Influences of Fluctuating Releases on Stream Habitats for Brown Trout in the Smith River below Philpott Dam

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Fred Leckie
Virginia Department of Game and Inland Fisheries
4010 W. Broad Street
Richmond, VA 23230-1104

Prepared by:

Principle Investigator: Donald J. Orth
Co-PI's: Tammy J. Newcomb, C. Andrew Dolloff, Panos Diplas
Research Associate: Colin W. Krause
Graduate Research Assistants: M. Anderson, A. Hunter, Y. Shen
Department of Fisheries and Wildlife Sciences
Virginia Polytechnic Institute and State University
Blacksburg, VA 24061-0321

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EXECUTIVE SUMMARY

State: Virginia Project Number: F-121-R

Project Title: **Influences of Fluctuating Releases on Stream Fishes and Habitat in the Smith River, below Philpott Dam**

This project consists of three job elements. Job 1 focuses on **Characteristics of Spawning and Rearing Habitats for Brown Trout**. The objectives are to characterize instream habitat conditions in areas where successful spawning and juvenile rearing of brown trout occurs. The Smith River tailwater supports a naturally reproducing population of brown trout, thus spawning patterns, age-0 emergence and growth, as well as spawning habitat quality is of interest to fisheries managers. Monitoring of spawning activity, age-0 emergence and growth, and habitat quality continued during 2003 to build on the findings from 2000, 2001, and 2002. Continued monitoring and studies conducted during 2003 provide insight into the working hypotheses of Job 1, which address factors (i.e. high flows and pulsing nature of high flows) that could potentially impair spawning success.

These hypotheses were addressed in 2003 by focusing effort on the following activities: 1) describing spatial and temporal patterns of spawning brown trout, 2) measuring attributes of individual redds within a subset of spawning areas, 3) measuring intragravel permeability, dissolved oxygen, and temperature within redds, 4) assessing sediment intrusion within Vibert boxes placed in artificial redds in the Smith River, 4) describing temporal patterns of emergence and growth of age-0 brown trout from a subset of spawning areas, and 5) analyzing the occurrence, magnitude, and duration of the peaking flows in the Smith River over the past 13 years.

During multiple years of this study we have observed spawning to initiate shortly after water temperatures fall below 9ºC. Additionally, redd development has proceeded from downstream to upstream reaches with time. The cause for this trend is the earlier onset of water temperatures below 9°C downstream before upstream.

Peaks in redd occurrence occurred at similar river kilometer locations downstream of Philpott dam in 2002 and 2000. In 2000 the upper reaches of the river (3 - 8 km) had the greatest number of redds (n = 147 or 78%). In 2002 there was also a peak in redd occurrence from 3 - 8 km, but it only represented 42% (n = 47) of redds observed throughout the tailwater. During 2001 only 3 redds were observed most likely from spawning activity occurring during a shorter than typical time-frame prior to commencing redd surveys. Despite not observing many redds in 2001 there was successful spawning based on the high recruitment observed in the spring.

Emergence of age-0 trout occurs first in upstream reaches because water temperatures during incubation are warmer near the dam than those downstream, which are cooled by winter air temperatures. Age-0 occupied edge habitat after emergence and during the initial growth period (~ March - June). In 2003 the recruitment of age-0 trout was lower than in previous years of study, coincident with increased occurrence, magnitude, and duration of peak-flow during the emergence period.

Intragravel permeability was greater within the redd egg-pocket than in the undisturbed substrate beside the redd for all redds sampled but one. Mean permeability in (4499 cm/hr; SD = 2055) and beside (2166 cm/hr; SD = 1781) redds was statistically different (based on ± 2 SE). Permeability in and beside individual redds was statistically different (based on 95% confidence intervals) for 27 of the 30 redds measured. These results demonstrate that the spawning trout modifies the substrate, thereby enhancing permeability.

Fine sediment intrusion into Vibert boxes increases with distance downstream of Philpott dam. Fine sediment intrusion into redds at downstream sites may be resulting in lowered survival of eggs and thus a lower abundance of age-0 fish when compared with upstream locations

Sampling to obtain age-0 population densities on May 14-15th and July 10th and 14-18th, 2003 revealed that density was greatest upstream (with the exception of immediately below the dam at 0.7 km) and diminished with distance from the dam. The same trend has occurred in all previously sampled years. Additionally, May 2003 age-0 density estimates are significantly lower (based on 95% confidence intervals) than in May 2001 and 2002 at sites 6.0, 12.9, 14.1, and 22.8 km. This is also the case for July 2003 where densities were significantly lower at seven of 11 sampled sites compared to June of 2000, 2001, and 2002. The cause of reduced population density is potentially due to increased occurrence, magnitude, and duration of peak-flow during the emergence period in 2003. Peak-flows were more prevalent in 2003 due to frequent rain-events which caused the US Army Corps of Engineers to release more water to maintain a normal reservoir level.

During the years of this study starting in fall 1999 through 2002, the average percent time (based on annual and water year) that flow has been peaked is less than during the previous eight years for which we have 15-min interval discharge data.

Job 2 focuses on **Determinants of Brown Trout Growth and Abundance**. The objectives of Job 2 are (1) To collect biological data to quantify relative abundance of trout in the Smith River from Philpott Dam to Martinsville and monitor annual variation in brown trout recruitment success; (2) To assess longitudinal and seasonal shifts in brown trout diet composition, and (3) To evaluate the bioenergetic constraints on trout growth under existing temperature regimes.

Within the tailwater, thermal and flow regimes are predicted to influence the brown trout population. Variations in the flow and thermal regimes can also influence food availability and ultimately the amount of food that trout can consume. Brown trout diet composition can vary on spatial and temporal scales based on available prey and high numbers of brown trout may be causing competition among trout for food resources. To determine the role that food consumption is having in structuring trout growth rates, a study was initiated to determine daily consumption rates of brown trout in four reaches, which will be used in subsequent bioenergetics models.

In 2002, the number of age-0 brown trout (2002 year class) was significantly higher than in previous sampling years. The reduction in generation peaking flows may have lead to the increase in the number of age-0 brown trout. In 2003, peak generation flows were greatly increased in both magnitude and duration over the flows that occurred in 2002. The 2003 brown trout year class is significantly lower than in previous years.

Aquatic macroinvertebrates are an important component in the diets of brown trout in the Smith River. In the Dam Reach, macroinvertebrates comprised the majority of the diet. Fish were common in the diets of trout from all of the reaches during December, and were also common during May and September in the Bassett and Kohler reaches.

Absolute growth rates in length (mm day^{-1}) and weight (g day^{-1}) were significantly different among sampling sites ($P<0.0001$) with lowest growth rates near the dam and increased growth rates at intermediate sites. Growth rates varied seasonally ($P<0.0001$) with highest growth rates from June to October, and lowest growth rates occurred during October to April. Regression models for trout growth failed to explain a large portion of the variation in specific growth rates; however, the models give an indication that temperature and trout numbers were influencing trout growth in the Smith River.

Consumption rates will be incorporated into bioenergetics models to help determine factors limiting the growth of brown trout in the Smith River. Both diet information and oxygen consumption parameters will be combined in a bioenergetics model to fully evaluate potential constraints and predict trout growth under alternative thermal regimes.

A Masters thesis by Anne K. Hunter was completed on the nonsalmonid component of the fish community in the tailwater. Based on this analysis, flow and temperature directly influence fish community patterns in the Smith River and the patterns are persistent over space and time even though numbers of individuals vary. The thesis is available at: http://scholar.lib.vt.edu/theses/available/etd-04082003-215009/.

Job 3 focuses on **Hydraulic Model Development and Application to Smith River Tailwater.** The objective is to design a field survey and modeling protocol to measure effects of varying flows on the shear stress, mobilization of streambed gravels, and relate discharge to the amount of redd scouring or brown trout fry displacement that would occur at sites in the tailwater. This information coupled with flow records should permit prediction of catastrophic year-class failures and flow ranges that provide for acceptable reproduction.

One-dimensional (1-D; PHABSIM model) and two-dimensional (2-D; RMA2 model) hydraulic numerical models were used to predict brown trout spawning habitat at an upstream site (4.2 km below Philpott dam) at various water discharges. A RiverCAT (Acoustic Doppler Current Profiler) was employed to measure the three-dimensional (3-D) water velocities and water surface elevations along selected river cross-sections. Both models were calibrated for the moderate flow and validated for the base-flow and bankfull flows. From the simulation of certain physical conditions (i.e. water discharge, channel topography, and substrate) by the hydraulic models, water depth and velocity predictions were transferred into quality indices of spawning habitat through a habitat suitability model.

The relative error between field measurements of water surface elevation and predictions from both models was less than 10% at 108 randomly surveyed locations at various flows. The mean absolute error of the water surface prediction throughout the site was 0.02 m for the 1-D model and 0.03 m for the 2-D model at base-flow. At bankfull flow, the prediction error of water surface elevation was 0.04 m for the 1-D model and 0.035 m for the 2-D model. The greatest errors were at the stream margins.

River reaches with more redds had higher habitat quality indices, which implied a positive relationship between redd densities observed in the river and habitat quality predicted the models. Specifically, 78% of the locations with redds were identified as areas having high composite suitability indices ($CSI>0.9$) by the 2-D model, whereas 69% of the total redd locations where predicted with the same high CSI values by the 1-D model. The distribution of redds was predicted more accurately by the 2-D model. At base-flow which is the dominant discharge during fish spawning, the calculated Spearman correlation coefficient (r) of redd density and habitat quality was 0.744 ($P = 0.009$, $n = 10$) for 1-D PHABSIM, whereas for the 2-D RMA2 model the coefficient was 0.875 ($P = 0.001$, $n = 10$).

Our results show that both models performed reasonably well with regard to the prediction of the water surface elevations and velocity distributions. However, the 1-D model cannot correctly simulate local velocity patterns around boulders.

PROJECT NARRATIVE

State: Virginia Project Number: F-121-R

Project Title: **Influences of Fluctuating Releases on Stream Fishes and Habitat in the Smith River, below Philpott Dam**

<u>Introduction</u>

Need: This study was designed in response to discussions with fisheries biologists from the Virginia Department of Game and Inland Fisheries and the U.S. Army Corps of Engineers. These agencies are interested in determining the feasibility of enhancing habitat for wild brown trout (*Salmo trutta*) in the Smith River below Philpott Dam, Henry County, Virginia. A three-mile special trout regulation area is regulated by a 16-inch minimum, 2-fish-per-day limit. The full potential of the brown trout fishery is limited by a flow regime that fluctuates from 45 cfs to 1280 cfs on a daily basis; the minimum flows are now significantly lower than before dam construction. The Smith River supported over 36,000 anglers hours of trout fishing in 1995 and trout anglers indicated they were willing to pay more for opportunities to catch wild trout and not cancel fishing plans due to generation flows (Hartwig 1998). Presently trout catches are dominated by catchable rainbow trout, with the exception of the special regulations section where catches of wild brown trout exceed that of rainbow trout by thirteen to one. Only 3.6% of brown trout caught by anglers exceed 16 inches. Doubling an angler's chance of catching a large brown trout would more than double the net economic value of the riverine fishery (Hartwig 1998). High mortality coupled with modest growth rates of 1+ and older brown trout appear to limit recruitment of trophy fish. While it appears clear that the flow regime could be improved to benefit the trout fishery, it is not clear what changes in flows or channel enhancements should be proposed. The working hypothesis is that the brown trout recruitment is limited by spawning and rearing habitat and adult growth rates are constrained by reliance on small drifting invertebrate prey base and reproductive costs. This study will (1) determine timing and flow levels needed to enhance spawning and rearing habitats to benefit the wild brown trout population, (2) develop a protocol for estimating brown trout population characteristics (growth, mortality, population density), (3) survey the nongame fishes along a continuum of temperature and fluctuating flow levels, and (4) develop a hydraulic model to evaluate the effect of different releases on physical habitat conditions during spawning, incubation, and fry rearing periods. We assume long-term investment in monitoring the effects of any actions to enhance this fishery; therefore, the study elements (Jobs 1 and 2) will provide the framework for adaptive management of this important tailwater fishery.

Significant advances have been made in developing assessment tools for analyzing flow effects on stream fauna (Stalnaker 1994, Van Winkle et al. 1998) and dam operators are more routinely reconsidering their operations in response to the demands of anglers and recreationists who use tailwaters. The altered conditions in tailwaters have a variety of effects (Cushman 1985, Hunter 1992) and create novel habitat conditions that permit the establishment of valuable salmonid fisheries in regions where these resources are limited. The predictive reliability of instream flow assessment tools are most limited in situations where streamflow may vary by several orders of magnitude over short (hourly or more) time periods (Gore et al. 1989), such as the Smith River tailwater. Furthermore, reliance on single factors to predict population responses is unreliable (Jager et al. 1999). It is clear from initial correspondence with staff of the U.S. Army Corps of Engineers that more specific proposals for reservoir releases will be needed before they are able to respond and evaluate the feasibility of changes in the flow releases.

Objectives: To conduct research to validate and discover new fish-population and habitat relationships and provide defensible fish-habitat relationships to be used for developing specific management actions to improve the fisheries resources of the Smith River tailwater. Specific working hypotheses to be tested under this study include:

1. High flows during the incubation period for salmonids can scour spawning gravels causing catastrophic mortality of a year class depending on timing and magnitude of floods. This short-term event represents a typical habitat bottleneck on population abundance.
2. Short-term pulsing flows create localized areas of high shear stress that disturb benthic habitat, thereby limiting growth and production of young trout and their prey.
3. These impacts are spatially variable and characterizing the extent of sensitive and nonsensitive locations at different flows will suggest suitable flow regimes or mitigation strategies. For example, middle sections of the Smith R. (special regulations section) support reproduction ("source") while upstream and downstream reaches are "sink" populations.
4. These impacts can be more adequately modeled with 2-dimensional finite element or finite difference methods than traditional 1-dimensional models and mitigation measures (e.g., boulder placements) can be evaluated with this modeling technology.

Expected Results and Benefits: The Smith River supports over 36,000 angler hours of trout fishing annually; total economic value of the trout fishery was $440,000/yr (1995 dollars, Hartwig 1998) under current suboptimal conditions. Anglers report a highest willingness to pay for catching larger trout, wild trout, and more fishable flows. It is likely that management actions could enhance the value of this fishery with minor influence on the value of power production ($670,000/yr). Essential data will be collected to permit managers to coordinate and cooperate with other agencies and utilities interested in optimal management of the flowing water resources of the Smith River. Information generated will provide the Fish Division with a reliable modeling tool for evaluating effects of flow on trout habitat.

Approach: Flow release schedules in tailwaters may influence salmonid populations through at least two major pathways: (1) disturbance during early life history or (2) impoverishment of the prey base. Adult trout seem to be quite adaptable at dealing with the flow fluctuations (Niemala 1989, Pert and Erman 1994). Disturbance causing high mortality early in life would create a habitat bottleneck constraining population abundance. The bottleneck could occur if high flood flows during incubation scour eggs from redd pockets. The descriptions of brown trout redds (Dechant and West 1985, Crisp and Carling 1989, Grost and Hubert 1991) do not currently permit the prediction of susceptibility to scour at high flows. Even if redds are protected, however, high flows may cause downstream displacement and mortality in the fry stage (Heggenes 1988, Heggenes and Traaen 1988, Crisp and Hurley 1991), unless the bottom topography provides hydraulic refugia during flow pulses (Lobón-Cerviá 1996). In severe cases the trout fishery would have to be sustained via stocking. Furthermore, siltation of redds is greater when flows are fluctuating (Carling and McCahon 1987). The indirect pathway limits the quality, quantity, or stability of habitat for prey organisms, thereby depressing the growth and ultimate size of resident trout. Both of these effects may play a role in a tailwater. In the Smith River tailwater, brown trout may be influenced via both of these pathways. In addition, nongame fishes may be affected by similar habitat bottlenecks due to fluctuating flow. Operation of Philpott Dam also restricts non-game fishes from much of the tailwater reach (~10 km below the dam) due in part to cold temperatures. However, some of the nongame species in the Smith River often occur with trout in other drainages and may be similarly affected by the pulsing flow regime. Because of their spawning and rearing habits, these fishes (e.g., rosyside dace, bluehead chub, Roanoke hog sucker, Roanoke darter, and fantail darter) may be more vulnerable than brown trout to disruption of spawning habitat (Smith 1999). Minor changes in operation of the tailwater or other mitigation strategies (e.g., boulder placements, Shuler and Nehring 1993, or changes in releases) may reduce the effects of these limitations. Another unanswered question is the extent to which the apparent high mortality is due to movement of brown trout outside the special regulation area; previous studies elsewhere indicate that large brown trout move longer distances (Clapp et al. 1990; Bunnell et al. 1998). To address these distinct problems this study consists of three distinct jobs.

Site Description: The proposed study site is the Smith River below Philpott Dam in the Roanoke River drainage. Philpott Dam is operated by the U. S. Army Corps of Engineers and is operated in a peaking mode, depending on energy demands and water availability. Hypolimnetic releases range from 4 to 14ºC annually below the dam, but approach 25ºC in reaches more than 20 km downstream. Temperatures between 12-19ºC result in optimal growth of juvenile brown trout (Ojanguren et al. 2001) and temperatures > 19ºC results in visible thermal stress (e.g. cessation of feeding; Elliott 1981). Lethal temperatures for brown trout range from 25 to 30ºC. Considering the information available on brown trout response to temperature, we hypothesize that higher temperatures during extended low flows would induce stress and perhaps increased movements of brown trout (McMichael and Kaya 1991). Generation flows of 1400 cfs (USGS gage 02072000) are typically released at peak demand times during week days, and minimum flows of 45 cfs are released at other times. Flows increase from base-flow to maximum levels in approximately 15 minutes. This may be accompanied by rapid declines in temperature (10ºC in 1 hr) in downstream reaches during summer months.

Despite daily fluctuation in flows and temperature, a reproducing population of brown trout exists from the dam downstream to Martinsville (32 km), and densities decrease with distance from the dam in response to increasing warm-season temperatures. There is also a gradient in sediment characteristics. The channel immediately below the dam has highly armored streambed sediments while numerous tributaries between Bassett and Koehler increase sediment loading to the stream. Generation flows likely cause displacement of young brown trout (Heggenes 1988) and invertebrates immediately below the dam. Brown trout recruitment is variable from year to year, presumably due to variation in flow during incubation and/or emergence and early rearing stages. Qualitative sampling of trout indicate that brown trout are most abundant in the middle sections from 4 to 10 km below the dam. Age-0 brown trout are rare near the dam (possible flow disruption effect) and downstream of Bassett (possible sedimentation or temperature effect; Smith 1994, 1998; Orth 2000, 2001). Previous studies in other streams indicate that redd densities are patchily distributed and correlated with densities of age-0 and older brown trout (Beard and Carline 1991). Highest redd densities are expected in glides and riffles and a high proportion of riffles facilitates production of fry (Baran et al. 1997). However, in hydropeaking situations the flow fluctuations may limit successful reproduction in otherwise suitable spawning habitat (Liebig et al. 1996). This research focuses on understanding the local topographical influence that would lead to displacement of young brown trout; experiments have shown that young grayling (*Thymallus thymallus*) may not be substantially displaced by flow increases depending on the availability of refugia and shelter seeking behavior (Valentin et al. 1994).

Most work that is routinely done to assess the suitability of spawning gravels for trout may have limited predictive power under conditions of pulsing and nonuniform flow (i.e. varied depth of flow). Reiser et al. (1989) describe some the approaches for developing a window of acceptability for flows that will protect spawning gravels and indicate wide variability in recommendations based on different methods. Suitability of spawning areas depends on at least four factors: (1) streamflows that continuously infiltrate the gravels during incubation and larval development, (2) location where local depth and velocity conditions are within ranges where spawners can construct redds and complete mating, (3) flushing of fine sediments that intrude the interstices of gravel at least once per year prior to spawning season, (4) flows must be less than those sufficient to mobilize and transport gravel.

Job 1. Characteristics of Spawning and Rearing Habitats for Brown Trout

<u>Job Objective</u>: To characterize the instream habitat conditions in areas where successful spawning and juvenile rearing of brown trout occurs.

The Smith River tailwater supports a naturally reproducing population of brown trout, thus spawning patterns, age-0 emergence and growth, as well as spawning habitat quality is of interest to fisheries managers. Monitoring of spawning activity, age-0 emergence and growth, and habitat quality continued during 2003 to build on the findings from 2000, 2001, and 2002. Continued monitoring and studies conducted during 2003 provide insight into the working hypotheses of Job 1, which address factors that could potentially impair spawning success. Those factors are:

1. High flows during the incubation period for salmonids scour spawning gravels causing catastrophic mortality of eggs and embryos depending on timing and magnitude of high flows. Over the long-term, peaking flows have had the effect of concentrating and removing spawning gravels and thus limited potential spawning habitat. Tributaries have delivered fine sediment leading to highly impacted substrates downstream of their confluence with the mainstem.
2. Short-term, pulsing flows create localized areas of high shear stress that disturb benthic habitat, thereby limiting growth and production of young trout and their prey. Daily peaking operations may therefore lead to rapid deterioration and/or failure of redds and displacement of newly-emerged age-0.

These hypotheses were addressed in 2003 by focusing effort on the following activities: 1) describing spatial and temporal patterns of spawning brown trout, 2) measuring attributes of individual redds within a subset of spawning areas, 3) measuring intragravel permeability, dissolved oxygen, and temperature within redds, 4) assessing sediment intrusion within vibert boxes placed in artificial redds in the Smith River, 4) describing temporal patterns of emergence and growth of age-0 brown trout from a subset of spawning areas, and 5) analyzing the occurrence, magnitude, and duration of the peaking flows in the Smith River over the past 13 years.

<u>Procedures</u>

Spatial and Temporal Patterns of Spawning

Redd surveys were conducted during the 2002 spawning season to evaluate spawning activity and redd characteristics. Two observers noted the presence of redds in reaches typically 200-400 m long at 12 locations within the tailwater (Figure 1). This amounted to a total search distance of 8 km or 33% of the 24 km stretch of tailwater under study. Searches were conducted by two individuals wading in the river during daylight hours at base-flow conditions. Redd surveys were conducted during October (30th), November (5-7, 14, 15, and 18-20th) and December (3, 6, 12, 19, and 20th) 2002. The presence of brown trout on redds and redd condition (e.g. new vs. old condition) was noted to ascertain spawning activity. Redd locations were noted as a GPS position, distance (m) to nearest water's edge, and drawn on a detailed map of the river reach. Redd characteristics measured for a subset of the observed redds were 1) length and width (cm) of the pit, length of the tailspill, and width of the upstream edge of the tailspill, 2) depth (cm) at the upstream edge of the pit, middle of the pit, top of the tailspill, and downstream edge of the tailspill, and 3) water velocity (m/s) taken at 2 cm above the bottom at locations measured for depth, and mean column velocity taken at 60% depth over the middle of the pit.

Substrate Permeability

Substrate permeability (i.e. intragravel flow) was measured to evaluate whether redd construction improves egg incubation habitat, whether redd condition changes dependant on river km, to compare Smith River permeability quality with values in literature, and to establish baseline permeability conditions for potential long-term trend monitoring. Permeability in the egg-pocket region of redds and beside redds (i.e. in unmodified substrate within one meter of the redd) was measured with a MARK VI standpipe and vacuum pump/chamber assembly (Barnard and McBain 1994, McBain and Trush 2000) The standpipe was driven into the gravel with a "post-driver" (specifically built for the standpipe) until the upper row of intake holes were 8 cm below the substrate surface. Thus permeability measurements are of intragravel water 8-14 cm deep, which falls within the 8-22 cm range of brown trout egg-pocket depths (Chapman 1988, Devries 1997). At each location that the standpipe was driven into the gravel, 10 permeability measurements were made without removing the standpipe. These multiple measurements were averaged to reduced measurement variability. In addition to permeability, dissolved oxygen (DO) and temperature of intragravel water was measured in the egg-pocket region of redds and beside redds with an YSI meter model 95. DO and temperature were also measured in the free-flowing channel-water above the redd.

Sediment Intrusion

Fines < 2 mm can detrimentally effect incubating eggs in redds by blocking intragravel flow and DO (Chapman 1988, Maret et al. 1993). To determine the magnitude, duration, and longitudinal trends of fine sediment intrusion into artificial redds in the Smith River a study using vibert boxes was performed. Substrate compositions representative of spawning redds at five spawning areas located 4.2, 6.2, 13.1, 14.9, and 22.8 km downstream from Philpott Dam were placed in vibert boxes. These compositions were collected from five redds in each of the five spawning areas in January 2001 using a McNeil bulk core sampler (Orth et al. 2001). From these compositions, fines < 2 mm were removed, thus intruded fines into the vibert boxes could be assessed. At each of the five spawning areas nine vibert boxes were installed where redds had been observed during fall 2000. Vibert boxes were buried under 8-10 cm of sediment that had been tossed repeatedly with a shovel in the water column to remove fines (Garrett and Bennett 1996). The 8-10 cm of sediment overlying the vibert boxes was level with the surrounding channel bottom. The burial depth represents the egg pocket depth (8-22 cm) of brown trout (Chapman 1988, Devries 1997). Vibert boxes were placed in a grid pattern of three boxes per row (rows spaced 1.0 m and boxes within rows spaced 0.5 m) where the most upstream row was installed first followed by the second then third downstream rows to prevent deposition of fines on vibert boxes when tossing sediment. Burial locations were marked with flagging tape tied to rebar and large rocks placed at the end of each row. At each spawning site three vibert boxes (starting with the most downstream row) were retrieved after 0.5, 1, and 1.5 months from June 25 to August 7, 2002. This study was carried out a second time retrieving vibert boxes after 1, 2, and 3 months from November 6, 2002 to February 8, 2003. Vibert boxes were carefully excavated until the tops of the boxes were exposed. A zip-lock bag was held open along the river bottom immediately downstream of the vibert box. The vibert box was pulled out of the sediment and placed directly into the zip-lock bag. In the laboratory, samples were dried to constant mass in a 60^{o}C oven, fines < 2 mm were separated by manual shaking (for 45 sec) through a #10 (2 mm) mesh sieve, and dry weight (0.01 g) was recorded.

