In: Advances in Environmental Research. Volume 14 ISBN 978-1-61209-584-4
Editor: Justin A. Daniels, pp. 351–364

Chapter 13

UNIONID MUSSEL MORTALITY FROM HABITAT LOSS IN THE SALMON RIVER, NEW YORK, FOLLOWING DAM REMOVAL

John E. Cooper
Cooper Environmental Research
1444 County Route 23, Constantia, New York 13044, USA

ABSTRACT

Habitat loss generally involves alteration of the environment by the physical additions of bridges and dams, or removal of resources, such as dredging or timber cutting, that result in the loss of specific conditions that were required by aquatic organisms. Habitat alteration can also be caused by the removal of past alterations, which can result in the altered habitat being perturbed again with a concomitant loss of specific habitat conditions. The Fort Covington Dam was removed from the Salmon River in June, 2009, nearly 100 years after construction. The draining of the 1.3-hectare reservoir required only 25 hours and resulted in a lowering of the water level by 47 cm at the reservoir center (3.3 m at the dam). The rapid draining of the reservoir stranded, and subsequently killed, more than 2800 unionid mussels of eight species along the shorelines and in two adjacent ponds. Loss of shallow-water river habitat was estimated to be 30%, with a 66% loss of habitat in the ponds. The removal of the dam increased water velocity from 12 cm/s to 34 cm/s during low flow, and scoured sand deposits from the upstream riffle, shorelines, and sand bars that had formed in the reservoir. The volume of scoured sand deposited within the river 5 months after dam removal was estimated at 42,480 m^3, which covered 1097 m of the river bottom to a depth of up to 1.5 m. Continued scouring has resulted in deposition in downstream portions of the river, leaving steep-sided shorelines with unstable coarse sand as the primary mussel habitat. Dam removal has been utilized in restoring rivers to a more natural state but can have unintended consequences: in this case the sudden reduction in the mussel population.

INTRODUCTION

Unionid mussel populations have declined dramatically over the past 30 years primarily through habitat loss and degradation associated with human-induced alterations. Dams have been cited as one of the more prominent agents of habitat change resulting in a loss of up to 60% of the associated mussel fauna (Layzer et al. 1993; Williams et al. 1993; Strayer 2008).

Dam removal has become a useful tool for habitat restoration and is likely to be employed more often as many dams are reaching the end of their useful life (>50 years). The financial cost to upgrade deteriorating dams generally exceeds the cost of removal. More than 500 dams have been removed in the United States over the past century (Thomson et al. 2005) and the trend is expected to increase, although primarily with small dams (<5m height; Poff and Hart 2002). Reconnection of fragmented river sections will increase migration of fishes, some of which are essential as hosts for larval stages (glochidia) of mussels, thereby facilitating mussel reproduction if suitable habitats are present.

Mussel relocation has been used to mitigate the effects of structural alterations (bridge construction, dam removal) but with relatively low mussel survival (Cope and Waller 1995) in early attempts. Advances in technique and selection of relocation sites since 1994 have increased mussel survival to >97% (Cope et al. 2003).

Lower energy habitats (lesser slope) would require a longer recovery period than those habitats that have greater slope and energy. Habitats that are altered by sand deposition would require greater energy for sand removal than those habitats altered by silt since moving coarse-grained deposits would require greater water velocity (Doyle and Harbor 2003), which is likely only in extreme rain events or from snowmelt.

Unionid mussels have developed several behavioral adaptations to changing water level, such as drought (McMahon 1991), but have no defense against permanent emersion. The dewatering of a reservoir has obvious consequences for mussels (Haag and Warren 2008) especially where dam removal eliminates the potential for restoring previous water levels.

STUDY SITE

The Fort Covington Dam was an abandoned run-of-river dam located on the first riffle of the Salmon River, approximately 8 km upriver from its confluence with the St. Lawrence River, and was one of eight dams that blocked the Salmon and Little Salmon rivers (Figure 1). The Fort Covington Dam had been used for hydroelectric power and as a gristmill. The original dam was built in the late 1800s as a wood crib structure but was damaged in a freshet in 1911. It was rebuilt in 1912 as a low-head concrete gravity dam (spillway 27 m in length, 3 m high). Concrete abutments were present at both ends of the dam, which housed discharge ports. An engineering survey of the dam in 2002 revealed that the ports were partially blocked with debris and the remains of the stop logs, with many cracks and spalled areas in the abutments, spillway, and gate areas (Milone and MacBroom 2004). The dam created a reservoir extending approximately 1600 m upstream. The river width ranged from 20 to 70 m (mean = 30 m) prior to dam removal. The channel bed was flat with a bankfull depth of 1.2 m (surface area of 4 to 6 ha, not including ponds) with a distinct thalweg at the outside of bends.

The Salmon and Little Salmon rivers are 4th-order and have primarily coarse sand-cobble/boulder substrate in the riffles and sand-silt substrate in the glides. Water discharge in the Salmon and Little Salmon rivers is flashy and responds rapidly to inputs of precipitation: for example, flow increased from 0.7 to 14 m^3/s over a 12-hour period from a single rain event. Mean daily streamflow can reach 93 m^3/sec (USGS gage 04270000). Flooding occurred upstream of the dam when ice floes collected at the dam, and occurred downstream of the dam site in 2009 due to an ice jam during the first winter after dam removal.

