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Authors: Heise, Ryan J., Cope, W. Gregory, Kwak, Thomas J., and Eads, Chris B.

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SHORT-TERM EFFECTS OF SMALL DAM REMOVAL ON A FRESHWATER MUSSEL ASSEMBLAGE

Ryan J. Heise[1]

North Carolina Wildlife Resources Commission, 1718 NC Highway 56
Creedmoor, North Carolina 27522 U.S.A.
ryan.heise@ncwildlife.org, 919-707-0368

W. Gregory Cope

Department of Environmental and Molecular Toxicology, North Carolina State University
Raleigh, North Carolina 27695-7633 U.S.A.
greg_cope@ncsu.edu

Thomas J. Kwak

U.S. Geological Survey, North Carolina Cooperative Fish and Wildlife Research Unit, Department of Biology
North Carolina State University, Raleigh, North Carolina 27695-7617 U.S.A.
tkwak@ncsu.edu

Chris B. Eads

College of Veterinary Medicine, North Carolina State University
Raleigh, North Carolina 27606 U.S.A.
chris_eads@ncsu.edu

[1]Corresponding Author

ABSTRACT

Dam removal is increasingly used to restore lotic habitat and biota, but its effects on freshwater mussels (family Unionidae) are not well known. We conducted a four-year study to assess short-term effects on mussels after removal of a small hydropower dam on the Deep River (Cape Fear River drainage), North Carolina, USA, in 2006. We conducted annual pre- and post-removal monitoring of mussel density, richness, and survival (post removal only) with transect surveys and quadrat excavation, and assessed changes in substrate composition at two impact sites (tailrace and impoundment) and two reference sites. Before-after-control-impact (BACI) analyses of variance did not detect a significant change in mussel density (total or individually for the three most abundant species), species richness, Eastern Elliptio (*Elliptio complanata*) mean length, or substrate composition in the tailrace or drained impoundment following dam removal. Apparent annual survival estimates of Eastern Elliptio at the tailrace site did not differ among sampling periods and were similar to control sites. We observed minimal mussel mortality from stranding in the dewatered reservoir. These results demonstrate that adverse short-term impacts of dam removal on downstream mussel assemblages can be minimized with appropriate planning, timing, and removal techniques, but additional monitoring is warranted to determine long-term effects on mussels within the restored river reach.

KEY WORDS Apparent survival, BACI, *Elliptio*, imperiled species, mussel density, quantitative sampling, restoration, Unionidae

INTRODUCTION

The diverse freshwater mussel fauna of the southeastern U.S. is highly imperiled, and loss of habitat and other effects of dams are among the most important factors in the decline of these animals (Richter et al., 1997; Strayer et al., 2004; Cope et al., 2008). The ecological costs of dams are well documented (Dynesius & Nilsson, 1994; Watters, 2000; Bednarek et al., 2001) and removal is becoming a common river restoration tool (Bednarek et al., 2001; Poff & Hart, 2002), especially as the financial cost of maintaining these aging structures exceeds their benefits (Stanley & Doyle, 2003).

Much of the research guiding dam removal has been conducted on large, high dams, but the vast majority of future removal projects concern small or mediumsized dams (Heinz Center, 2002). Much less research is available on the effects of these dams on physical and biological components of river ecosystems, and thus, dam removals commonly occur without sufficient information to predict their outcome. Habitat restoration through dam removal is an important conservation strategy in the long-term, but potential negative short-term effects of dam removal on mussels have been rarely investigated. For example, large quantities of sediment are often deposited and stored within an impoundment, and sediment mobilization during dam removal may impact or extirpate downstream mussel populations (Sethi et al., 2004). In some cases, tailraces of small dams or even the reservoirs themselves support important mussel assemblages (Nedeau et al., 2000; Singer & Gangloff, 2011), and these habitats are especially vulnerable to negative effects of dam removal.

Based on available information (e.g., Sethi et al., 2004) and input from state and federal agencies, removal of a small hydropower dam (Carbonton Dam) on the Deep River, North Carolina, was conducted following procedures designed to minimize adverse effects on fish and mussels. These procedures included a gradual drawdown of the impoundment and dam removal during the fall-winter, a season regarded as less stressful to aquatic biota. We conducted a four-year study to examine the effectiveness of these measures on reducing negative effects on the mussel assemblage in the Deep River. We examined changes in mussel density, species richness, length, and survival (post removal only) and substrate characteristics at impacted and non-impacted sites prior to and after dam removal.