Age-0 Emergence and Growth

To characterize the timing of age-0 brown trout emergence, size at emergence, short-term growth, and population estimates we have routinely monitored five known spawning sites (4.2, 6.0, 12.9, 14.1, and 22.8 km downstream of Philpott dam) since 2001. At each site we electrofished within 3 m of the riverbank for an average length of 90 m on March 12th, 19th, April 2nd, and 29th 2003. Once age-0 were present at all five sites on April 29th we allowed two weeks for more age-0 to emerge before commencing with population density sampling on May 14-16th 2003. We conducted 3-pass depletion samples within 3 m of the riverbank in 25 m reaches located at the downstream boundary of the site's spawning area (0 - 25 m), 75 m downstream (75 - 100 m), and 150 m downstream (150 - 175 m). We sampled the side of the channel that offered slower flows for potential refugia for age-0 (e.g. inside of meander bends and the non-thalweg side of the channel). Counts of age-0 per pass, total length (mm), and weight to 0.1 g was recorded. These locations and sampling methods were identical to those used in 2001 and 2002. Additionally, a single electrofishing pass in the middle of the channel paralleling all three 25 m sections at site 4.2 km was conducted to verify age-0 prefer edge habitat.

Visual assessment of the limited avoidance capabilities of age-0 < 50 mm in length supports the validity of depletion samples without a blocking mechanism. These river reaches were also sampled in conjunction with Job 2 in July using protocols and gear described in the Job 2 section of this report.

Discharge Analysis

Analysis of flow data was conducted to gain an understanding of the peak-flow release regime from Philpott dam over the past 12 years. We obtained 15-min interval discharge data dating back to January 1999 and daily mean discharge data dating back to 1946 from the U.S. Geological Survey (USGS). USGS data is from gage # 02072000 located 0.5 km below Philpott dam. Daily mean inflow into Philpott reservoir (calculated based on reservoir water elevation changes) from 1952 to present was obtained from the U.S. Army Corps of Engineers (USACE).

The 15-min interval data was analyzed for occurrence, magnitude, and duration of peaking flow. For calculation purposes we classified peak-flow as any discharge >100 cfs to conservatively designate it as any discharge greater than the typical 45-65 cfs base-flow. Occurrence is the percent time that peak-flows occurred, magnitude is the average discharge of peak-flows in cubic feet per second (cfs), and duration is the average time in hours per day that peak-flow was released. Mean daily USGS gage station data and USACE reservoir inflow data were assessed graphically.

Results and Discussion

Spatial and Temporal Patterns of Spawning

Spawning (i.e. redd development) was successfully monitored during the fall and winter of 2000 and 2002. In 2001, we initiated redd surveys based on dates of spawning occurrence the previous year. However, due to different weather conditions in 2001 the peak spawning period was earlier than in 2000 causing our surveys to miss the majority of the spawning activity. In 2000 and 2002 spawning was observed shortly after water temperatures fell below 9ºC. Redd development proceeded from downstream to upstream reaches with time in 2000 and 2002. The cause for this trend is the onset of water temperatures below 9°C downstream before upstream. Redds were first discovered on November 6-7th 2002 at downstream sites (14.9 and 19.5 km) (Table 1). A week later (Nov 14-15th) additional downstream sites (16.5 and 23.7 km) were searched and redds found. During this time frame (Nov 15 and 18-20th) redds were also found at upstream sites (3.4, 4.2, 6.2, 9.2, and 13.1 km). Redds were not found at upstream sites on previous surveys (Oct 30, Nov 5, 6, and 14th, or during Job 2 trout sampling Oct 21-25th). On the last survey days in December (19-20th) new redd development had declined, but was still occurring at upstream sites (0.5, 3.4, 4.2 km). One interesting note is the late occurrence of redd development at 22.8 km (sewage treatment plant area) where redds were observed on December 6th, but not on November 5th or 19th.

Peaks in redd occurrence occurred at similar river kilometer locations downstream of Philpott dam in 2002 and 2000 (Figure 2). In 2000 the upper reaches of the river (3 - 8 km) had the greatest number of redds (n = 147 or 78%) (Figure 2). In 2002 there was also a peak in redd occurrence from 3 - 8 km, but it only represented 42% (n = 47) of redds observed throughout the tailwater. The higher total number of redds observed during 2000 (n = 189) than 2002 (n = 113) could be from less area surveyed for redds in 2002 (33% of the tailwater) than 2000 (62% of the tailwater). Though more time (14 days) was spent at the river in 2002 than 2000 (11 days), much of that time was spent on permeability sampling. During 2001 only 3 redds were observed most likely from spawning activity occurring during a shorter than typical time-frame prior to commencing redd surveys. Despite not observing many redds in 2001 there was successful spawning based on the high recruitment observed in the spring (Orth et al. 2002).

Redds were found as close as 0.6 m to the water's edge (i.e. near the bank) as well as in the middle of the channel. Distance from redds to the nearest water's edge averaged 7.5 m, with 15.6 % of the redds being within 2 m of shore (n = 96). The width of the Smith River is fairly consistent throughout the 24 km study area and its width averages 28 m. Therefore, the majority of redds were found either near the bank (2 m) or between 5-12 m from the water's edge. We observed that some redds built close to the water's edge had low permeability due to clay substrate present in the stream bank, however, no correlation between redd distance from water's edge and substrate permeability was apparent from the data.

Redd dimensions, water depths, and water velocities were measured (n = 44) during 2002 using the same methods as in 2000 (Figures 3, 4, and 5). Water depth and mean column velocity over redds were within the generalized suitable or optimum values reported in the literature (Raleigh et al. 1986), which suggests the trout are selecting appropriate spawning habitat. Redd dimensions of upstream redds measured in 2002 are within the same size range as the majority of those measured in 2000 (Figure 3). However, the 2002 redd dimension and water depth data for downstream reaches does not reveal the trend of larger redds in deeper water downstream where larger fish are typically sampled (Figures 3 and 4). This may be an artifact of less data collected in 2002 (n = 44) than in 2000 (n = 137-148). The water velocity over redds between years 2000 and 2002 was similar in magnitude and trend (Figure 5). The water velocity data of year 2002 is more evenly distributed with distance from the dam than the clustered data of year 2000, which is due to collection of data at more locations throughout the tailwater in 2002 (Figure 5).

Substrate Permeability

Intragravel permeability was greater within the redd egg-pocket than in the undisturbed substrate beside the redd for all redds sampled but one (Figure 6). Mean permeability in (4499 cm/hr; SD = 2055) and beside (2166 cm/hr; SD = 1781) redds was statistically different (based on ± 2 SE). Permeability in and beside individual redds was statistically different (based on 95% confidence intervals) for 27 of the 30 redds measured (Figure 6). These results demonstrate that the redd modifies the substrate, in-turn enhancing permeability for increased egg survival. The lack of a trend in permeability from up to downstream is not due to similar substrate conditions throughout the tailwater. Both the sediment intrusion data (Figure 7) and bottom coverage data (Orth et al. 2001) shows that percent fine sediment increases with downstream distance from the dam. The lack of a longitudinal trend is from brown trout selecting suitable substrate locations throughout the tailwater.

The DO level (mg/l) followed a consistent trend, being greatest in the free-flowing channel-water, followed by the egg pocket, with intragravel water beside the redd having the least DO (Figure 8). Like permeability, no longitudinal trend is apparent. DO measured during daylight hours of November and December in the redd egg pocket averaged 11 mg/l (SD = 0.52) which is not limiting to embryo survival (Raleigh et al. 1986, Chapman 1988). Water temperature was very similar among the in-redd, beside-redd, and in-channel locations; among these locations temperature never differed more than 0.2ºC.

Sediment Intrusion

Fine sediment intrusion into vibert boxes increases with distance downstream of Philpott dam (Figure 7). Though this trend is not significant based on ±2 standard error bars, it parallels the trend of bottom coverage data of sand (< 2 mm) measured in 2001 (Orth et al. 2001). This trend also corresponds with age-0 trout recruitment, which is greatest upstream where there is the least fine sediment intrusion. Interestingly, there is no trend of increased intrusion over time. This most likely indicates that intrusion quickly reaches an equilibrium point. Sites 13.1, 14.9, and 22.8 km had samples with >10 % fines (i.e. weight of vibert box contents >2 mm divided by the weight of fines <2 mm intruded) (Figure 7). Survival of salmonid embryos is known to decrease as the percentage of fines increases above 10-20% (Chapman 1988, Maret et al. 1993). Fine sediment intrusion into redds at downstream sites may be resulting in lowered survival of eggs and thus a lower abundance of age-0 fish when compared with upstream locations (Figures 9 and 10).

Age-0 Emergence and Growth

In 2001 and 2002 age-0 emergence occurred from late February to early March. The timing of emergence is linked to the water temperature during the incubation period (Crisp 1981, 1988). Emergence as early as late February has also been reported for the White river tailwater system in Arkansas (Pender and Kwak 2002). Early emergence in tailwaters is caused by reservoir release water being warmer than free-flowing river water which is cooled by cold winter air temperatures. However, in 2003 emergence did not occur until late March due to a cold winter causing greater cooling to the released water below the dam. The 2002/2003 incubation period had less degree days (i.e. coldest temperatures) throughout the majority of the tailwater than all other monitored years (1999-2002) (Figure 11). In 2003 it was not until March 19th that we first found one age-0 on March 19th which had the majority of its yolk present, six age-0 on April 2nd, and 57 age-0 on April 29th. Emergence occurred first at upstream sites 4.2

and 6.0 km below the dam, and emergence was not observed at 12.9, 14.1, and 22.8 km until April 29th. In 2001 emergence also occurred upstream first. This occurs because water temperatures near the dam are warmer than those downstream cooled by winter air temperatures (Figure 11, 12, and 13).

In July and August 2003 during the growth period for age-0, water temperatures near the dam (0.7-5.1 km) were warmer than in all previous year monitored (Figure 12). Temperatures downstream from 10.2 to 24.3 km were more synchronous than in previous years (Figure 13). The unique temperature trends in summer 2003 are most likely from a lack of strong stratification in the reservoir caused by 30-52% more rain during 2003 than in 1999, 2000, 2001, or 2002 (percentage calculated for months January through June from rain gage data at Philpott dam).

Sampling to obtain population densities on May 14-15th and July 10th and 14-18th, 2003 revealed the density of age-0 was greatest upstream (with the exception of immediately below the dam at 0.7 km) and diminished with distance from the dam (Figures 9 and 10). The same trend has occurred in all previously sampled years (Figures 9 and 10). Additionally, May 2003 age-0 density estimates are significantly lower (based on 95% confidence intervals) than in May 2001 and 2002 at sites 6.0, 12.9, 14.1, and 22.8 km (Figure 9). This is also the case for July 2003 where densities were significantly lower at seven of 11 sampled sites compared to June of 2000, 2001, and 2002 (Figure 10). The cause of reduced population density is potentially due to increased occurrence, magnitude, and duration of peak-flow during the emergence period in 2003 (Table 2). Peak-flows were more prevalent in 2003 due to frequent rain-events which caused the USACE to release more water to maintain a normal reservoir level. During the typical age-0 emergence months of February, March, and April the occurrence of peak-flow averaged over those months in 2003 was 40% compared to 5% in 2002, 6% in 2001, and 7% in 2000 (Table 2). The magnitude during 2003, 2001, and 2000 was similar at 1281, 1018, and 1043 cfs respectively, but lower in 2002 at 572 cfs. Therefore, it is possible the lower magnitude in 2002 enabled the high recruitment seen that year and the high occurrence of peak-flow in 2003 caused the poor recruitment (Figures 9 and 10).

Another possibility for poor recruitment in 2003 is that egg survival during the incubation period of approximately November 15, 2002 through March 15, 2003 was worse than in previous years. Egg survival is related to water temperature, fine sediment intrusion, and DO (Chapman 1988, Pender 1998, McBain and Trush 2000). DO levels in redds recorded in November and December 2002 were above those required by brown trout eggs for survival (Raleigh et al. 1986, Chapman 1988, Maret et al. 1993). Fine sediment intrusion measured in summer and fall 2002 occurred rapidly and increased with distance downstream of the dam. Though these measurements were not done in 2000 or 2001 there has been no noticeable large scale changes to the river substrate composition from up to downriver to effect DO and fine sediments. Thus, water temperature is a likely factor to affect egg survival from year to year. The 2000/2001 and 2002/2003 incubation periods experienced 225-262°C less degree days (i.e. sum of daily mean water temperature) on average throughout the Smith River tailwater than in 1999/2000 and 2001/2002 (Figure 11). Whereas, the 1999/2000 and 2001/2002 incubation periods experienced nearly the same degree days (8°C difference). Because 2000/2001 had a cold winter like that of 2002/2003, and 2000/2001 had better emergence and recruitment, the reason 2003 had low recruitment is most likely the high occurrence of peak-flows.

The length and weight of age-0 brown trout at emergence in March was similar throughout the tailwater (Figure 14). By June/July, the age-0 trout had grown larger at downstream locations due to warmer water temperatures. Age-0 length and weight increased significantly between March and May, and between May and June/July; with the exception of May 2003. Due to the later emergence of age-0 trout in 2003, the average length and weight of age-0 sampled in May 2003 were similar to samples from March 2001 and 2002 (Figure 14).

To verify that age-0 prefer edge habitat and confirm the legitimacy electrofishing only within 3 m of the riverbank, we electrofished the middle of the channel parallel to the three 25 m sections normally sampled at site 4.2 km. Site 4.2 km was chosen because it consistently produces the most age-0. Only four age-0 trout were found and all were under the same rock in the 0-25 m sampled section. Four age-0 found in the middle of the channel is substantially lower than the 47 age-0 which were found in the edge habitat for this section. In the edge habitat 109 and 44 age-0 were found in sections 75-100 and 150-175 m respectively, compared to zero age-0 in the middle of the channel. These results confirm that age-0 prefer edge habitat and validate our sampling method (Pender and Kwak 2002). The four age-0 found in the middle of the channel could have been recently emerging from a redd. We electrofished over known mid-channel redd locations at this site, but saw no other age-0.

Discharge Analysis

During the years of this study starting in fall 1999 through 2002, the average percent time (based on annual and water year) that flow has been peaked is less than during the previous eight years for which we have 15-min interval discharge data (Table 2). Specifically, the average percent time that flow was peaked based on the water-year for 1999-2002 was 10%, versus 25% for 1992-1998. The percent time of peak-flow release in 2003 will likely be greater or at least similar to years during the 1990's due to the high occurrence of release during January-July 2003. Additionally, for the majority of 2001 and 2002 the magnitude of peak-flows were approximately half that which is typical of most years. Since multiple years of this study have experienced flow regimes atypical of the 1991-1998 norm, we must consider that conclusions drawn from the Smith River study may not represent all flow conditions within the operating range of the USACE. However, the wide range of flow conditions experienced during the Smith River study have provided us with somewhat of a natural experiment. The change in flow conditions from one year to the next, changing between high and low release occurrence, magnitude, and duration, have enabled us to evaluate the relevance that flow regime played, for example, on the recruitment of age-0 trout from year to year.

Mean daily flow data also depicts the more "mild" release regime during 1999-2002 compared to the history of the flow operations of Philpott dam dating back to 1953 (Figure 15). The lessened discharge in first few years following 1953 are due to filling the reservoir. Prior to the completion of Philpott dam in 1952 the USGS gage data recorded the unregulated flow regime of the Smith River (Figure 15). The flow regulation and flood prevention capabilities of Philpott dam are obvious when the mean daily discharge before and after 1952, as well as between the USACE computed inflow and USGS discharge are compared. Flow prior to 1952 and inflow into the reservoir is typified by flood events where as the outflow from the dam has consistent low and high flows.

Preliminary Conclusions

Spatial and Temporal Patterns of Spawning

- During multiple years of this study we have observed spawning to initiate shortly after water temperatures fall below 9ºC, which concurs with the literature (Raleigh 1986). Additionally, redd development has proceeded from downstream to upstream reaches with time. The cause for this trend is the onset of water temperatures below 9°C downstream before upstream. Initiating our spawning surveys and observing the appropriate reaches at the correct time during October-December can be based on monitoring the water temperature in the Smith River.

- The water depth and mean column velocity over redds were within the generalized suitable or optimum values reported in the literature, which suggests the trout are selecting usable spawning habitat. This is further suggested by the lack of a diminishing trend in redd intragravel permeability from up to downstream despite increasing fine sediments downstream. The lack of a longitudinal trend is from brown trout selecting suitable substrate locations throughout the tailwater.

Substrate Permeability

- Intragravel permeability was greater within the redd egg-pocket than in the undisturbed substrate beside the redd, which confirms that the redd modifies the substrate to enhance egg survival. The DO level in the redd egg pocket averaged 11 mg/l which is not limiting to embryo survival (Chapman 1988). Despite acceptable DO levels at the time of measurement and brown trout selecting the best spawning locations available, there is still a decline in age-0 recruitment with increasing distance below Philpott dam.

Sediment Intrusion

- Fine sediment intrusion into redds at downstream sites may be resulting in lowered survival of eggs and thus a lower abundance of age-0 fish when compared with upstream locations. With increasing distance from Philpott dam fine sediment intrusion increases, whereas age-0 trout recruitment declines. In the upper reaches (4.2 km) age-0 recruitment is at its highest within the tailwater and coincides with the lowest fine sediment intrusion recorded within vibert boxes. Immediately below the dam there is no recruitment due to strong peak-flows and lack of suitable substrate.

Age-0 Emergence and Growth

- In 2003 the recruitment of age-0 trout was lower than in previous years of study; potentially due to increased occurrence, magnitude, and duration of peak-flow during the emergence period.

- Emergence in 2003 occurred approximately a month later than in 2001 and 2002. The cause is most likely from a decline in degree days due to a cold winter. Age-0 trout emerged first in upstream reaches because water temperatures near the dam are warmer than those downstream cooled by winter air temperatures. Despite age-0 first emerging upstream, by May it is the age-0 downstream that have grown the largest due to warmer water temperatures downstream during Spring.

- We confirmed that age-0 occupy edge habitat after emergence and during the initial growth period (~ March - June).

Discharge Analysis

- The first years of this study (1999-2002) experienced flow regimes atypical of the 1991-1998 norm. Occurrence of peak-flows during the study has been less than typical, thus we must consider that conclusions drawn from the Smith River study may not represent all flow conditions within the operating range of the USACE. This year (2003) the flow regime has had a much higher occurrence of peak-flows and will provide a basis for comparison.

<u>Future Research and Job Schedule</u>

Data collection in the field will continue to document spatial and temporal characteristics of spawning, emergence, and recruitment via redd surveys and age-0 sampling during fall 2003 and spring 2004. The habitat sections sampled to determine recruitment success will be characterized by measuring parameters such as cover, depth, water velocity, substrate, and embeddedness. A study to evaluate the potential downstream displacement of age-0 trout has been postponed to spring 2004 due to frequent peak-flows and poor recruitment in 2003. In addition to and in case the displacement study is unsuccessful we plan to measure the near-bank water velocity during the base to peak-flow transition to better describe the age-0 trout rearing habitat. Comparing this velocity data to age-0 trout swimming speeds in the literature will help determine whether displacement of age-0 is occurring. During the water velocity and/or downstream displacement fieldwork we also hope to visually observe the reaction of age-0 trout to the rising flow.

From the data job 1 has collected during the years of this study we have numerous ideas to evaluate and test statistically during the final year of the project. Further analysis of the substrate permeability data will involve comparing the magnitude of the permeability measurements (cm/hr) to those in other rivers presented in the literature; as well as statistical tests for difference between permeability in and beside redds (e.g. non-parametric paired test). The weight or percent of intruded sediment into the Vibert boxes will be related to literature reported embryo survival rates within redds. The downstream trend of increasing sediment intrusion paralleling trends in the bottom coverage data will be further evaluated (e.g. repeated measures tests). To determine that recruitment success is driven primarily by flow and/or temperature in the Smith River and not varying spawner biomass we hope to evaluate the constancy of spawner biomass over the years of study. Age-0 recruitment density declines longitudinally in correlation with habitat suitability. Another consideration to be evaluated is that the body condition and/or density of adult female brown trout differs longitudinally in correlation age-0 densities. We plan to statistically test whether the age-0 population densities differ or correlate with the water temperature and flow data over the years of study (or other possible variables such as fine sediment intrusion, redd substrate composition, intragravel permeability, water quality, and/or habitat characteristics). Additionally, we will test for age-0 population difference among sites, within sites, and over time. The redd dimensions, water depth, and water velocity data over redds requires statistical analysis (e.g. analysis of covariance; tests of correlation) to determine if differences between years and up to downstream trends exist.

Redd densities in the Smith River will be compared to literature reported densities of "healthy" brown trout populations. To build on an earlier Smith River water quality analysis presented in Orth et al. (2001) we hope to obtain updated alkalinity data from the DEQ to assess the low alkalinity levels as a cause for poor trout productivity and reproduction.

Job 1 Schedule. All aspects of Job 1 are on schedule with no significant changes anticipated at this time. Reporting period extends to bold line.