Two ponds were created by the damming of the river (Figure 2). The east pond was fed by a small stream and was shallow (<1m depth). The west pond was fed by surface runoff and was deeper (2.5 m) at the downstream end.

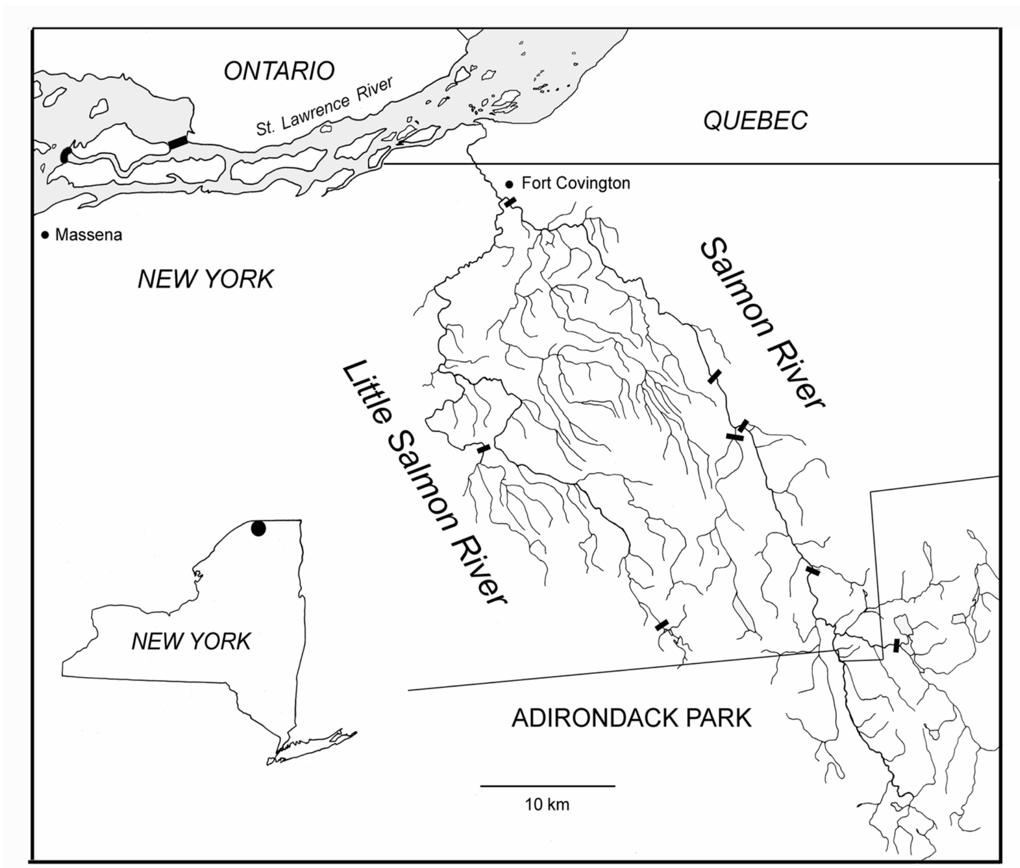

Figure 1. Watershed of the Salmon and Little Salmon rivers. The Fort Covington Dam (black rectangle) is the most downstream dam. The Long Sault Dam (left) and Robert Moses Power Dam (right) are in the St. Lawrence River.

The Fort Covington Dam was removed in June, 2009, nearly 100 years after construction. A cobble apron was constructed on both sides of the river to allow access by a hydraulic hammer and excavator to remove sections of the concrete gates and dam. The cobble apron was then removed and both banks were graded and stabilized with boulders.

