Flaming Gorge Tailwater Aquatic Biota Monitoring Program, 1994-2005

Funding provided by:

U.S. Department of the Interior
Bureau of Reclamation
Upper Colorado Regional Office
125 South State Street
Salt Lake City, Utah 84138

Utah Division of Wildlife Resources
Flaming Gorge Project
Post Office Box 158
Dutch John, Utah 84023

Work performed by:

National Aquatic Monitoring Center
Department of Watershed Sciences
Utah State University
Logan, Utah 84322-5210

Report written by:

Mark R. Vinson, Eric C. Dinger, and Michelle Baker

May 2006
Data collection dates: January 1994 to September 2005
Laboratory sample processing dates: January 1994 to December 2005
Data analysis dates: January 2006 to April 2006
Draft report completed: January 2006
Final report completed: May 2006

Personnel involved and their primary responsibilities:

National Aquatic Monitoring Center
Department of Fisheries and Wildlife
Utah State University
Logan, UT 84322-53210
- Dave Axford, field data collection and laboratory sample processing
- Eric Dinger, data analysis and report writing
- Joe Kotynek, field data collection, laboratory sample processing
- Leslie Ogden, data entry
- Matt Tagg, field data collection and sample processing
- Erin Thompson, field data collection and sample processing
- Mark Vinson, overall project design, implementation, and report writing
- Tom Wauters, field data collection
- Chris Webster, field data collection, laboratory sample processing
- Dan Zamecnik, office administration

Biology Department
Utah State University
- Michelle Baker, New Zealand mud snail experiment

U.S. Forest Service
- Jeff Kershner, project design

Utah Division of Wildlife Resources
- Don Archer, project administration
- Steve Brayton, fish collections, logistics, project coordination
- Lowell Marthe, fish collections, logistics
- Roger Schneidervin, fish collections, logistics, project coordination

U.S. Bureau of Reclamation
- Tom Chart, Project administration and coordination
Acknowledgments
Financial and logistical support for this work was provided by the Utah Division of Wildlife Resources, the U.S. Bureau of Reclamation, the U.S. Forest Service, the U.S. Bureau of Land Management, and Utah State University. Ted Angradi provided considerable advice throughout the study. Utah State University students assisting with field and laboratory work over the years included Eric Billman, Greta Burkart, Brooke Bushman, Warren Collier, Nathan Etherington, Tarita Harju, Krista Keuster, Ben Kuhns, Greg Parry, Kaori Sasakura, and Andree’ Walker. We thank them all for their hard work and good spirits.
# Table of Contents

Executive summary ........................................................................................................... v

Chapter I. Flaming Gorge Tailwater Aquatic Biota Monitoring
Introduction ....................................................................................................................... 1
Study site ............................................................................................................................ 2
Methods ............................................................................................................................ 4
Results ............................................................................................................................... 7
Literature Cited .................................................................................................................. 25

Chapter II. Growth of rainbow trout fed New Zealand mud snails
Introduction ...................................................................................................................... 2
Methods ............................................................................................................................ 4
Results ............................................................................................................................... 7
Discussion
Literature Cited .................................................................................................................. 25
Executive summary
Flaming Gorge Tailwater Aquatic Biota Monitoring
There is long-term effort underway in the Upper Colorado River Basin to restore and protect native fish populations downstream of Flaming Gorge Dam. Toward this goal, a Biological Opinion was issued in 1992 specifying that releases from Flaming Gorge Dam be changed from a hydroelectric power-maximizing type scenario to a regime that more closely resembled the pre-dam natural hydrograph. Specifically, streamflows were to be higher in spring and lower during mid-late summer and winter as compared to streamflow releases in years prior to 1994. The purpose of this project was to assess the effects of this change in flow regimes on aquatic plants, invertebrates, and fish diets. The study area was the Green River from Flaming Gorge Dam downstream to Swinging Bridge.

Between 1994 and 2005, mean daily streamflow releases from Flaming Gorge Dam more closely approximated the pre-dam discharge regime than streamflows had since dam closure in December 1962. On average, streamflows were higher in the spring and lower the rest of the year and daily fluctuations were limited to a single peak event. In addition, greater than power plant capacity floods occurred in 1997, 1999, and 2005. Flows in 1997 were 8,600 CFS (244 m$^3$ sec$^{-1}$), in 1999 they were 10,900 CFS (297 m$^3$ sec$^{-1}$), and in 2005 they were 6,890 CFS (195 m$^3$ sec$^{-1}$). Prior to this, only three streamflow events had been above power plant capacity (ca. 4600 CFS) and released through the dam’s bypass tubes because of high spring runoff since the dam was closed in 1962. These occurred in 1983 (13,700 CFS, 388 m$^3$ sec$^{-1}$, 1984 (8,600 CFS, 244 m$^3$ sec$^{-1}$, and 1986 (8000 CFS, 227 m$^3$ sec$^{-1}$).

The major measured ecological affects of this change in dam operations and the three flood events on stream biota were:

1. an immediate, but short-term, ca. 6 month, decrease in aquatic plant biomass following above power plant capacity streamflows,
2. an immediate, but short-term, ca. 6 month, decrease in aquatic invertebrate abundance following above power plant capacity streamflows,
3. a long-term steady increase in aquatic invertebrate taxa richness,
4. mid-term, 1 to 3 year, changes in the relative composition of dominant invertebrate taxa following above power plant capacity flows, namely, an increase in Ephemeroptera populations and a decrease in Amphipoda,
5. little to no effect on fish populations as measured by fish diet composition over the study period, and there were no significant changes in fish condition over the study period as well (UDWR unpublished data).

These results suggest to us that the overall effects of these more natural flow regimes on the biotic assemblages of the Green River were positive. No lasting negative effects of these streamflows on aquatic biota were detected. Our results also suggest that continued streamflow releases of this timing and magnitude should continue to have positive effects on stream biota. The increase in invertebrate taxa richness in particular benefits the long-term health and stability of the overall aquatic ecosystem, as it provides additional forage opportunities for fish.

The monitoring data collected and reported on in this and our 1998 report represents one of the largest and most comprehensive data sets available to assess the long-term effects of dam operations on aquatic biota. We advocate that the monitoring of aquatic biota assemblages in the Green River downstream from Flaming Gorge Dam be continued at similar intensity levels as those of the last 12 years. We believe it would also be beneficial to expand this monitoring program downstream to compliment the fish and habitat monitoring currently being conducted by the native fish recovery program downstream from Swinging Bridge. The positive benefits of the post-1994 streamflow regime that we report on upstream of Swinging Bridge have likely occurred in downstream reaches as well, but to date insufficient monitoring has been done on aquatic plants and aquatic invertebrates to document these benefits.

