely by irrigation needs in the summer, with little or no concern for fish and invertebrate populations. Thus, bull trout can potentially experience changes in water level, habitat, and forage availability due to summer drawdown, particularly if reservoir volume is reduced to run-of-river levels. Decreased forage fish populations could negatively impact the bull

[1] Author to whom correspondence should be addressed.
Email: brose@usgs.gov

trout population in the area by contributing to decreased growth, fecundity, and survival. Recent recognition of the potential effects of summer drawdown on the prey base for bull trout in the reservoir has led to requests for a minimum pool level requirement (U.S. Fish and Wildlife Service 2002). However, studies are needed to provide resource managers with information to minimize any adverse effects of drawdown.

In this paper, we describe the effects of a typical summer drawdown and a complete dewatering of Beulah Reservoir on changes in catch rates and size structure of chironomid larvae (the most abundant invertebrate taxa) and fishes, both of which are important forage items for bull trout (Wilhelm et al. 2011, Beauchamp and Van Tassell 2001). Our results should be useful for the management of Beulah Reservoir and may help assess potential indirect impacts of reservoir drawdown on the local bull trout population.

Methods

Study Site and Sampling Design

Agency Valley Dam was built by the BOR on the North Fork Malheur River (NFMR), Oregon, during 1934–35 at river kilometer 29 and formed Beulah Reservoir. The impoundment provides irrigation water to local farms and some flood control, and has no facilities for up- or downstream passage of fish. The reservoir surface elevation is 1020 m above sea level at full pool and has an average width of 1.9 km and a length of about 4 km. Surface area is about 770 ha. The NFMR and Warm Springs Creek enter from the north and water exits from the south end of the reservoir. The north end of the reservoir is relatively shallow (<10 m deep) with a low gradient bottom and the south end drops quickly from shore and reaches a maximum depth of about 23 m. Although summer temperatures exceed 20 °C at all depths throughout the reservoir (Petersen et al. 2003), it cools rapidly in the fall and typically ices over in winter. Beulah Reservoir is productive with high abundances and diverse size classes of redside shiners [*Richardsonius balteatus*], redband trout [*Oncorhynchus mykiss gairdneri*], suckers [*Catostomus* spp.], and northern pikeminnow [*Ptychocheilus oregonensis*] (Petersen et al. 2003). The reservoir is stocked with hatchery rainbow trout (RBT; [*Oncorhynchus mykiss*]) each spring.

We sampled fish and aquatic insects in Beulah Reservoir during the spring and fall of 2006 and 2007, and the spring of 2008 but, because of dramatically different seasonal water volumes, we only present findings for spring here. Our intent was to evaluate changes in selected fish and insect metrics relative to drawdown level. Reservoir volume and NFMR discharge data were obtained from BOR Hydromet stations (USBR 2012) and we plotted these data for the years 1999–2008 (Figure 1) for comparative purposes.

Aquatic Insect Sampling

We estimated the density of invertebrates in the water column during the spring of each year (Table 1) using a 0.5 m^2, 500-μm mesh conical zooplankton net towed vertically from about 1 m above the bottom to the water surface. First, at the laboratory, we divided the reservoir into four unequally sized quadrants (i.e., northwest, northeast, southwest, and southeast) using the program ArcView (Esri, Redlands, CA). We then divided each quadrant into several smaller sections (each about 300 × 300 m), numbered them, and chose three sections in each quadrant for sampling using a random number generator prior to arriving in the field. We used a GPS device on our boat to locate the center of each section and, if for some reason upon arrival (e.g., low water level) a section could not be sampled, we chose another. We used a depth finder to determine reservoir depth at our location and lowered the net to the approximate starting position. We combined catches from the three tows into a single composite sample for each quadrant and preserved samples in the field with 70% ethanol. In the laboratory, organisms were identified to family and the number and total mass (blotted dry and adjusted to approximate living mass, Howmiller 1972) were recorded. Densities for each composite sample (i.e., 8 samples in 2006, 16 in 2007, and 20 in 2008) were estimated by dividing the total number of individuals collected per taxa by the total water volume sampled and used to derive metrics for the reservoir in each year. Because our catches were dominated by