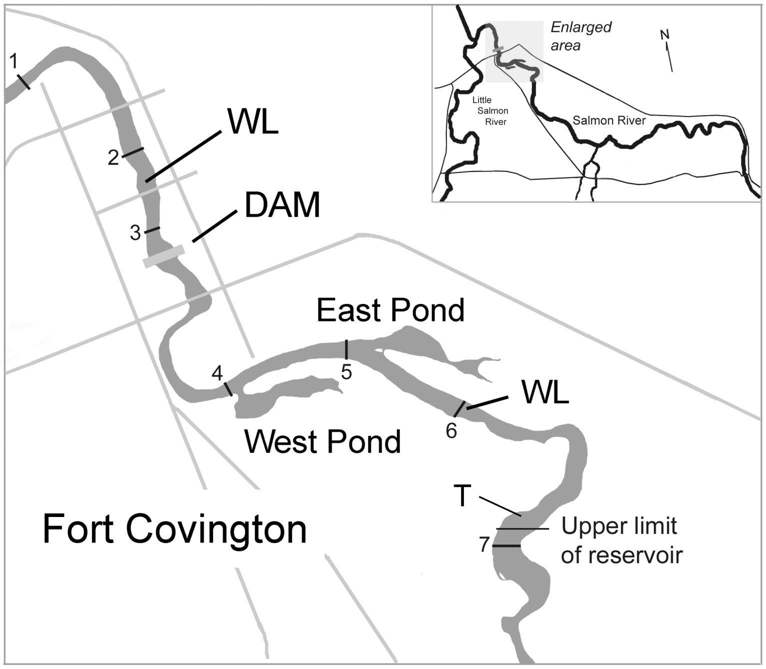

Figure 2. Salmon River reservoir area as it appeared prior to dam removal. The river flows through the village of Fort Covington. Numbered lines are transects from the pre-removal study; WL and T represent the sensors used to collect water level and temperature data. The inset map shows the study area used in the pre- and post-removal studies. Roads are depicted as gray lines (not all roads are shown).

Data Collection

The Salmon River was used as the experimental river and the Little Salmon River as a control in the dam removal assessment: no effect was expected from dam removal on the Little Salmon River. Fifteen transects were established, 9 in the Salmon River, and 6 in the Little Salmon River, divided between riffles and glides. Transects were paired across rivers with the exception of transects 1 through 3, which did not have analogous reaches in the Little Salmon River (Cooper et al. 2004). Sampling of sediment, macroinvertebrates (primarily insects), fish, and plants was started in 2002 and ended in 2004, with the expectation that the dam would be removed in 2005. However, the dam was not removed until June 2009. The intervening years were used to estimate the unionid mussel population within the study area. Systematic sampling of mussels (a probability-based design; Strayer and Smith 2003) was done at 10 transects from 2005 through 2008. Forty quadrats of 0.25 m^2 were sampled at each transect using visual and tactile methods. Ten percent of the glide quadrats were excavated to a depth of 20 cm. Population estimates of reservoir mussels were made using transects 5 and 7 (Figure 2).

Water level and temperature was monitored hourly from 6 May to 20 October 2009, using automatic recorders at transect 6 in the reservoir, and downstream of the dam at transect 3. Water temperature data was also recorded hourly at transect 7. Each water level sensor was hung freely in a 3.8 cm-dia PVC pipe. Water level data was corrected for barometric pressure by a separate sensor. Water velocity was estimated at transect 5 (N = 7) using a mini current meter.

Sediment constituents (sand, silt, clay) were determined from three composite samples taken by ponar dredge at glide transects (Cooper et al. 2004) and expressed as percent dry weight (Folk 1980). The extent of sand migration after dam removal was estimated visually each month and plotted on aerial photographs taken in 2004 (NYSDEC).

The reservoir shoreline was divided into 13 sections of arbitrary length to estimate sand surface area, shoreline surface area, and to make mussel counts. Sand surface and shoreline surface area were estimated using Adobe Photoshop by relating pixel number within colored polygons of estimated sand coverage, or exposed shoreline area, to the number of pixels contained in known areas measured in the river. The volume of sand was estimated by multiplying the area by the average depth of deposited sand. The exposed shoreline was defined as the width between the normal reservoir level (shown by a vegetation line) and that at low water after dam removal; the section width was then multiplied by the linear distance.

Field counts of mussels in the former reservoir were made on 7–8, 13–14, and 17 July, 2009. Mussel species were segregated into six categories (Table 1) according to their observed physical condition. Living mussels were defined as those that showed a response to an attempt to force the valves open. Living mussels were returned to the water.

RESULTS

The opening of the dam lowered the reservoir water level by 47 cm over a 25–hr period (Figure 3). Changes in water level were correlated between transect 3 and transect 6 prior to the opening of the dam (6 May to 24 June; Spearman R = 0.91) but less so after dam removal (19 August to 20 October; Spearman R = 0.41) as water level continued to decline within the reservoir. The sensors were affected by sand deposition from 24 June to 19 August after which they were relocated. Relative water level recorded at transect 3 was increased artificially by the deposition of sand, which raised the water level approximately 0.5 m at the sensor. Average water velocity increased at the center of the reservoir from 12 cm/s (pre-removal) to 34 cm/s (post-removal) at low discharge (0.5 m^3/sec).

Water temperature was similar between transect 3 and transect 6 (Spearman R = 0.99) for the periods of 5 May to 16 July and 15 August to 20 October. Deposition of sand around the temperature sensor at transect 6 created an unplanned but useful estimate of sand temperature (Figure 4). The sensor became buried 23 cm under the sand surface and recorded progressive temperature extremes that would be similar to that experienced by stranded and buried mussels (9.7 to 38.1 °C). Spearman correlation of water temperature to air temperature at transect 6 from 17 July to 14 August was 0.70 compared to only 0.51 for water temperature correlation between transect 6 and transect 7 for the same period, which suggests that the sensor was recording more effect from air temperature than water temperature.