METHODS

Study Site

The Deep River is a fourth-order tributary of the upper Cape Fear River drainage in the Piedmont physiographic province in central North Carolina (Fig. 1). The drainage area upstream of Carbonton Dam is about 2,600 km^2 and the estimated average annual discharge at the dam site is 38 m^3/s (based on the difference

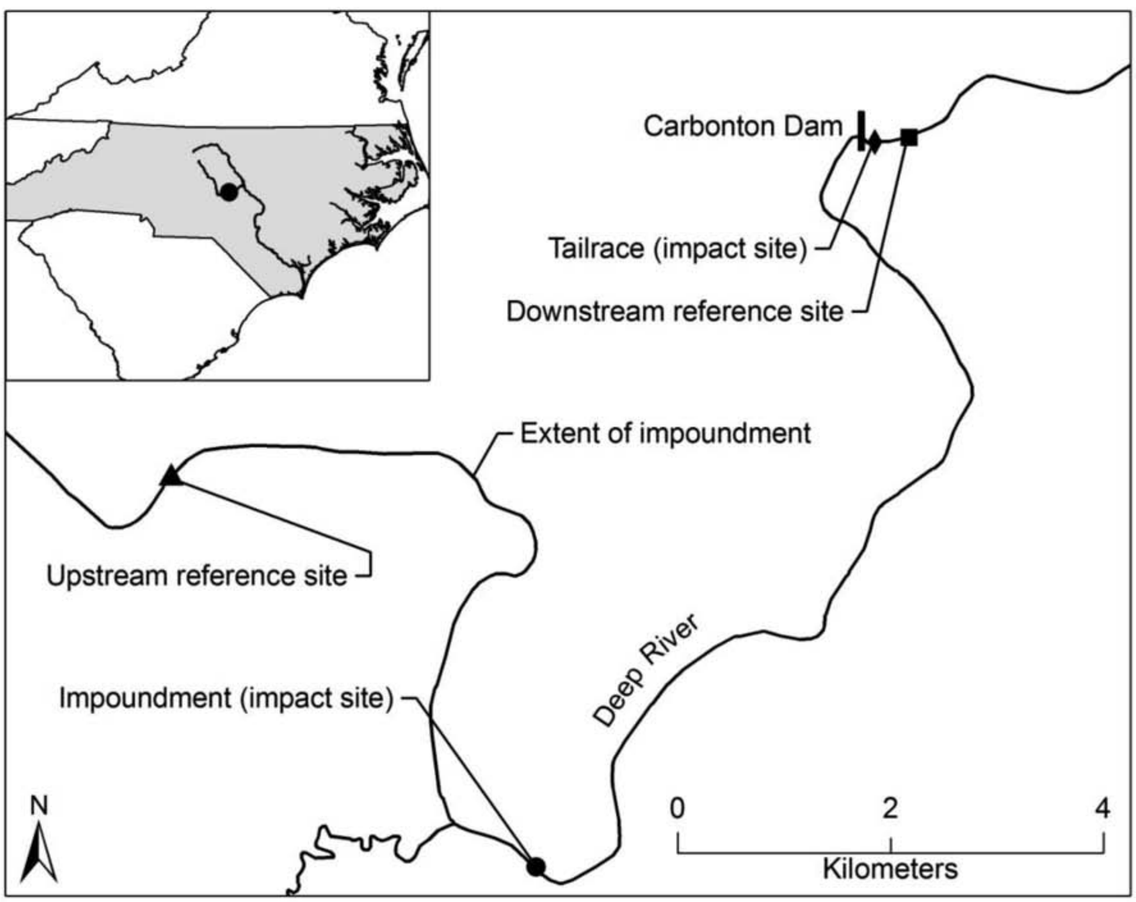

FIGURE 1

Location of Carbonton Dam in North Carolina (inset) and mussel sampling sites on the Deep River, North Carolina.

between discharge at USGS gaging stations 2102000 and 2101726). The watershed is primarily forested (63%), with smaller percentages of agricultural (12% pasture, 8% crops) and urban (11%) land use (NCDNER, 2005). Carbonton Dam was constructed in 1921 southwest of Sanford, North Carolina. The dam was 5 m high and 80 m wide with a hydropower generation capacity of one megawatt. The impoundment created by the dam was narrow and was contained within the banks of the Deep River (width = 45-80 m) and extended about 15 km upstream with a maximum depth of 8 m (Restoration Systems and Ecoscience Corporation, 2006). At least 14 other dams exist upstream on the Deep River, the closest being about 38 km upstream of Carbonton Dam (NCDENR, 2004).

Carbonton Dam was not navigable (i.e., without a lock) and lacked any engineering for fish passage. Water quality degradation, including low dissolved oxygen and excessive algal production, also had been recorded within the impoundment (NCDENR, 2005) and presumably contributed further to fragmentation of riverine habitat. State and federal environmental agencies prioritized the dam for removal to restore connectivity between populations of several state threatened and endangered mussel species (Table 1) and a federally endangered fish species (Cape Fear Shiner, *Notropis mekistocholas*) that occurred in this segment of the river. The dam was operated in a run-of-the-river flow regime (minimal water storage capacity) until June 2004. In 2005, a private environmental restoration company purchased the dam for removal to provide stream mitigation. Sufficient lead time between the purchase of the dam and the target removal date allowed state and federal resource agencies and the company to develop and recommend procedures that could minimize the impact of dam removal on aquatic life. These procedures included a gradual drawdown of the impoundment (over a two- to three-week period) and removal in the fall-winter, a time regarded as less stressful to aquatic biota in the region because large precipitation events are less frequent, dissolved oxygen concentrations are highest, and water temperatures are cool. These procedures also were attempts to minimize erosion and transport of sediment and large woody debris stored in the impoundment (see Bednarek et al., 2001). Drawdown of the impoundment, using existing powerhouse gates, began on October 15, 2005, and proceeded for approximately three weeks, and removal was completed in February 2006 (Restoration Systems and Ecoscience Corporation, 2006).