Calendar Year	1999	2000				2001				2002				2003				2004			
Project Year	Year 1			Year 2			Year 3			Year 4				Year 5							
Quarter	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2
Purchase supplies		X																			
Recruit students	X																				
Assemble supplies		X																			
Redd surveys					X				X				X				X				
Fine sediment intrusion													X	X							
Substrate permeability													X	X							
Downstream displacement																		X			
Redd and fry monitoring			X			X	X			X	X			X	X			X	X		
Rearing habitat surveys				X				X				X				X					
Data analyses								X	X			X	X			X	X	X	X		
Manuscript preparation										X	X			X	X			X	X		
Final report																				X	

Table 1. Redd surveys were conducted at 12 locations (km) below Philpott dam from October 30th to December 20th, 2002. The date and distance below Philpott dam (km) at which a redd survey took place is indicated by the bold squares. The numbers within the squares are the number of redds found which had not been present during previous surveys

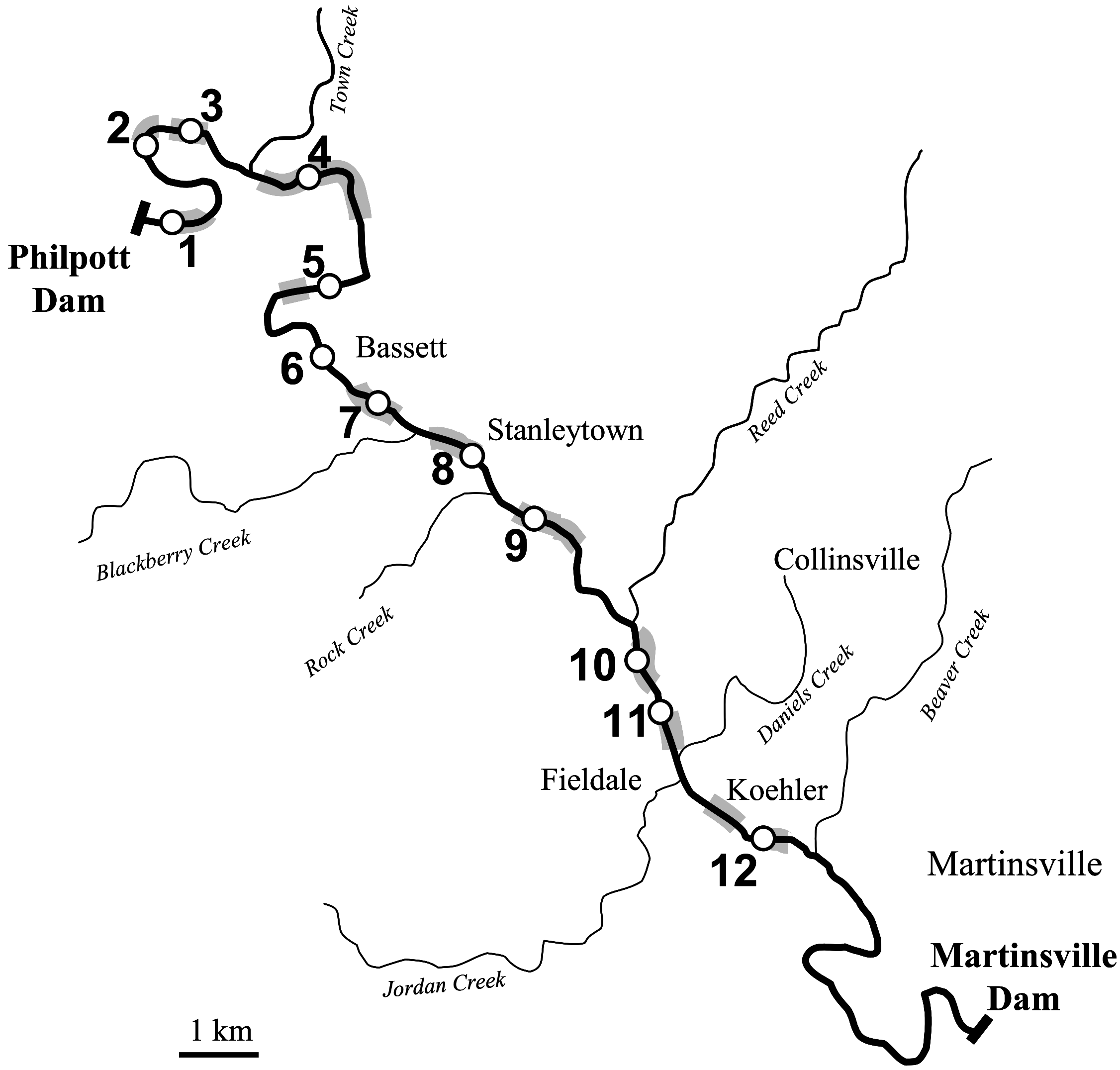
Date	Distance below dam (km)											
	0.5	3.4	4.2	6.2	9.2	13.1	14.9	16.5	19.5	20.6	22.8	23.7
10/30/02			$\boxed{}$									
11/05/02						$\boxed{}$				$\boxed{}$	$\boxed{}$	
11/06/02			$\boxed{}$	$\boxed{}$			$\boxed{4}$					
11/07/02							$\boxed{}$		$\boxed{3}$			
11/14/02						$\boxed{}$		$\boxed{2}$	$\boxed{1}$			$\boxed{4}$
11/15/02				$\boxed{1}$				$\boxed{4}$				$\boxed{1}$
11/18/02		$\boxed{1}$	$\boxed{3}$			$\boxed{2}$						
11/19/02				$\boxed{}$	$\boxed{2}$	$\boxed{2}$					$\boxed{}$	
11/20/02				$\boxed{3}$		$\boxed{9}$	$\boxed{9}$					
12/03/02	$\boxed{1}$	$\boxed{6}$	$\boxed{9}$									
12/06/02				$\boxed{12}$		$\boxed{2}$	$\boxed{2}$				$\boxed{10}$	
12/12/02	$\boxed{1}$											
12/19/02			$\boxed{7}$			$\boxed{5}$						
12/20/02	$\boxed{2}$	$\boxed{5}$										

Table 2. Discharge statistics evaluating 12.5 yrs of 15-min interval USGS data from the gage #02072000 near Philpott dam on the Smith River, VA. Statistics calculated evaluate the occurrence, magnitude, and duration of peaking flow (for calculations peak-flow was classified as >100 cfs). Occurrence is the percent time that peak-flows occurred. Magnitude is the average discharge in cubic feet per second (cfs) of the peak-flow. Duration is the average time in hours per day that peak-flow was released. Data is also averaged annually based on the calendar year (Jan 1 - Dec 31) and the water year (Oct 1 - Sept 30).

	1991	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**
Occurrence (% time)													
January	40%	27%	37%	22%	30%	32%	53%	11%	7%	7%	6%	14%	32%
February	16%	10%	28%	32%	18%	26%	59%	51%	7%	8%	6%	8%	26%
March	30%	20%	58%	42%	19%	22%	78%	39%	7%	6%	6%	4%	35%
April	32%	32%	53%	35%	9%	23%	56%	38%	10%	6%	6%	5%	58%
May	34%	28%	31%	16%	15%	32%	52%	28%	12%	9%	20%	5%	35%
June	23%	55%	18%	14%	16%	28%	31%	18%	13%	6%	20%	4%	63%
July	17%	19%	18%	17%	21%	16%	17%	16%	16%	6%	17%	5%	38%
August	16%	18%	15%	35%	16%	29%	15%	16%	29%	6%	32%	5%	
September	18%	23%	18%	21%	21%	40%	19%	20%	18%	6%	17%	5%	
October	7%	13%	14%	9%	7%	40%	12%	11%	7%	6%	17%	5%	
November	3%	25%	6%	11%	4%	53%	8%	10%	7%	6%	19%	5%	
December	4%	24%	11%	10%	4%	69%	7%	10%	7%	6%	13%	8%	
Annual Avg.	**20%**	**24%**	**26%**	**22%**	**15%**	**34%**	**34%**	**22%**	**12%**	**7%**	**15%**	**6%**	
Water-Yr Avg.		**20%**	**28%**	**22%**	**16%**	**22%**	**45%**	**22%**	**12%**	**7%**	**12%**	**9%**	
Magnitude (cfs)													
January	1249	1145	1228	1129	1351	1413	655	1162	1153	1118	1033	1017	638
February	1203	1155	1233	1179	1251	1225	650	1220	1205	1054	1040	641	1280
March	1223	1219	1601	1351	1251	1071	658	1240	1169	1028	1006	558	1260
April	1212	2156	1265	1270	1211	1223	653	1232	1281	1048	1007	518	1304
May	1196	1223	1247	1227	1230	1068	658	1222	749	693	753	519	1294
June	1232	1437	1242	1138	1240	1681	757	1222	1287	1035	647	525	1358
July	1197	1217	1255	1190	1131	1197	1187	1230	1255	1044	650	536	1350
August	1227	1227	1238	1261	1263	1255	1219	1228	1059	1060	638	490	
September	856	1114	1023	801	767	1263	867	829	894	1046	606	544	
October	1151	931	841	1220	1181	657	1110	1120	1075	1055	623	509	
November	1116	1237	1063	1146	1120	659	1112	1307	1234	1014	735	526	
December	1095	1238	1223	1192	1101	826	1171	1279	1193	1006	1001	684	
Annual Avg.	**1187**	**1333**	**1278**	**1198**	**1185**	**1058**	**748**	**1195**	**1103**	**1006**	**737**	**650**	
Water-Yr Avg.		**1271**	**1228**	**1139**	**1188**	**1233**	**787**	**1165**	**1147**	**1052**	**871**	**642**	

Table 2. Continued.

	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003
Duration (hrs)													
January	10	6	9	5	7	8	13	3	2	2	2	3	8
February	4	2	7	8	4	6	14	12	2	2	1	2	6
March	7	5	14	10	5	5	19	9	2	2	2	1	9
April	8	8	13	9	2	5	14	9	2	2	2	1	14
May	8	7	7	4	4	8	12	7	3	2	5	1	8
June	5	13	4	3	4	7	7	4	3	2	5	1	15
July	4	5	4	4	5	4	4	4	4	2	4	1	9
August	4	4	4	9	4	7	4	4	7	2	8	1	
September	4	6	4	5	5	10	4	5	4	2	4	1	
October	2	3	3	2	2	10	3	3	2	1	4	1	
November	1	6	1	3	1	13	2	2	2	2	5	1	
December	1	6	3	2	1	17	2	2	2	2	3	2	
Annual Avg.	**5**	**6**	**6**	**5**	**4**	**8**	**8**	**5**	**3**	**2**	**4**	**1**	
Water-Yr Avg.		**5**	**7**	**5**	**4**	**5**	**11**	**5**	**3**	**2**	**3**	**2**	



Site #	1	2	3	4	5	6	7	8	9	10	11	12
km below dam	0.5	3.4	4.2	6.2	8.9	11.9	13.1	14.9	16.5	19.5	20.6	23.7

Figure 1. Map of the Smith River tailwater between Philpott Dam and Martinsville Dam with sampling sites numbered upstream to downstream. Gray areas indicate reaches surveyed for redds during October 30th to December 20th, 2002.

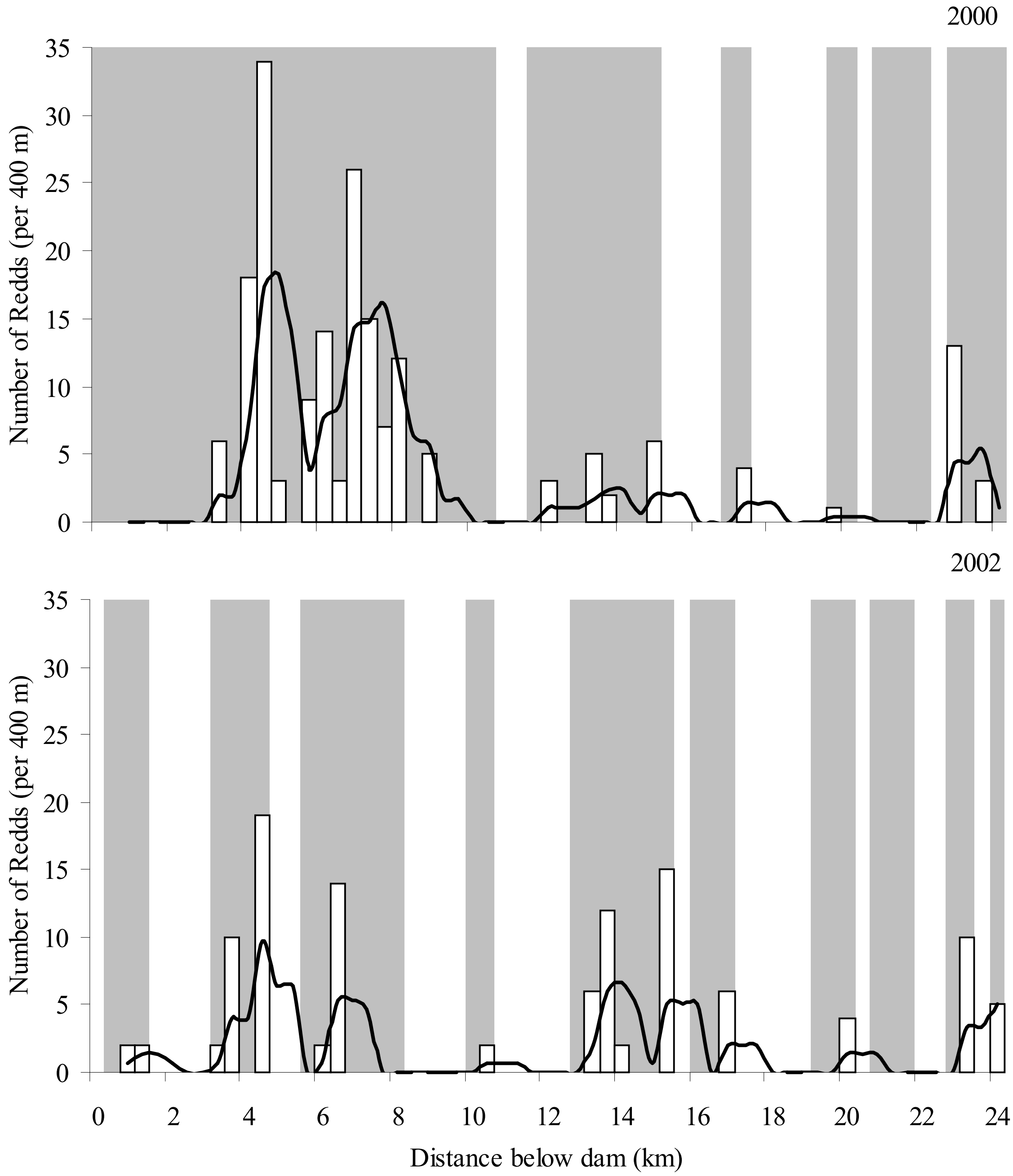


Figure 2. Distribution of redds downstream of Philpott Dam discovered from November 7th, 2000 to January 9th, 2001, and from November 7th to December 20th, 2002 (2001 redd survey data is not shown because only three redds were found most likely due to spawning occurring before surveying). Bars indicate the actual count of redds in 400 m reaches, trend lines are 3-point moving averages on the actual count, and the gray areas represent the approximate areas surveyed.

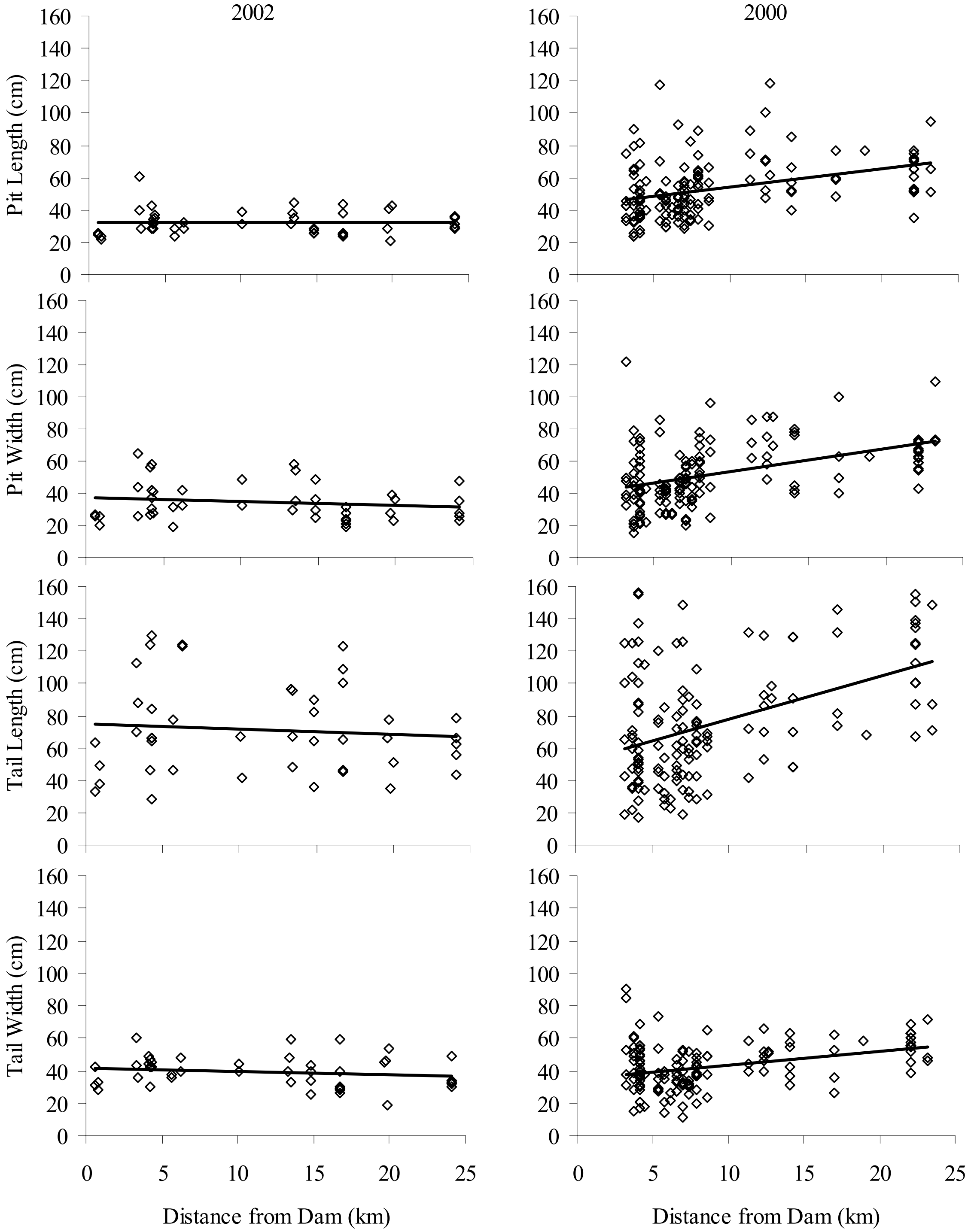


Figure 3. Redd dimensions; length and width (cm) of Redd pit and tailspill during 2002 (n = 44) and 2000 (n = 137-148) with increasing downstream distance (km) from Philpott dam, Bassett, VA .

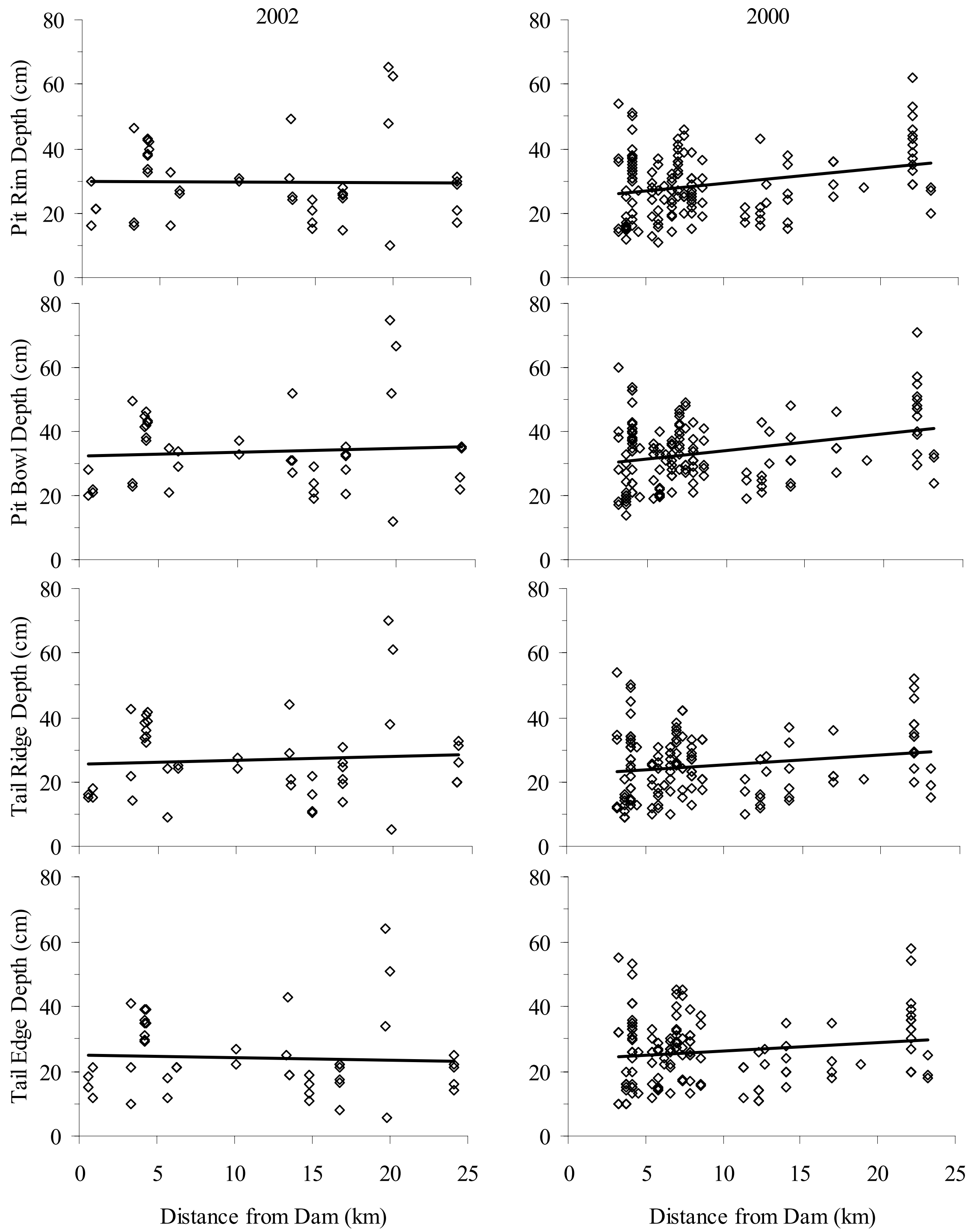


Figure 4. Water depth (cm) over redds during 2002 (n = 44) and 2000 (n = 138-148) with increasing downstream distance (km) from Philpott dam, Bassett, VA .

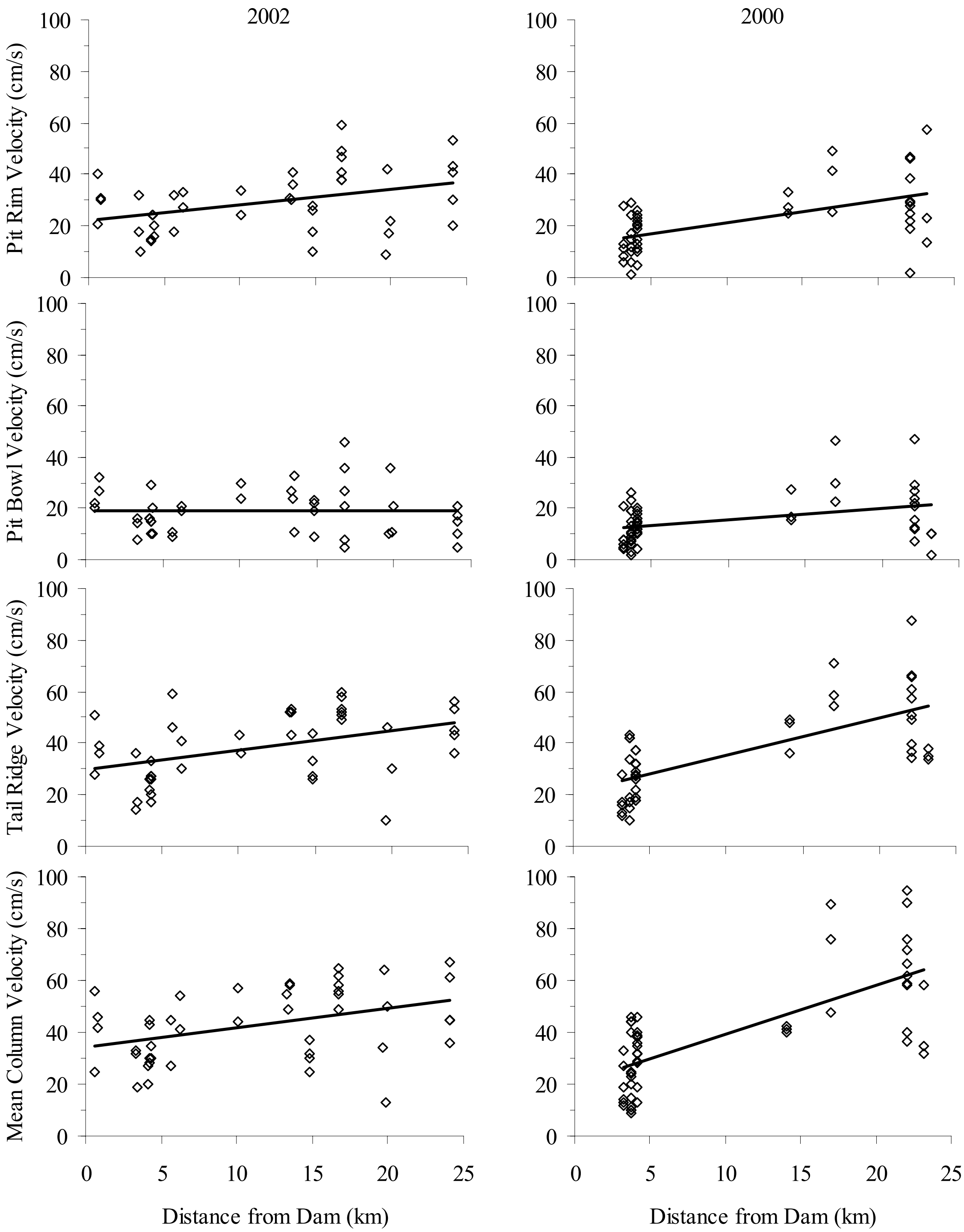


Figure 5. Water velocity (cm/s) over redds during 2002 (n = 44) and 2000 (n = 46-54) with increasing downstream distance (km) from Philpott dam, Bassett, VA .