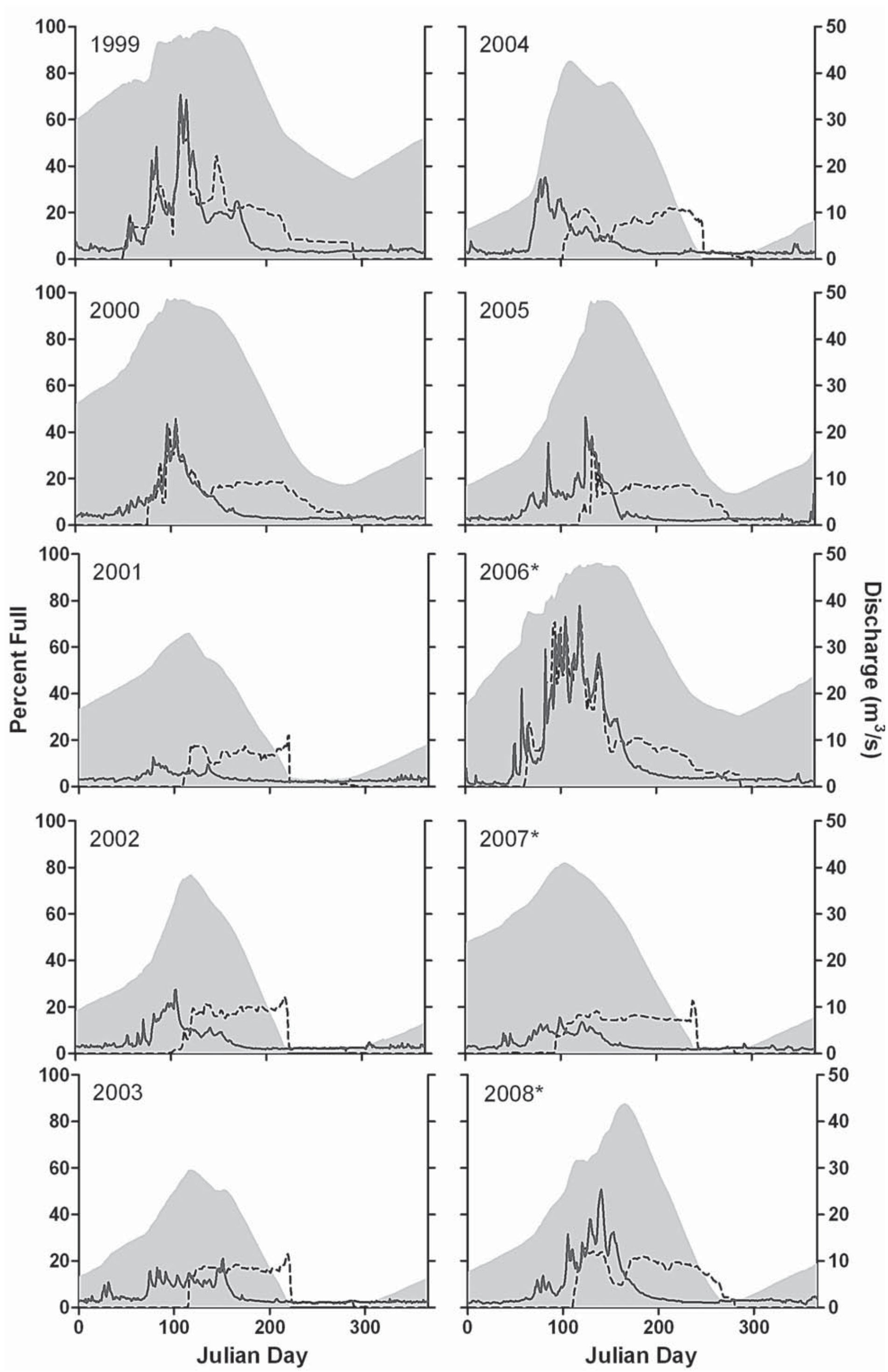

Figure 1. Discharge of water entering (grey solid line) and exiting (black dashed line) Beulah Reservoir and reservoir volume (histogram), 1999–2008. Our study occurred during years marked with an asterisk.

TABLE 1. Dates that fish and invertebrates were sampled in Beulah Reservoir from 2006 to 2008. Fish were sampled on all dates shown, and the dates when benthic (B) and pelagic (P) chironomids were sampled are indicated by a letter following the date.

2006	2007	2008
4/12–4/16 B	3/20–3/26 P	3/25–3/31 B, P
4/25–5/01 B	4/3–4/9 B, P	4/8–4/13 B, P
5/9–5/15 P	4/18–4/24 B, P	4/23–4/27 B, P
5/23–5/26	5/2–5/8 B, P	5/6–5/12 B, P
6/5–6/11 B, P	5/16–5/22	5/19–5/23 B, P

chironomid larvae, we focused our analysis on these taxa only.

We also estimated the density and biomass of benthic chironomid larvae in substrates that experienced different levels of dewatering over the past 30 years or so. Again, this sampling was done during the spring of each year. We used BOR data from 1977 to 2005 on the minimum pool elevation of Beulah Reservoir after summer drawdown to determine the number of years substrates at different elevations were dewatered. We determined that substrates above 1010 m were dewatered in 96% of the years (i.e., substrates above this elevation were dewatered in 27 of the 28 years examined) and substrates lower than 997 m were dewatered in only 4% of the years. Substrates ranging in elevation from 997 to 1010 m were partially dewatered in 92% of the years. These three elevation bands (i.e., 1010 to 1020 m [full pool], 997 to 1010 m, and less than 997 m) represented about 25, 62, and 13% of reservoir substrates. Prior to sampling, we divided each elevation band into sections of about 300 × 300 m each and randomly selected two from each elevation band to sample. During selected field visits (Table 1), we again used a GPS device on our boat to locate the center of each section and collected samples with a full-sized Ponar dredge. In the end, we collected 6 samples per elevation band in 2006 and 2007, and 10 in 2008. We calculated density (N/m^2) by dividing the total number of individuals collected per taxa by the area sampled and weight of individuals by dividing the combined mass of insects by total number per sample.