**Growth of rainbow trout fed New Zealand mud snails**

New Zealand mud snails are an invasive species that colonized the Green River tailwater in 2001. In many streams, NZMS reach high population densities and out-compete native fauna for space on the substrate and food. There has been speculation that the snail’s thick operculate shells make them difficult to digest and fish may gain
little nutritional value from eating them. The objective of this study was to evaluate the ability of rainbow trout to assimilate New Zealand mud snails (NZMS). We found that juvenile rainbow trout starve when fed NZMS. A stable isotope experiment showed that trout fed $^{15}$N-labeled NZMS had muscle isotopic signatures 30% lower than trout fed a $^{15}$N-labeled amphipod. In a second experiment, trout fed NZMS lost 0.14-0.48%/d of their initial body weight, while trout fed amphipods gained 0.64-1.37%/d of their initial body weight. Fifteen percent of NZMS that passed through the intestinal tracts of trout were empty and were assumed to have been digested, 32% were dead, but present in their shells and were assumed to be undigested, and 53% were alive. Our results confirm that North American trout fisheries face potential negative impacts from NZMS invasion.
Introduction
There is long-term effort underway in the Upper Colorado River Basin to restore and protect native fish populations downstream of Flaming Gorge Dam. Toward this goal, a Biological Opinion was issued in 1992 specifying that releases from Flaming Gorge Dam be changed from a hydroelectric power-maximizing type scenario to a regime that more closely resembled the pre-dam natural hydrograph (Figure 1). Specifically, streamflows were to be higher in spring and lower during mid-late summer and winter as compared to streamflow releases in years prior to 1992. The purpose of this project was to assess the effects of this change in flow regimes on aquatic plants, invertebrates, and fish diets. The study area was the Green River from Flaming Gorge Dam downstream to Swinging Bridge.

Figure 1. Mean daily discharge (CFS) of the Green River downstream from Flaming Gorge Dam for the period of record for the United States Geological Survey (USGS) gage near Greendale, Utah (Station 092345000).
The tailwater reach of the Green River downstream of Flaming Gorge Dam is one of the most productive and popular trout fisheries in America. Changes in temperature or flow regimes, such as higher spring discharges could be detrimental to the trout fishery by influencing trout food resources. Rainbow trout (*Oncorhynchus mykiss*) in the Flaming Gorge tailwater have fairly specific diets. Filbert (1991) found that rainbow trout fed primarily on Chironomidae and *Gammarus lacustris*. This same lack of diet diversity (compared with trout diet composition in other North American streams, e.g., Reed and Bear 1966, Griffith 1974, Tippets and Moyle 1978, Allan 1981, Allan 1982, Angradi and Griffith 1989) has also been shown for rainbow trout below other high hypolimnetic release dams, such as Glen Canyon Dam (Persons et al. 1985, Maddux et al. 1987, Angradi et al. 1992). Reduced availability and selection of different food items makes trout populations in tailwater systems particularly susceptible to changes in production of the few available invertebrate taxa they feed upon.

In the Colorado River downstream of Glen Canyon Dam, Chironomidae and *G. lacustris* abundance was directly linked to the extent and health of periphyton communities (Leibfried and Blinn 1987, Blinn and Cole 1991, Pinney 1991). Chironomid and *G. lacustris* fed almost exclusively (>90%) on periphytic diatoms attached to *Cladophora* filaments (Pinney 1991). *Cladophora* and other large aquatic plants can also provide an important refuge for many macroinvertebrate taxa. Streamflow fluctuations can affect macroinvertebrate and periphyton communities through scouring, reduction in colonization rates following disturbance events, and by inhibiting photosynthesis due to increased turbidity.

The resistance of Green River phytobenthos communities to disturbance is likely a function of disturbance frequency, magnitude, and duration. The threshold at which biologically significant impacts occur is related to flood magnitude, stream competence, substrate particle size, and stream channel morphology. Community resilience is a function of the frequency, timing, and the area of disturbance.

Alternatively, high flow events may also benefit stream invertebrate assemblages and ultimately fish populations. Higher streamflows in the springtime and lower streamflows the remaining of the year are the norm for most unregulated salmonid
rivers. Higher than normal streamflows can enhance habitat and invertebrate assemblages by 1) rearranging substrates, thereby improving habitat quality, 2) reducing densities of abundant taxa, which may lessen interspecific competition among species and increase taxa diversity, and 3) resuspend nutrients buried in the substrate which can lead to increased primary production.

**Study design**

Our objective was to monitor benthic plant standing stocks, aquatic invertebrate assemblages, and fish diets downstream from Flaming Gorge Dam to assess the effects of the operations of Flaming Gorge Dam on biota. These data were collected seasonally (January, April, July, and September) for benthic plants and invertebrates since 1994 and semi-annually and then later annually for fish diets since 1998. This study design was based on the belief that (1) primary production (plants) affects secondary production (invertebrate assemblages), (2) secondary production effects trout production, and (3) streamflow disturbances affect primary and secondary production, as well as fish diets. Impacts were likely to vary longitudinally in response to changes in channel morphology and dampening of the flood wave downstream. This report describes data collected between January 1994 and September 2005 at 7 sites along the Green River from Flaming Gorge Dam downstream to Swinging Bridge.

**Study Area**

This study was conducted on the Green River downstream from Flaming Gorge Dam in northeastern Utah (40° 54’N, 109° 25’W). The Green River is the largest tributary of the Colorado River. It originates in the Wind River Range in Wyoming and flows south to its confluence with the Colorado River in southern Utah (Fig. 1). The elevation of the river at the base of the dam is 1835 m a.s.l. The contributing drainage basin is 7500 km2. Construction of Flaming Gorge Dam began in 1959 and was completed in December 1962. The dam rises 149 m above the original river channel. The dam is principally used for hydroelectric power generation and flood control. It creates a 146 km long, 1.5 million ha3 capacity reservoir.
Overall, the Flaming Gorge tailwater reach can be characterized as clear and cold (annual dam release temperatures of 2-14 °C), with deep runs (2 to 4 m) and pools (5 to 12 m), large cobble and boulder substrates and abundant growth of aquatic plants which includes moss (*Amblystegium riparium*), *Cladophora glomerata*, *Chara*, *Elodea canadensis*, *Potamogeton crispus*, *P. pectinatus*, *Ranunculus*, and *Spirogyra*. Aquatic invertebrate assemblages are dominated by Chironomidae, Simuliidae, Amphipoda, and *Baetis* mayflies (Vinson 2001). Fish populations are dominated by rainbow trout (*Oncorynchus mykiss*) and brown trout (*Salmo trutta*), with fewer numbers of Rocky Mountain whitefish (*Prosopium williamsoni*) and sculpin (*Cottus bairdi*) present. Trout biomass exceeds 800 kg/ha (Filbert and Hawkins 1995). Vinson (2001) describes the hydrology and biology of this reach of the river in detail.