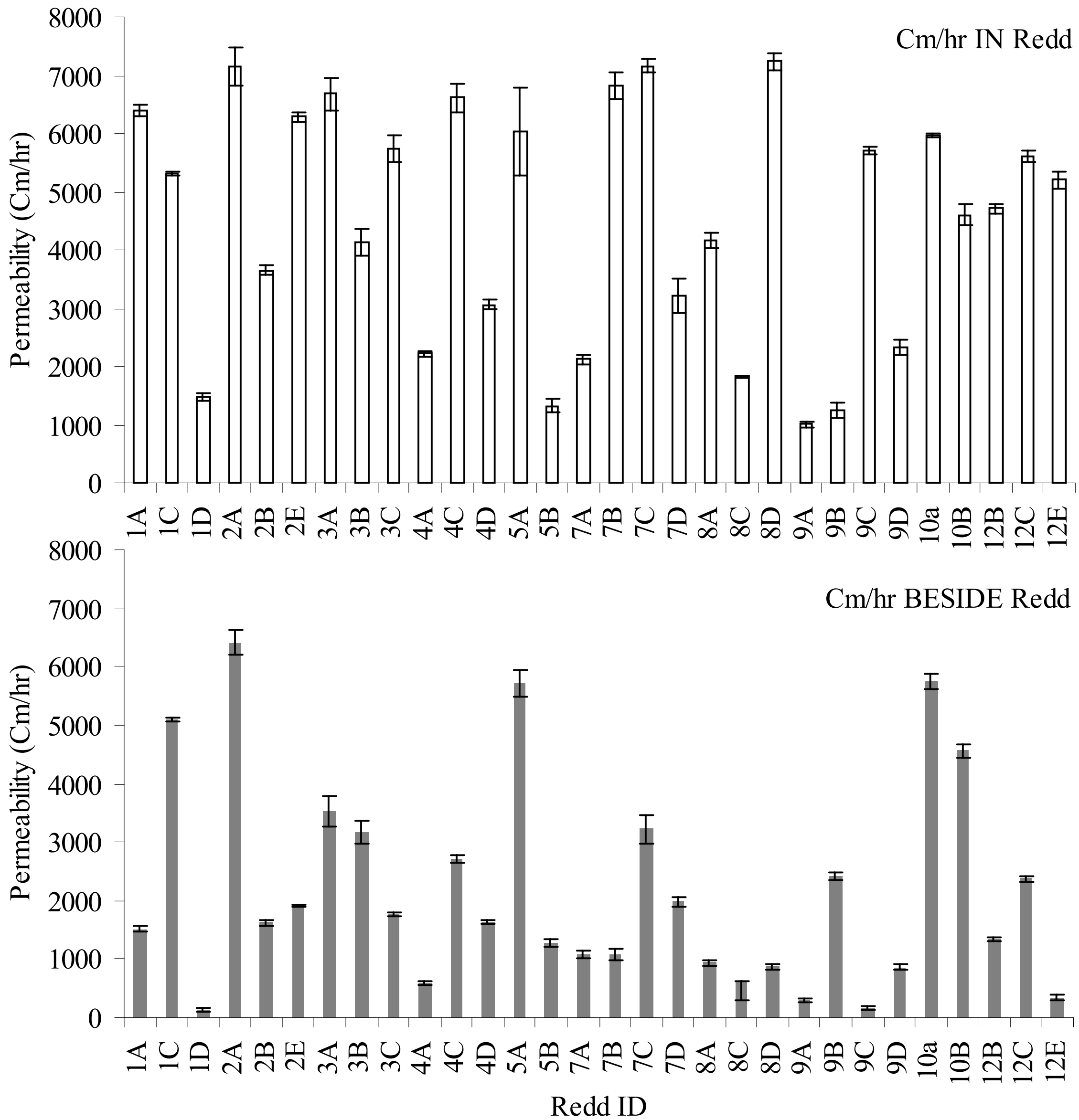


Figure 6. Permeability (cm/hr) within the redd egg-pocket area (top panel) and in undisturbed substrate beside the redd (bottom panel). Error bars represent 95% confidence intervals. The redd ID number corresponds to the sample sites shown in figure 1.

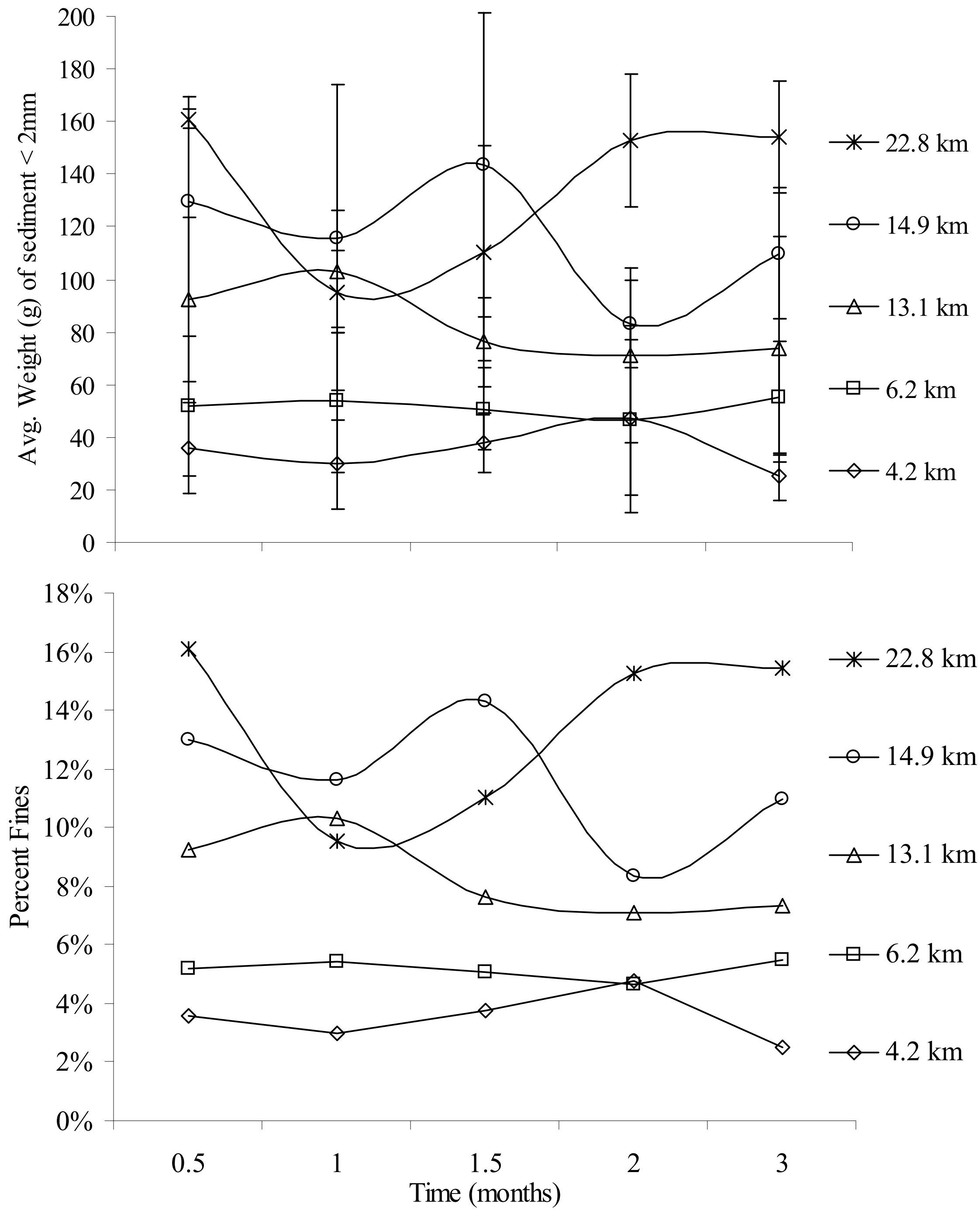


Figure 7. Average weight (g) of sediment <2 mm (top panel) and percent fines (lower panel) intruded into vibert boxes at 4.2, 6.2, 13.1, 14.9, and 22.8 km below Philpott dam over time (0.5, 1, and 1.5 months from 6/25 to 8/7/02, and 2 and 3 months from 11/06/02 to 02/08/03). Error bars represent 2±SE.

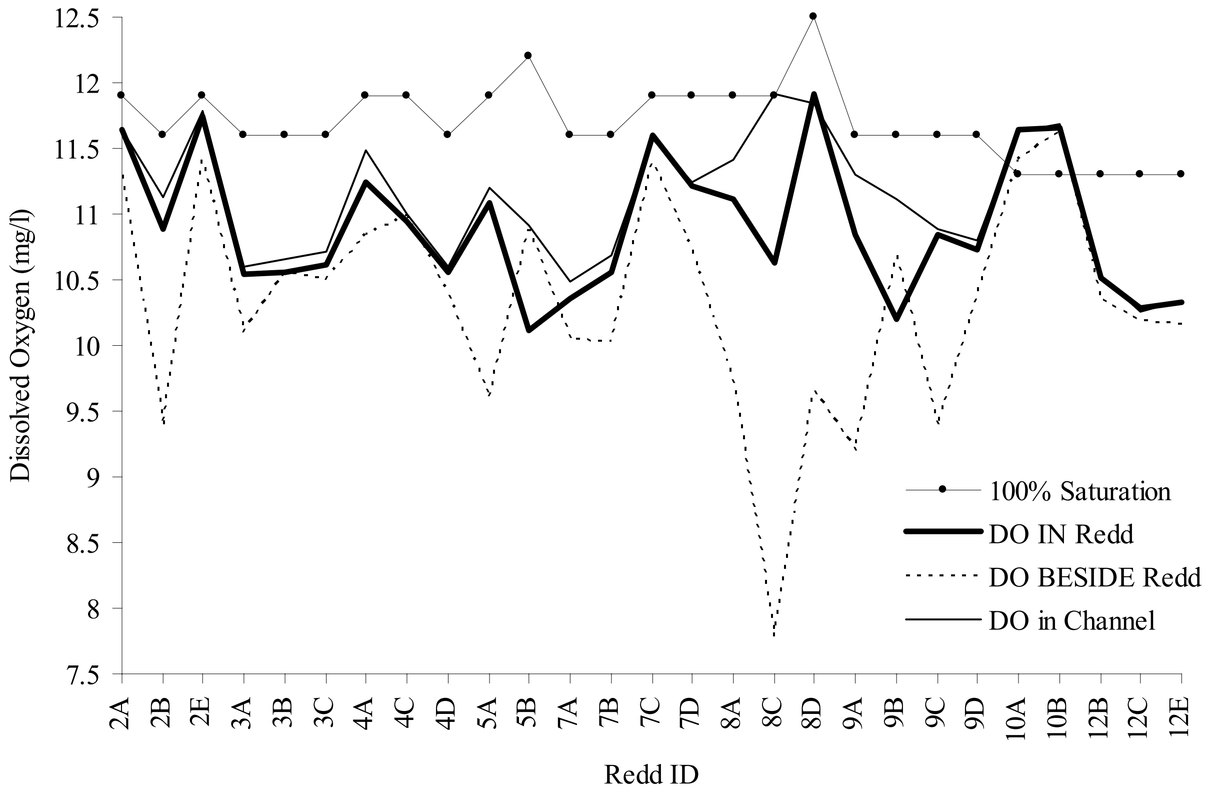


Figure 8. Dissolved oxygen (DO) (mg/l) was measured at 27 redds, specifically within the redd egg pocket area, beside the redd (~1 m from the redd pit), and in the free-flowing channel-water. For comparison, mg/l of DO if water is 100% saturated is shown. The redd ID number corresponds to the sample sites shown in figure 1.

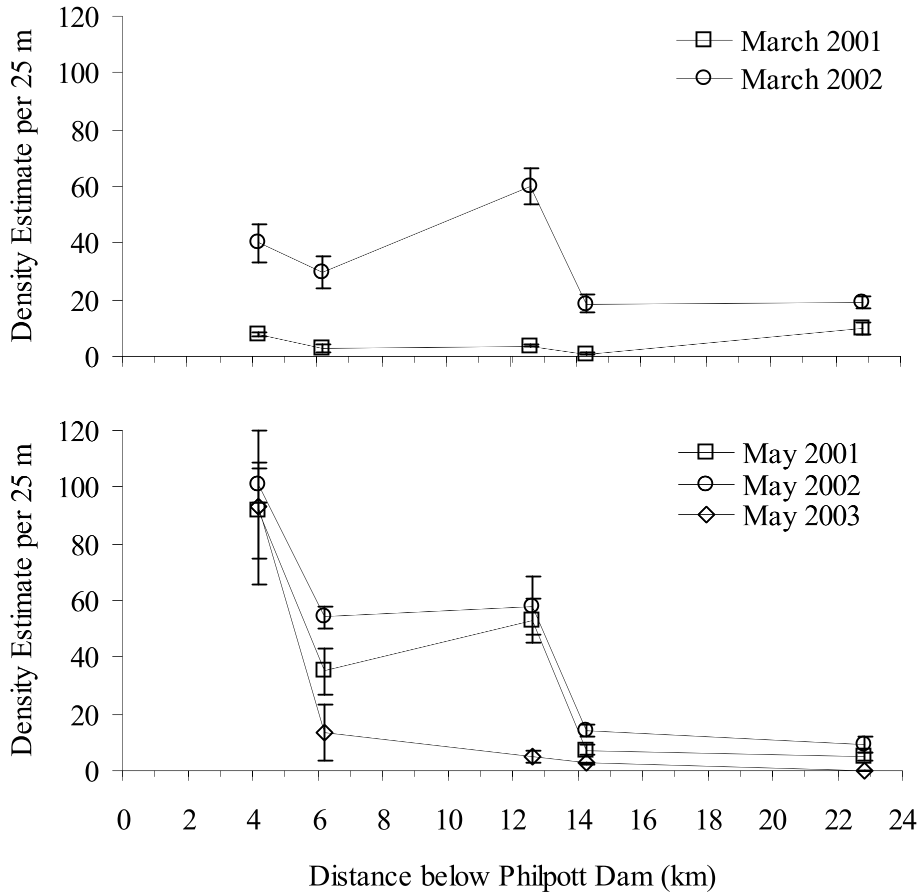


Figure 9. Density estimates (Microfish software) for age-0 brown trout sampled by 3-pass depletion backpack electrofishing in March 2001 and 2002, and May 2001-2003. Error bars represent 95% confidence intervals.

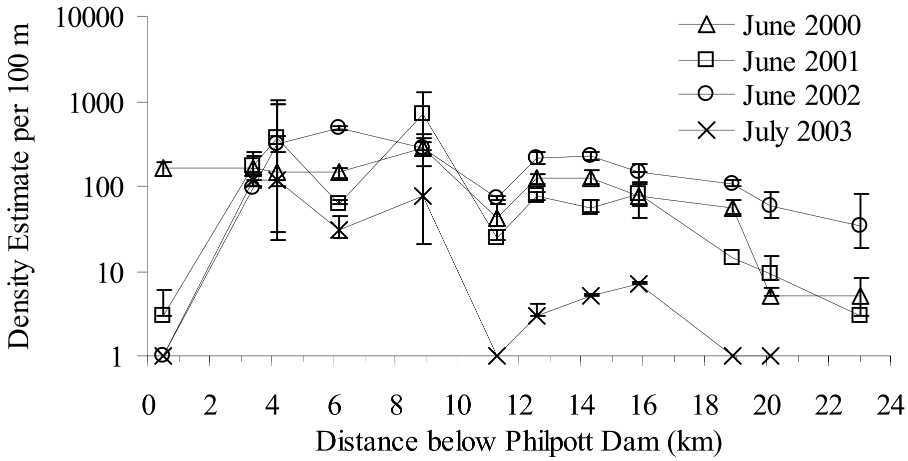


Figure 10. Density estimates (Microfish software) for age-0 brown trout sampled by 100 m, full channel, 3-pass depletion barge electrofishing in June 2000-2002. Error bars represent 95% confidence intervals. Density estimates are shown on a log scale to better differentiate between sampling periods.

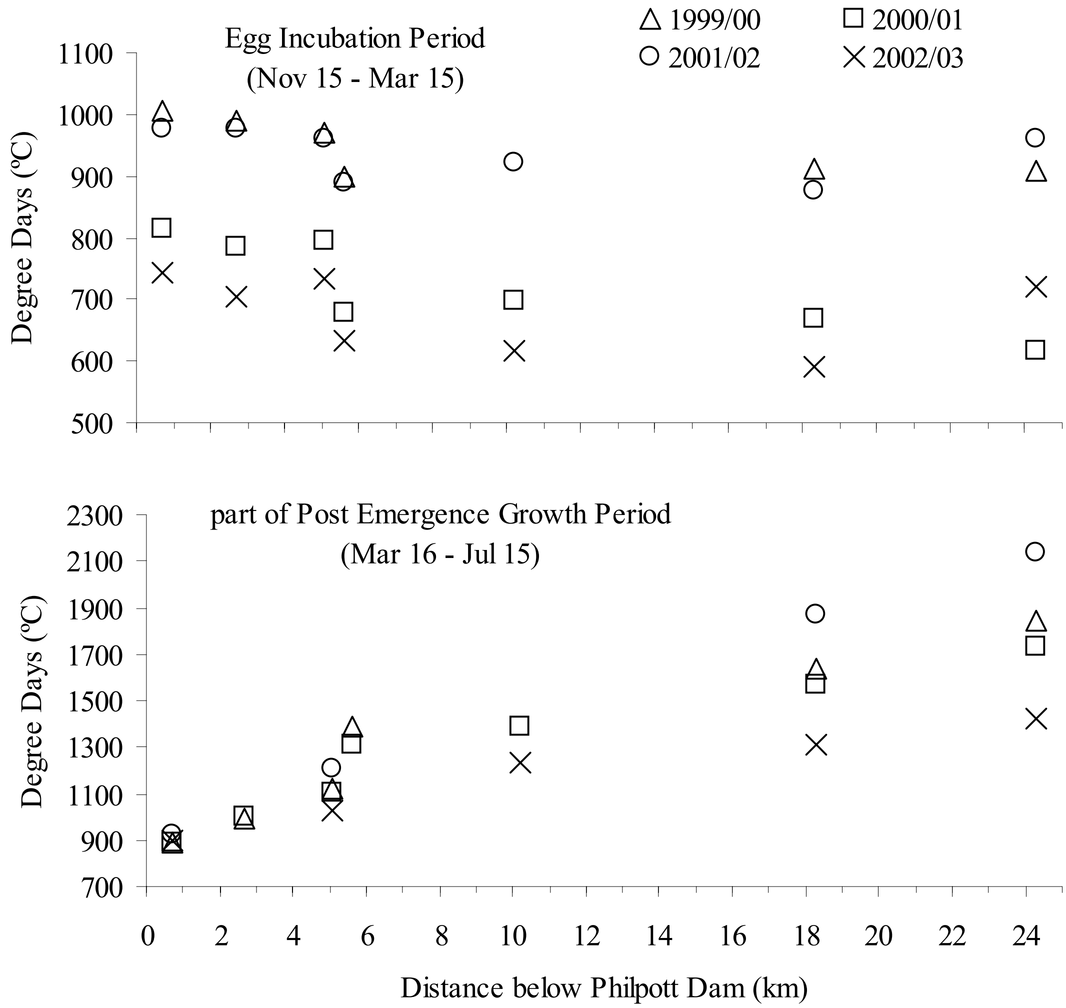


Figure 11. Degree days (°C) (i.e. sum of daily mean water temperature) at 0.7, 2.7, 5.1, 5.6, 10.2, 18.3, and 24.3 km below Philpott dam in the Smith River, VA during the typical egg incubation period (11/15 - 03/15; n = 122) and part of the growth period after emergence (03/16 - 07/15; n = 122) during 1999/00, 2000/01, 2001/02, and 2002/03.

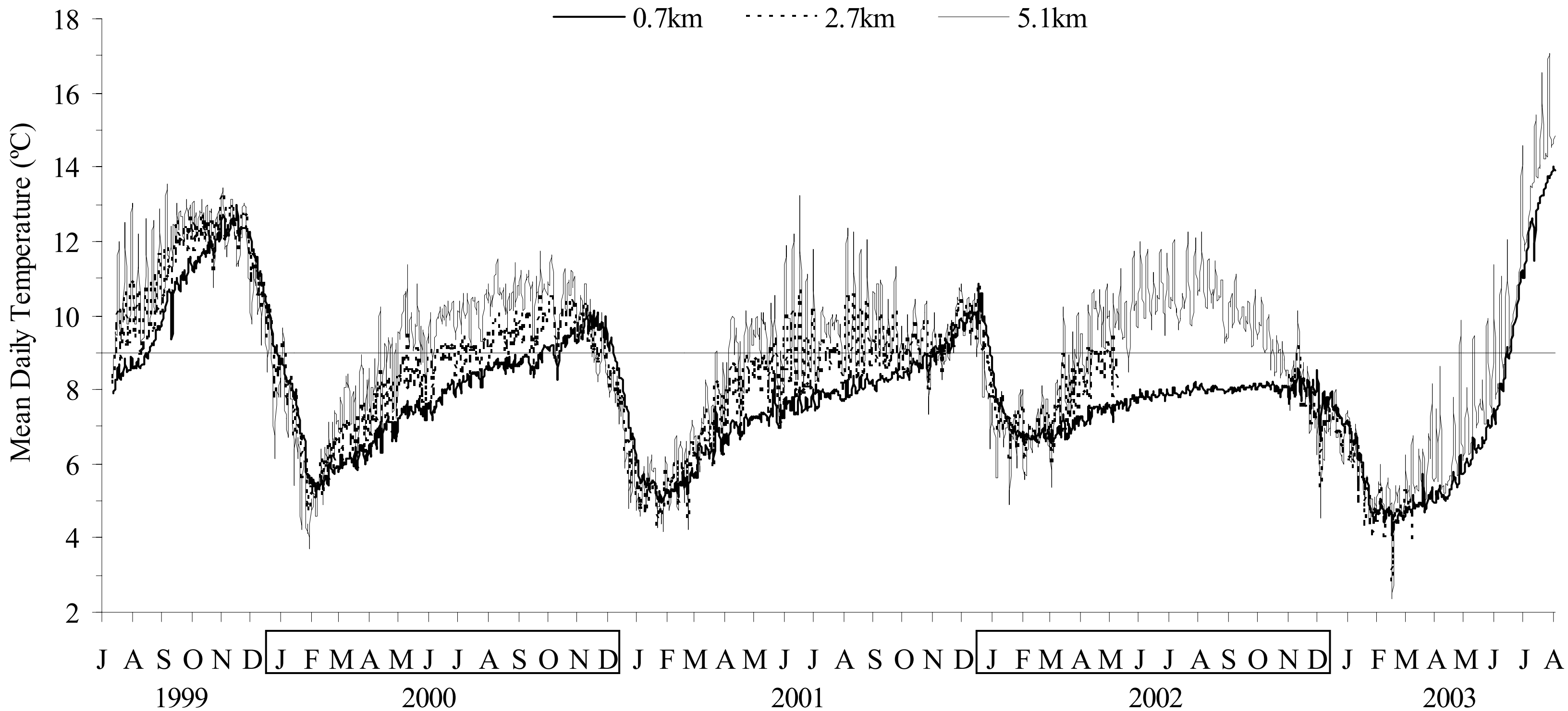


Figure 12. Mean daily water temperature (ºC) at 0.7, 2.7, and 5.1 km below Philpott dam in the Smith River, VA from July 1999 to August 2003. Horizontal line displays 9ºC, which initiates spawning.

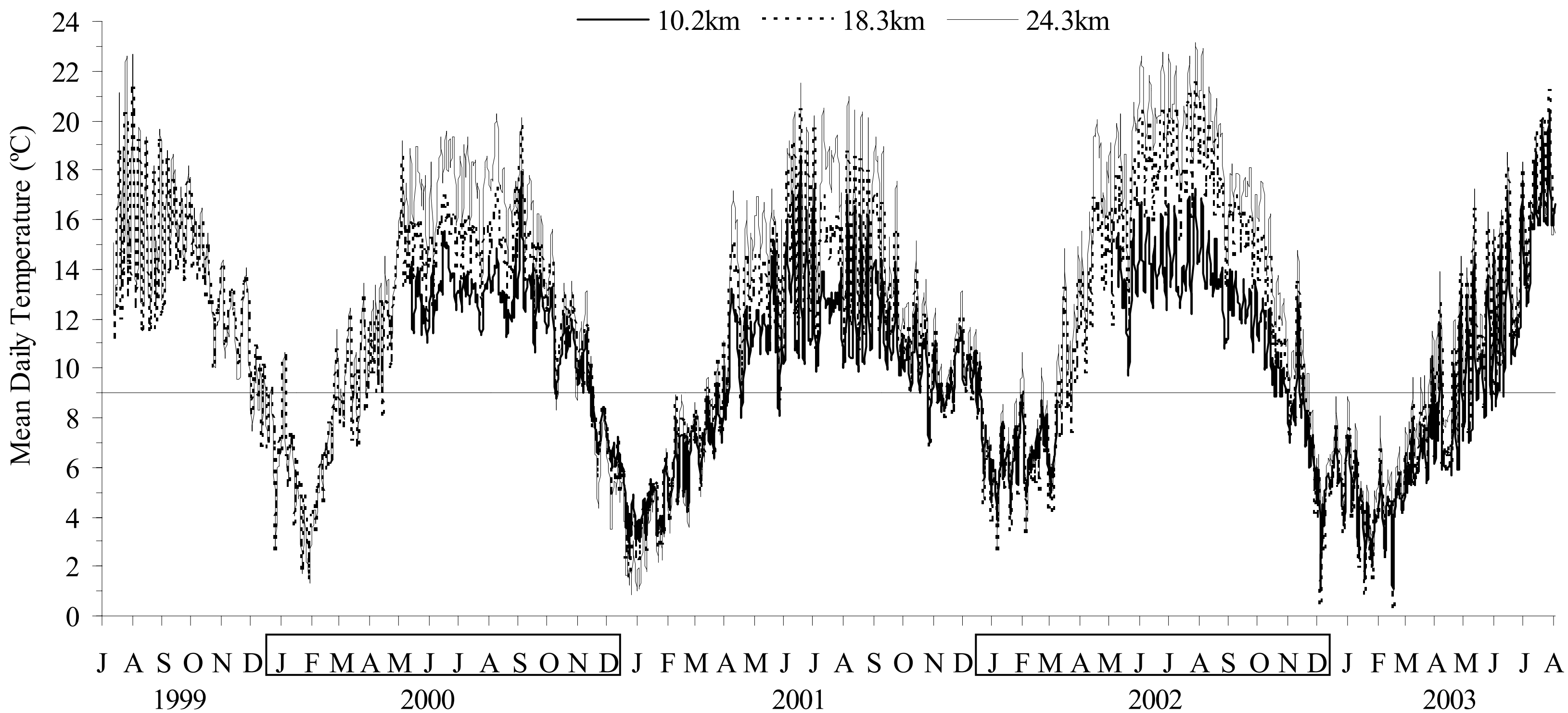


Figure 13. Mean daily water temperature (ºC) at 10.2, 18.3, and 24.3 km below Philpott dam in the Smith River, VA from July 1999 to August 2003. Horizontal line displays 9ºC, which initiates spawning.