Sampling design

We conducted annual pre- and post-dam removal mussel surveys at two impacted sites and two reference sites in June from 2005 to 2008 (1 pre-removal and 3 post-removal samples; Fig. 1). However, we were unable to conduct a pre-dam removal mussel survey at the impoundment site (see below) prior to the drawdown and dam removal because of excessive depth. Criteria used to select sites included the presence of mussels (based on preliminary mussel surveys) and accessibility. One impacted site was located within the impounded reach, 10 river km (rkm) upstream of the dam, and the other was in the dam tailrace, 70 m downstream of the dam. One reference site was located 18 rkm upstream of the dam, beyond the influence of the impoundment, and the other was 400 m downstream of the dam. In larger rivers with high dams, 400 m may not be sufficient distance to qualify as a reference site (e.g., Vaughn & Taylor, 1999). Because of the small size of the Deep River and other channel morphological features, this distance appeared adequate to isolate the site from short-term effects of dam removal, and the high substrate stability we observed at the site during the study (see Results) supported the use of this site as a reference. We attempted to standardize sampling area within the boundaries of mussel aggregations. Sampling areas were 48 m long (upstream to downstream) by 5 m wide (bank toward channel) at all sites except for the tailrace site, which was 48 m long by 17 m wide; a wider sampling area was used at the tailrace site because mussels were broadly distributed across the wider river channel.

We used a combination of visual-tactile and quadrat excavation sampling to maximize the accuracy of mussel species richness and density estimates. Visual-tactile sampling is preferred for estimating species richness because it allows rapid coverage of large areas and collection of large numbers of individuals, but it can underestimate density by failing to detect burrowed and small individuals (Smith et al., 2000; Strayer & Smith, 2003). In contrast, quadrat excavation provides less biased density estimates, but because it is slower and more laborious, it is less effective for estimating richness.

At each site, we established 12, 3-m wide transects spaced 4 m apart (on center) positioned perpendicular to the shoreline. Sample areas were relocated in subsequent years by GPS coordinates and permanent markers on the river bank. Nine of the transects were randomly selected for sampling by visual-tactile methods, and the three remaining transects were sampled with quadrats. During sampling, a white metal chain was placed along the center line of the transect to indicate its location, and the transect length was measured to allow determination of transect area (length x 3 m). For visual-tactile surveys, three experienced personnel simultaneously snorkeled along each transect, each searching about 1 m of the width of the transect. Snorkelers searched for mussels visually and by feeling through the substrate (a heterogeneous mixture of silt, sand,

and gravel), and then placed all mussels into individually labeled dive bags. The annual mean sampling time among transects (all three snorkelers combined) ranged from 23-113 min among sites and years (overall mean = 58 min, coefficient of variation among sample dates at a site averaged 19.8%). After completing each transect, all mussels were identified to species, measured (total length, mm), and batch marked with a Dremel® tool by etching the periostracum of each valve of the mussel with a mark unique to the survey year (i.e., 1-4). After processing, each mussel was placed into the substrate within the transect from which it came. In addition to annual visual-tactile sampling, we conducted weekly visual surveys (1-2 h each sampling event) of the newly exposed river banks at the impoundment site during much of the drawdown process (October 2005 – February 2006) to document the extent of mussel stranding and mortality.

For the three quadrat-sampled transects, 10 randomly selected points were located within each transect according to coordinates based on the distance from shore and the width of the transect (total of 30 quadrats per site). At each sampling point, a 0.25-m^2 metal quadrat frame was placed on the substrate surface, and all sediment within the quadrat was excavated down to aggregated substrate or to 10 cm and placed in a 20-L container and processed on shore. Mussels from quadrat samples were processed as described for those from visual-tactile samples.

We developed site-specific calibration factors to account for bias in density estimates from visual-tactile sampling relative to quadrat sampling, and to standardize density estimates from these two methods. Calibration factors were computed and used to adjust density estimates after Peterson & Paukert (2009) as follows. First, the quadrat density estimate was divided by the visual density estimate (using combined data from all mussel species) for each year, and then an arithmetic mean among years was calculated. This mean calibration factor was then multiplied by each visual density estimate to yield an adjusted visual density estimate that represented a complete census mussel estimate comparable to quadrat sampling. The adjusted density estimates from visual data were used in all analyses. We considered stream sites as the experimental unit rather than transects (sensu Hurlbert, 1984). Estimated site density was expressed as the mean density among all 12 transects; thus N=1 density estimate per site per year.

We analyzed substrate removed from quadrats during mussel sampling to examine potential changes in substrate composition associated with dam removal. We fractioned substrate samples into particle size categories of cobble/boulder (>64.0 mm diameter), gravel (64.0 - 2.0 mm), and sand/silt/clay (<2.0 mm) (Bovee & Milhous, 1978) by passing the sediment through a set of nested sequential sieves. We then determined the relative percentage of each of these three particle size categories by measuring the total wet weight (nearest kg) of each fraction with an analog hanging balance (Viking Pelouze® Model 7810, Pelouze Scale Company, Evanston, IL).