Two small (< 1 m3 sec-1) perennial tributaries, and a large intermittent tributary, enter the Green River 2.4, 12, and 18 km downstream from the dam. Red Creek is an intermittent tributary that causes several changes in the physical and biotic nature of the river. One or more sediment mobilizing flash floods occur annually in Red Creek in response to summer thunderstorms. Suspended sediment concentrations up to 87 g L-1 have been measured in Red Creek and its contribution of sediment into the Green River is estimated to be 77 x 106 kg year-1 (Andrews 1986). Bed material in riffles consists of coarse gravel, cobbles, and boulders. Downstream from Red Creek the river leaves the canyon and enters Brown’s Park Valley. River width increases to 200 m and the depth decreases to 0.5-2 m. Bed material in the upper reaches of Brown’s Park consists of gravel and cobbles overlain by seasonal deposits of sand and silts exported from Red Creek. In downstream reaches bed material is composed almost entirely of sand.

Sampling sites were selected based on previous data collection efforts at several of these locations, access throughout the year, and their location downstream from the dam and Red Creek. Red Creek is an intermittent tributary of the Green River that significantly changes the character of the Green River downstream from this point. Four benthic biota sampling sites were located upstream of Red Creek and 3 were located downstream from Red Creek (Table 1).
Table 1. The location of benthic biota sampling sites downstream from Flaming Gorge Dam. All sampling locations were on the Green River.

<table>
<thead>
<tr>
<th>Site ID</th>
<th>Location</th>
<th>Up or Downstream from Red Creek?</th>
<th>Latitude</th>
<th>Longitude</th>
<th>Elevation (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>GRFGR1</td>
<td>Upstream from tailrace boat ramp</td>
<td>Upstream</td>
<td>40.84</td>
<td>109.422</td>
<td>1710</td>
</tr>
<tr>
<td>GRFGR2</td>
<td>Secret Riffle</td>
<td>Upstream</td>
<td>40.902</td>
<td>109.37</td>
<td>1701</td>
</tr>
<tr>
<td>GRFGR3</td>
<td>just upstream from Little Hole</td>
<td>Upstream</td>
<td>40.91</td>
<td>109.327</td>
<td>1695</td>
</tr>
<tr>
<td>GRFGR4</td>
<td>Grasshopper Island</td>
<td>Upstream</td>
<td>40.91</td>
<td>109.265</td>
<td>1681</td>
</tr>
<tr>
<td>GRFGR5</td>
<td>Taylor Flat Bridge</td>
<td>Downstream</td>
<td>40.915</td>
<td>109.173</td>
<td>1660</td>
</tr>
<tr>
<td>GRFGR6</td>
<td>Upstream end of Swallow Rapid</td>
<td>Downstream</td>
<td>40.865</td>
<td>109.14</td>
<td>1646</td>
</tr>
<tr>
<td>GRFGR7</td>
<td>Swinging Bridge</td>
<td>Downstream</td>
<td>40.834</td>
<td>109.049</td>
<td>1635</td>
</tr>
</tbody>
</table>

Field methods

Hydrology

Streamflow data were obtained from United States Geological Survey (USGS) near Greendale, Utah (Station 092345000), located 0.8 km downstream from the dam. Water temperature data were collected about every 4 h with Hobo Temp® electronic temperature recorders (Onset Inc., Pocasset, MA). Thermographs were deployed at the dam, near Secret Riffle, just upstream from Little Hole, at the Taylor Flat Bridge, at the upstream end of Swallow Canyon, and at Swinging Bridge. Data for this report are presented for 4 of these sites; the dam, just upstream from Little Hole, at the Taylor Flat Bridge, and at the upstream end of Swallow Canyon. All data are available upon request from Mark Vinson.

Benthic biota

Epiphytic aquatic plants and benthic aquatic invertebrate samples were collected seasonally (January, April, July, and September) each year at 7 locations (Table 1). Epiphytic aquatic plant standing stock was estimated by collecting samples from ten $D_{50}$ sized rocks at each sampling site. The attached plants were removed from a small area
(5.3 cm$^2$) of the upper surface of each rock. Samples were kept cold while on the river, then frozen and returned to the laboratory. Benthic macroinvertebrate samples were collected in riffle habitats with a Hess net (0.08 m$^2$, 250 micron mesh). Eight samples were collected at each site and composited to form a single sample from each site on each sampling date. When possible, samples were collected below the 800 cfs (28 cms) minimum flow water line.

**Drifting invertebrates**

Invertebrate drift was collected at two stream reaches: 1) 0.8 km downstream from Flaming Gorge Dam and 2) 12 km downstream from the dam at Little Hole. Drift was collected each September in conjunction with fish stomach content sampling. Invertebrate drift was collected at dusk (ca. 1900 h) at the downstream end of riffle habitats with 15 cm diameter round nets. Seven nets were set along a transect perpendicular to shore. Nets were set throughout the water column for 1 hour. Samples were preserved in the field in 95% ethanol and returned to the laboratory.

**Fish stomach contents**

Between April 1998 and September 2002, trout stomach contents were collected in April and September. Between 2002 and 2005, these samples were collected only in September; due to changes in how often fish were collected by the Utah Division of Wildlife Resources. Samples were collected from two stream reaches: 1) downstream from Flaming Gorge Dam and 2) downstream from Little Hole on two consecutive nights.

Fish were collected by Utah Division of Wildlife personnel by electrofishing. All fish were collected at night between 2000 and 2200 h. Approximately 1,200 brown and rainbow trout, 100 Rocky Mountain whitefish, and 20 sculpin were collected over two nights with electrofishing equipment. Stomach contents were collected from a minimum of 50 of these fish each night. Once collected, fish were anesthetized with Tricaine Methanesulfonate (MS-222), identified, measured to the nearest millimeter, weighed to the nearest 0.1 kg, and had their stomach contents removed by pulsed gastric irrigation.
Stomach contents were preserved in the field in 95% ethanol and returned to the laboratory.