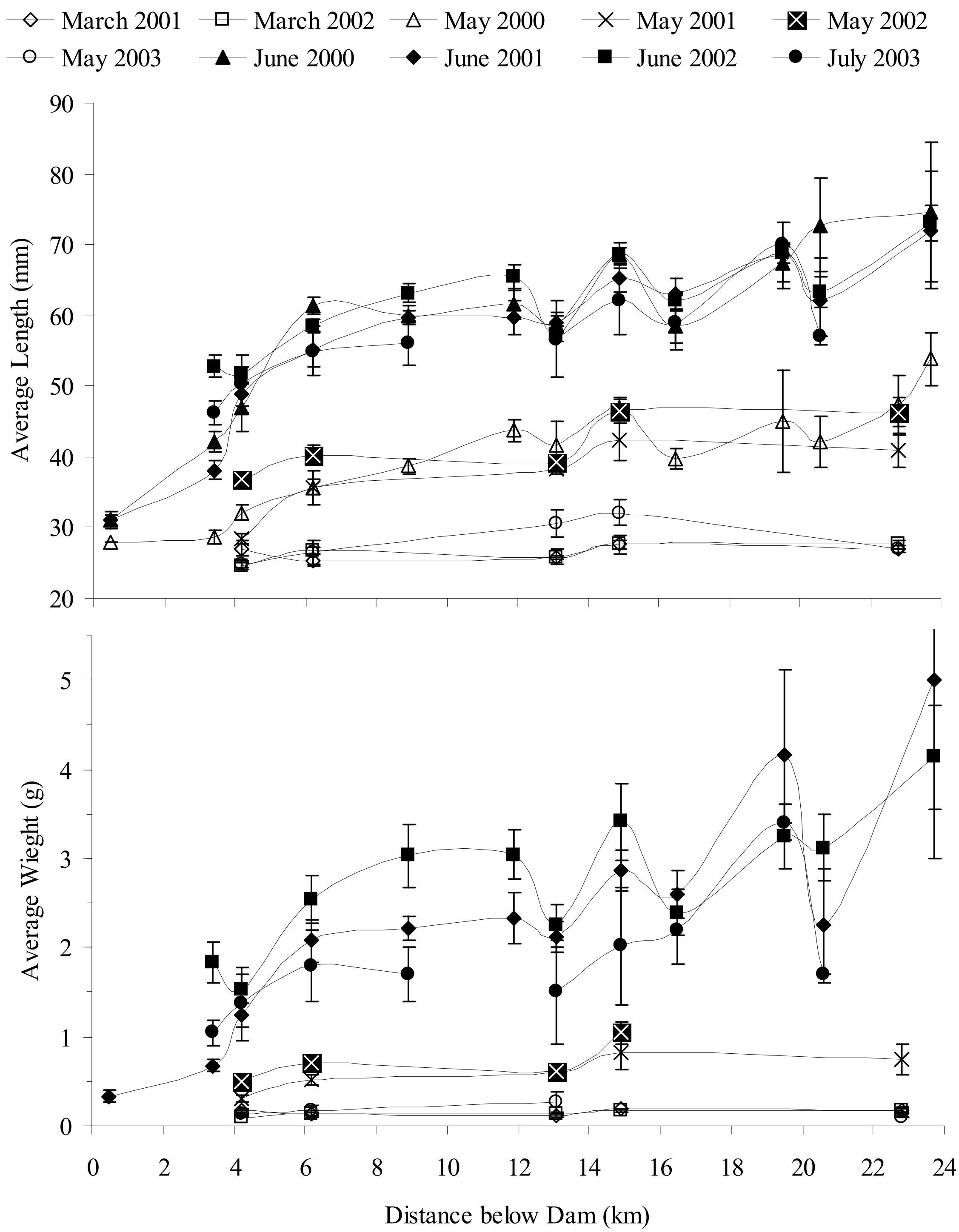


Figure 14. Average total length (mm) and average weight (g) of age-0 brown trout in the Smith River, VA sampled in March 2001-2002, May 2000-2003, and June/July 2000-2003. Error bars represent 2±SE. Note, lines connecting data points do not denote an association between distances below the dam, but are shown to provide clarity between sampling periods.

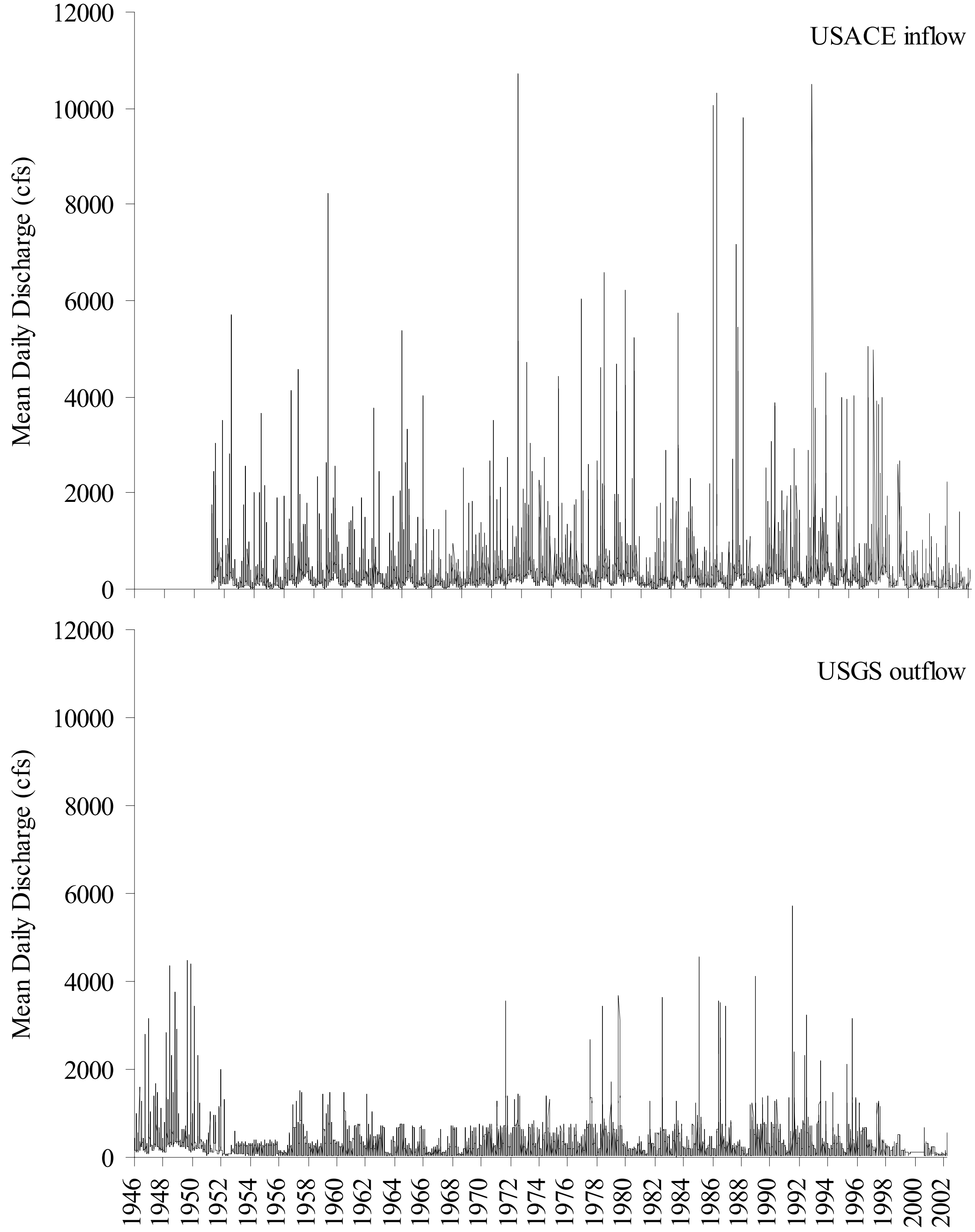


Figure 15. Mean daily discharge (cfs) above (upper panel) and below (lower panel) Philpott dam, Smith River, VA. Inflow is calculated by the U.S. Army Corps of Engineers based on reservoir elevation changes. Outflow is measured by a U.S. Geological Survey gage station. Flows were unregulated prior to the completion of Philpott dam in 1952.

Job 2: Determinants of Brown Trout Growth and Abundance

Job Objective: To collect biological data to quantify relative abundance of trout in the Smith River from Philpott Dam to Martinsville and monitor annual variation in brown trout recruitment success. To assess longitudinal and seasonal shifts in brown trout diet composition. To evaluate the bioenergetic constraints on trout growth under existing temperature regimes.

Specific objectives outlined in Job 2 evaluate the Smith River brown trout population and possible constraints on the population's growth and distribution. These objectives include; assessment of longitudinal trends in population dynamics of brown trout in the Smith River, evaluation of longitudinal and seasonal diet composition of the brown trout in the tailwater, evaluation and modeling of forage and thermal constraints on the brown trout, and assessment of the metabolic response of brown trout under two acutely fluxing thermal regimes.

Within the Smith River tailwater, thermal and flow regimes are predicted to influence the brown trout population. However, variations in flow and thermal regimes can also influence food availability and ultimately the amount of food that trout can consume. In addition, brown trout diet composition can vary on spatial and temporal scales based on the available prey. In addition, high numbers of brown trout may be causing competition among trout for food resources. To determine the role that food consumption is having in structuring the growth rates of trout, a study was initiated to determine daily consumption rates of brown trout in four reaches, which will be used in subsequent bioenergetics models. The inclusion of the diet study will lead to a greater understanding of the limitations in the tailwater.

Procedures

Trout Population Sampling

Brown trout populations were assessed at 12 locations from Philpott Dam (Rkm 0.5) to Martinsville (Rkm 23.0) beginning in June 2000 and continuing through July 2003. Trout populations were scheduled for sampling during June 2003 as in previous study years; however, dam generation schedules forced a delay until July. In previous years, fish were sampled in June and October 2000, April, June, and October 2001, and April, June, and October 2002, and July 2003. Fish were collected with two pulsed DC barge electrofishers that each had three mobile anodes. During the June/July sampling periods, three-pass depletion electrofishing was conducted on 100-m sections enclosed with block nets, while single pass electrofishing was conducted in 200-400 m sections during April and October. In July 2003, the site located 23 km downstream of Philpott Dam was not sampled to due high water levels.

During June 2000, brown trout larger than 100 mm were implanted with PIT (Passive Integrated Transponder) tags (Biomark™, Inc.). Tag recapture rates were low during the June 2001 sampling period, thus additional brown trout (>70 mm) were tagged in October 2001 (Table 1). All trout during each sampling period were measured to the nearest mm (total length), weighed to the nearest 1 gram, and scanned for the presence of a PIT tag.

Trout otoliths were collected to assess length-at-age of brown trout in the Smith River. Sagittal otoliths were removed from the trout, adhered to microscope slides, and sanded to allow light to pass through the otolith. The otolith was then viewed using an imaging system (Image

Pro-Plus ® software), which allowed digital enhancement of the image to aid in identification of annuli. All annuli were marked and measurements were taken along the dorsal axis from the focus to each annulus.

Diet Composition Study

The tailwater was divided into four reaches to account for variations in physical attributes of the channel that occur on a longitudinal pattern in the tailwater. Brown trout were collected from each of the four reaches during February, May, September, and December 2002 via backpack electrofishing. Five trout were collected four times per day over a 24-hour period from each reach for a total of 20 trout per reach per sample month. Efforts were made to maximize the total length distribution of trout collected to obtain a full array of diet items and consumption levels. Upon capture, trout were measured to the nearest mm (TL) and weighed to the nearest gram and sacrificed for collection of stomachs and otoliths. Stomach contents were removed, preserved in a 10% formalin solution, and returned to the lab for processing and analysis. Food items were identified, enumerated and weighed to the nearest 0.001 g. Fish in the stomach contents were identified to species when possible, aquatic invertebrates were identified to family when possible, and terrestrial insects and other aquatic organisms were identified to order. To aid in analysis, food items were grouped into 12 categories: Ephemeroptera, Plecoptera, Trichoptera, Diptera, partially digested unidentified aquatic insect matter (consists of insect parts), Isopoda, Crayfish, Gastropoda, Fish, Other (comprised of families that occurred sporadically in the diets), Terrestrial insects, and Vegetation.

Analyses

For the trout population analysis, the tailwater was divided into four reaches to account for variations in the physical attributes of the channel that occur on a longitudinal pattern in the tailwater. The reaches are as follows: Dam Reach (0-5.3 Rkm), Special Regulations Reach (5.3-10 Rkm), Bassett Reach (10-15.9 Rkm), and Kohler Reach (15.9-23.0 Rkm).

Maximum-likelihood population estimates and 95% confidence intervals were calculated for June sampling periods based on catch rates from three passes using Microfish 3.0 to estimate age-0 brown trout population estimates (Van Deventer and Platts 1983). Relative trout abundance was calculated for all sampling periods as the number of age-1 and older brown trout per 100 m by the equation: (# of trout/ Distance shocked) x 100; where distance shocked is in meters. The number of trout caught on the first pass was used for the June sampling periods. Relative abundance was calculated for each site. To test for significant differences between reaches within a sampling period and within a reach between sampling periods, data was rank transformed and a one-way analysis of variance (ANOVA) was performed. This test is the parametric equivalent of the nonparametric Kruskal-Wallis test. If significant differences existed, a Tukey's Honestly Significant Difference (HSD) test was used to determine which reaches or sampling periods were significantly different from the others. All tests were significant at alpha=0.05.

Relative stock density (RSD) indices were used to assess the length-frequency distribution of trout sampled by electrofishing. Relative stock density (Wege and Anderson 1978) was calculated by the formula: RSD = (number of fish≥specified length/number of fish≥minimum stock length) x 100; where the specified lengths were 230 mm (quality length) and 300 mm (preferred length) and minimum stock length is 150 mm (Milewski and Brown

1994). Annual means and variances were calculated for each site. Relative stock density indices were rank transformed. Differences in RSD indices between reaches within years and between years within reaches were tested by an ANOVA on the ranks. If significant differences existed within a year or reach, Tukey's HSD test was used to determine which reach or year was significantly different from the others. All tests were considered significant at alpha=0.05.

All length and weight data was examined for errors by log transforming the data. Data points having studentized residuals of greater than ±4 were checked against data sheets to check for recording or data entry errors.

Length-weight regressions were calculated for each of the four reaches during each sampling period by log transforming the length and weight data. Significant differences in slope (*a"*) of the length-weight equation were tested using PROC GLM in SAS to test for differences between reaches within the sampling period. All tests were considered significant at alpha=0.05.

Seasonal specific growth rates were calculated for recaptured PIT tagged trout that were caught during two successive sampling periods. Specific growth rates were calculated from the formula: G=ln l2- ln l1/t2-t1; where l2 and l1 are the lengths of the fish in mm at times t1 and t2 (Ricker 1979). Regression analysis was used to assess specific growth rates in length with environmental factors (Table 2). Analyses were conducted using site-specific data from the 12 sampling sites. Stepwise regression was used to determine which environmental characteristics were significant at alpha=0.05. Analyses were conducted in SAS version 8.0.

Brown trout annuli measurements from otoliths were used to back-calculate length-at-age using the formula: Focus to annulus distance / Focus to edge distance x Total length; where all measurements are in mm. Back-calculated lengths at age were examined for differences between reaches using analysis of variance on ranked data. The 1999, 2000, 2001, and 2001 year classes were examined for differences in mean back-calculated length-at-age 1 among years within each reach using analysis of variance. If differences existed, a Tukey's HSD test was used to determine which reaches or years were significantly different. All tests were considered significant at alpha=0.05.

Mean weight of stomach contents was assessed for seasonal and longitudinal shifts with PROC ANOVA in SAS. If differences between seasons or between reaches existed, a Fisher's Least Significant Difference (LSD) test was used to determine which seasons or reaches were significantly different. Tests were considered significant at alpha=0.05. Food categories were expressed as a percentage of the overall total weight of the stomach contents to assess seasonal and longitudinal changes in diet composition (Hyslop 1980).

Results and Discussion

Brown trout abundance

Population estimates for age-0 brown trout were lower in 2003 than in previous sampling years (Figure 1). In 2002, age-0 population estimates were much higher than in 2000 and 2001. The increased number of age-0 in 2002 may be the result of reduced occurrence and magnitude of generation flows. In 2003, peak generation flows were greatly increased, in both magnitude and duration, compared to flows in 2002. Changes in age-0 abundance during 2002 and 2003 indicate that age-0 abundance is strongly influenced by generation flow.

Brown trout relative abundance of age-1 and older trout was significantly different between the four reaches during all sampling periods except the June 2002 sampling period (P=0.1377). Relative abundance was greatest in the Dam and Kohler reaches during all sampling periods (Table 3). The relative abundance of brown trout in the Bassett and Kohler reaches were not significantly different from each other during any of the ten sampling periods. The Dam and Special Regulations reaches were not significantly different from each other during any of the sample dates.

Within the Dam, Bassett, and Kohler reaches, trout relative abundance was not significantly different between sampling dates. In the Special Regulations Reach, relative abundance was significantly different between sampling dates (P=0.0158) with abundance in April 2002 (170 trout/100 m) being significantly higher than abundance in June 2002 (52 trout/100 m). Although the relative abundance of brown trout in the Dam Reach was not significantly different between June 2000 and July 2003, there was a 40% increase in age-1 and older brown trout abundance. The increase in age-1 and older trout in the Dam Reach may be the result of the strong 2002 year class in the Dam Reach recruiting to the age-1 age group.

Brown trout Relative Stock Density

No significant differences were found between reaches within 2000 (P=0.3109), 2001 (P=0.9482) or 2002 (P=0.1980) for brown trout greater than 230 mm (Table 4). In addition, no significant differences were found between years in the Dam (P=0.7191), Special Regulations (P=.9128), Bassett (P=0.1766), and Kohler (P=0.7610) reaches.

Relative stock density indices for brown trout greater than 300 mm were significantly different between reaches in 2000 (P=0.0062), with the Dam Reach being significantly different from the Bassett and Kohler reaches (Table 5). No significant differences were observed between reaches in 2001 (P=0.0844). There was a significant difference between reaches in 2002 (P=0.0112), with the Dam Reach being significantly different than the Kohler Reach (Table 5). No significant differences were observed between years in the Dam (P=0.8247), Special Regulations (0.8267), Bassett (P=0.0616), and Kohler (P=0.8118) reaches; however, it should be noted that RSD values for 300 mm brown trout have declined annually (40-50% annually) since 2000 in the Bassett Reach (Table 5).

Length-weight regressions

Brown trout length-weight regressions were significantly different between reaches within each sampling period during all periods except October 2002. The Dam Reach was significantly different from the Kohler Reach during six of the ten sampling periods, with the Kohler Reach having a steeper length-weight regression slope than the Dam Reach (Table 6). Although the brown trout length-weight regression slope was significantly different during three sampling periods between the Bassett and Kohler reaches (June and August 2000 and July 2003), the Bassett and Kohler reaches were most similar in length-weight relationships during all time periods when compared to the relationships observed between the other reaches (Table 6).

The Kohler Reach had the greatest regression slopes during seven of the ten sampling periods. Length-weight regressions with higher slopes mean that for a given length, trout will weigh more than fish of the same length but having a smaller regression slope. The lowest regression slopes occurred in the first 10 Rkm during eight out of the ten sampling periods, which indicates that brown trout in the upper 10 km have a lower body condition than trout in the lower 13 km of the tailwater.

It should be noted that in July 2003, only 38 brown trout were collected and used in the calculation of the length-weight regression in the Kohler Reach. In all other sampling periods, all reaches had more than 200 length-weight observations that were used to calculate the length-weight regression equation.

Brown trout growth and environmental characteristics

Thirteen variables (Table 2) were regressed against specific growth rates in length for 2000-2001. The regression model for specific growth in length was a two variable model and included minimum temperature ($P<0.0001$) and number of trout per 100 m ($P=0.0003$) and is explained by the equation: mm day^{-1}= -0.00000261 x (# of trout 100 m^{-1})+0.00007532 x (minimum water temperature) + 0.00055154; where temperature is in ^{o}C (Figure 2). Although the variables were significant in the model, the regression model only explains a small percentage of variation in trout growth (R^2=0.0743). With number of trout per 100 meters being negatively related to trout growth, it indicates that trout growth may be density dependent in the Smith River. If trout growth is density dependent, growth rates of the 2002 year class should be low; however, because trout from the 2002 year class were not tagged, growth rates for individual trout will not be obtained.

Brown trout otoliths

Brown trout otoliths were collected from 1,056 brown trout since June 2000, which includes 319 otoliths from the brown trout diet study. To date, 956 otoliths have been aged. Brown trout otoliths from the Smith River were difficult to age due to thermal checks that appear on the otoliths based on dam generation schedules (Figure 3). The checks can be misidentified as annuli, thus potentially over-aging the trout. Of the trout that have been aged, only 2% have been 4 years of age or greater. Back-calculated length-at-age for brown trout exceeded 300 mm for age-4 brown trout at Rkm 20.1 and 23.

Mean back-calculated length-at-age 1 was significantly different between reaches ($P<0.0001$) with trout in the Bassett Reach being significantly larger than the other reaches (Figure 4). In addition, brown trout from the Dam Reach are significantly smaller at age 1 than all other reaches. Mean back-calculated length-at-age 2 was also significantly different among reaches ($P=0.0009$), with trout from the Dam Reach being smaller than trout from all other reaches (Figure 4). Significant differences were also found for mean length-at-age 3 ($P<0.0001$), with brown trout from the Dam Reach being significantly smaller than trout in the Bassett and Kohler reaches (Figure 4).

Mean back-calculated length-at-age 1 was not significantly different between the 1999, 2000, 2001, and 2002 year classes in the Dam ($P=0.7259$) and Special Regulations ($P=0.0876$) reaches (Table 7). In the Bassett Reach, mean back-calculated length-at-age 1 was significantly different between the year classes ($P=0.0039$), with the 2000 year class being significantly smaller than the 2001 and 2002 year classes (Table 7). Significant differences in back-calculated length-at-age 1 were observed in the Kohler Reach ($P=0.0006$), with trout from the 2001 year class being significantly larger than the 1999 year class (Table 7).

Brown trout diet composition

In 2002, a total of 320 brown trout were collected from four reaches of the Smith River for stomach content analysis (Table 8). Only three brown trout had empty stomachs. Fifty-two aquatic invertebrate families and 26 orders of aquatic invertebrates, terrestrial invertebrates, and aquatic organisms were identified in the stomach contents of the brown trout. Although most fish found in the trout stomach contents were unidentifiable, those that could be identified included fantail darter *Etheostoma flabellare* and bluehead chub *Nocomis leptocephalus*.

Longitudinal shifts in trout diets.—Mean weight of stomach contents in February was significantly different among the four reaches (P=0.0092) with the mean weight of stomach contents in the Special Regulations Reach being significantly greater than the Dam and Kohler reaches (Figure 5). Mean weight of stomach contents in brown trout were not significantly different between reaches in May (P=0.1672). In September, mean weights of stomach contents were significantly different between reaches (P=0.0256) with the Special Regulations Reach being greater than the Dam and Kohler reaches (Figure 5). During December, mean weights of stomach contents were not significantly different among reaches (P=0.2976).

In February, brown trout from each reach consumed a large proportion of Ephemeroptera; however, trout diets in the Dam reach consisted of a smaller proportion (28%) than diets from the lower three reaches (>50%; Table 9). While Trichoptera was observed in the diets from brown trout in the three lower reaches, Trichoptera was not observed in trout stomach contents from the Dam reach (Table 9). Isopoda was the second most dominant item in the trout diet from the Dam reach (25%), but only consisted of 6% or less of the total diet from the lower three reaches (Table 9).