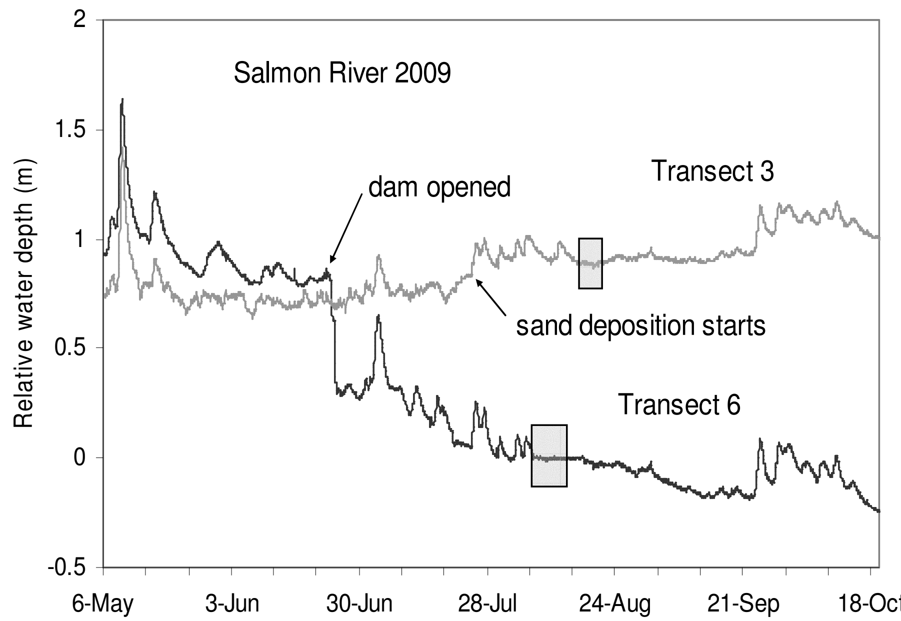

Figure 3. Relative water level in the Salmon River reservoir in 2009. Sand deposition affected the water level sensor readings (shaded boxes), which also increased the relative water level at Transect 3. Erosion of the sand sediment was accelerated by 2.7 cm of rain from 1–3 July.

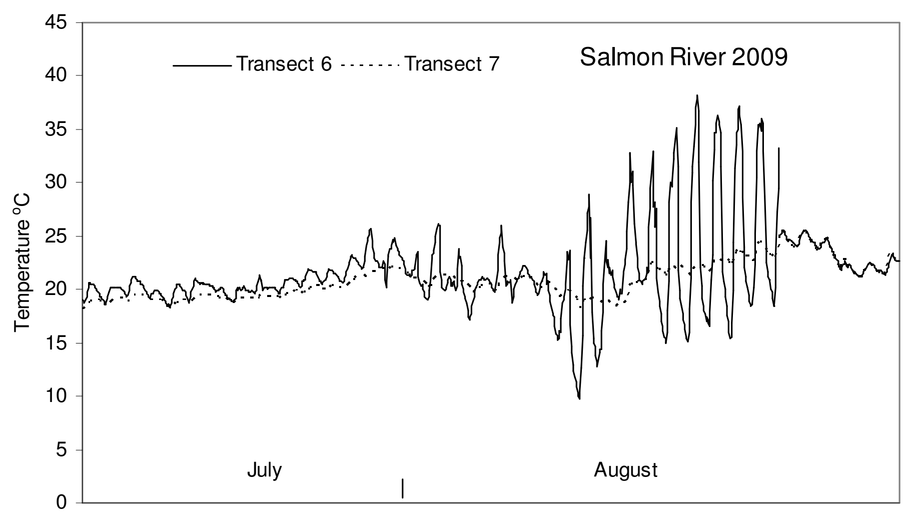

Figure 4. Comparison of water temperature at transect 6 and transect 7 during the period of 16 July to 22 August, 2009. Sand displaced the water within the pipe holding the sensor until only sand temperature was recorded. The sensor was relocated on 19 August.

Sand Movement

The volume of redistributed sand was estimated to be 42,480 m^3 in November 2009. Sand movement into the reservoir from the transect 7 riffle was about 20% of the total and moved a distance of 335 m (equivalent to 3.8 m/day). The remainder of the sand was from existing sandbars and shoreline deposits, which moved 762 m out of the reservoir (equivalent to 8.7 m/day), accumulating to a depth of up to 2.7 m. Sand deposits moved an additional 295 m downstream by June, 2010, reducing the average lower river depth from 2.7 m to about 0.5 m. Sand removal expanded the transect 3 riffle in an upstream direction by 107 m, and the riffle at transect 7 expanded downstream by 55 m. A riffle was exposed between transects 6 and 7 in spring 2010 (Figure 5). Water depth at transect 2 was reduced from 3 m to 0.4 m, and reduced at transect 3 from 0.5 m to 0.1 m. Sand deposition continued downstream beyond the confluence of the Salmon and Little Salmon rivers reducing the average depth from 5 m to <1 m by May, 2010. A sandbar was created across the mouth of the Little Salmon River, which was removed by high water in October, 2010.