**Laboratory methods**

**Aquatic plants**

Aquatic vegetation samples were thawed, dried at 60 C, weighed, and then fired in a muffle furnace at 550 C for 2 hours to obtain ash-free dry mass (AFDM).

**Benthic invertebrates**

The general procedures followed for processing invertebrate samples were similar to those recommended by the United States Geological Survey (Cuffney et al. 1993) and are described in greater detail and rationalized in Vinson and Hawkins (1996). A minimum of 300 organisms were removed and identified from each sample under a dissecting microscope. The remaining portion of each sample was then searched for rarer organisms that were not collected during the sample splitting process. Aquatic invertebrates were identified to the lowest taxonomic level possible based on organism maturity, except for Chironomidae, which were identified to sub-family and some non-insect groups, primarily Annelida, which were identified to Class or Order. Insects were most frequently identified to species or genus. Small, immature, or damaged specimens were generally identified to family. Species level identifications were based on identification keys, distributions records, and the author’s experience. All identified invertebrates in each sample were composited into a single museum-grade glass screw-top vial with a polypropylene lid and polypropylene liner. Sample labels were written with fade proof permanent black carbon ink on waterproof paper. Information on each label included the sampling location, sampling date and a unique catalog number. Specimens were preserved in 70% ethanol. Since September 2001, all samples were retained in our collection and are available upon request. Prior to that, samples were dried to determine aquatic invertebrate dry biomass.
**Drifting invertebrates**

All invertebrates were identified from each sample. Invertebrates were identified to similar taxonomic levels as benthic samples. Invertebrate drift was standardized as the number of invertebrates per $100\text{m}^3$ of water.

**Fish stomach contents**

All invertebrates were identified from each sample. Invertebrates were identified to similar taxonomic levels as benthic samples. Trout stomach content volume was determined by water displacement (0.1 ml). Trout stomach content data (invertebrate volume and abundance) were standardized by dividing them by trout length. Fish dietary preference was examined using Ivlev's Electivity Index (Ivlev 1961). This index characterizes the degree of prey selection for a certain prey type, on a range of -1 to +1, with negative values indicating avoidance and positive values indicating an active preference for the prey item. Ivlev's electivity index (IEI) is defined as:

$$\text{IEI} = \frac{(r_i - p_i)}{(r_i + p_i)}$$

Where $r_i$ is the relative abundance of prey item $i$ in the fish gut and $p_i$ is the relative abundance of the item in the environment (in this case prey items in the benthos and drift).

The cumulative number of unique taxa collected annually at all sites for benthic, drift, and stomach samples were determined. For benthic samples, data were summarized for January, April, July, and September samples. For stomach and drift samples, data were summarized for all samples collected each September.
**Results**

*Streamflow*

Mean daily discharges generally exhibited a more typical Rocky Mountain snowmelt driven pattern between 1994 and 2005 (Figure 2), as compared to daily flow patterns prior to 1994 (Figure 1). Specifically, streamflows were highest in the spring and lowest in the winter (Figure 3). Since 1994, streamflows greater than power plant capacity occurred in 1997, 1999, and 2005 (Figure 3). Flows in 1997 were 8,600 CFS (244 m$^3$ sec$^{-1}$), in 1999 they were 10,900 CFS (297 m$^3$ sec$^{-1}$), and in 2005 they were 6,890 CFS (195 m$^3$ sec$^{-1}$, Figure 3).

*Figure 2. Mean (± 1 SE) monthly flows in the Green River near Greendale during 3 major dam operation time periods.*
Figure 3. Mean daily discharge (CFS) of the Green River downstream from Flaming Gorge Dam. Data are from the United States Geological Survey (USGS) gage near Greendale, Utah (Station 092345000).
**Water temperatures**

Instantaneous maximum daily water temperatures were typically 8 to 10 C and minimums were generally 2 to 4.5 C (Figure 4). The diel range in water temperature was < 2 C. Water temperature regimes were similar among years. Water temperatures increased in the summer as the distance downstream from the dam increased and decreased in the winter in the downstream direction. This change was fairly gradual and overall there were only slight differences in temperatures among the sampling sites.

*Figure 4. Instantaneous water temperatures of the Green River at 4 locations downstream from Flaming Gorge dam. Data were collected every 4 hours using Hobo-Temp electronic temperature recorders.*
Aquatic plants

Aquatic plant biomass peaked in spring and was generally lowest in late-summer. Above power plant capacity flow events caused an immediate decrease in aquatic plants (Figure 5). These high flows in 1997 and 1999 caused plant biomass to drop to minimal levels, whereas the higher flows in 2005 appeared to reduce plant biomass to levels commonly measured between 1997 and 2003. Following high flow events, plant biomass recovered to near pre-high flow levels within about 6 months. Overall there appeared to be no appreciable decrease in plant biomass throughout the 11 year study period. Within riffles, *Amblystegium* (moss) was common throughout the year, *Chara* peaked in the fall and was lowest in early spring, and *Cladophora* was most abundant during the winter and declined throughout the summer and fall.

![Figure 5. Mean (+ 1 SE) aquatic plant biomass (moss, vascular plants, and algae) between 1994 and 2005. Arrows indicate the occurrence of above power plant capacity flows.](image-url)
**Benthic invertebrate assemblages**

**Taxa Richness**

Taxa richness increased fairly consistently from about 7 to 14 taxa between 1994 and 2005, an approximate rate of 0.7 taxa per year (Figure 6). Longitudinally, richness generally doubled on average from about 7 to 15 taxa, between upstream and downstream sites. Streamflow restoration changes appeared to have significant positive effects on taxa richness. Post-dam taxa richness prior to 1994 (1963 – 1993) averaged 8.7, whereas taxa richness after 1993 averaged 11.6 ($t = 2.19$, df = 30 $p = 0.03$; Figure 7A). Since 1994, richness was significantly higher in years following the above power plant capacities (1997 to 2005) then prior to 1997 ($t = 5.63$, df = 48, $p < 0.0001$, Figure 7B).

![Figure 6. Mean (+ 1 SE) aquatic invertebrate taxa richness between 1994 and 2005.](image-url)
The increase in aquatic invertebrate taxa richness since 1994 is thought to be the result of both more natural seasonal discharge regimes and the higher than power plant capacity flows that occurred in 1997, 1999 and 2005. These two factors cannot be separated. Overall, both of these factors appear to have caused disturbances to the streambed that have reduced aquatic vegetation and fine sediment, and increased substrate heterogeneity, which have all promoted increased taxa richness.