Data analysis

A before-after-control-impact (BACI) design was used to assess temporal differences in mussel density and length, species richness, and substrate composition among sites. Species richness was defined as the total number of species observed using both visual and quadrat methods at a given site and year. We followed an analysis of variance (ANOVA) procedure for asymmetric BACI described by Underwood (1991, 1994) and Smith (2002) with a significance probability of 5% ($\alpha = 0.05$). The response variables included in the BACI analysis were density (all mussel species, and separately for Eastern Elliptio (*Elliptio complanata*), Eastern Creekshell (*Villosa delumbis*), and Eastern Pondhorn (*Uniomerus carolinianus*)), Eastern Elliptio mean length, total mussel species richness, and percentage of each substrate category (boulder/cobble, gravel, and sand/silt/clay). The general model for this analysis was

$$y = \mu + BA + T(BA) + CI + L(CI) + (BA \times CI) + \text{error},$$

where y is the measured response variable (i.e., mussel density, species richness, mean mussel length, substrate composition), μ is the grand mean of the measured response variable, BA is the mean effect of the before or after period (i.e., 2005 before, 2006–2008 after), T(BA) is the effect of sampling date (i.e., year) within the before or after period, CI is the mean effect of the control (i.e., upstream or downstream reference sites) or impact treatment (i.e., tailrace or impoundment sites), L(CI) is the effect of location (i.e., site) within the control or impact treatment, and (BA x CI) is the effect of the before or after period in the control or impact treatment (i.e., the BACI effect). The response variables conformed to the normality assumption for ANOVA (Shapiro-Wilk W test, $P>0.05$; Zar, 1996).

We conducted two separate BACI analyses; one in which the tailrace (impact) response variables were compared to those of the two reference sites (controls), and another in which the impoundment (impact) variables were compared to those of the reference sites. Because we were unable to conduct a pre-dam removal mussel survey in the impoundment, we used data from the first post-removal sample (taken 4 months after removal in 2006) to represent pre-removal (2005) conditions so as to conform to the balance needed for the BACI data analysis. We assumed that the mussel assemblage in the impoundment before dam removal was similar to that seen 4 months after removal because minimal bank erosion was observed, major changes in sediment composition in the former impoundment were not observed during our study, and we found few dead mussels that were stranded by receding water (see Results). In addition to BACI analyses including mean mussel length, we also compared length distributions of Eastern Elliptio sampled at the upstream reference, tailrace, and downstream reference sites between years before (2005) and after dam removal (2006–2008; all methods combined) with pairwise Kolmogorov-Smirnov two-sample tests.

Apparent survival of the Eastern Elliptio at each site was estimated with the Cormack-Jolly-Seber model (with model averaging) in the software program MARK (Lebreton et al., 1992; White & Burnham, 1999), using recapture rates of marked individuals (visual and quadrat data combined) in successive annual samples. Apparent survival was defined as the probability that an individual mussel was alive and available for recapture (White & Burnham, 1999). The Cormack-Jolly-Seber model is an open population model, which we considered most applicable because it allowed for immigration or recruitment and emigration or death occurring between sampling periods. Capture probability was estimated to describe the chance of capturing an individual that is present during the study period. We did not estimate survival or capture probability for any other species because of their rarity and low sample sizes (see Results).

RESULTS

Trends in mussel and habitat parameters

Among all sites and years, a total of 11 mussel species were collected, including one state endangered, three state threatened, and one significantly rare species (Table 1). Cumulative richness among all years was highest at the tailrace site (10 species), and lowest at the impoundment site (5). Estimates of species richness over time were variable and showed no clear pattern at most sites except for the impoundment site where richness appeared to decline gradually after dam removal (Fig. 2). Eastern Elliptio was the numerically dominant species at all sites and accounted for 88-95% of the mussels. Eastern Pondhorn, Eastern Creekshell, and Triangle Floater (*Alasmidonta undulata*) were also found at all sites but in much lower densities (Table 1). Most species were represented by four or fewer individuals at each site. Density of Eastern Elliptio was highest but most variable, both within and among years, at the downstream reference site (5.8 to 11.6 mussels/m^2 among years; Fig. 2). Density of Eastern Elliptio was lowest at the previously impounded site (0.52 to 0.57 mussels/m^2; Fig. 2).

We observed only minimal short-term changes in substrate composition at the two impact sites among years (Fig. 3). The percentage of fine sediment (i.e., sand/silt/clay) at the tailrace site appeared to increase slightly from 38.3% pre-dam removal to 49.4% the first year after removal, but it then declined to 24.7% by the third year after removal; however, standard error for estimates of fine sediments overlapped among most years. Similarly, the percentage of fine sediment at the impoundment site appeared to increase from 30.1% immediately after removal (2006) to 49.6% in 2007, but it then decreased to 36.4% by 2008. The substrate composition at the two reference sites remained stable among years (Fig. 3A, B).