During May, Ephemeroptera was still common in the diets of trout in the Dam and Bassett reaches, but they were less common in the Special Regulations and Kohler reaches (Table 9). As in February, Trichoptera was not observed in the diets of trout from the Dam reach, but were common in the diets of trout from the lower three reaches, with the highest percentage (25%) being observed in the Special Regulations reach (Table 9). While Isopoda was not in the diets from the lowest three reaches during May, it composed the largest percentage of the diet in the Dam reach (25%; Table 9). Crayfish were observed in the trout diets from the lower three reaches, but were not observed in the diets from trout in the Dam reach (Table 9). Fish were also observed in the diets of trout from the Bassett and Kohler reaches, but they were not present in the diet in the Dam and Special Regulations reaches (Table 9).

In September, Trichoptera were again present in the diets of trout from the lower three reaches, but were absent in the diets from the Dam reach (Table 9). Diptera was more common in the diets of trout from the Dam reach than the other reaches (Table 9). As in the previous sampling periods, Isopoda was a dominant item in trout from the Dam reach, but were not observed in the diets from the lower three reaches (Table 9). As in May, crayfish was a common diet item from the lower three reaches, but were not in the diets from the Dam reach (Table 9). In addition, fish were present in the diets of trout from the Bassett and Kohler reaches, but were absent in the diet from the other reaches (Table 9).

During December, Plecoptera was consumed in all reaches, but trout in the Dam reach consumed a smaller percentage than trout from the other reaches (Table 9). Although Trichoptera was observed in the diets in trout from the Dam reach, it was a smaller percentage than in the other reaches. Isopoda continued to be a dominant diet item of trout from the Dam reach (37%), but was not a common item in the lower three reaches. Crayfish were common diet items in trout from the lower three reaches, but were not observed in diets from the Dam reach (Table 9). Fish were observed in the diets from all of the reaches in December, but composed a smaller percentage of the diet in the Special Regulations reach (Table 9).

Seasonal shifts in diet.—Mean weight of stomach contents did not significantly change between seasons in the Dam (P=0.6329), Bassett (P=0.5956), and Kohler (P=0.3346) reaches. In the Special Regulations Reach, mean stomach weights were significantly different between seasons (P=0.0459) with mean stomach weights being greater in May than in February and December (Table 10).

Ephemeroptera was a common diet item in February and May, but during September and December, it composed a smaller percentage of the overall diet of the trout (Table 9). Plecoptera was observed in higher percentages in February and December than in the other two sampling periods (Table 9). Crayfish were not observed in February, but were common in the diet in May, September, and December for trout in the Special Regulations, Bassett, and Kohler reaches (Table 9). Gastropoda were more common in September and December than in other months (Table 9). Fish were common in the diets of trout from all of the reaches during December, and were also common during May and September in the Bassett and Kohler reaches (Table 9). During May and September, terrestrial insects comprised a high percentage of the diet from all of the reaches; however, during February and December, terrestrial invertebrates were observed in very small percentages (Table 9).

Preliminary Conclusions

Brown Trout Population Assessment

- In 2003, age-0 brown trout abundance was lower than in 2002. In 2002, the age-0 brown trout population was significantly higher than in previous sampling years.

- Relative abundance of age-1 and older brown trout was significantly different among reaches within a sampling date. The Dam and Special Regulations reaches had higher relative abundances than the Bassett and Kohler reaches.

- Relative abundance was not significantly different between sampling dates in the Dam, Bassett, and Kohler reaches, but the Special Regulations Reach was significantly different between dates.

- Relative stock density indices of brown trout greater than 230 mm were not significantly different between reaches or sampling years indicating that the proportion of trout greater than 230 mm is equal among the four reaches.

- Relative stock density indices for trout greater than 300 mm were significantly different between reaches in 2000 and 2002 with the Dam Reach being lower than the Bassett (2000) and Kohler (2000 and 2002) reaches. No significant differences were observed in RSD for trout larger than 300 mm between years within any of the reaches. This indicates there is a larger proportion of trout greater than 300 mm in the Bassett and Kohler reaches than in the Dam Reach.

- Brown trout length-weight regression slopes were significantly different between reaches in all sampling periods except October 2002. The greatest regression slopes were found in the Kohler Reach during seven of the ten sampling periods, while the lowest regression slopes were observed in the Dam and Special Regulations reaches. With steeper regression slopes, fish in the lower reaches of the tailwater have a higher body weight than fish in the Dam and Special Regulations reaches at a given length.

- Although the regression model failed to explain a large portion of the variation in specific growth rates, the models indicate that temperature and trout population numbers are important in explaining trout growth.

- Mean back-calculated length-at-age 1 was significantly lower in the Dam Reach (116 mm) than the other reaches (124-137 mm). Mean back-calculated length-at-age 2 was significantly lower in the Dam Reach (172 mm) than the other reaches (184-189 mm). For age-3 trout, mean back-calculated length-at-age was significantly lower in the Dam Reach (217 mm) than in the Bassett (237 mm) and Kohler (250 mm) reaches.

- No significant differences were observed between year classes for length-at-age 1 within the Dam and Special Regulations reaches. The Bassett and Kohler reaches had significant differences in back-calculated length-at-age 1.

Brown Trout Diet Analysis

- Mean weight of stomach contents were significantly different in February and September, with the Special Regulations Reach being significantly higher than the Dam and Kohler Reaches. No significant differences were observed in mean stomach content weight between reaches during May and December.

- Aquatic macroinvertebrates are an important component of brown trout diets. Isopoda was common in the diets of trout from the Dam Reach (17-37% of the total stomach content weight), but were not observed in a large proportion of the diet from trout in the other reaches (0-6%). Trichoptera was common in the trout diets from the Special Regulations (9-25%), Bassett (5-30%), and Kohler (4-14%) reaches but were less common in the trout diets from the Dam Reach (0-4%).

- Fish were more common in the trout diets from the Bassett and Kohler reaches than in the Dam and Special Regulations reaches.

- Mean weight of stomach contents were significantly different between seasons within the Special Regulations Reach, with the mean stomach content weight being greatest in May. No significant differences in mean stomach content weight were observed in the Dam, Bassett, and Kohler reaches between seasons.

- Seasonal changes were observed in the brown trout diets. Ephemeroptera was an important diet item during the February sampling period in all reaches (28-62%), but were found in smaller percentages (0-34%) during the other collection months. Terrestrial insects were common in the diets during the September sampling period (10-26%) but were less common during other sampling periods (0-12%).

- In December, fish were observed in the diets of trout in all reaches. In previous sampling months, fish were only observed in the Bassett and Kohler reaches.

Future Research and Job Schedule

Additional analysis will be conducted to further assess growth on spatial and temporal scales. Abundance and mortality will be assessed between seasons and years on a spatial scale. Additional statistical analysis will be conducted on the seasonal and longitudinal diet data to assess potential differences in diets based on trout size. Maximum consumption rates will be calculated for the different reaches and seasons. Consumption rates will then be incorporated into bioenergetics models to help determine factors limiting growth of brown trout in the Smith River. In addition, a laboratory study on oxygen consumption rates of brown trout under two acutely fluxing thermal regimes will allow a greater understanding of the effects of fluxing temperatures on brown trout. Both the diet information and the oxygen consumption parameters will be combined in a bioenergetics model to fully evaluate potential constraints and predict trout growth under alternative thermal regimes. In addition, regression analysis will be used to evaluate potential factors (including temperature variables, flow variables, trout and nongame fish abundance, and habitat variables) on growth limitations.

Job 2 Schedule. All aspects of Job 2 are on schedule with no significant changes anticipated. Reporting period extends to bold line.

Calendar Year	1999			2000				2001				2002				2003				2004	
Project Year	Year 1			Year 2				Year 3				Year 4				Year 5					
Quarter	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2
3-pass depletion				X				X				X				X					
PIT tagging				X					X				X								
Habitat mapping				X					X	X											
Data analysis				X	X	X	X	X				X	X	X	X	X	X	X			
Recapture				X	X		X	X	X		X	X	X		X	X	X				
Stomach sample collection										X	X	X	X								
Diet analysis										X	X	X	X	X							
Monthly temp monitoring								X	X	X	X	X	X	X	X	X	X				
Otolith preparation and aging								X	X	X	X		X	X	X						
Bioenergetics modeling									X	X			X	X			X	X			
Fluxing thermal regime (lab)										X	X	X	X	X	X	X	X				
Work plan preparation						X	X														
Annual report				X				X				X				X					
Dissertation writing																	X	X	X		
Manuscript preparation																	X	X	X		

Table 1. Number of brown trout recaptured in June 2001 that were tagged in June 2000 and the number of brown trout tagged in October 2001 and recaptured in June 2002 and July 2003 from 12 sampling sites in the Smith River, Virginia, tailwater.

Distance from Philpott Dam (km)	Number tagged in June 2000	June 2001 Recaptures for trout tagged in June 2000	Number tagged in October 2001	June 2002 Recaptures for trout tagged in October 2001	July 2003 Recaptures for trout tagged in October 2001
0.5	77	6	105	26	6
3.4	252	46	235	114	3
4.2	423	71	238	75	14
6.2	235	44	238	49	11
8.9	270	65	317	56	4
11.3	170	14	226	55	12
12.6	17	3	147	26	7
14.3	101	6	112	20	5
15.9	90	1	89	6	1
18.9	71	22	91	6	0
20.1	82	0	56	12	0
23	86	6	98	21	-
Total	1874	284	1952	466	63

Table 2. Environmental characteristics assessed using regression analysis to evaluate growth rates of trout in the Smith River.

Variable
Average water temperature
Minimum water temperature
Maximum water temperature
Portion of time with temperature <12^{o}C
Portion of time with temperature 12-19^{o}C
Portion of time with temperature >19^{o}C
Average flow magnitude
Minimum flow magnitude
Maximum flow magnitude
Trout abundance
Trout biomass
Non-salmonid fish abundance
Macroinvertebrate abundance

Table 3. Relative abundance (standard errors; SE in parentheses) of brown trout in the Smith River, VA, collected from four reaches from June 2000-July 2003. Abundance estimates are number of trout per 100 m. Abundances in the same row with the same letter are not significantly different (alpha=0.05).

	Reach			
Sampling Date	Dam (SE)	Special Regulations (SE)	Bassett (SE)	Kohler (SE)
June 2000	114 (30.4) a	100 (26.5) a	42 (17.9) a	22 (7.5) a
August 2000	86 (13.3) ab	106 (17.5) a	35 (8.5) bc	19 (3.9) c
October 2000	75 (16.7) a	92 (7.0) a	27 (2.2) b	16 (3.2) b
April 2001	71 (14.7) a	80 (5.0) a	48 (11.5) ab	20 (3.3) b
June 2001	50 (17.0) ab	137 (9.0) a	49 (18.7) ab	13 (3.6) b
October 2001	67 (20.5) ab	79 (2.0) a	35 (11.0) ab	17 (2.4) b
April 2002	131 (21.1) a	170 (4.5) a	52 (10.2) b	21 (6.9) b
June 2002	75 (13.9) a	52 (17.0) a	67 (26.1) a	29 (5.9) a
October 2002	92 (10.0) a	115 (2.0) a	36 (9.3) b	13 (3.0) b
July 2003	124 (27.5) a	60 (21.0) ab	41 (26.7) ab	6 (1.0) b

Table 4. Mean relative stock density indices (variance; var in parentheses) for brown trout greater than 230 mm from 4 reaches collected during 2000, 2001, and 2002 from the Smith River, VA. Mean values within a column followed by the same letter are not significantly different (alpha=0.05).

	Year		
Reach	2000 (var)	2001 (var)	2002 (var)
Dam	37.1 (39.72) a	38.5 (174.76) a	33.1 (115.32) a
Special Regulations	46.9 (248.65) a	46.5 (438.08) a	37.2 (220.5) a
Bassett	47.1 (11.83) a	44.1 (82.06) a	34.8 (94.23) a
Kohler	49.9 (63.84) a	44.4 (147.25) a	49.4 (99.23) a

Table 5. Mean relative stock density indices (variance; var in parentheses) for brown trout greater than 300 mm from 4 reaches collected during 2000, 2001, and 2002 from the Smith River, VA. Mean values within a column followed by the same letter are not significantly different (alpha=0.05).

	Year		
Reach	2000 (var)	2001 (var)	2002 (var)
Dam	1.4 (2.92) b	2.1 (2.43) a	1.0 (1.00) b
Special Regulations	3.2 (9.25) ab	5.4 (37.85) a	3.2 (12.5) ab
Bassett	12.2 (26.23) a	7.2 (16.03) a	3.6 (8.36) ab
Kohler	16.9 (97.15) a	11.3 (18.51) a	12.2 (52.03) a

Table 6. Length-weight regression slopes and intercepts for brown trout collected from four reaches and ten sampling periods between 2000 and 2003 in the Smith River, VA, tailwater. Slopes in the same row with the same letter are not significantly different (alpha=0.05).

	Reach							
	Dam		Special Regulations		Bassett		Kohler	
Date	Slope	Intercept	Slope	Intercept	Slope	Intercept	Slope	Intercept
June 2000	2.9124 a	-4.8221	2.9422 c	-4.8927	2.9856 b	-4.9682	3.0048 ac	-5.0273
August 2000	2.9919 b	-5.0033	2.9465 b	-4.9032	3.0375 b	-5.1024	3.0618 a	-5.2133
October 2000	2.9817 bc	-4.9807	3.0126 ab	-5.0541	2.9864 ac	-4.9857	3.0263 a	-5.0773
April 2001	2.8544 b	-4.6852	3.3158 a	-5.7147	3.0392 a	-5.0868	2.9958 a	-4.9887
June 2001	2.9611 b	-4.9279	2.9547 c	-4.9139	2.9590 a	-4.9247	2.9891 a	-4.9916
October 2001	2.9793 b	-4.9643	3.0055 b	-5.0375	3.0247 a	-5.0713	3.1046 a	-5.2462
April 2002	2.9574 a	-4.9137	2.9564 b	-4.9163	2.9763 b	-4.9590	2.9794 ab	-4.9767
June 2002	2.9599 a	-4.9224	2.9234 b	-4.8539	2.9262 b	-4.8565	2.9385 ab	-4.8913
October 2002	2.9376 a	-4.8751	2.9395 a	-4.8875	2.9373 a	-4.8859	2.9435 a	-4.8993
July 2003	3.0290 b	-5.0785	2.9862 c	-4.9914	3.0875 a	-5.2174	2.9686 c	-4.9330

Table 7. Mean back-calculated length-at-age 1 (mm) for brown trout collected from four reaches for the 1999, 2000, 2001, and 2002 year classes. Means in the same column with the same letter are not significantly different (alpha=0.05).

	Reach			
Year Class	Dam	Special Regulations	Bassett	Kohler
1999	115 a	124 a	135 ab	116 b
2000	120 a	134 a	133 b	128 ab
2001	115 a	128 a	148 a	136 a
2002	117 a	-	144 a	-

Table 8. Size distribution of brown trout collected during the diet study in the Smith River tailwater below Philpott Dam, Virginia, in 2002. N_T= number of fish collected. N_F= number of trout with food in their stomach.

Month	Reach	N_T	N_F	Avg. total length (range) mm
February	Dam	20	20	167.8 (81-280)
	Special Regulations	20	20	178.0 (104-252)
	Bassett	20	19	163.8 (98-303)
	Kohler	20	20	162.1 (102-294)
May	Dam	20	20	183.7 (100-275)
	Special Regulations	20	19	208.6 (140-318)
	Bassett	20	19	206.3 (128-298)
	Kohler	20	20	217.9 (142-400)
September	Dam	20	20	181.5 (120-262)
	Special Regulations	20	20	221.9 (179-301)
	Bassett	20	20	212.3 (98-279)
	Kohler	20	20	219.5 (128-323)
December	Dam	20	20	197.7 (139-266)
	Special Regulations	20	20	216.4 (134-272)
	Bassett	20	20	209.6 (121-327)
	Kohler	20	20	193.1 (135-297)

Table 9. Percent by weight of food items from brown trout collected during February, May, September, and December, 2002, in four reaches in the Smith River below Philpott Dam, Virginia.

	Reach															
	Dam				Special Regulations				Bassett				Kohler			
	Feb	May	Sep	Dec	Feb	May	Sep	Dec	Feb	May	Sep	Dec	Feb	May	Sep	Dec
% Ephemeroptera	27.9	33.9	0.2	0.2	54.4	6.9	1.5	9.0	62.3	13.7	2.0	4.6	50.6	4.1	9.3	6.9
% Plecoptera	4.9	0.0	0.0	3.5	9.9	1.4	2.6	13.8	5.9	0.3	5.5	8.0	7.5	0.2	4.4	16.7
% Trichoptera	2.1	0.0	0.4	3.7	8.5	24.6	24.5	16.4	4.8	17.1	29.3	10.7	4.1	13.6	10.3	12.9
% Diptera	4.3	9.5	4.5	8.6	4.1	1.6	3.4	1.8	1.3	4.7	1.2	4.6	2.9	6.3	0.0	2.5
% Fish	0.0	0.0	0.0	9.0	0.0	0.0	0.0	1.1	0.0	3.4	5.2	6.3	0.7	11.6	9.3	26.6
% Crayfish	0.0	0.0	0.0	0.0	3.2	10.1	6.3	4.8	0.0	5.0	7.3	12.6	0.0	5.9	11.1	8.4
% Isopoda	25.2	25.0	17.1	36.6	0.6	0.3	0.0	2.3	6.1	0.0	0.0	0.8	0.3	0.0	0.0	0.0
% Gastropoda	2.0	7.2	15.6	18.2	0.6	15.9	3.1	10.7	0.8	2.5	7.4	24.5	1.6	2.1	3.6	16.9
% Terrestrial Ins.	0.6	0.6	24.5	0.2	0.1	0.2	26.1	0.0	0.0	7.3	9.7	0.0	0.0	12.2	16.5	0.0
% Insect matter [a]	22.1	12.2	32.9	9.1	15.9	16.3	26.8	17.6	9.8	29.9	20.0	12.8	23.6	34.3	27.5	6.3
% Other	10.3	8.5	2.1	6.3	2.6	15.1	4.9	12.4	2.2	5.1	10.7	6.1	4.7	8.3	7.7	0.4
% Vegetation	0.7	3.0	2.7	4.7	0.1	2.7	1.0	10.1	1.8	6.1	0.4	5.6	4.1	1.5	0.4	2.5

[a] Comprised of partially digested unidentified insect matter

Table 10. Mean weight (standard error; SE in parentheses) of stomach contents collected from brown trout from four reaches in the Smith River, VA over four sampling periods in 2002. Mean weights in the same column with the same letter are not significantly different (alpha=0.05).

	Reach			
Month	Dam	Special Regulations	Bassett	Kohler
February	0.135 (0.024) a	0.438 (0.098) b	0.304 (0.085) a	0.179 (0.034) a
May	0.259 (0.090)a	1.328 (0.564) a	0.432 (0.151) a	0.787 (0.410) a
September	0.250 (0.063)a	0.701 (0.152) ab	0.530 (0.084) a	0.390 (0.106) a
December	0.284 (0.135)a	0.202 (0.059) b	0.551 (0.204) a	0.606 (0.249) a

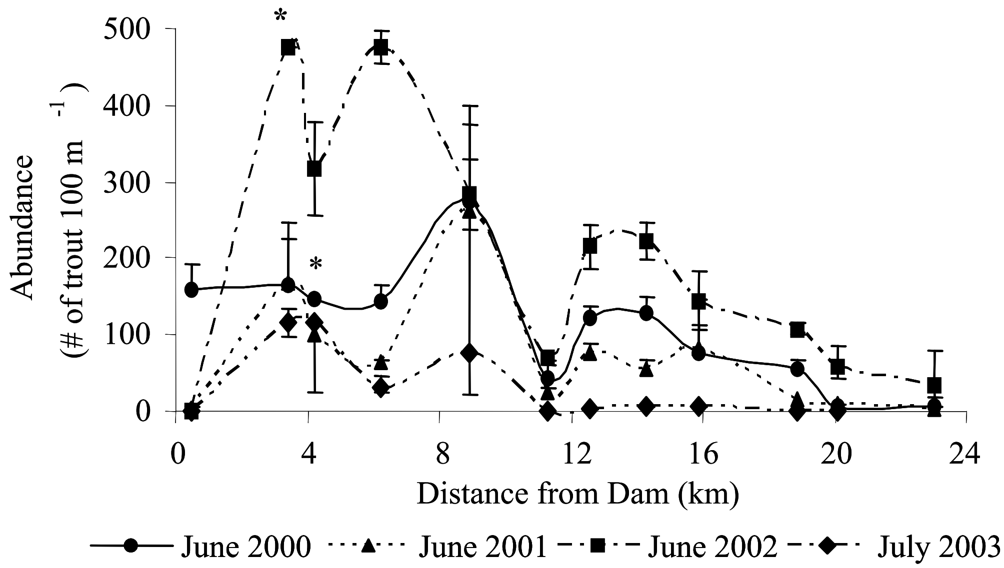


Figure 1. Population estimates (with 95% confidence intervals) for age-0 brown trout from the Smith River, Virginia, sampled from 12, 100-m sections by 3-pass depletion electrofishing in June 2000, June 2001, June 2002, and July 2003. Asterisk indicates that the population estimate for the site is not reliable because of non-descending catch data.

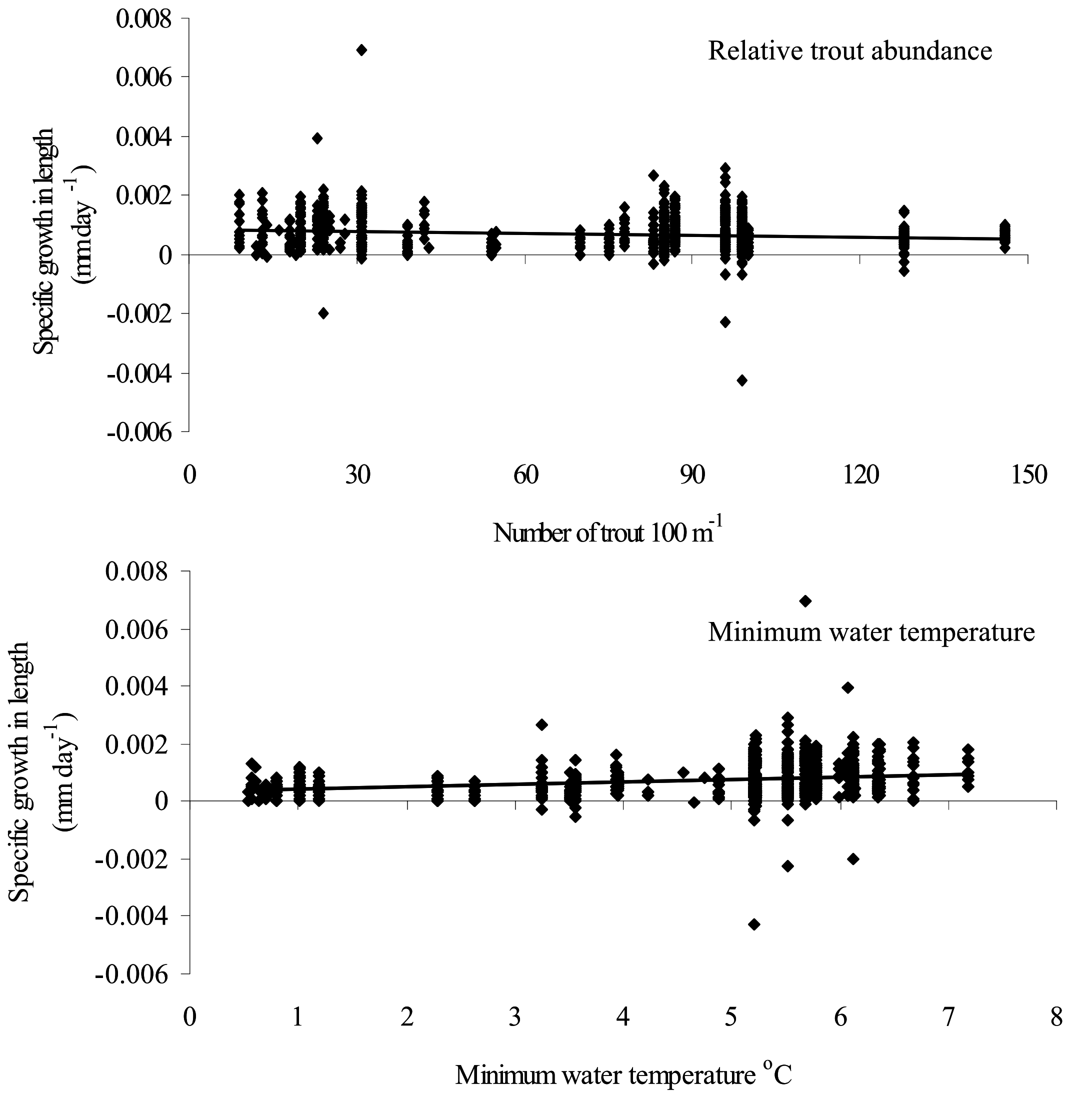


Figure 2. Relationship between specific growth in length for PIT-tagged brown trout in the Smith River, VA, and the relative abundance of trout (upper panel) and minimum water temperature (lower panel).

Figure 3. Image of a brown trout otolith (5X magnification) from a 115-mm brown trout collected on 9 February, 2002, 18.9 km downstream of Philpott Dam. Arrows are pointing to checks from dam generation periods.