**Macroinvertebrate abundance**

Aquatic invertebrate abundance varied considerably within and among years, but mean annual densities were similar throughout the study period (Figure 8). The highest mean densities among all sites recorded were 18,634 individuals per m$^2$ in October.
1995 and the lowest were 4,386 individuals per m² in April 1997. Within years, densities were generally highest in spring prior to high flows and lowest in winter. Invertebrate abundances were reduced immediately following the high flow events in 1997, 1999 and 2005. This effect was more pronounced in 1997 than in 1999 and 2005, possibly because a larger than power plant capacity flow had not occurred since 1986, 11 years previously. Post-high flow abundances appeared to return to pre-high flow levels within 6 to 9 months. Invertebrate abundances decreased steadily from up-to downstream sites (Figure 9).

![Graph showing benthic invertebrate densities across all sampling sites between 1994 and 2005.](image)

*Figure 8. Mean (± 1 SE) benthic invertebrate densities across all sampling sites between 1994 and 2005.*
New Zealand mud snail populations

New Zealand mud snails were first collected near Swallow Canyon on 18 September 2001 (Vinson 2004). Presently they can be found in fairly high densities from just downstream from Flaming Gorge Dam downstream to the Colorado State line, a distance of about 50 km. They inhabit all habitats and substrate types, but the snails appear to prefer habitats with abundant aquatic vegetation with lower than reach average stream velocities, as their abundances were always higher within aquatic vegetation beds as compared to mineral substrates. Local population densities were highest within beds of sago pondweed (*Potamogeton pectinatus*). Local population
abundances have fluctuated both seasonally and annually and to date there are no clear trends, other then they have become more abundant in the river overall, as the number of sites they are collected at has increased over time (Figure 10).

Figure 10. Mean (+ 1 SE) abundances of New Zealand mud snails (bars) and the number of sites (solid line) they were collected at out of 7 sites from 2002 to 2006.
Assemblage composition

Composition of invertebrate assemblages was fairly consistent among years, although there were more Ephemeroptera present in 2000 and 2001 (Figure 11). Across all years and sites 4 orders dominated invertebrate abundances: Amphipoda (mostly *Hyallela azteca*) accounted for 54%, Diptera (Chironomidae and Simuliidae) for 31%, Ephemeroptera for 11%, and Coleoptera for 4%. There were patterns in relative abundances in downstream versus upstream sites. Downstream from Red Creek, invertebrate assemblages were more balanced. At these sites, Ephemeroptera accounted for about 26%; Diptera, primarily Chironomidae and Simuliidae, accounted for 41%; and amphipods accounted for about 26% of the total assemblage abundance (Figure 12). At upstream sites, Amphipoda made up the majority of organisms (62%); Diptera made up 28%; Ephemeroptera accounted for 7%; and Coleoptera accounted for 3%.

![Figure 11. Composition of invertebrate assemblages between 1994 and 2005.](image-url)
Invertebrate Composition upstream from Red Creek

Figure 12. Composition of aquatic invertebrate assemblages up (top graph) – and downstream (bottom graph) from Red Creek between 1994 and 2005.

Invertebrate composition downstream from Red Creek
Invertebrate drift

The composition of invertebrate drift was similar to that found in benthic samples, with the exception that terrestrial insects were present in drift samples. Like the benthic samples, the primary invertebrates were Dipterans (Chironomidae and Simuliidae), Amphipoda, and Ephemeroptera (Figure 13). Other invertebrates such as Trichoptera, Oligochaetes, Coleoptera and terrestrial insects were a minor component of drift. Ephemeroptera had a higher representation in the drift following the 1997 flood, but tailed off by 2001. From 1999 to 2001, drift was dominated by Diptera, but Amphipoda was dominant in all other years.

Drift abundance was variable from year to year, with large peaks in 1998 and 2003 (Figure 14). Both periods of high drift abundance were dominated by *H. azteca*. Because these samples were based on a single sampling date each year, we were unable to reliably ascribe this variability to any environmental or biotic factors, as they did not match the trends we observed in benthic data, for which we feel more accurately describes invertebrate assemblages. We believe the higher drift abundances in 1998 and 2003 were caused by fisherman induced disturbances upstream of the sampling sites. These data are probably best used as relative indicators of fish diet selection based on available invertebrates in the drift and benthos (next section).
Figure 13. Composition of aquatic invertebrate drift samples by major invertebrate Orders for both sampling locations between 1998 and 2005.

Figure 14. Mean drift abundances (numbers per 100 m$^3$) for all sites between 1998 and 2005.
Fish diet composition and selectivity

Fish stomach content volumes were generally similar throughout the study period, except for higher values in 1998 (Figure 15). Stomach content volume was unrelated to the amount of invertebrates in the drift – sometimes high amounts of drift were associated with high stomach volume, but other times high amounts of drifting invertebrates were associated with low stomach volume.

![Standardized fish stomach volumes](image)

*Figure 15. Standardized stomach content volume between 1998 and 2005.*

All fish species exhibited strong dietary preference for certain prey types, though all fish fed predominantly on *Hyalella azteca*, Chironomidae midges, and Ephemeroptera (Figure 16). For example, Rocky Mountain whitefish consumed more Simuliidae blackflies than the other species, whereas mottled sculpin fed largely on *Hyalella azteca*.

These results were mirrored by Ivlev electivity values (Figure 17). Positive IEI values indicate that fish ate a prey item in proportions greater than the prey item occurred in the environment, i.e., preferred that item, and negative IEI values indicate that the prey item was consumed in a proportion less than the prey item occurred in the environment.
environment, i.e., tended to avoid that prey item. Brown trout showed a preference for *Baetis tricaudatus*, *G. lacustris*, New Zealand Mud Snails (NZMS – *Potamopyrgus antipodarum*), terrestrial insects, Physidae snails, and Dipterans other than Chironomidae and Simuliidae. Brown trout tended to avoid *Hyalella azteca*, Chironomidae, and Elmidae riffle beetles. Rainbow trout showed strong preference for *B. tricaudatus*, terrestrial insects and Dipterans. They avoided Elmidae and *H. azteca*. Rocky Mountain whitefish preferred *B. tricaudatus*, Simuliidae, and NZMS. Mottled sculpin preferred, almost exclusively, *B. tricaudatus*, *G. lacustris*, and Trichoptera; and avoided most other prey types.