Fish Sampling

Fish were sampled about once every two weeks during the spring (Table 1) with modified fyke nets and sinking experimental gillnets. At the start of each season, we chose 4–5 suitable fyke net locations for each of the four quadrants (i.e., northwest, northeast, southwest, and southeast) within the reservoir. During sampling, one fyke net (91-cm-high, 122-cm-wide, and 0.6-cm mesh size with a 13 m center lead extending to shore) was set in each quadrant at a randomly selected location and fished overnight. The gillnets were 36.5-m-long by 3.0-m-deep and contained six 6 m panels with stretch mesh sizes of 8.9, 7.6, 6.3, 5.1, 3.8, and 2.5 cm and we used a stratified (i.e., by quadrant/section as described above) randomized design to establish sampling locations. For sampling, 1–3 gillnets were fished simultaneously on the bottom for 30 min or less during daylight hours. Sample sites for both net types were replaced at the start of each session. Captured fish were anesthetized (using either 50 mg/L MS-222 for non-game fish or one tablet of Alka-Seltzer Gold in 2.5 L of water for game fish), identified to genus or species, measured for fork length (FL), weighed (0.1 g), and released. We excluded RBT from all analyses due to the seasonal influence of stocked unmarked fish.

Data Analysis

We compared the mean ranks of pelagic chironomid densities in the spring of each year using a Kruskal-Wallis test followed by Dunn's post-test procedure. For each sample period, we ended up with four composite samples (i.e., one from each quadrant), determined the density of chironomids (N/m^3) for each, and compared them among years. For our benthic chironomid data, we calculated the density (N/m^2) of each sample and analyzed the data in two ways: (1) between elevation bands by grouping the data within each band across years and comparing between elevation bands; and (2) between years by grouping data from each elevation band within a year, and comparing amongst years. We also compared the mass of benthic chironomid larvae from each depth strata using data from all years combined. For these comparisons, we also used Kruskal-Wallis tests and the level of significance for all tests was set at $P \leq 0.05$.

We compared the mean ranks of fyke net and gillnet catch rates per net set (catch per unit of effort [CPUE] as fish/h, or FPH) of dace, northern pikeminnow, redside shiners, suckers, and white crappies [*Pomoxis nigromaculatus*] between years with Kruskal-Wallis tests as described above. We assumed equal catchability of fish between years. We also evaluated the size composition of our catches by arbitrarily dividing the total catch into 3 or 4 size classes (e.g., 50–100, 101–150, 151–200, and > 200 mm) and qualitatively comparing changes between years.

Results

Magnitude of Reservoir Drawdowns

The volume of Beulah Reservoir and discharge from the NFMR varied greatly during and preceding the study (Figure 1). In the spring of 2006, Beulah Reservoir was filled to 99% of its 0.07 km^3 capacity and was reduced from June through September to 0.023 km^3. This represented a drawdown of about 68%, which was less than the mean drawdown of 85% that occurred from 1998 to 2008. Mean (± SD) daily discharge from the NFMR upstream of Beulah Reservoir was 7.0 ± 8.9 m^3/s, which was about twice the ten-year (1998–2008) average of 3.6 ± 3.7 m^3/s. In 2007, Beulah Reservoir filled to 84% of capacity and was reduced to run-of-river level (<1% of full pool) on 30 August. During this dewatering, most of the reservoir was shallow (<0.2 m) and maximum depths were about 1 m. Mean discharge from the NFMR was 2.0 ± 1.4 m^3/s, or about 55% of the ten-year average. The run-of-river conditions lasted until late October. During the spring of 2008, Beulah Reservoir was filled to 89% of capacity and drawdown started in early June, after we had finished sampling.

Chironomid Densities and Size

The densities of pelagic chironomids in 2006 were significantly higher than those in 2008 (H = 14.66, df = 2, $P = 0.0007$); other yearly comparisons were not significant. The mean (± SD) density of pelagic chironomids was 0.57 ± 0.67 individuals/m^3 in 2006 ($N = 8$) and 0.15 ± 0.18 individuals/m^3 in 2007 ($N = 20$). None were captured in 2008.

The density of benthic chironomids was significantly lower in areas of the reservoir that were frequently dewatered (substrate elevations from 1010 to 1020 m) compared to areas that were partially dewatered most of the time (997 to 1010 m) or rarely dewatered (elevations less than 997 m; H = 21.16, df = 2, $P < 0.0001$; Figure 2a). Also, chironomids from areas that were frequently dewatered were significantly smaller than those