Habitat Changes

The width of the exposed shoreline habitat ranged from 0.5 m at the upper end of the reservoir to 3.3 m at the dam. The loss of shoreline habitat within the reservoir was estimated to be 3,963 m^2, which was about 30% of the reservoir surface area. The former mussel habitat in the reservoir was primarily along the shorelines: 83% of all living mussels collected from 2005 to 2008 were within 3 m of either shore. The former mussel habitat was composed of 54–80% sand, 16–31% silt, and 6–14% clay compared with the channel that was 99% sand and <1% silt. The channel now constitutes the greatest proportion of habitat. Several isolated sediment deposits were uncovered in the channel in spring, 2010, which are more similar in sediment fractions to the former habitat (81% sand, 11% silt, and 8% clay) but experience about three times greater flow velocity.

Mussel Mortality

Eight of twelve species of mussels known from the Salmon River (Cooper, unpublished data) were stranded within the former reservoir (Table 1). Those species not stranded were *Alasmidonta marginata*, *A. undulata*, *Lampsilis ovata*, and *Lasmigona costata*. Mussels were collected from 80% of the reservoir shoreline upstream from the Route 37 bridge; the remainder of the shoreline was too steep and unstable to be searched. The majority (74%) of all mussels collected were stranded between transects 4 and 6. Shell lengths of stranded mussels ranged from 17.5 to 128 mm (median = 62 mm). More than 400 living mussels were relocated from drying sediments to open water during the survey period.

The reservoir mussel population was estimated to be 3,808 (95% CI = 3,681 to 3,940). The number dead (DS and RD from Table 1) was 2,954 or about 77% of the estimated population. No estimate could be made of the mussels covered by sand. Mussel mortality was also observed in two adjacent ponds (Table 2) that lost water after the dam was removed. The pond on the east side of the river was drained of nearly all water by August 2009 (except for stream input) and accounted for 76% of the dead mussels counted from the ponds.

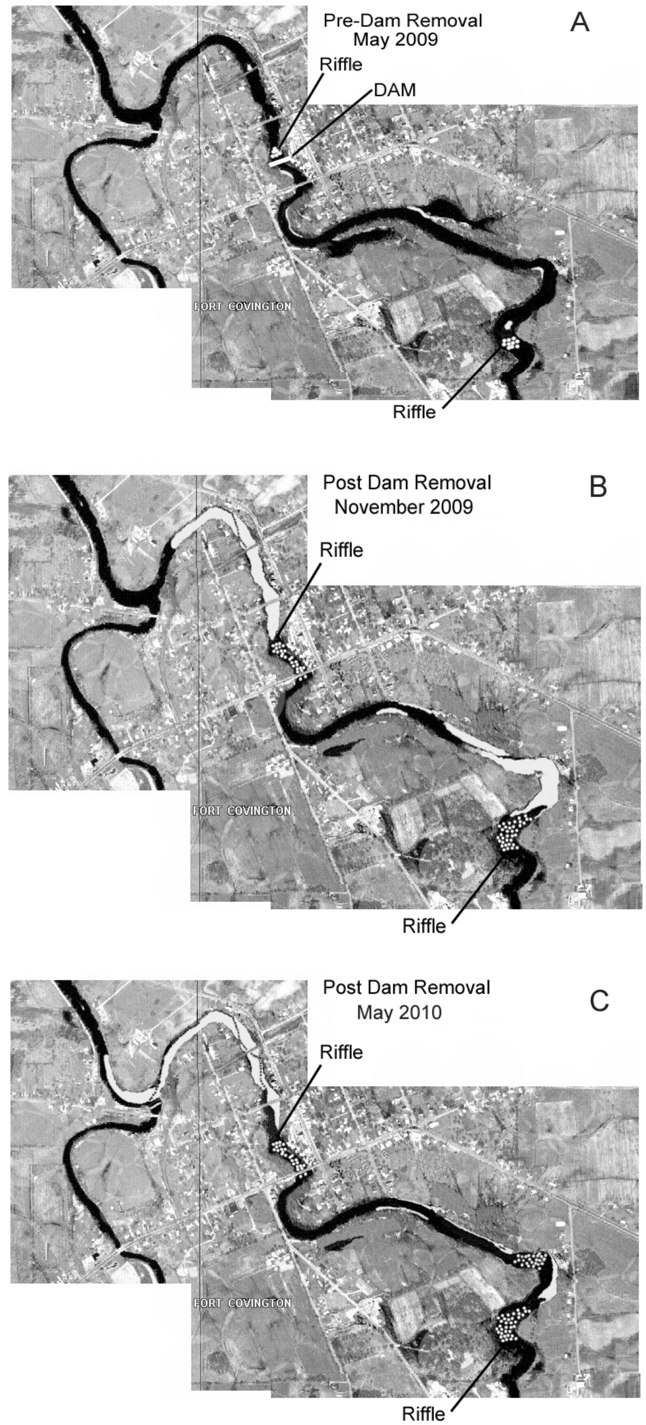

Figure 5. Sand deposition movement in the Salmon River from pre-removal, 2009, (A) through post-removal in November (B) and May, 2010 (C). Existing sandbars are shown in white (A) as is re-distributed sand in B and C. Riffles are shown as white dots. The horizontal distance across each panel is 2297 m.