Piscivory was documented in 199 of the 1320 fish sampled. Among species, 20% of brown trout (117 of 578 fish) and 12% of rainbow trout (67 of 564 fish) had fish in their stomachs. Twenty-five percent of the fish consumed were identified as rainbow trout, 11% as mottled sculpin, and 64% of the bones could not be identified to the species of fish. Eighteen rainbow trout and 14 brown Trout had salmonid eggs in their stomachs, whereas no fish eggs were collected from mottled sculpin or Rocky Mountain whitefish. Brown trout were also found to consume mice; 7% of the brown trout sampled in 2004 and 10% of the brown trout sampled in 2005, had mice in the stomachs. Few mice were found in stomachs prior to 2004. No mice were found in rainbow trout stomachs. The number of New Zealand mud snails that were found in trout stomachs increased steadily after they colonized the river in 2001 (Figure 18).

Fish species diets varied among years, with fish tending to eat whatever was most abundant that year (Figure 19). When Ephemeroptera were relatively abundant (1999 – 2001), rainbow trout and brown trout incorporated more of them into their diets. When Ephemeroptera were less abundant, rainbow trout and brown trout consumed more Amphipoda. Rocky Mountain whitefish diets varied less predictably. Mottled sculpin consumed mostly Amphipoda in all years. The number of aquatic invertebrate taxa we collected in fish stomachs increased throughout the study period (Figure 20). This increased dietary breadth occurred for all fish species. As benthic invertebrate diversity continues to increase, we expect fish diet diversity to increase as well.
Figure 16. Relative composition of benthic, drift, and fish stomach contents for all sites between 1998 and 2005.
Figure 17. Ivlev electivity values for dietary preference for all study years and sites.
Figure 18. The percentage of fish that had New Zealand mud snails in their stomachs between 1998 and 2005.
Figure 19. Fish diet composition between 1998 and 2005.
Figure 20. The cumulative number of unique taxa collected annually at all sites for all sampling dates for benthic, fish stomach, and drift samples between 1998 and 2005.
Conclusion
Since 1992, streamflow releases from Flaming Gorge Dam more closely approximated the pre-dam discharge regime with respect to streamflow timing than had occurred since dam closure in December 1962. Principally, streamflows were higher in the spring and lower the rest of the year and daily fluctuations were limited to a single peak event. In addition, above power plant capacity streamflows occurred in 1997, 1999, and 2005.

The major measured ecological affects of this change in dam operations and the three flood events on stream biota were:

6. an immediate, but short-term, ca. 6 month, decrease in aquatic plant biomass following above power plant capacity streamflows,
7. an immediate, but short-term, ca. 6 month, decrease in aquatic invertebrate abundance following above power plant capacity streamflows,
8. a long-term steady increase in aquatic invertebrate taxa richness,
9. mid-term, 1 to 3 year, changes in the relative composition of dominant invertebrate taxa following above power plant capacity flows, namely, an increase in Ephemeroptera populations and a decrease in Amphipoda,
10. little to no effect on fish populations as measured by fish diet composition over the study period, and there were no significant changes in fish condition over the study period! (UDWR unpublished data).

These results suggest to us that the overall effects of these more natural flow regimes on the biotic assemblages of the Green River were positive. No lasting negative effects of these streamflows on aquatic biota were detected. Our results also suggest that continued streamflow releases of this timing and magnitude should continue to have positive effects on stream biota. The increase in invertebrate taxa richness in particular benefits the long-term health and stability of the overall aquatic ecosystem, as it provides additional forage opportunities for fish.

The data collected and reported on in this and our 1998 report represents one of the largest and most comprehensive data sets available to assess the long-term effects of dam operations on aquatic biota. We advocate that the monitoring of aquatic biota assemblages in the Green River downstream from Flaming Gorge Dam be continued at
similar intensity levels to those of the last 12 years. We believe it would also be beneficial to expand this monitoring program downstream to compliment the fish and habitat monitoring currently being conducted by the native fish recovery program downstream from Swinging Bridge. The positive benefits of the post-1994 streamflow regime that we report on upstream of Swinging Bridge have likely occurred in downstream reaches as well, but to date insufficient monitoring has been done on aquatic plants and aquatic invertebrates to document these benefits.
Literature cited


Leibfried, W.C. and D.W. Blinn. 1987. Effects of steady versus fluctuating flows on aquatic macroinvertebrates in the Colorado River below Glen Canyon Dam,


Chapter II. Growth of rainbow trout fed New Zealand mud snails

By Mark Vinson and Michelle Baker

The New Zealand mud snail (NZMS, *Potamopyrgus antipodarum* Gastropoda: Hydrobiidae) is a small (<7 mm long) operculate freshwater snail (Winterbourn 1970) that has spread from New Zealand throughout the world. Since its arrival in the United States in 1987 it has spread rapidly throughout the western U.S. (http://www.esq.montana.edu/dlg/aim/mollusca/nzms/). The NZMS flourishes in a variety of aquatic habitats, including springs, rivers, lakes, and estuaries. It has been found across a wide range of water temperatures (0-30 C, Winterbourn 1969; Hylleberg and Siegismund 1987, Michaelis 1977), substrates (Heywood and Edwards 1962; Cogerino et al. 1995; Richards et al. 2001), water depths (Zaranko, et al. 1997), productivity (Schreiber et al. 2003), and salinities (Winterbourn 1970). A probability survey of perennial streams in 12 western U.S. states conducted in 2000-2004 found that 10% of all stream miles in 42% of the conterminous U.S had NZMS (US EPA Environmental Monitoring and Assessment Program, unpublished data). In many streams, NZMS reach high population densities and out-compete native fauna for space on the substrate and food (Hall et al. 2003; Kerans et al. 2005). In Polecat Creek, Montana, for example, NZMS population densities exceeded 100,000 individuals per square meter and comprised >95% of the invertebrate biomass in some stream sections (Hall et al. 2003). At these densities they replace the native invertebrate forage base which could affect fish populations through their diet. Apparently, the snail’s thick operculate shells make them difficult to digest and fish may gain little nutritional value from eating them. The objective of this study was to quantitatively determine if juvenile rainbow (*Oncorynchus mykiss*) trout assimilate NZMS into their muscle tissue.
Methods

Study Design
We conducted two experiments to determine if trout assimilate nutrients from NZMS: 1) the addition and tracking of the stable isotope $^{15}$N though trophic pathways including $^{15}$N enriched NZMS; and 2) an evaluation of trout growth on an unlimited diet of NZMS. Changes in weight over 3-month trials were compared to trout fed a native amphipod, *Hyalella azteca*, and to that estimated weight loss for starved trout calculated using the Wisconsin Bioenergetics Model (Hanson et al. 1997). NZMS digestibility was also evaluated by collecting trout feces and back-calculating consumption rates based on trout weight changes using the bioenergetics model.