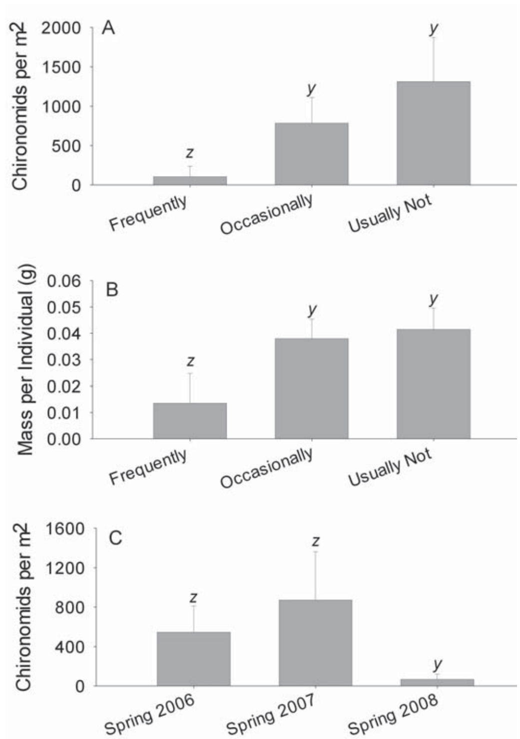

Figure 2. Mean (95% confidence interval) catch per m^2 (upper panel) and mass (middle panel) of benthic chironomids collected from locations in Beulah Reservoir that ranged from 1010 to 1020 m (full pool) and were dewatered in 96% of years (frequently dewatered), substrates lower than 997 m and were dewatered in only 4% of years (usually not), substrates that ranged in elevation from 997 to 1010 m and were partially dewatered in 92% of years (occasionally) during summer drawdown, and catch per m^2 of benthic chironomids collected during the spring, 2006–2008 (bottom panel). Bars with letters in common indicate that the mean rank values did not differ significantly ($P > 0.05$).

from other areas (H = 14.71, df = 2, $P = 0.0006$; Figure 2b). The densities of benthic chironomids in 2008 were significantly lower than in 2006 and 2007 (H = 25.34, df = 2, $P < 0.0001$; Figure 2C).

Fish Catches and Size Structure

In total, we captured 24,162 fish in 5954 h of sampling. The most common fish species we caught were redside shiners (58% of total catch) and northern pikeminnow (31% of total catch). Other taxa collected included bridgelip sucker [*Catostomus columbianus*], longnose dace [*Rhinichthys cataractae*], speckled dace [*Rhinichthys osculus*], white crappie, sculpins [*Cottus* sp.], chiselmouth [*Acrocheilus alutaceus*], mountain whitefish [*Prosopium williamsoni*], and bull trout. We excluded sculpins, chiselmouth, mountain whitefish, and bull trout from our analysis because few of them were captured (<1% of the total catch).

Catch rates in spring of fyke nets for dace ranged from 0.02 ± 0.07 FPH (mean ± SD) during spring 2008 to 0.25 ± 0.56 FPH in 2006 (Figure 3). Catch rates of dace in 2006 were significantly higher than those in 2007 and 2008 (H = 22.16, df = 2, $P < 0.0001$; Figure 3). The CPUE did not differ between 2007 and 2008. Dace from 32 to over 70 mm were present in most of our catches and the percentage of fish captured in different size classes varied between seasons (Table 2). From 2006 to 2007, the percentage of dace in our catches from 32–50 mm increased slightly, while fish greater than 50 mm showed a slight decrease (Table 2). From 2007 to 2008, the percentage of dace from 32–50 mm decreased by 35% and fish from 51–70 mm increased by 147% in our catches (Table 2). The percentage of fish greater than 70 mm in our catches increased each year and the percentages of fish in other size classes were similar between years (Table 2). Dace were not captured by gillnets.

Catch rates of fyke nets for northern pikeminnow ranged from 0.28 ± 0.95 FPH during spring 2008 to 2.68 ± 4.33 FPH in 2006 and differed significantly for all comparisons (H = 75.15, df = 2, $P < 0.0001$; Figure 4a). Northern pikeminnow less than 101 mm represented most of our catches (90–94%) and the percentage of fish in each length

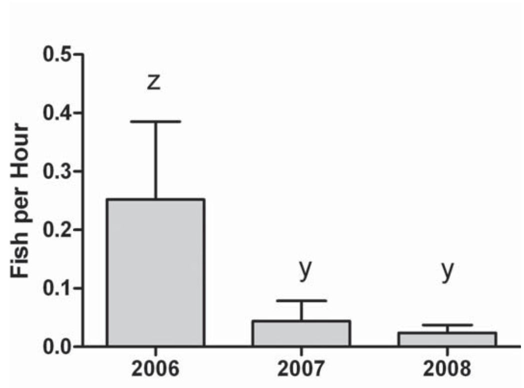

Figure 3. Mean (95% confidence interval) catch per hour of fyke nets for species of dace in Beulah Reservoir, 2006–2008. Bars with letters in common indicate that the mean rank values did not differ significantly ($P > 0.05$).