Table 1. Mortality estimate of eight species of mussels from the Salmon River reservoir shoreline in 2009. Total mortality is the sum of categories DS and RD. Key to categories: RE = relic shells; DS = dead stranded, shells with tissue remaining inside; RD = recently dead, no tissue but no evidence of internal erosion; REP = relic shells with evidence of predation; DSP = dead stranded shells with evidence of predation; RDP = recently dead shells with evidence of predation. Shells classified as 'P' are not included in the mortality estimate.

	Category						
Species	RE	DS	RD	REP	DSP	RDP	Total
Elliptio complanata	483	1259	1287	34	22	54	3139
Lampsilis radiata	8	24	14	0	2	1	49
Strophitus undulatus	3	18	30	0	0	0	51
Pyganodon cataracta	6	12	16	0	0	2	36
Pyganodon grandis	0	0	1	0	0	0	1
Pyganodon sp.	0	0	2	0	0	0	2
Lasmigona compressa	0	0	1	0	0	0	1
Lampsilis cariosa	0	2	0	0	0	0	2
Anodontoides ferussascianus	0	3	0	0	0	0	3
Total	500	1318	1351	34	24	57	3284

Table 2. Mortality estimate of four species of mussels from the east and west ponds in the Salmon River in 2009.

Species	RE	DS	RD	REP	DSP	RDP	total
Elliptio complanata	0	18	1	0	1	0	20
Lampsilis radiata	0	1	1	0	0	0	2
Pyganodon cataracta	25	96	87	1	6	9	224
Pyganodon grandis	0	15	0	0	0	4	19
Total	25	130	89	1	7	13	265

The west side pond was reduced by about half in surface area by August 2009 but retained water in the deeper downstream end until September 2010. No estimate was made of the mussel population in the ponds.

CONCLUSIONS

Habitat Changes

Correlation of upriver and downriver water temperature was nearly 100% showing that the presence of the dam did not alter the water temperature profile of the downriver habitats. This would be expected from a run-of-river dam in contrast with hypolimnion releases where coldwater discharges reduce downriver temperature (Layzer et al. 1993). The effect of decreasing water level within the reservoir was evident in the increasing variation in water temperature at transect 6.

The effects of dam removal on stream morphology include increasing the slope in the former reservoir and shifting of channels (Stanley et al. 2002), both of which have occurred in the Salmon River. The final channel morphology will not be apparent for many years. Opening the dam resulted in exposure and drying of the shoreline habitat and redistribution of eroded sand, which alternately covered and uncovered the existing habitat as the sand moved downstream. The erosion of the shorelines was accelerated by a rain event but this material would likely have eroded over time without the rain by the undercutting of banks. Water velocity within the reservoir was increased at low flow by a factor of three and any additional input would magnify the erosion along the shorelines. Erosion was also accelerated during the following spring snowmelt. The redistributed sand represented nearly 100 years of accumulation since river flow rates were not great enough to carry the sand over the dam.

Mussel Mortality

Rapid dewatering of the reservoir was a contributing factor in the mortality of mussels due to stranding, dessication, and the facilitation of predation. Similar mortality effects were described in the dewatering of the reservoir in Koshkonong Creek, Wisconsin (Sethi et al. 2004). The preferred location of mussels close to the shoreline in the Salmon River was similar to that found by Brim Box et al. (2002) in southeastern US rivers. Larger streams maintain base flow and only those mussels at the stream margins are affected by emersion. In the Salmon River, most mussels live at the margins of glides so a greater percentage was affected even though base flow was maintained. Mussels move in apparent response to changing water levels and temperature (Amyot and Downing 1997; Schwalb and Pusch 2007; Cyr 2009) associated with reproduction and season. Average movement is considerable compared to body lengths (4.2 to 11 cm/wk) but would have been much too slow to adjust to dewatering of the Salmon River reservoir: mussels would have needed to move at a rate equivalent to 84 cm/wk to remain immersed. Some mussel species can survive for long periods (>40 days) if the relative humidity remains greater than 60% but survival declines rapidly at less than 55% (Deitz 1974). The relative humidity in the former reservoir area was >64% during July, August, and September 2009 (NOAA Climate Data), which might have prolonged survival affording the stranded mussels the opportunity to move but no indication of movement was observed. Mussels have behavioral mechanisms to deal with drought conditions such as decreased metabolism, burrowing, and anaerobic respiration (McMahon 1991; Golloday et al. 2004) but not with permanent emersion. Mussels are constrained by behavioral responses to drought in the absence of movement, thus mortality increases when declining water level does not increase later. Mortality also increases as water temperature (inside of valves) increases as water level declines and air and sediment temperatures rise. Although some mussel species can survive emersion of up to an hour at 35 °C (Bartsch et al. 2000), the mussels in the Salmon River were exposed to temperatures of >35 °C for periods of 2 to 6 hours on five occasions. Habitat characteristics can act as a physical trap: compacted soil due to loss of pore water (sediment hardening) was evident during the mussel counts, and it is unlikely that these mussels could have forced their way out of the dry sediment. The inability of mussels to cross over cobble and boulders could also lead to stranding. Stranding was observed in several riffles in the Salmon and Little Salmon rivers in previous surveys where decreasing water level left mussels behind boulders.