Stable Isotope Experiment
Filamentous algae (*Spirogyra*) and diatoms were grown in 19 L aquaria that were spiked with 75 mg of 98 atom percent $^{15}$NH$_4$Cl. After 7 days >100,000 NZMS and amphipods were each introduced into spiked aquaria. NZMS were also raised in a control not enriched with $^{15}$N. NZMS and amphipods used in the experiment were collected from the Green River, downstream from Flaming Gorge Dam in northern Utah, USA. Every few weeks, additional NZMS and amphipods were collected and placed into alternate aquaria and allowed to assimilate the heavy isotope before feeding.

At the start of the experiment, 27 juvenile rainbow trout (total length = 89 – 113 mm, weight = 8.7 – 13.8 gm) were marked with a passive integrated transponder tag (BioSonics, Inc. Seattle, Washington), measured and weighed. The fish were placed in a large flow-through trough at 10 C and fed Silver Cup trout diet (Nelson and Sons, Inc. Tooele, Utah) twice daily. After 23 days, 3 fish were euthanized (pre-treatment control fish) and 24 fish were evenly and randomly assigned, 6 each, to one of four feeding treatments; 1) enriched NZMS, 2) enriched amphipods, 3) non-enriched NZMS, and 4) commercial fish diet. Fish were fed twice daily. We fed trout an amount of food that was consumed in about 5 minutes (about 80 amphipods, 60 pieces of fish food, and 70 NZMS at each feeding). Twice daily, about 80% of the water and the fish feces in each
aquarium were siphoned out and replaced with fresh well water. The feeding trial was run for 84 days following the 23 day acclimation period.

Isotope samples were collected from invertebrates prior to putting them in the isotope-enriched aquaria and after 23 days. Trout tissue samples were collected from 3 fish prior to starting the feeding trials and at the end of the experiment.

**Weight Change Experiment**

Six juvenile trout (total length = 106 - 142 mm, weight = 13.1 to 38.6 gm, mean = 29 gm) were randomly placed in each of two large flow through troughs (2 m long x 0.6m wide x 0.3 m deep) supplied with well water at 10 C. Both troughs were stocked with aquatic macrophytes, principally *Ranunculus, Elodea,* and *Lemna minor.* Invertebrates attached to the macrophytes were removed. One trough was stocked with amphipods and the other with NZMS collected from the Green River downstream from Flaming Gorge Dam. Additional amphipods and NZMS were transplanted into the troughs every few weeks. Amphipod and NZMS populations remained high throughout the experiment, averaging more then 130,000 m$^{-2}$. Trout were weighed approximately weekly to the nearest 0.01 g.

After 108 days the fish were removed from the troughs and quarantined for 24 h in individual aquaria to allow them to clear their digestive tracts. Fish were then placed in the opposite treatment trough from which they had spent the previous 108 days. Fish feces were collected from each aquarium, kept in water, and the NZMS were tabulated as alive, dead, or empty shells under a dissection microscope. Live snails were very active and easily separated from dead and empty shells. The feeding experiment was run as before. At the end of 107 days the above procedure was repeated and the experiment was run for another 98 days.

**Bioenergetics Model**

The weight loss of starved fish was modeled using the Wisconsin Bioenergetics Model (Hanson et al. 1997). The model estimates the rate of change for juvenile rainbow trout not assimilating any prey energy. The model was run for juvenile rainbow trout with an
initial starting weight of 29 gm at 10 C for 108 days. NZMS digestibility was estimated for each fish for each feeding trial by calculating the proportion of maximum consumption based on the observed weight changes. The model was run with a caloric energy value for NZMS of 4,500 j g⁻¹ dry mass (Ryan 1982).
Laboratory Methods

A white muscle tissue sample was collected from each fish from below the dorsal fin with a 5 mm diameter dermal biopsy punch (Miltex Instrument Company, Bethpage, New York). Samples were dried at 60 °C for 48 h and ground using a mortar and pestle. Approximately 2 mg of sample material was placed into 4 X 6 mm tin capsules. ¹⁵N content was measured using a Europa Hydra 20/20 continuous flow isotope ratio mass spectrometer (PDZ Europa Ltd., Cheshire, U.K.) at the University of California-Davis Stable Isotope Facility, Davis, California, USA. Stable isotope content is reported as δ¹⁵N (units of ‰), defined as:

\[
\delta^{15}N = \left( \frac{R_{\text{SAMPLE}}}{R_{\text{STANDARD}}} \right) - 1 \times 1000
\]

Where \( R_{\text{SAMPLE}} \) is the \(^{15}\text{N}:{ }^{14}\text{N} \) ratio in the sample and \( R_{\text{STANDARD}} \) is the \(^{15}\text{N}:{ }^{14}\text{N} \) ratio in the atmospheric \( \text{N}_2 \) standard (\( R_{\text{SAMPLE}} = 0.0036765 \)). Values of \( \delta^{15}N \) were converted to the isotope mole fraction (\(^{15}\text{N}/^{14}\text{N} \)) by the equation:

\[
\frac{^{15}\text{N}}{^{14}\text{N} + ^{15}\text{N}} = \frac{\left( \delta^{15}N \right) + 1}{1 + \left( \delta^{15}N \right) \times 0.0036765} \times 0.0036765
\]

Isotope mole fraction (MF) is used to calculate the mass of \(^{15}\text{N} \) in food and fish tissues by multiplying the MF times the mass of total N in the sample.
Results

Stable Isotope Experiment
The addition of $^{15}$N-NH$_4$ to the aquaria resulted in enriched invertebrate tissues after 23 d. Mean $\delta^{15}$N values varied significantly (ANOVA p < 0.0001) among fish food items; non-enriched NZMS averaged 9.9 ± 0.8 ‰, (mean ± SD), enriched NZMS averaged 668 ± 60 ‰, and enriched amphipods averaged 1299 ± 68 ‰. On a per gram dry weight basis the $^{15}$N content of the labeled NMZS and labeled amphipods were not different. Labeled NZMS contained 9.8 ± 0.8 $\times$ $10^{-4}$ μg$^{15}$N/μg dry mass and labeled amphipods contained 9.9 ± 1.0 $\times$ $10^{-4}$ μg$^{15}$N/μg dry mass

At the end of the experiment, the mean isotopic signature, reported as $\delta^{15}$N, of fish muscle tissue varied significantly among diet treatments (ANOVA p<0.0001); pre-treatment control fish averaged 11.4 ± 0.4 ‰; fish fed commercial feed averaged 10.9 ± 0.8 ‰; fish fed non-labeled NZMS averaged 12.6 ± 1.2 ‰; fish fed $^{15}$N-labeled NZMS averaged 69.9 ± 16 ‰; and fish fed $^{15}$N-labeled amphipods averaged 99.6 ± 10.6 ‰. The mean isotopic signature of fish fed $^{15}$N-labeled NZMS was 80% higher then trout fed unlabeled NZMS and 30% lower then trout fed $^{15}$N-labeled amphipods.