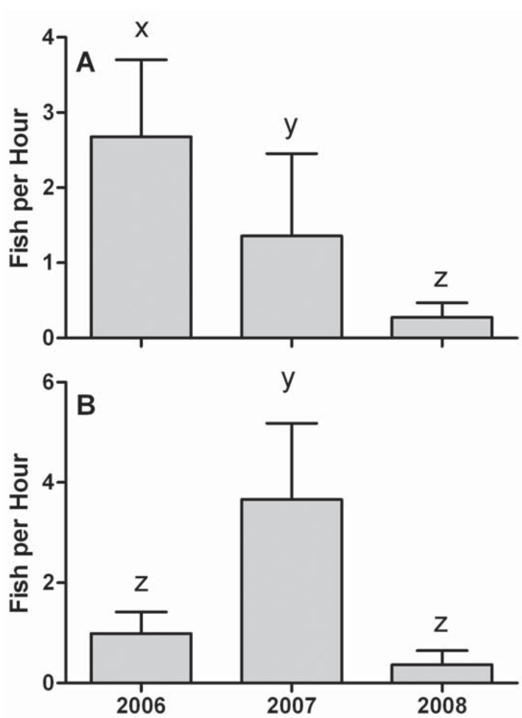

Figure 4. Mean (95% confidence interval) catch per hour of fyke nets (a) and gillnets (b) for northern pikeminnow in Beulah Reservoir, 2006–2008. Bars with letters in common indicate that the mean rank values did not differ significantly ($P > 0.05$).

TABLE 2. The percent of catch by size class for common fishes captured by fyke nets and gill nets in Beulah Reservoir, 2006–2008.[1]

	Percent of catch by size class					
	Spring 2006 (100%)		Spring 2007 (84%)		Spring 2008 (89%)	
Species & FL	Fyke net	Gill net	Fyke net	Gill net	Fyke net	Gill net
Dace	**406**	**0**	**75**	**0**	**55**	**0**
32–50	76	-	79	-	51	-
51–70	23	-	17	-	42	-
>70	1	-	4	-	7	-
N. pikeminnow	**4180**	**57**	**2467**	**94**	**665**	**12**
22–100	92	0	90	0	94	0
101–150	6	33	5	11	5	0
151–200	2	37	4	80	1	0
>200	0	30	0	10	0	100
Redside shiner	**4384**	**12**	**7766**	**7**	**1751**	**1**
22–50	73	0	28	0	28	0
51–70	23	0	57	0	26	0
>70	4	100	16	100	46	100
Suckers	**729**	**11**	**963**	**11**	**146**	**12**
32–100	67	8	61	0	47	0
101–150	22	9	4	0	16	0
151–200	7	9	10	27	11	0
>200	4	73	24	73	27	100
White crappie	**43**	**1**	**213**	**13**	**0**	**0**
68–100	60	100	41	46	-	-
101–150	40	0	55	54	-	-
151–200	0	0	4	0	-	-
>200	0	0	0	0	-	-

[1] Percentages in parentheses below the seasons indicate the reservoir volume during each sample period. Numbers in bold within each column are the sample sizes for that species, gear, and sample period; FL = fork length (mm).

class did not change appreciably among sampling periods (Table 2).

The catch rates of gillnets for northern pikeminnow ranged from 0.37 ± 1.26 FPH during spring 2008 to 3.66 ± 6.52 FPH in 2007 (Figure 4b). Catch rates showed a significant increase (272%) from 2006 to 2007 and decreased significantly (by 90%) from 2007 to 2008. (H = 25.17, df = 2, $P < 0.0001$; Figure 4b). The catch rates of gillnets for northern pikeminnow were 63% lower in 2008 than in 2006 but did not differ significantly (H = 25.17, df = 2, $P < 0.0001$; Figure 4b). From 2006 to 2007, fish from 101–150 mm and those greater than 200 mm showed a 67% decrease, whereas fish from 151–200 mm showed a 116% increase in our catches (Table 2). From 2007 to 2008, fish from 101–200 mm were not captured and the percentage of fish in our catches greater than 200 mm increased ten-fold (Table 2).

The CPUE of fyke nets for redside shiners ranged from 0.74 ± 1.71 FPH during 2008 to 4.40 ± 9.81 FPH during 2007 (Figure 5a). The CPUE of redside shiners in 2008 differed significantly from values in 2006 and 2007 (H = 37.72, df = 2, $P < 0.0001$; Figure 5a); other comparisons were similar. From 2006 to 2007, fish from 22–50 mm showed a 62% decrease and fish >50 mm increased almost 3-fold in our catches (Table 2). From 2007 to 2008, fish from 51–70 mm showed a 46% decrease in our catches and large fish (> 70 mm) a 188% increase in our catches (Table 2). Only redside shiners greater than 70 mm were captured by gillnets (Table 2). The mean CPUE of these larger fish ranged from 0.02 ± 0.21 during 2008 to 0.33 ± 1.17 FPH during 2007 and no yearly catch comparisons differed significantly (Figure 5b).