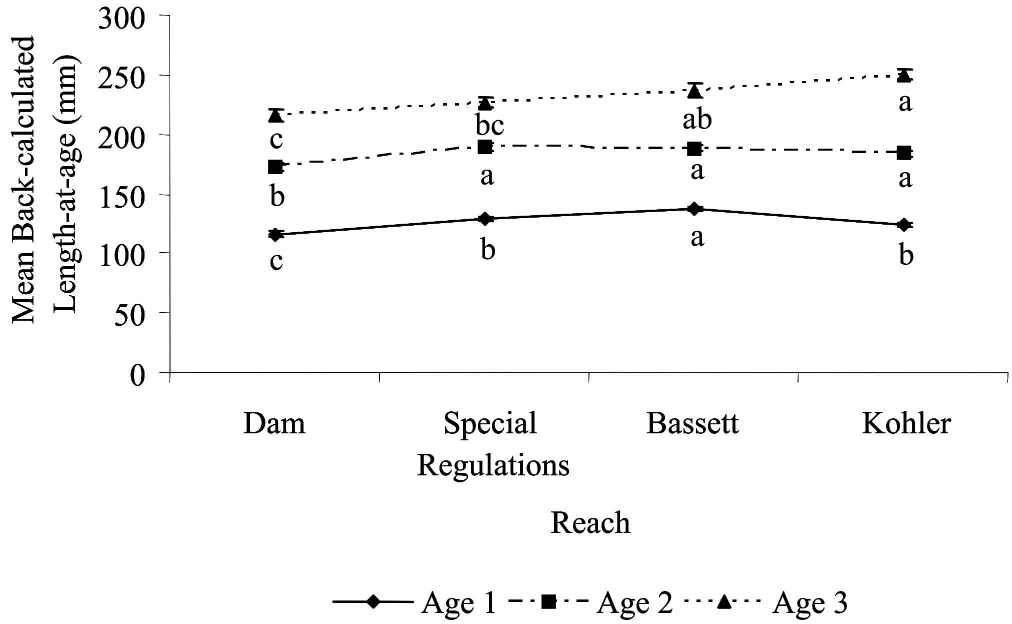


Figure 4. Mean back-calculated length-at-age and standard error for brown trout collected from 12 sites in the Smith River, Virginia, from June 2000 through July 2003. Means in the same age group with the same letter are not significantly different (alpha=0.05).

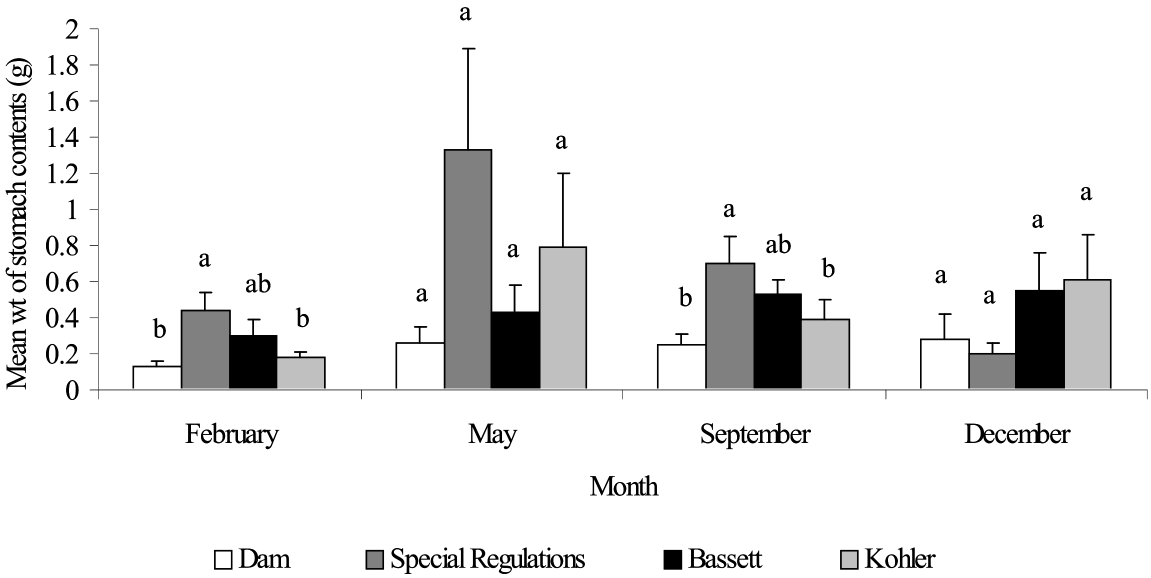


Figure 5. Mean weight and standard error of stomach contents collected from brown trout from four reaches in the Smith River, VA over four sampling periods in 2002. Mean weights in the same month with the same letter are not significantly different (alpha=0.05).

Job 3. Hydraulic Model Development and Application to Smith River Tailwater

<u>Job Objective</u>: To design a field survey and modeling protocol to measure effects of varying flows on the shear stress, mobilization of streambed gravels, and relate discharge to the amount of redd scouring or brown trout fry displacement that would occur at sites in the tailwater. This information coupled with flow records should permit prediction of catastrophic year-class failures and flow ranges that provide for acceptable reproduction.

Efforts devoted to Job 3 during 2003 focused on the ability of a one-dimensional (1-D; PHABSIM model) and a two-dimensional (2-D; RMA2 model) hydraulic numerical model to predict brown trout spawning habitat at an upstream site (4.2 km below Philpott dam) at various water discharges. A RiverCAT (Acoustic Doppler Current Profiler) was employed to measure the three-dimensional (3-D) water velocities and water surface elevations along selected river cross-sections. Velocities were averaged over the channel water depth and used to calibrate and validate the numerical models at various flows. Based on field measurements, a stream-specific brown trout spawning habitat suitability model was developed. From the simulation of certain physical conditions (i.e. water discharge, channel topography, and substrate) by the hydraulic models, water depth and velocity predictions were transferred into quality indices of spawning habitat through a habitat suitability model. Discussion is provided regarding the differences in hydraulic simulations and habitat qualities predicted by the two models. Comparing the 1-D and 2-D models provides insight into the ability of the models to simulate fish habitat in streams and is expected to facilitate further sediment transport simulations using the 2-D model.

<u>Procedures</u>

Flow field measurements using RiverCAT

Hydraulic data at the upstream study site were surveyed at bankfull flow (42 m^3/s), a moderate flow (19 m^3/s), and at base-flow (1.79 m^3/s). These three discharges represent the range of flow conditions commonly encountered in the Smith River. At base-flow, water velocity was measured using a hand-held wading rod and flow meter. During bankfull flow, the river can not be waded, which required a SonTek RiverCAT attached to a rope-pulley system to record the water depth and velocity profiles along selected channel cross-sections (Figure 1).

Stream-specific Habitat Suitability Model

Development of an accurate biological model for a target species is an important step to insure a successful implementation of habitat analysis. To serve this purpose, a stream-specific brown trout spawning habitat suitability model was created, which involved the following procedures. First, water depth, mean column velocity, and substrate data were obtained at randomly selected locations with and without redds. This data reflected the specific environments preferred by brown trout for spawning. Field limitation of these suitable spawning environments was also analyzed so that a fish preference index could be estimated based on the utilization and availability of these environmental conditions. To evaluate these indexes according to their relative importance for trout habitat preferences, Principal Component Analysis (PCA) was performed to adjust the relative weight for every preference index represented by each physical variable (i.e. water depth, velocity, and substrate). The final format of the habitat suitability model is:

$$CSI = I_V^{0.49} \times I_S^{0.34} \times I_D^{0.17}$$

where CSI is the composite suitability index, I_V, I_S, and I_D are the individual suitability preference index for mean column velocity, substrate type, and depth at the cell (PHABSIM) or element node (RMA2), respectively. The CSI value ranges from 0 to 1, with low values indicating poor habitat and high values denoting good preference by fish.

Comparison between 1-D and 2-D model predictions on brown trout spawning habitat

The two cross-sections of interest within the study reach for comparing 1-D and 2-D model velocity predictions are shown in Figure 2. The upstream transect is characterized by a large number of irregularly distributed boulders that force the flow towards the left bank (looking upstream). As a result, pronounced transverse flows and vortices persist within the transect. In contrast, at the downstream transect there are no large obstructions and the local bed topography is relatively flat. Compared to the upstream flow, the flow patterns are fairly smooth and dominated by streamwise parallel flows in the downstream transect. Geometric and hydraulic features of these two locations present two different flow conditions available in this reach, which help identify what kind of flow situation will cause significant difference between the two modeling approaches.

Both models were calibrated for the moderate flow and validated for the base-flow and bankfull flows. An assumption made is that the reach topography would not be markedly changed during these different flows. Predicted water surface elevations from the 2-D model were averaged across each transect to compare to the 1-D model and field measurements along the study site. The comparison also applied to the distribution of depth-averaged velocities (downstream in 1-D and total velocity direction in 2-D) at the two channel cross-sections. The Kolmogorov-Smirnov (KS) two-sample test (Gordon et al. 1992) was employed to evaluate velocity-prediction accuracy for each model (Table 1) based on field measurements ($P \leq 0.05$). The assessment can help identify the ability of the two models in reproducing flow characteristics (Ghanem et al. 1996, Waddle et al. 2000).

Distribution of potential fish habitat quality within the study reach was generated by transferring the outputs of both hydraulic models to the stream-specific habitat suitability model. A comparison of habitat quality was also made, which provided an alternative way to estimate performance of the numerical models as tools for biological application. Assessing the biological performance of each model was implemented at the microhabitat level (i.e. cells in 1-D & elements in 2-D model) and macrohabitat level (reach scale).

The relationship between redd availability at the microhabitat level and the prediction of habitat quality was examined through the Spearman statistical analysis (Gibbons 1985). This involved counting the observed number of redds on each cell of the 1-D model and each element of the 2-D model (the cell and element are used to discretize the flow field of the study site). The model outputs (i.e. water depth and velocity) were coupled with substrate observations and imported into the biological model to estimate CSI for all the tiles (a tile is a cell or an element) in the study site at base-flow, which was the dominant flow when observed redds were surveyed. The computed CSI of each tile was classified into 10 equally spaced groups, ranging from 0 to 1. The relationship between redd's density and predicted habitat quality was analyzed through the Spearman statistic with an adjustment to account for tied observations.

Weighted Usable Area (WUA) was calculated at the macrohabitat level to evaluate effects of flow regulations on physical spawning habitat in the Smith River. WUA was calculated individually by multiplying the wetted surface area of each tile to its corresponding cell- or element-averaged CSI then all values were summed to constitute a total WUA for the entire study site. WUA was computed at multiple flow conditions to reflect the range discharges in the Smith River. These simulations were used to evaluate the relationship between discharge magnitude and its corresponding habitat quality at the reach scale.

Results and Discussion

Comparison between 1-D and 2-D model predictions on brown trout spawning habitat

Flow field simulation comparison – Calibration and validation procedures were applied at all study site locations where corresponding field measurements of the water surface elevations and velocities were available.

The relative error between field measurements of water surface elevation and predictions from both models was less than 10% at 108 randomly surveyed locations at various flows (Figure 3). The mean absolute error of the water surface prediction throughout the site was 0.02 m for the 1-D model and 0.03 m for the 2-D model at base-flow. At bankfull flow, the prediction error of water surface elevation was 0.04 m for the 1-D model and 0.035 m for the 2-D model.

Though both models performed reasonably well at base-flow and bankfull flow, there was greater error near the water's edge (Figure 4a and 4b). The predictive error near the water edge may be attributed to the lack of information on river-bank friction and vegetation cover. Compared to the 2-D model, velocity output of the 1-D model was less accurate, especially around complex channel geometry (i.e. near boulders) where intricate flow patterns persists at the upstream transect (see shaded area in Table 1). This is because the PHABSIM (1-D) analysis was based completely on the local Manning's n roughness coefficient without considering lateral momentum exchange, which was caused by turbulent flows surrounding the boulders.

Fish habitat quality simulation comparison - River reaches with more redds had higher habitat quality indices, which implied a positive relationship between redd densities observed in the river and habitat quality predicted the models (Figure 5). Specifically, 78% of the locations with redds were identified as areas having high suitability indices (CSI>0.9) by the 2-D model, whereas 69% of the total redd locations where predicted with the same high CSI values by the 1-D model. There were no redds found in the area where the 2-D model predicted regional-averaged CSI values below 0.3. There were three redds in the location where the 1-D model predicted CSI values less than 0.1 (Table 2). The reason the 1-D model predicted lower CSI values at redd locations is mainly attributed to lower substrate indices obtained at these areas. The 1-D model assumes that physical conditions within each rectangular cell are uniform. Hence, it is possible to use only part of the unsuitable substrate to represent the whole tile, although the remaining part of that tile may be good for spawning. For example, channel areas having CSI values < 0.1 had varying substrate conditions, but ranked as uniform unsuitable substrates in the 1-D model. Consequently, lower substrate indices obtained at those tiles distorted the availability of suitable environments.

Our Spearman statistical analysis showed that the distribution of redds was predicted more accurately by the 2-D model (Figure 5). For instance, at base-flow which is the dominant discharge during fish spawning, the calculated Spearman R correlation coefficient of redd density and habitat quality was 0.744 ($P = 0.009$, $n = 10$) for 1-D PHABSIM, whereas for the 2-D RMA2 model the coefficient was 0.875 ($P = 0.001$, $n = 10$).

The 1-D model predicted more total wetted area than the 2-D model at base-flow, but differences of the wetted area were reduced at higher flows (Table 3). Limited by the ability of the 1-D model in bathymetric approximation, the stream obstructions such as boulders between cross-sections were treated as flat locations and estimated as being submerged underwater even at base-flow. Under the same base-flow condition as for the 1-D model, the 2-D model represented boulders as dry areas where in field the boulders protrude over the free water surface. Dry areas can be excluded from usable habitat due to no suitable water depth. Thus, the 2-D model is more accurate than the 1-D model when dealing with local obstructions.

The comparison results of WUA between two models are listed on Table 3. At base-flow, both models generated nearly identical WUA. This is because the 1-D model predicted more wetted area and less reach-averaged CSI value than the 2-D model. However, at higher flows, the wetted area from both models were very close, but the CSI values of the 1-D model were less than the 2-D model due to unsuitable substrate represented in the 1-D model. Consequently, at moderate flow the 1-D model predicted total WUA of the study reach as 331 m^2 less than the value computed by the 2-D model. The 1-D model predicted 83% WUA of the 2-D model at bankfull condition.

Both models showed the ratio of total WUA to wetted area decreased as discharge increased, which was probably caused by high velocity values not suitable for fish spawning at higher flows (Figure 6). For example, at moderate flow, the reach-averaged velocity was 0.64 m/s, corresponding to a velocity suitability index of 0.8. However, when the reach-averaged velocity increased to 1.02 m/s at bankfull flow, the velocity suitability index was reduced to 0.2.

Discrepancy in habitat prediction by the models was attributed to shallow water areas. The 1-D model only considered rectangular cell-averaged velocity and depth, which would unavoidably exaggerate velocity and depth predictions along the water edge. Where as the finite element mesh in the 2-D model was adjusted to adapt to the highly irregular channel boundaries. For the 2-D model nodes were setup along the banks and island edge, and corresponding flow values on the nodes were estimated independently from other nodes in deeper water. Use of rectangular cells to represent habitat environments by the 1-D model may cover only part of the actual area surrounding a redd. The redd may be located on a common boundary of two adjacent cells with different suitability index values. The 2-D model avoids this problem by employing small elements with flexible shapes and sizes to better replicate the geometry of a spawning location.

Preliminary Conclusions

- Both the 1-D and 2-D model performed reasonably well with regard to the prediction of the water surface elevations and velocity distributions. However, the 1-D hydraulic model cannot correctly simulate local velocity patterns around boulders. The result is improved as flow becomes more uniform at higher flows.

- By incorporating river obstructions into bathymetric data the 2-D hydraulic model captures complex flow patterns more accurately than the 1-D model.

- Results from the habitat simulations not only imply the significant relationship between redd density and habitat quality, but clearly indicate that the relationship can be approximated more closely by a 2-D model.

Future Research and Job Schedule

Future research efforts will focus on modeling sediment scour and fill patterns at different flows using a 2-D sediment model currently under development. Numerical output from this 2-D sediment model will be used to predict survival rates of brown trout embryos through empirical relationships obtained from literature.

We plan to evaluate the influence of different discharges on redd scouring. Typical egg burial depths within brown trout redds will be based on our field measurements and the 8 to 25 cm egg burial depth reported in DeVries (1997). By assuming embryos are distributed linearly within the burial depth range, we can assess embryo mortality based on computed local scour depth at different flows. For example, we would assume no egg mortality if the local scour depth of a potential redd location is less than 8 cm and 100% mortality if scour depth is more than 25 cm.

Survival rates of embryos will be modeled for a range of fine sediment intrusion levels into the redd gravel. To obtain the composition of particles on the riverbed (i.e. particle size ratio of fines to gravels) we will run the numerical simulation of the sediment model. Permeability of the predicted sediment composition will be derived from an empirical formula (Sakthivadivel 1966). By employing Darcy's law, the intragravel velocity can be calculated based on the estimated permeability and water head loss. With an empirical relationship proposed by Cooper (1965) we will be able to estimate embryo survival rate from the calculated intragravel velocity.

Job 3 Schedule. All aspects of Job 3 are on schedule with no changes anticipated at this time. Reporting period extends to bold line.

Calendar Year	1999	2000				2001				2002				2003				2004			
Project Year	Year 1			Year 2				Year 3				Year 4				Year 5					
Quarter	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2	3	4	1	2
Bathymetric surveys					X	X	X				X										
Hydraulics measurements				X						X	X										
Model calibration														X	X						
Time series analysis												X	X				X				
Data analysis and graphics						X	X				X			X		X					
Manuscript preparation								X	X			X	X	X	X	X	X				

Table 1. Comparison of velocity simulations at two selected transects using the Kolmogorov-Smirnov (Ks) (Gordon et al. 1992) two-sample test (Significance level α = 0.05).

		Downstream Section		Upstream Section	
		Ks	*P*	*Ks*	*P*
Base-flow	1-D	0.29	0.216	0.41	0.040
	2-D	0.29	0.216	0.23	0.563
Bankfull-flow	1-D	0.33	0.109	0.32	0.247
	2-D	0.35	0.135	0.33	0.310

Note: shaded area shows there is a significant difference between measured velocities and predicted values ($P < 0.05$).

Table 2. Relationship between redd density and CSI computed by the 1-D and the 2-D models (The redds density is calculated based on a unit area of 1000 m^2).

CSI	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	1.0
Redd density (1-D)	1.8	0.0	0.0	0.0	0.0	11.0	22.0	4.4	4.8	25.0
Redd density (2-D)	0.0	0.0	0.0	7.9	0.0	3.9	4.8	8.5	13.3	28.4

Table 3. Comparison of habitat quantity at the multiple flows between the 1-D (PHABSIM) and the 2-D (RMA2) models

Model	Q (m^3/s)	Wetted Area (m^2)	WUA (m^2)	WUA / Wetted Area (%)
1-D	1.79	4360.03	2041.09	46.81
	10	4550.20	2375.62	52.21
	19	4629.11	2096.74	45.29
	30	4629.11	1702.41	36.78
	42	4629.11	841.74	18.18
2-D	1.79	4151.21	2039.55	49.10
	10	4633.61	2733.00	58.98
	19	4600.32	2428.31	52.79
	30	4633.61	2017.47	43.54
	42	4633.61	1011.71	21.80

Figure 1. Rope-pulley system setup across a selected transect at the upstream study site to which RiverCAT (ADCP) is attached. Software installed on a notebook computer was running to receive wireless data transmitted from RiverCAT.

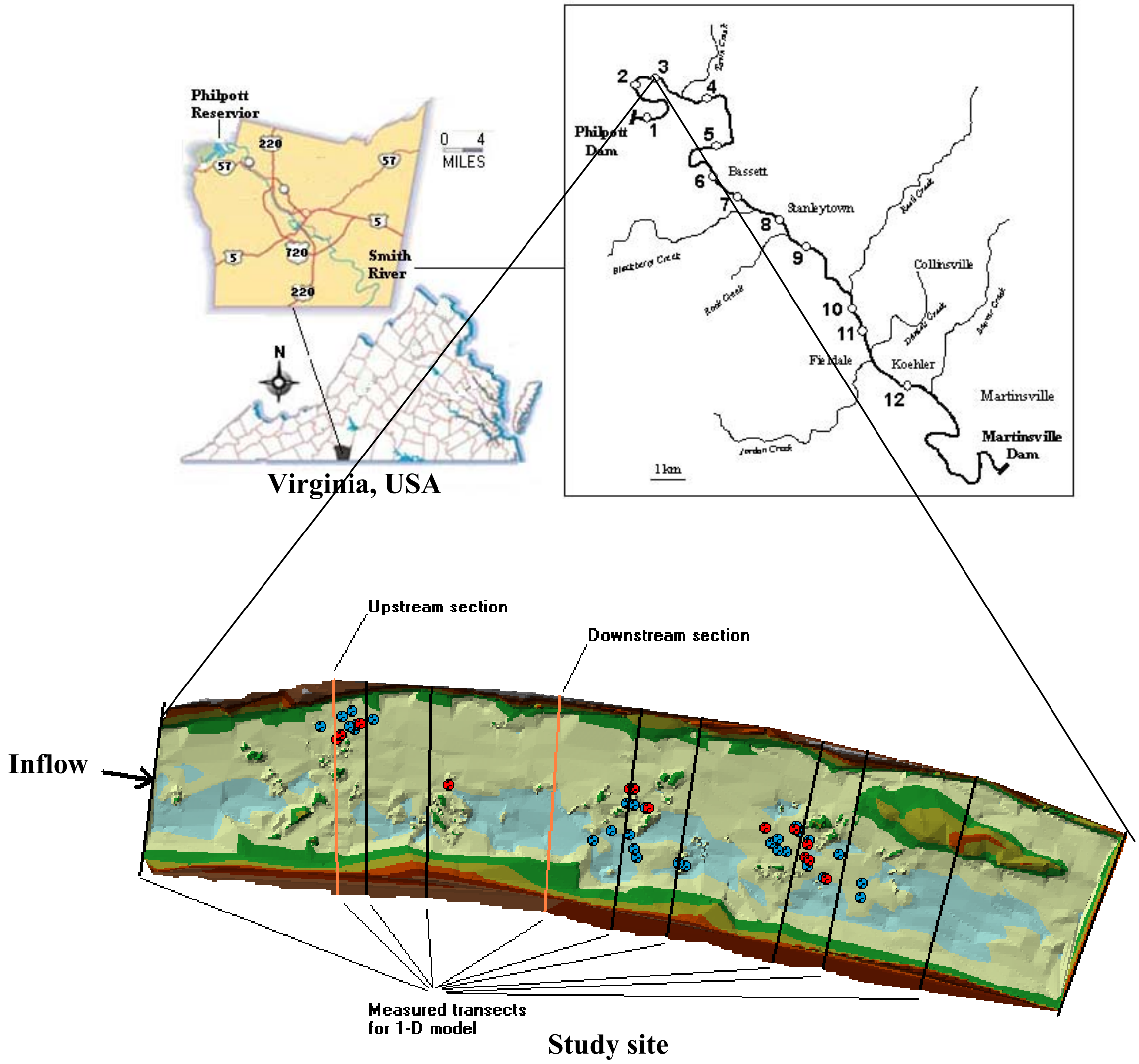


Figure 2. Map of the Smith River tailwater between Philpott Dam with sampling sites numbered upstream to downstream (Site 3 is the upstream study area). The two selected cross-sections (i.e. upstream and downstream sections) shown at the upstream study site were used for velocity comparison between the 1-D and 2-D models. Color dots are redds surveyed during the year of 2000 (blue) and 2002 (red).