The increase in water velocity combined with the unstable nature of the channel sediment would reduce the suitability of the substrate for colonization by mussels (Strayer 2008). There is potential for the channel habitat to improve for mussels in those sediment deposits that are more similar to the previous shoreline habitat, and where areas of reduced flow develop, such as downstream of downed trees or other obstructions.

Mussel habitat might increase in the lower velocity areas of riffles, particularly at transects 3 and 7, and in the riffle between transects 6 and 7. These gains in habitat in the riffles will be offset, in the short term, by the loss of the former glide habitat at transect 1 that was covered by sand to a depth of 2.7 m: the ultimate fate of this previously rocky habitat, as well as those habitats downriver, will depend on water velocity sufficient to remove the sand. The continued movement of the downriver sand will gradually cover the existing habitat, a process that might continue for many years.

Habitat loss has been mitigated by relocating mussels to suitable habitat. Some early relocation efforts resulted in low mussel survival but successful relocations were described by Waller et al. (1995), Havlik 1997, and Dunn and Seitman (1997). A more recent study has emphasized *in situ* relocations to reduce mortality (Cope et al. 2003), but moving mussels to open water that is undergoing changes in sediment and velocity might result in burial or additional stress. Some unionids can survive burial by moving upward while other species remain in position and die (Brim Box and Mossa 1999). Moving them upstream to occupied habitats might also be a solution as increasing the density of mussels was found to not affect survival of the relocated or previous populations (Cope et al. 2003, Havlik 1997). The question arises as to why relocation was not attempted in the Salmon River? A relocation program would require planning with appropriate funding and personnel, none of which were available. Coordination between those doing the dam removal and those who could relocate mussels was also lacking; the dam removal contract allowed for altering the date of reservoir dewatering, which occurred one month earlier than anticipated.

The creation of the reservoir might have allowed the mussel population to increase by providing suitable habitat (creation of ponds, retention of silt, decreased water velocity) to a level that was found in only a few glides elsewhere within the study area. The creation of the ponds might have allowed the expansion of the *Pyganodon* spp. population; fewer *Pyganodon* were collected in other areas of the rivers. The collection of *P. grandis*, a species not reported from the Salmon River previously, would indicate that this species has been present since the dam was constructed but not noticed. This species is known from Quebec, Canada, on the north side of the St. Lawrence River (Clarke 1981). To what extent the loss of *P. grandis* will do to the reproducing population is unknown. The mussel population that results after dam removal might be a closer approximation of the mussel population prior to dam construction. Upriver glides were characterized by narrow, silty, low-velocity shorelines and coarse sand substrate in the channel, which the former reservoir might come to approximate in future years.

At present, shorelines continue to erode with steep banks being undercut, alternately creating and removing sand deposits, resulting in an unstable substrate, which could restrict mussel repopulation. Migration of fish hosts is no longer constrained and might result in more successful reproduction when substrates become more stable, although no evidence of constraint was found in the pre-removal study (Cooper et al. 2004). The removal of the dam was done partly to reconnect two segments of the Salmon River with the expectation that a more natural environment would result, at least in the lower river. The fulfillment of this expectation will depend on time and great enough water flow to remove the sand deposits, which were not anticipated. One immediate effect of dam removal has been the sudden reduction of the mussel population in the former reservoir and downriver habitats. The long life and slow reproductive rate of mussels will likely mask any perceptible recovery in the former reservoir for many years, and perhaps decades.

ACKNOWLEDGEMENTS

I would like to thank David L. Strayer and Kurt J. Jirka for their assistance in confirming the identity of several mussel species.

REFERENCES

Amyot, J-P. and J.A. Downing. 1997. Seasonal variation in vertical and horizontal movement of the freshwater bivalve *Elliptio complanata* (Mollusca: Unionidae). *Freshwater Biology* 37:345–354.

Bartsch, M.R., D.L. Waller, W.G. Cope, and S. Gutreuter. 2000. Emersion and thermal tolerances of three species of unionid mussels: survival and behavioral effects. *Journal of Shellfish Research* 19:233–240.

Brim Box, J. and J. Mossa. 1999. Sediment, land use, and freshwater mussels: prospects and problems. *Journal North American Benthological Society* 18:99–117.

Brim Box, J., R.M. Dorazio, and W.D. Liddell. 2002. Relationship between streambed substrate characteristics and freshwater mussels (Bivalvia: Unionidae) in coastal plain streams. *Journal North American Benthological Society* 21:253–260.

Clarke, A.H. The freshwater mollusks of Canada. National Museums of Canada, Ottawa. 1981. 446 p.

Cooper, J.E., J.M. Farrell, and J.A. Toner. 2004. Predicting the effects of dam removal on aquatic communities in the Salmon River, New York. Final Report Grant 671. SUNY-ESF, Syracuse, NY.