Mean CPUE of fyke nets for suckers ranged from 0.06 ± 0.12 FPH in 2008 to 0.53 ± 1.22

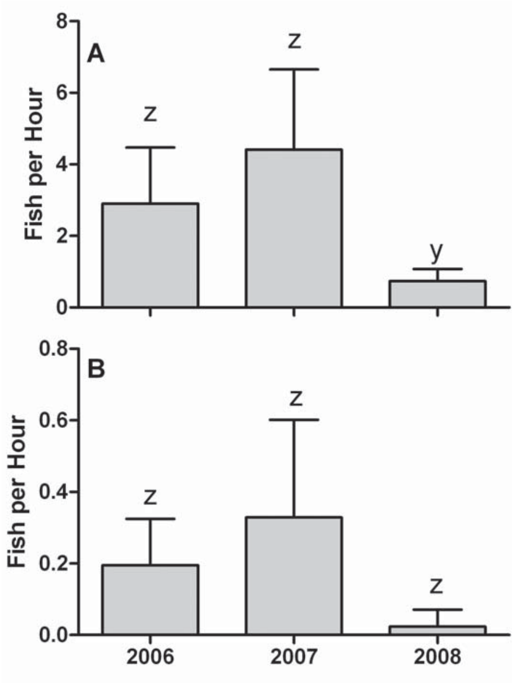

Figure 5. Mean (95% confidence interval) catch per hour of fyke nets (a) and gillnets (b) for redside shiners in Beulah Reservoir, 2006–2008. Bars with letters in common indicate that the mean rank values did not differ significantly ($P > 0.05$).

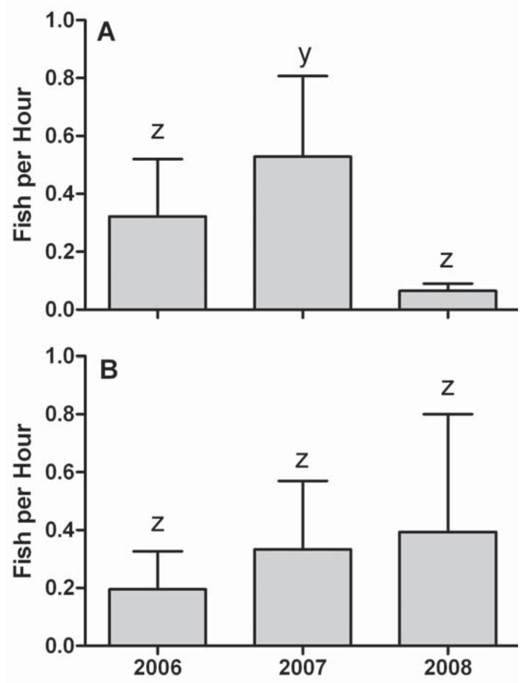

Figure 6. Mean (95% confidence interval) catch per hour of fyke nets (a) and gillnets (b) for species of suckers in Beulah Reservoir, 2006–2008. Bars with letters in common indicate that the mean rank values did not differ significantly ($P > 0.05$).

FPH in 2007 (Figure 6b). The CPUE of suckers increased significantly from 2006 to 2007 and decreased significantly from 2007 to 2008 (H = 42.46, df = 2, $P < 0.0001$). Catch rates of suckers in 2006 and 2008 did not differ significantly. From 2006 to 2007, suckers from 101–150 mm showed an 82% decrease and fish > 200 mm increased six-fold in our catches (Table 2). Fish from 32–100 mm comprised similar percentages of our catch. From 2007 to 2008, after the complete dewatering, fish from 32–100 mm showed a 23% decrease and fish 101–150 mm a four-fold increase in our catches (Table 2). Fish > 150 mm comprised similar percentages of our catch during these years.

The CPUE of gillnets for suckers ranged from 0.20 ± 0.79 FPH during 2006 to 0.39 ± 1.83 FPH during 2008 (Figure 6B). The CPUE of gillnets for suckers did not differ significantly between any yearly comparisons (H = 1.02, df = 2, P = 0.60). Only suckers larger than 150 mm were consistently captured in gill nets and small sample sizes precluded further analysis.

Mean CPUE of fyke nets for white crappie was highest during 2007 (0.12 ± 0.23 FPH) and they were not collected after the reservoir was dewatered (Figure 7a). The CPUE of fyke nets for white crappie differed significantly for all yearly comparisons (H = 108.1, df = 2, $P < 0.0001$). From 2006 to 2007, fish from 68–100 mm showed a decrease (32%) in our catches, whereas all other size classes increased (Table 2).

Mean CPUE of gillnets for white crappie was highest during 2007 (0.48 ± 1.46 FPH) and low for other years (2006: 0.01 ± 0.16 FPH; 2008: 0.0 ± 0.0 FPH). The CPUE of gillnets for white crappie was significantly higher in 2007 than catch rates from other years (H = 23.87, df = 2, $P < 0.0001$). In 2006, only fish from 68–100 mm were captured in gillnets (Table 2). After this, in 2007, this size