The mean length of Eastern Elliptio varied widely among sites, but not among years within sites. Mean length of Eastern Elliptio over the four years ranged from 51.8–53.9 mm (52.6 mm overall mean) at the upstream reference site, 53.1–54.7 mm (53.7 mm overall mean) at the impoundment site, 73.4–77.1 mm (75.4 mm overall mean) at the tailrace site, and 69.6–73.0 mm (71.1 mm overall mean) at the downstream reference site. Length distributions differed significantly among years (before and after dam removal) at the tailrace and downstream reference sites, but not at the upstream reference site (Table 2). The differences at the tailrace site were the result of variable shifts in length frequency about the mode, but at the downstream reference site, the differences reflected a change in the mode and skewness, indicating a reduction in modal size and increased numbers of smaller individuals.

BACI effects

At the tailrace site, no significant dam removal effects were detected by the BACI analysis for any mussel or substrate response variable. We found significant control/impact effects in the mean length of Eastern Elliptio ($P<0.0001$) and proportions of boulder/cobble ($P=0.003$) and sand/silt/clay ($P=0.002$), indicating consistent differences between the control and impact sites that did not change after removal of the dam (see Fig. 3). We also detected significant location effects (nested within control/impact, $P<0.05$) in all mussel variables (except for total species richness) and in the proportions of boulder/cobble ($P=0.004$) and sand/silt/clay ($P=0.007$), reflecting differences between the two control sites.

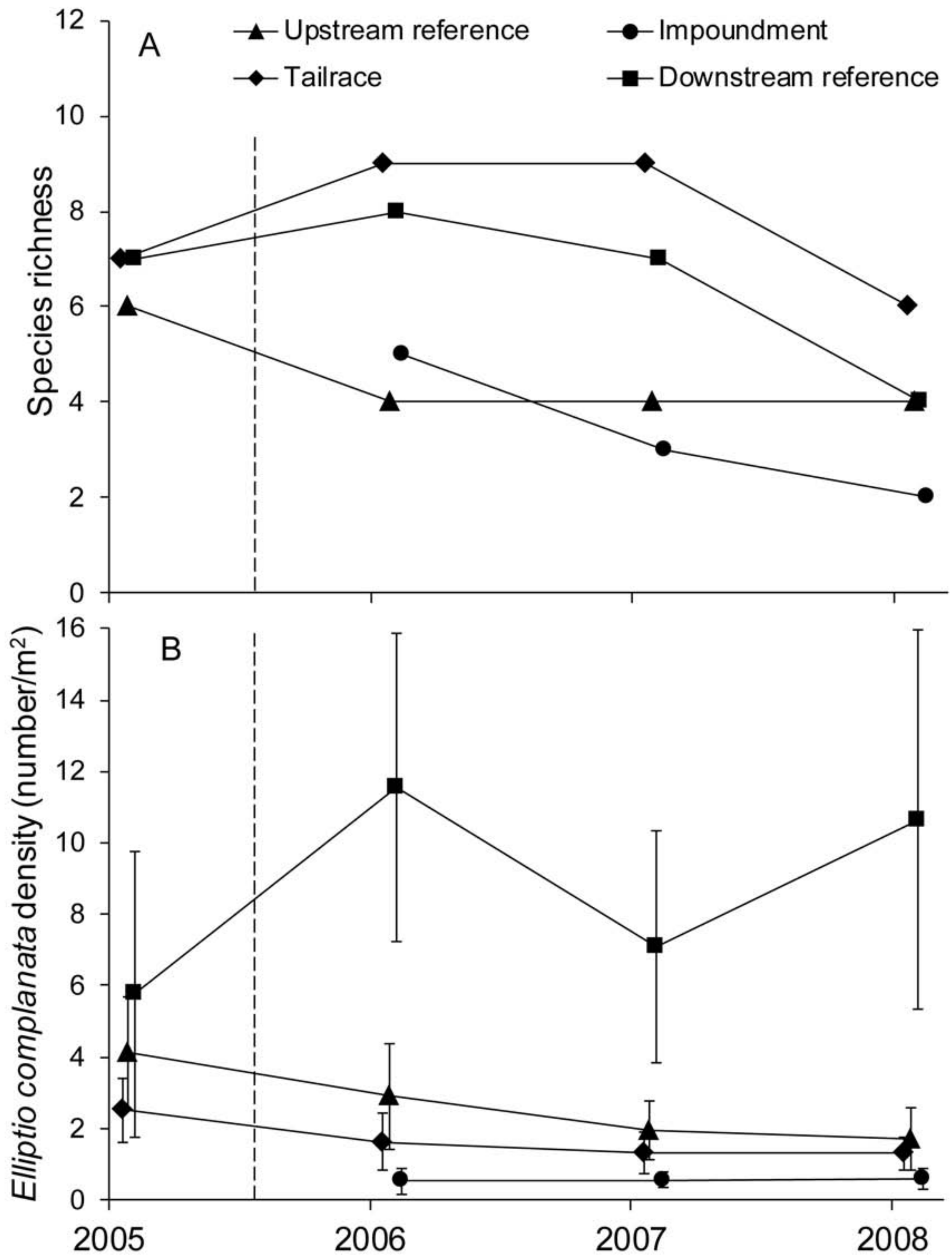

FIGURE 2

(A) Total species richness and (B) mean (among sampling transects) Eastern Elliptio (*Elliptio complanata*) density at each site on the Deep River, North Carolina, from 2005-2008. Error bars are 95% confidence intervals; the vertical dashed line represents the date of removal of the Carbonton Dam. Only means were included in the BACI analyses.