Weight Change Experiment
The two diet treatments produced opposite trends in trout growth over all 3 trial periods (Figure 1). Trout fed NZMS lost on average 0.15%, 0.14%, and 0.48% of their initial body weight per day; whereas fish fed amphipods gained on average 0.64%, 1.37%, and 0.99% of their initial body weight per day. During the 3$^{rd}$ feeding trial, 3 fish in the NZMS treatment died after day 227. The bioenergetics model predicted that starved trout would have lost 0.53% of their initial body weight per day. This was similar to what we observed in the 3$^{rd}$ trial for trout fed NZMS, but more then we observed in the first 2 trials.

Snail Survival through Trout Digestive Tract
Fish were quarantined when switched between food treatments and allowed to void their stomach contents. Of the 723 NZMS that passed through the intestinal tracts of 13
trout, 15% of the shells were empty and were assumed to have been digested, 32% were dead, but present in their shells and were assumed to be undigested, and 53% were alive. The mean modeled digestibility of NZMS based on weight changes was 9% and ranged from 4 to 11%.

Figure 1.—The relative change in juvenile rainbow trout weight from initial weight when fed *Hyalella azteca* and *Potamopyrgus antipodarum* (NZMS). Trout were moved between the two treatments after 108 and 213 days. Trout in food group A were sequentially fed *H. azteca*, then NZMS, and then *H. azteca*. Trout in group B were fed the opposite. The dashed line describes trout weight loss over time for fish that were starved and was determined using the Wisconsin Bioenergetics Model (Hanson et al. 1997).
Discussion

Our results showed that in a laboratory setting, juvenile rainbow trout readily ingested NZMS, but received little nutritional value from them. There are three aspects of trout life history and behavior that are relevant to our results and the effects NZMS may have on wild trout populations. First, there is strong correlation between aquatic invertebrate abundance and trout growth, abundance, distribution, and condition, (Elliot 1975; Shirvell and Dungey 1983; Wilzbach 1985; Filbert and Hawkins 1995; Johansen et al. 2005). Second, trout typically consume prey in proportion to their abundance in the environment (Bres 1986; Cada 1987; Angradi and Griffith 1990; Hildebrand and Kershner 2004). Third, resident trout commonly move throughout river basins (Gowan et al. 1994; Gowan and Fausch 1996; Schmetterling and Adams 2004) and more mobile fish tend to have lower condition than the general population (Gowan and Fausch 1996; Hildebrand and Kershner 2004).

The rapid dispersal of NZMS within basins has been attributed to passive downstream drift on aquatic vegetation and woody debris, upstream movement (Haynes et al. 1985; Lassen 1975) and recreationists (Ribi 1986). More than half of the snails ingested by trout in our experiment survived passage through the digestive tract. We assume that many if not most of these snails would survive and reproduce. Clearly, there is a high potential for trout themselves to influence the distribution of NZMS in rivers as they carry the snails in their guts. Fish that consume NZMS may have lower condition than the general population and thus may be more mobile (Gowan and Fausch 1996; Hildebrand and Kershner 2004) thus increasing the spread of NZMS.

Once NZMS colonize a site they often numerically dominate stream invertebrate assemblages, with populations >100,000 individuals m\(^{-2}\) regularly reported (Dorgelo 1987; Richards et al. 2001; Kearns et al. 2005). At these high population densities they can modify stream ecosystems. Hall et al. (2003) found NZMS accounted for 97% of the invertebrate biomass and consumed 75% of gross primary productivity in a Rocky Mountain stream. Our results suggest they will also impact higher trophic levels. In streams where NZMS become established, they will likely become a dominant component of trout diets to the detriment of trout populations. Initial evidence for this is
available from the Green River downstream from Flaming Gorge Dam where NZMS were first detected at a site 35 km downstream from the dam in 2001. Since 2001, NZMS have expanded their populations throughout a 50 km river reach from the dam downstream to the Utah–Colorado state line. Stomach content sampling of trout has detected a sharp increase in the number of trout consuming NZMS since 2001 (Figure 2a) and the condition of brown trout (Salmo trutta) and rainbow trout with NZMS in their guts has been significantly less then for trout without NZMS in their stomachs (ANOVA: brown trout; $F = 8.26; \text{df} = 1, 576; P = 0.0042$, rainbow trout; $F = 12.0; \text{df} = 1, 561; P = 0.0006$; Figure 2b).

The increasing occurrence of NZMS in trout diets in the Green River, a decrease in trout condition for trout known to be ingesting NZMS, the low digestibility of the snails, and their high dispersal capabilities has negative implications for trout fisheries throughout North America. Future work should 1) determine the potential distribution of NZMS in North America (sensu Bossenbroek et al. 2004), 2) quantify competitive interactions between NZMS and native invertebrates (sensu Kerans et al. 2005) and 3) determine the effects of NZMS on higher trophic levels in natural systems (sensu Vander Zanden et al. 1999).
Figure 2a top) The percentage of trout collected from the Green River downstream from Flaming Gorge Dam with *Potamopyrgus antipodarum* in their stomachs and b, bottom) the mean ± SE condition of trout with and without NZMS collected from their stomachs.
contents. Trout were collected as part of the Utah Division of Wildlife’s annual September trout population survey. Each year, approximately 100 brown trout (*Salmo trutta*), Rocky Mountain whitefish (*Prosopium williamsoni*), and rainbow trout were collected over two nights with electrofishing equipment. Once collected, fish were anesthetized with Tricaine Methanesulfonate (MS-222), identified, measured to the nearest millimeter, weighed to the nearest gram, and had their stomach contents removed by pulsed gastric irrigation (Foster 1977). Stomach contents were preserved in the field and identified in their entirety in the laboratory. Trout condition was determined following (LeCren 1951) where $\log_{10}(W) = \log_{10}(a) + b \times \log_{10}(L)$. $W$ is fish weight (g), $L$ is fish total length (mm), $\log_{10} a$ is the $y$-intercept, and $b$ is the slope.
References