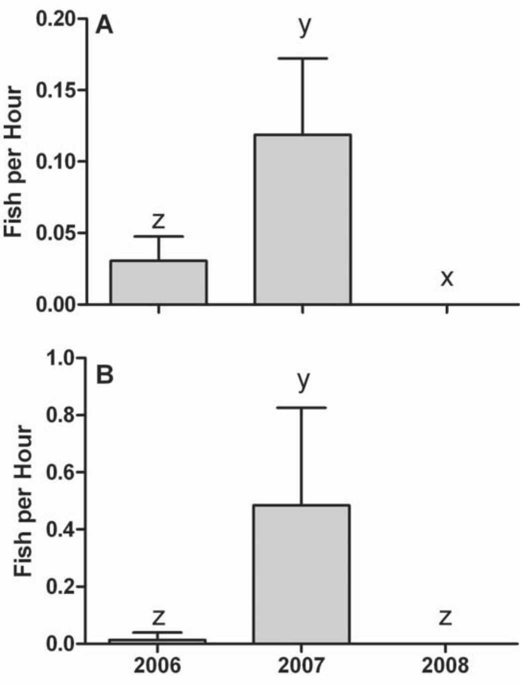

Figure 7. Mean (95% confidence interval) catch per hour of fyke nets (a) and gillnets (b) for white crappies in Beulah Reservoir, 2006–2008. Bars with letters in common indicate that the mean rank values did not differ significantly ($P > 0.05$).

class of fish decreased and larger fish comprised 54% of our catch (Table 2). White crappies were not captured in gillnets in 2008.

Discussion

Our results indicate that summer drawdown of Beulah Reservoir impacts the fish assemblage and larval chironomid population but the severity of effects were dependent on the extent of drawdown. In 2006, when the drawdown (68%) was mild compared to the mean drawdown of 85% from 1997–2008, we detected few significant changes in the CPUE and size composition of fish in our catches from 2006 compared to those in 2007. We did note that the densities of benthic chironomid larvae in substrates that were frequently dewatered during 1997–2008 were lower and their average mass smaller than those in areas that remained inundated. In 2007, Beulah Reservoir was reduced to run-of-river levels and the amount of habitat available for fish and invertebrates was less than one percent of that at full pool and deep-water habitat was eliminated. As such, in the spring of 2008 (after the reservoir refilled), the catch of pelagic chironomid larvae decreased to near zero and the relative density of benthic forms decreased by 96%. Further, the catches of most fish species decreased, and the sizes of fish in our catches changed depending on gear type and species. Our results are similar to those of Petersen et al. (2002), who also reported a decrease in overall CPUE and changes in species composition of gillnet catches a year after a 96% drawdown of Beulah Reservoir. However, they also concluded that the fish assemblage in Beulah Reservoir was fairly resilient to this type of water management, considering that the reservoir has been drawn down to run-of-river levels several years since 1936 and gill net sets from 1955 to 1970 have shown only temporary (1–3 years) decreases in catches. Our results tend to support this conclusion because our relatively high catches in 2006 came one year after three consecutive years of maximum drawdown (see Figure 1). Although our catches of fish in the spring of 2008 were relatively low following the maximum drawdown in 2007, we would expect—based on historical information (see Petersen et al. 2002 for a review)—fish catches to increase within a couple of years. Increases in fish catches would probably be hastened during a few (e.g., 2–3) consecutive high water years that would preclude the need to draw the reservoir down to minimum levels. If, however, the reservoir is drawn down to minimum levels for several (e.g., 2–4) consecutive years, increases in fish catches may take longer.

We only captured pelagic chironomids in relatively high numbers in 2006 and 2007—catches were essentially nil in 2008. Even so, our highest mean catch was 0.5 individuals/m^3 and our catch rates decreased from 2006 to 2007, suggesting that other factors besides drawdown, such as altered timing of adult emergence, unknown timing of larval dispersion, and increased predation, were affecting pelagic chironomid densities (Yamagishi and Fukuhara 1971, Hershey 1985, Pinder 1986, Mousavi et al. 2002, Takagi et al. 2005).

We observed fewer and smaller chironomids in sediments of Beulah Reservoir that were frequently dewatered compared to areas that were typically inundated, which is similar to the results of Furey et al. (2006), who studied the responses of benthic macroinvertebrates to more than 30 years of seasonal drawdowns in Sooke Lake Reservoir, British Columbia, Canada. That the density of benthic chironomids in Beulah Reservoir was highest below the drawdown exposure zone (i.e., the area of sediment exposed—and eventually re-inundated—during an annual drawdown cycle; Furey et al. 2006) was not surprising and has been reported elsewhere (Grimas 1965, Fisher and LaVoy 1972, Furey et al. 2006). Higher densities of benthic macroinvertebrates below the drawdown exposure zone were probably due to the movement of organisms during drawdown to more consistently inundated habitat. The periodic desiccation that occurs in the drawdown exposure zone can lead to the development of dispersal mechanisms, physiological adaptations, or life-history strategies that enable benthic macroinvertebrates to survive such ephemeral conditions (Williams 1987, Delettre 1989). We suspect that a more detailed analysis of the chironomid species assemblage in Beulah Reservoir would reveal insight into the attributes that allow these insects to persist, but this was beyond the scope of our study. The periodic disturbances caused by drawdown in Beulah Reservoir also explains the small body size of benthic chironomid larvae in the drawdown exposure zone since chironomids in disturbed habitats often have small body sizes (Statzner et al. 2001). A small body size, desiccant-resistant instars, and the short generation times of an *r*-selected life history strategy can facilitate the colonization and survival of chironomids in reservoirs (Furey et al. 2006). It is possible that minor to moderate manipulations of a drawdown strategy would reduce the impacts of such events on benthic chironomid larvae production, and we recommend future research explore this notion. For example, Benson and Hudson (1975) found that reducing fall drawdown by 4–5 m in Lake Francis Case, South Dakota, yielded a three-fold increase in benthic macroinvertebrate production during May.