Cope, W.G. and D.L. Waller. 1995. Evaluation of freshwater mussel relocation as a conservation and management strategy. *Regulated Rivers: Research and Management* 11:147–155.

Cope, W.G., M.C. Hove, D.L. Waller, D.J. Hornbach, M.R. Bartsch, L.A. Cunningham, H.L. Dunn, and A.R. Kapuscinski. 2003. Evaluation of relocation of unionid mussels to *in situ* refugia. *Journal of Molluscan Studies* 69:27–34.

Cyr, H. 2009. Substrate and fetch affect the emergence of freshwater mussels from lake sediments. *Journal North American Benthological Society* 28:319–330.

Deitz, T.H. 1974. Body fluid composition and aerial oxygen consumption in the freshwater mussel, *Ligumia subrostrata* (Say): effects of dehydration and anoxic stress. *Biological Bulletin* 147:560–572.

Doyle, M.W. and J.M. Harbor. 2003. A scaling approximation of equilibrium time-scales for sand-bed and gravel-bed rivers responding to base-level lowering. *Geomorphology* 54:217–223.

Dunn, H.L. and B.E. Seitman. 1997. Guidelines used in four geographically diverse unionid relocations. In: K.S. Cummings, A.C. Buchanan, C.A. Mayer, and T.J. Naimo, editors.

Conservation and management of freshwater mussels II: initiatives for the future. Upper Mississippi River Conservation Committee, Rock Island, IL. 1997, p. 176–183.

Folk, R.L. 1980. Petrology of sedimentary rocks. Hemphill Publishing Company, Austin, TX. 184 p.

Golloday, S.W., P. Gagnon, M. Kearns, J.M. Battle, and D.W. Hicks. 2004. Response of freshwater mussel assemblages to a record drought in the Gulf Coastal Plain of southwestern Georgia. *Journal North American Benthological Society* 23:494–506.

Haag, W.R. and M.L. Warren, Jr. 2008. Effects of severe drought on freshwater mussel assemblages. *Transaction American Fisheries Society* 137:1165–1178.

Havlik, M.E. Are unionid mollusk translocations a viable mitigation technique? The Wolf River, Wisconsin, experience, 1992–1995. In: K.S. Cummings, A.C. Buchanan, C.A. Mayer, and T.J. Naimo, editors. Conservation and management of freshwater mussels II: initiatives for the future. Upper Mississippi River Conservation Committee, Rock Island, IL. 1997, p. 184–195.

Layzer, J.B., M.E. Gordon, and R.M. Anderson. 1993. Mussels: the forgotten fauna of regulated rivers. A case study of the Caney Fork River. *Regulated Rivers: Research and Management* 8:63–71.

McMahon, R. Mollusca: Bivalvia. In: J.H. Thorp and A.P. Covich, editors. Ecology and classification of North American freshwater invertebrates. San Diego, CA., Academic Press, Inc., 1991, p. 315–399.

Milone and MacBroom, Inc. 2004. Report of findings, Fort Covington Dam mitigation. Cheshire, CT. 75 p.

NOAA Climate Data. 2009. [downloaded 2010/8/27], available from www1.ncdc.noaa.gov.pub/.

NYSDEC. 2009. [downloaded 2009/10/22], available from www.nysgis.state.ny.us/.

Poff, N.L. and D.D. Hart. 2002. How dams vary and why it matters for the emerging science of dam removal. *BioScience* 52:659–668.

Schwalb, A.N. and M.T. Pusch.2007. Horizontal and vertical movements of unionid mussels in a lowland river. *Journal North American Benthological Society* 26:261–272.

Sethi, A.A., A.R. Selle, M.W. Doyle, E.H. Stanley, and H.E. Kitchel. 2004. Response of unionid mussels to dam removal in Koshkonong Creek, Wisconsin. *Hydrobiologia* 525:157–165.

Stanley, E.H., M.A. Luebke, M.W. Doyle, and D.W. Marshall. 2002. Short-term changes in channel form and macroinvertebrate communities following low-head dam removal. *Journal North American Benthological Society* 21:172–187.

Strayer, D.L. Freshwater Mussel Ecology. A multifactor approach to distribution and abundance. University of California Press, Berkeley, 2008, 204 pp.

Strayer, D.L. and D.R. Smith. A guide to sampling freshwater mussel populations. AFS Monograph 8, American Fisheries Society, Bethesda, MD, 2003. 103 pp.

Thomson, J. R., D. D. Hart, D. F. Charles, T. L. Nightengale, and D. M. Winter. 2005. Effects of removal of a small dam on downstream macroinvertebrate and algal assemblages in a Pennsylvania stream. *Journal North American Benthological Society* 24:192–207.

Waller, D.L., J.J. Rach, W.G. Cope, and G.A. Miller. 1995. Effects of handling and aerial exposure on the survival of unionid mussels. *Journal Freshwater Ecology* 10:199–207.

Williams, J.D., M.L. Warren, K.S. Cummings, J.L. Harris, and R.L. Neves. 1993. Conservation status of freshwater mussels of the United States and Canada. Fisheries 18:6–22.