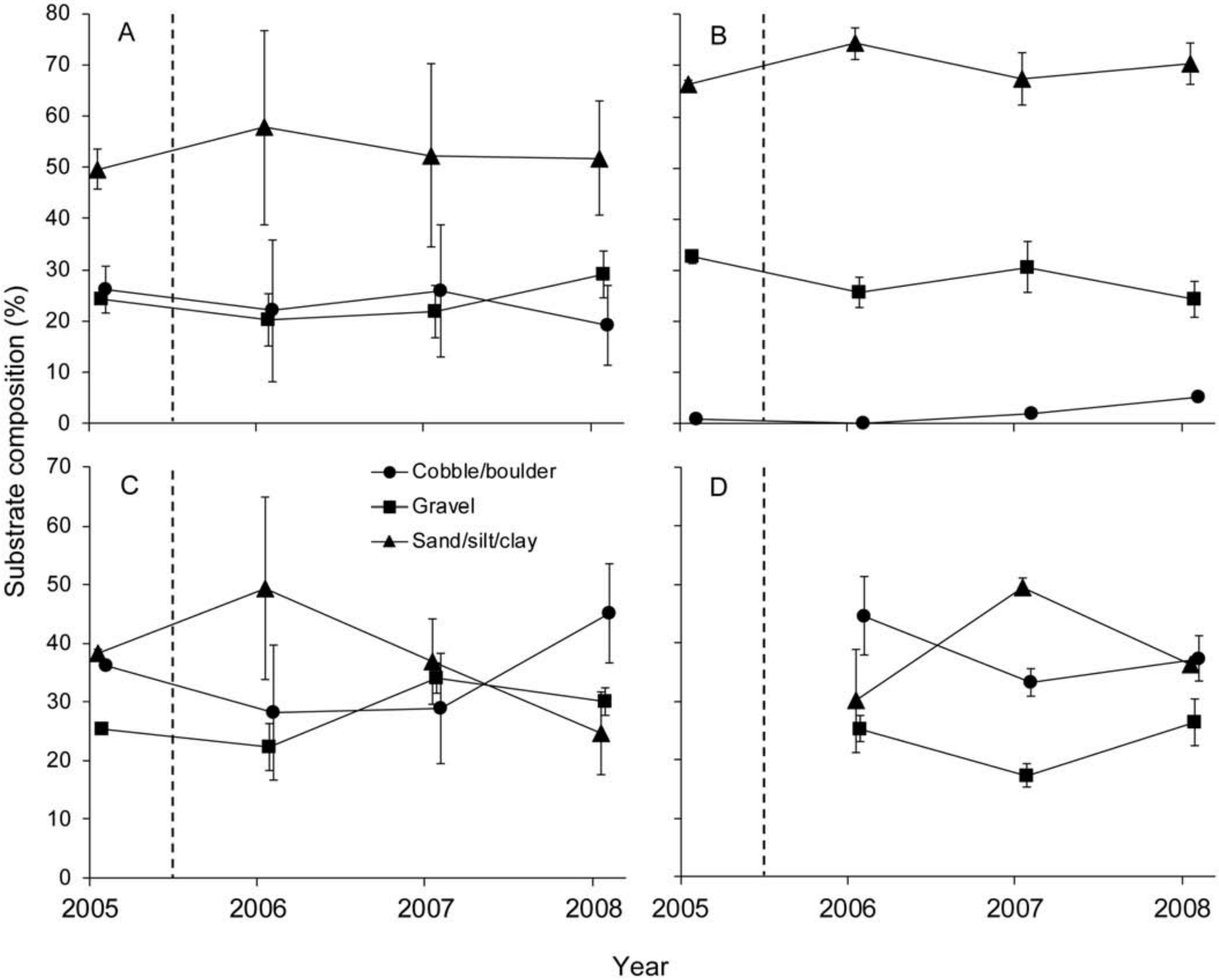

FIGURE 3

Mean (among sampling transects) percentage (± SE) of each sediment particle size class at the (A) upstream reference, (B) downstream reference, (C) tailrace, and (D) impoundment sites on the Deep River, North Carolina. The vertical dashed line represents the date of removal of the Carbonton Dam. Only means were included in the BACI analyses.

At the impoundment site, no significant dam removal effects were detected by the BACI analysis for any mussel or substrate response variable. Significant control/impact effects ($P<0.05$) were evident for all mussel variables (except for total species richness) and in proportions of boulder/cobble ($P=0.0003$) and sand/silt/clay ($P=0.003$) indicating differences between the control and impact sites that did not change after removal of the dam (see Figs. 2 and 3). We detected significant location effects (nested within control/impact, $P<0.05$) in all mussel variables (except for total species richness) and in boulder/cobble ($P=0.001$) and sand/silt/clay ($P=0.02$), indicating differences between the two control sites. During the drawdown process, only a few individuals (<10 total) of Eastern Elliptio and Paper Pondshell, *Utterbackia imbecillis*, were observed stranded on the newly exposed banks at the impoundment site.