Despite the reduction of nearshore habitat available for fish in the summer and fall of 2006, the CPUE of fish was similar between the springs of 2006 and 2007, indicating that fish densities in Beulah Reservoir did not change appreciably during a year of moderate drawdown. In 2007, however, which included a complete dewatering of the reservoir, the CPUE of most species decreased significantly from 2007 to 2008. Others have shown decreased catch rates of fish following drawdowns less severe than ours (Heman et al. 1969, Gaboury and Patalas 1984, Paller 1997), and we were not surprised to record decreased CPUE of fish after the dewatering of Beulah Reservoir. We surmise that, during the dewatering event, many fish left the reservoir, were stranded on dewatered substrates, or succumbed to increased predation pressure as the water volume slowly decreased. Evidence that many fish were entrained below the dam is apparent in a large fish kill that occurred below Agency Valley Dam late in the summer of 2007. This fish kill occurred directly downstream of the dam and involved thousands of fish and crayfish and it seems unlikely that such a large number of fish was present in the small tailrace of the dam prior to dewatering. It is also possible that large numbers of fish moved upstream. That our poor catch rates persisted into the spring of 2008 indicates that fish density remained low after re-filling of the reservoir and over the winter. Because we only evaluated two levels of drawdown in Beulah Reservoir and the responses of different species to these events varied, it is difficult to ascribe clear cause-effect relations among drawdown, water volume, and fish catches. As we mentioned earlier, however, the fish assemblage in Beulah Reservoir seems to be fairly resilient to the frequent and varied drawdowns that occur.

Besides impacting the CPUE of some species, drawdown of Beulah Reservoir had obvious effects on the sizes of fish captured. Specifically, the dewatering event of 2007 severely reduced the numbers of northern pikeminnow less than 200 mm, redside shiners from 20 to 70 mm, suckers less than 100 mm, and dace less than 50 mm. Also, white crappies were not captured after the severe drawdown. Paller (1997) reported changes in the size composition of fish in Par Pond following drawdown, particularly the loss of smaller largemouth bass, lake chubsuckers [*Erimyzon sucetta*], and bluegills [*Lepomis macrochirus*]. In our study, the loss of small fish was probably due to the same reasons described above—namely, emigration, stranding, and increased predation. Many fishes rely on the littoral zone and its associated vegetation for shelter and safety. Maintaining physical structure in the nearshore area—for example, by planting of natural or artificial vegetation on exposed drawdown shorelines (Ratcliff et al. 2009)—may help maintain the long-term availability and integrity of habitat for juvenile fishes.

Although our results from 2008 show that a single dewatering event can negatively impact fish populations in Beulah Reservoir, we think that the effects are probably short lived and the fish community is somewhat resilient to such disturbances. For example, our catch rates of fish in 2006 and 2007 were relatively high and the sizes of fish in those catches were indicative of populations with diverse age classes. Further, our work in 2006 was conducted about 1.5–2.5 years after consecutive dewatering events in Beulah Reservoir from 2002 to 2004—when no sampling occurred. Petersen and Kofoot (2002) and Petersen et al. (2003) sampled Beulah Reservoir during the summers of 2001 and 2002—prior to the three consecutive years of dewatering—and also reported that diverse age classes of fish were present. Thus, if severe drawdown events were having lasting effects on fish populations in Beulah Reservoir, we would have expected more evidence of this when we started sampling in 2006. Sampling of fish in Beulah Reservoir under a no or minimal drawdown scenario would provide some useful context for our results. The reasons why our catches in 2006 were high and contained many small fish could be due to higher reproductive success of species that persisted in the reservoir (e.g., suckers), increased recruitment in a relatively predator-free post-dewatering environment, increased availability of shoreline cover, or higher river flows prior to sampling, which may have displaced juveniles and increased emigration of fishes from areas upstream of the reservoir (e.g., dace and northern pikeminnow; Gonzalez et al. 1998).

A complete dewatering, or extensive drawdown (e.g., ca. 80-90%), of Beulah Reservoir produces the most pronounced short-term effects and could reduce the availability of forage fish for threatened bull trout in the fall, winter, and early spring. Indeed, management of the bull trout population in the NFMR and Beulah Reservoir is a priority for state, federal, and tribal agencies. Even though, over the long term, summer drawdowns may not cause irreparable harm to fish populations in Beulah Reservoir, avoiding or minimizing the occurrence of severe drawdown events will reduce short-term effects and likely ensure that an adequate prey base for bull trout is always present.

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