Mussel survival

The mean recapture rate of Eastern Elliptio ranged from 12.8% (SD=2.3) at the tailrace site to 24.5% (SD=11.6) at the impoundment site. Cormack-Jolly-Seber capture probabilities among years ranged 17-30% at the tailrace site, 34-52% at the downstream reference site, 19-34% at the upstream reference site, and 30-34% at the impoundment site. Annual apparent survival of Eastern Elliptio at the tailrace site was similar over time, including the interval that spanned dam removal and all subsequent intervals. Apparent survival at the tailrace site was similar to control sites and confidence intervals for all of these estimates overlapped broadly (Table 3). Survival estimates at the impoundment site for all intervals and for all sites from 2007-2008 were not informative due to the wide confidence intervals, which resulted from the model sensitivity to the low number of years sampled.

DISCUSSION

We found little evidence of short-term effects of dam removal on the mussel assemblage in the Deep River after removal of Carbonton Dam. There were no detectable differences that were attributable to dam removal in abundance of all species, and individually for the three most abundant species, nor in Eastern Elliptio mean length or length distribution. Our estimates of Eastern Elliptio survival were lower than previously reported survival rates for this species in a free-flowing stream (Villella et al., 2004), but survival in our study did not differ between impact and control sites. Species richness was variable among years at all sites due to sampling error, but it remained highest at the tailrace site throughout the study.

Sediment erosion, transport, and deposition in downstream reaches are among the most important negative physical effects of dam removal (Heinz Center, 2002), and release of fine sediment stored in the impoundment was a potential risk in the removal of Carbonton Dam. At the tailrace and impoundment sites, fine sediment appeared to increase slightly the year after removal, as expected, but by the end of the study (3 years after removal) it had declined to levels comparable to, or lower than, those observed before dam removal. However, BACI analysis showed no significant effects of dam removal on substrate composition. Several features of Carbonton Dam or the removal process appear to have minimized negative effects of downstream sediment transport. The reservoir apparently stored a relatively low volume of sediment because of its run-of-river flow regime and because numerous upstream impoundments capture sediment prior to its reaching the dam. Coarser materials, mostly sand and gravel, as well as woody debris, had accumulated immediately upstream of the dam, but these materials were graded with heavy equipment after drawdown to form a bench along one side of the river so that the material would erode more slowly (Restoration Systems and Ecoscience Corporation, 2006). The majority of the flow at the tailrace site during the drawdown and removal process was directed to the side of the river opposite from the diverse mussel bed that we monitored during this study, and this likely buffered the mussel bed from sediment deposition as well as scour. Also contributing to the lack of short-term adverse effects on mussels was a lack of organic and inorganic contaminant accumulation in the sediments stored in the impoundment (USFWS, 2005; Hewitt et al., 2006).

Stranding of mussels in dewatered impoundments can result in high mortality, in some cases affecting imperiled species (Nedeau et al., 2000). The reservoir behind Carbonton Dam was confined within the banks of the river, which prevented vast areas of the impoundment bottom from being exposed during drawdown. Therefore, stranding and aerial exposure of large numbers of mussels during impoundment draining was minimized — a result we verified through multiple qualitative observations throughout the drawdown process. Nevertheless, we observed a decline from 5 to 2 species in the dewatered impoundment over time; one of those undetected species (Eastern Floater, *Pyganodon cataracta*) is adapted to lentic environments. Changes in mussel assemblages in dewatered reservoirs after dam removal may be unavoidable as habitats revert from lentic to lotic characteristics, but we may expect additional stream species to colonize these restored habitats in the future. Because most imperiled species are dependent on lotic habitats, increases in habitat availability for these species can offset negative effects to previous reservoir assemblages.

To our knowledge, the only other study that addressed the effects of dam removal on unionoid mussels is Sethi et al. (2004), who reported substantial mussel mortality in both the former impoundment and tailrace reach after removal of Rockdale Dam on Koshkonong Creek, Wisconsin. Rockdale Dam appeared to have stored much larger amounts of sediment than Carbonton Dam, and it was dewatered rapidly (36 h), exposing large areas of substrate in the former reservoir and resulting in stranding and mortality of mussels. Moreover, downstream habitats were inundated with sediment as the newly forming river channel mobilized material in the former Rockdale impoundment. Three years after dam removal, mussel density downstream of Rockdale Dam had declined by about 32%.

Sethi et al. (2004) recommended that negative effects of dam removals could be minimized by a slow drawdown period (i.e., months to years) that would allow mussels to migrate with decreasing water levels and allow stabilization of reservoir sediments. Even though the impoundment above Carbonton Dam had a limited littoral zone and apparently held less sediment, the slower dewatering process (3 weeks) likely minimized mussel stranding in the former reservoir and sedimentation or scouring of downstream habitats. However, the effectiveness of a gradual drawdown on reducing mussel stranding and mortality may vary among species according to differences in mobility and burrowing behavior (Gough et al., 2012). Timing of dam removal also may influence the potential for negative effects. Rockdale Dam was dewatered in early September (Sethi et al. 2004), when water temperatures presumably remained high. Removal of Carbonton Dam in the fall-winter may have further reduced mussel mortality by minimizing heat and oxygen stress. Because of variation in factors such as dam configuration and river and impoundment morphology, the optimal timing and methods for reservoir dewatering and dam removal will be case-specific. Potential toxicity of sediments stored in the impoundment also is a critical factor in assessing potential negative effects of dam removal (USBR, 2006; Cope et al., 2008). Consideration of these variables during the planning and execution of future dam removals is necessary to ensure the best possible outcome for aquatic biota.

Positive effects of existing dams also should be evaluated prior to removal. Despite their many negative aspects, in some cases small dams may enhance downstream habitats by trapping toxicants or sediments or by increasing downstream oxygen and food concentration (Gangloff et al., 2011; Singer & Gangloff, 2011), and these reaches often support dense mussel assemblages or rare species, as we observed in the Deep River. In dam removal planning, consideration of potential short-term negative effects of dam removal on localized populations (e.g., in tailrace or impoundment reaches) should to be weighed against long-term negative effects of population fragmentation at the watershed scale. Even though we observed no short-term negative effects of dam removal in our study, we also detected no evidence of recolonization of the former impoundment or increased density downstream in this time frame. Because of the slow growth and low recruitment rates of many mussel species, such responses may not be evident for several years or even decades. Long-term monitoring of effects of dam removal on mussel populations has not occurred, but this is vital to assess the full benefits or risks of this conservation strategy.

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TABLE 1

Mean density (number/m^2) across years (and range) and cumulative species richness among all years at four study sites on the Deep River, North Carolina, from 2005-2008 and respective state conservation status in North Carolina (T = threatened, E = endangered, SR= significantly rare).

		Study site			
Species	Conservation status	Upstream reference	Impoundment	Tailrace	Downstream reference
Alasmidonta undulata Triangle Floater	T	0.003 (0-0.013)	0.003 (0-0.006)	0.013 (0.003-0.033)	0.017 (0.008-0.042)
Elliptio complanata Eastern Elliptio		2.67 (1.71-4.12)	0.541 (0.517-0.574)	1.69 (1.30-2.50)	8.77 (5.76-11.56)
Elliptio icterina Variable Spike		0.017 (0-0.054)			
Elliptio roanokensis Roanoke Slabshell	T	0.008 (0-0.033)		0.028 (0.015-0.043)	0.036 (0-0.067)
Elliptio sp. lance group		0.074 (0.013-0.174)		0.014 (0-0.036)	0.004 (0-0.017)
Lampsilis cariosa Yellow Lampmussel	E	0.024 (0-0.040)		0.009 (0-0.033)	
Pyganodon cataracta Eastern Floater			0.020 (0-0.040)	0.014 (0-0.037)	0.017 (0-0.025)
Strophitus undulatus Creeper	T			0.002 (0-0.003)	0.015 (0-0.025)
Uniomerus carolinianus Eastern Pondhorn		0.010 (0-0.027)	0.017 (0.007-0.028)	0.012 (0.003-0.033)	0.496 (0.372-0.584)
Utterbackia imbecillis Paper Pondshell				0.001 (0-0.003)	
Villosa delumbis Eastern Creekshell	SR	0.015 (0-0.033)	0.005 (0-0.007)	0.123 (0.094-0.143)	0.356 (0.190-0.510)
Total richness		8	5	10	8

TABLE 2

Pairwise comparisons of Eastern Elliptio (*Elliptio complanata*) length distributions before (2005) and after (2006–2008) dam removal at three study sites on the Deep River, North Carolina (impoundment not included due to lack of pre-removal data). Statistics are the Kolmogorov-Smirnov maximum difference in cumulative frequencies (D) and associated probability (P) of the paired distributions coming from the same population.

Years compared	Study site: Upstream reference D	P	Tailrace D	P	Downstream reference D	P
2005, 2006	0.1132	0.0960	0.0903	0.0281	0.1429	<0.0001
2005, 2007	0.1375	0.0773	0.0973	0.0362	0.1038	0.0012
2005, 2008	0.0965	0.4928	0.0746	0.1976	0.1665	<0.0001

TABLE 3

Estimated apparent survival and the 95% confidence interval (Cormack-Jolly-Seber model) of Eastern Elliptio (*Elliptio complanata*) between sampling dates from 2005-2008 at four study sites on the Deep River, North Carolina.

Study site	2005-2006	2006-2007	2007-2008
Upstream reference	0.77 (0.51-0.91)	0.80 (0.46-0.95)	0.68 (0-1)
Impoundment		0.95 (0-1)	0.61 (0-1)
Tailrace	0.69 (0.58-0.78)	0.72 (0.54-0.85)	0.64 (0-1)
Downstream reference	0.68 (0.61-0.73)	0.67 (0.61-0.73)	0.64 (0-1)