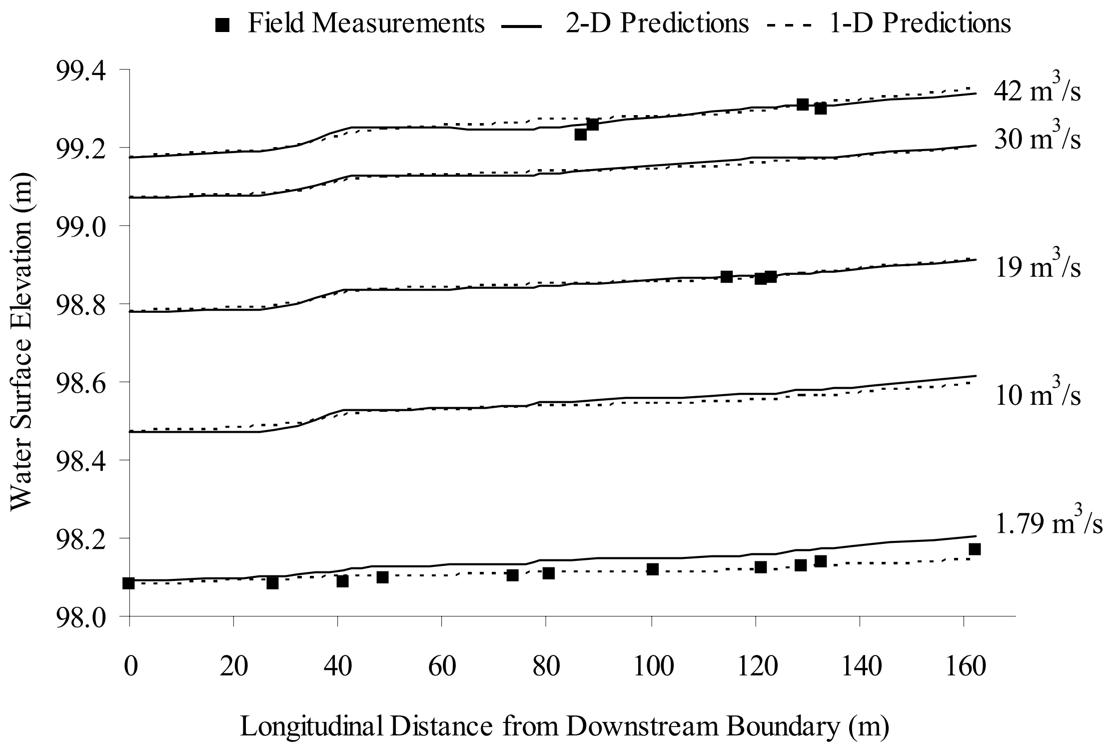


Figure 3. Comparison of longitudinal water surface elevation profiles along the study site, as predicated by the 1-D (broken line) and 2-D (solid line) models at 1.79 (base-flow), 10, 19 (moderate), 30, and 42 (bankfull) m^3/s. Solid squares depict measured data using the wading rod and RiverCAT. For convenience of comparison, the water surface elevations in the 2-D model were averaged in each cross-section.

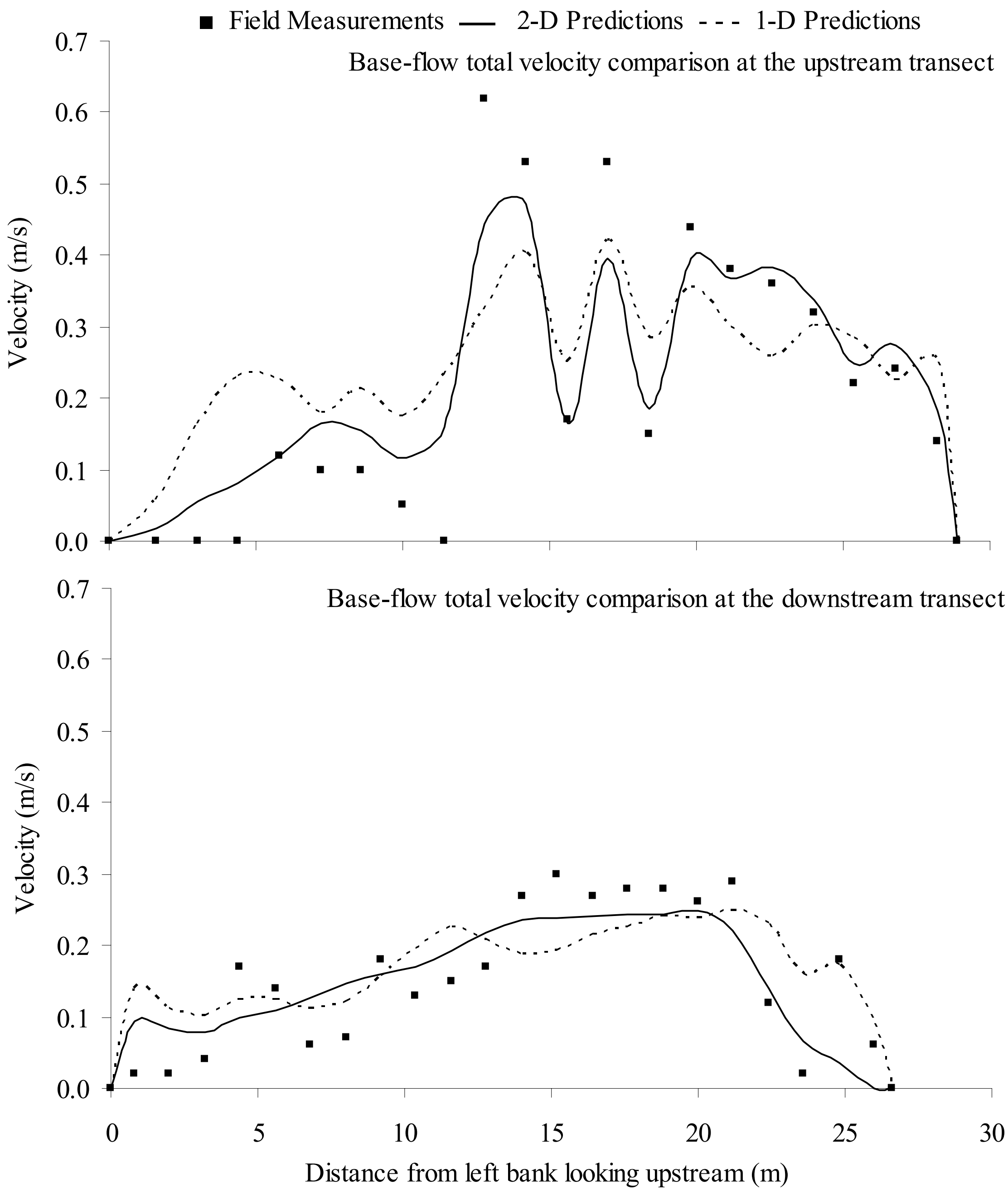


Figure 4(a). Comparison of the transect velocity profiles between the 1-D & 2-D models at base-flow. Velocities of the 1-D model are towards the downstream direction, while the velocities of the 2-D model are in their local maximum velocity directions.

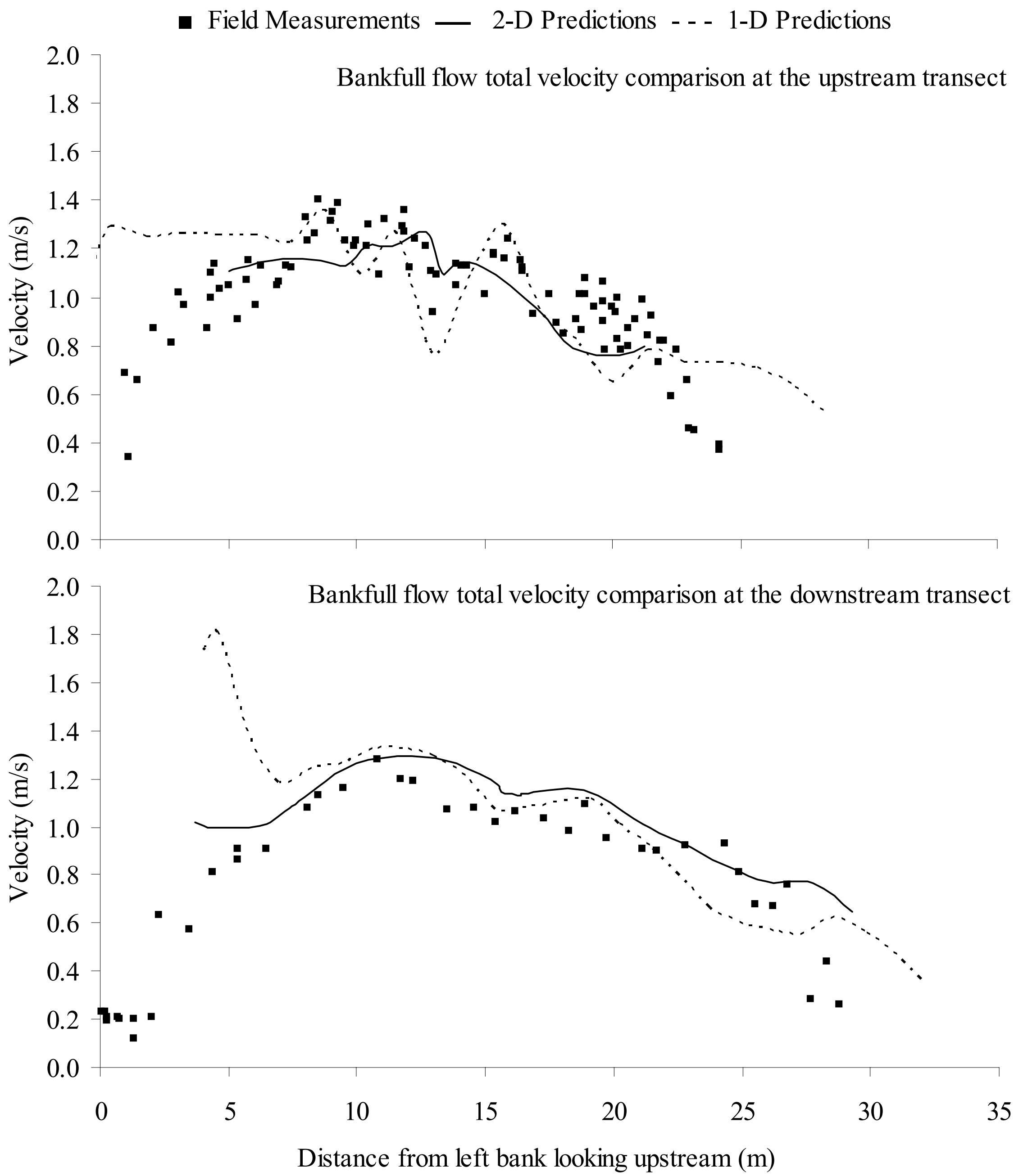


Figure 4(b). Comparison of the transect velocity profiles between the 1-D & 2-D models at bankfull flow. Velocities of the 1-D model are towards the downstream direction, while the velocities of the 2-D model are in their local maximum velocity directions.

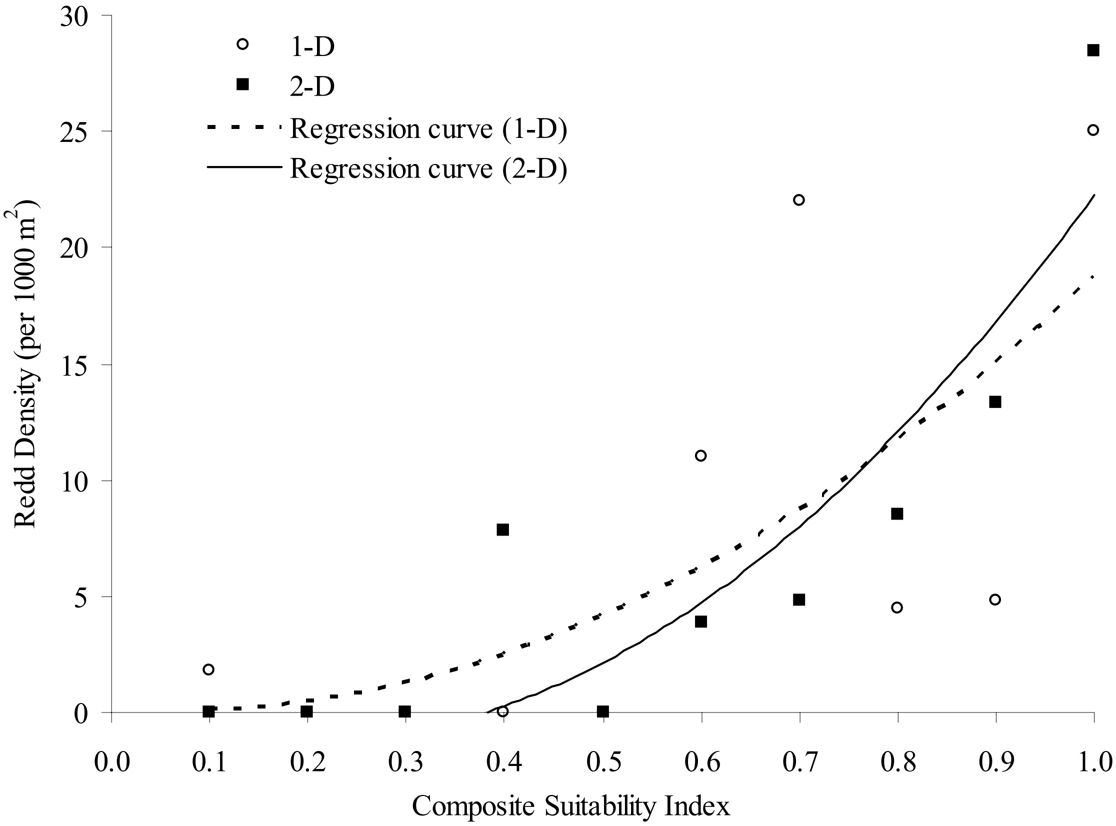


Figure 5. Comparison of habitat quality (CSI) at base-flow between the 1-D (PHABSIM) and 2-D (RMA2) models at the microhabitat level (i.e. cell or element). Redd density is calculated based on a unit area of 1000 m^2. Best fit curves (1-D R = 0.74; 2-D R = 0.88) are created through polynomial regression analysis by Microsoft Excel.

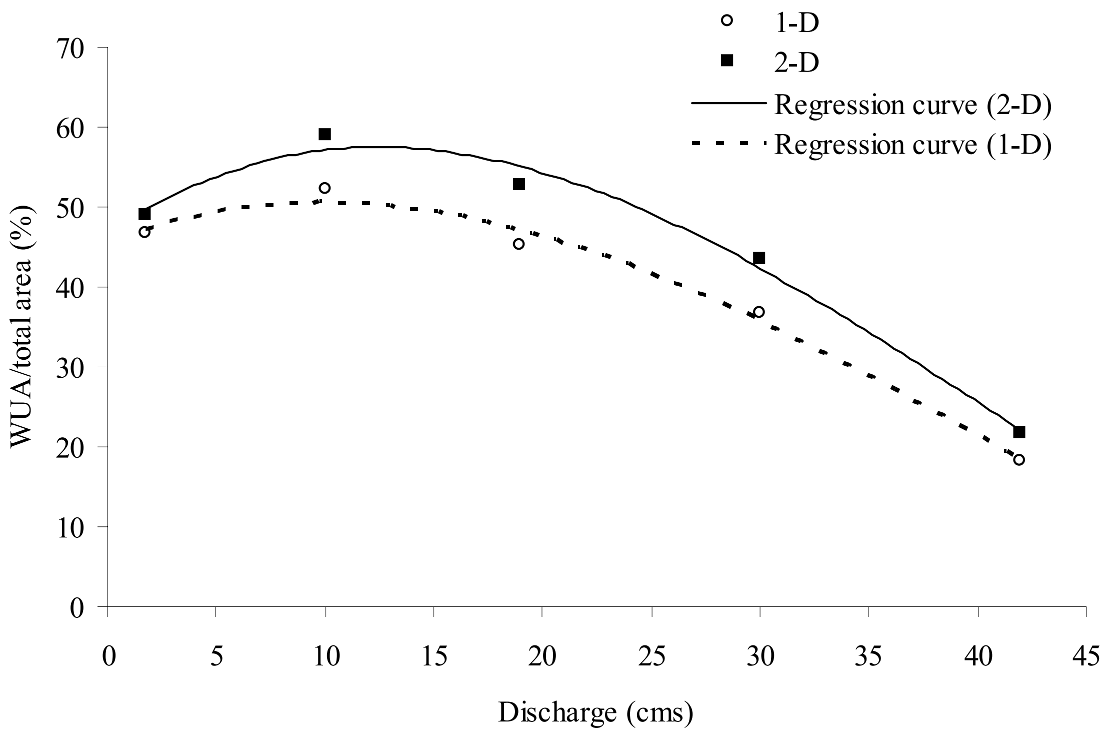


Figure 6. The WUA/Total Area ratio (%) predicted by both the 1-D and 2-D models at multiple flows. Best fit curves are created through polynomial regression analysis by Microsoft Excel.

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Appendix A. The following presentations and publications were completed in 2002 - 2003 by Smith River Study researchers.

1. Anderson, M. R., T. J. Newcomb, and D. J. Orth. 2002. Growth Rates of Brown Trout in the Smith River, Virginia, Tailwater. American Fisheries Society 132nd Annual Meeting, Baltimore, Maryland.

   Abstract: The Smith River tailwater below Philpott Dam is a highly valued naturalized brown trout fishery in southwestern Virginia. The tailwater historically produced trophy-sized brown trout. Today, however, trout rarely attain lengths (>356 mm) desired by managers and anglers. Our goal was to evaluate proximate and ultimate factors on brown trout growth for 24 km of the tailwater. Brown trout were captured by electrofishing in June 2000, tagged with passive integrated transponder tags, and recaptured in August and October 2000 and April and June 2001. Absolute growth rates in length (mm day^{-1}) and weight (g day^{-1}) were significantly different among sampling sites ($P<0.0001$) with lowest growth rates near the dam and increased growth rates at intermediate sites. Growth rates varied seasonally ($P<0.0001$) with highest growth rates from June to October, and lowest growth rates occurred during October to April. No linear trend in growth rates was observed with increasing distance from the dam. We developed multivariate, nonlinear models to identify factors that were contributing to observed growth patterns at different locations in the river. Determination of limiting factors will provide much needed information for improving brown trout growth and thus the fishery.

2. Hunter, A. K., and C. A. Dolloff. 2002. Longitudinal Patterns of Community Structure for Stream Fishes in a Virginia Tailwater. American Fisheries Society 132nd Annual Meeting, Baltimore, Maryland.

   Abstract: Artificial disturbances in flow impose changes outside the natural range experienced by most stream fishes, limiting their distribution and abundance. Such high environmental variability provides an opportunity to understand mechanisms shaping fish community structure in a regulated river. Philpott Dam located on the Smith River, VA is a peaking, hydropower facility with flows fluctuating from 30 to 1400 cfs and a hypolimnetic release which creates great thermal flux. A primary objective of our research is to describe nongame species distribution, abundance, and diversity and to relate these patterns to environmental conditions. Our study examines how brown trout abundance, mean monthly temperature, maximum hourly temperature flux, tributary location, and the difference between maximum and minimum daily flow explain longitudinal patterns of community structure for stream fishes in the Smith River tailwater. Understanding relationships between environmental gradients and fish community structure in regulated rivers will improve efforts to manage streamflow and preserve aquatic life.

3. Krause, C. W., T. J. Newcomb, and D. J. Orth. 2002. Thermal Habitat Assessment of Alternative Flow Scenarios in a Tailwater Fishery, American Fisheries Society 132nd Annual Meeting, Baltimore, Maryland.

Abstract: The Smith River tailwater (Patrick County, VA) offers a self-sustaining brown trout fishery managed for trophy trout (>406 mm), however trophy sized fish are rare. Limited food resources, physical habitat, and thermal habitat likely cause slow growth and small size. We assessed the potential for thermal habitat improvement with a one-dimensional hydrodynamic model coupled with a water temperature model. Temperature predictions from fifteen alternative flow regimes were evaluated for occurrence of optimal growth temperatures (12-19°C) and compliance with Virginia DEQ daily maximum (21°C) and hourly temperature change (2°C) standards. Optimal growth temperatures were increased by releasing water in the morning, decreasing duration of release, and maintaining existing base-flow. Maximum temperatures were decreased by releasing every day to prevent elevated temperatures on non-generation days, increasing base-flow, increasing duration of release, and releasing in the morning rather than evening. Hourly temperature change was decreased by increased base-flows, morning releases, and decreased release duration. Despite conflicting adjustments to improve all criteria concurrently, a 7-day, 7 am, 1-hour release regime improved all criteria compared to existing conditions. Integrating habitat assessment with hydropower operations via cost-benefit analysis could not be done because hydropower planning and operations at this ACOE impoundment are divorced from environmental planning.

4. Shen, Y., D. W. Crowder, and P. Diplas. 2002. Relationship between Spatial Hydraulic Metrics and Stream Habitat Availability, American Fisheries Society 132nd Annual Meeting, Baltimore, Maryland.

   Abstract: Flow complexity produced by topographic obstructions that are present in natural rivers appears to constitute an essential component of aquatic habitat. Little effort has been made to develop appropriate relationships between spatial flow complexity and areas fish and other aquatic organisms may use. Moreover, a direct comparison of the ability of one-dimensional and two-dimensional hydraulic models to predict and quantify spatial flow patterns of biological importance has yet to be made. A reach of the Smith River in Virginia where brown trout redd locations were observed is modeled, using the two-dimensional hydrodynamic model (RMA-2V), and the one-dimensional hydraulic modeling approach used by PHABSIM (Physical Habitat Simulations System). Recently developed spatial hydraulic metrics are computed throughout the study site based on the individual model results. A comparison is then made to determine the ability of RMA-2 and PHABSIM to identify potentially important flow complexity around redd locations. Using RMA-2 model results and the same spatial hydraulic metrics, flow complexity is also quantified around chub mound locations found in Mudlick Creek, Virginia. Results show that substantial flow complexity was found surrounding the locations of both brown trout and chub spawning locations and that the two-dimensional hydraulic model was better suited to quantify such flow complexity. Results are in agreement with established beliefs that a complex hydraulic environment is an important component of some fish habitats and suggest that the metrics evaluated here have the potential of becoming an important component of HSC (Habitat Suitability Criteria).

5. Anderson, M. R., and C. W. Krause. 2002. Trout Growth and Alternative Flow Regimes in the Smith River. Smith River Trout Unlimited, September 9th, Bassett, Virginia.

6. Orth, D. J. 2002. Responses of Fishes to Daily Flow Pulses in the Smith River, Virginia. Seminar to the Zoology Department at Southern Illinois University, December 2nd.

7. Hunter, A. K. 2003. Longitudinal Patterns of Community Structure for Stream Fishes in a Virginia Tailwater. M.S. thesis. Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

   Abstract: The Smith River, Virginia is a hydropeaking system with daily fluctuations in flow and temperature. We examined community structure in terms of abundance, composition, and distribution for 34 fishes within the first 24 km below Philpott Dam in the Smith River. Fish were sampled at 12 sites in 8 time periods ranging from 2000 to 2002 across 3 seasons, April, June, and October. Flows varied greatly during the duration of the study. We evaluated spatial and temporal change in fish community characteristics. Species demonstrated persistent trends in abundance, diversity, and composition throughout the duration of the study despite high environmental variability. Yet, our results indicated that numbers of individuals increased under the mildest flow regime. Fish abundance and diversity generally increased with increasing distance from the dam with peaks in abundance and diversity at tributary junctions. Fish composition changed minimally across seasons and years indicating consistent fish assemblages. Distributional patterns indicated a strong response to thermal gradients and presence of tributaries. I concluded that flow and temperature directly influence fish community patterns in the Smith River and that the patterns are persistent over space and time even though numbers of individuals vary.

   URL: http://scholar.lib.vt.edu/theses/available/etd-04082003-215009/

8. Hunter, A. K., M. R. Anderson, C. W. Krause, T. J. Newcomb, and D. J. Orth. 2003. Hydropeaking Flow Regime: A Determining Factor on Brown trout and Nongame Abundance. Southern Division American Fisheries Society, Spring Meeting, Wilmington, NC February 15th.

   Abstract: The Smith River tailwater (Bassett, VA) supports a self-sustaining Brown trout population and 34 nongame species. Hydropeaking regimes varied widely during 2000, 2001, and 2002. Corresponding electrofishing data shows population estimates and relative abundances (fish per 100 m) were greater in 2002 than 2001 and 2000 at the majority of 12 sampling sites 0.5 to 23.0 km below Philpott dam. June population estimates for age-0 brown trout were significantly greater at 8 of 12 sites and nongame were greater at 10 of 12 sites, though not all significant. Abundance during October was greater at 10 of 12 sites for age-0 and 8 of 12 sites for nongame; by as much as 117 age-0 and 641 nongame per 100 m. The hydropeaking regime from January 2000 through May 2001 was a 7-day/week, 1 hr, 1300 cfs release (50 cfs base-flow). The magnitude declined to a 5-day/week, 2-10 hr, 700 cfs release from June into November 2001. Flow increased to a 5-day/week, 3-4 hr, 1300cfs release in November until February 2002. Flow for the rest of 2002 was only a 5-day/week, 1 hr, 700 cfs release. This reduction in peak flow magnitude may be the cause of increased fish abundance in 2002.

9. Orth, D. J., C. W. Krause, and D. C. Novinger. 2003. Sediment Accumulation Patterns in a Hydro-peaking Tailwater in Virginia. Southern Dirt: Sedimentation in Southeastern Waters; Southern Division American Fisheries Society, Spring Meeting, Wilmington, NC February 15th.

   Abstract: Sediment characteristics are altered by river impoundment and this is illustrated by longitudinal sediment patterns in Smith River below Philpott Dam, operated with daily releases for hydropower production. Channel elevation near the dam has degraded since 1980. Upstream reaches were dominated by larger rocks (> 64 mm) and bedrock (80% bottom coverage), whereas pebble and gravel substrates covered a higher percentage of the streambed downstream (40%). Sand and smaller particles (< 2 mm) made-up a higher percentage in downstream reaches (> 12 km from dam; 20 - 50%). Fine sediment (< 2 mm) intrusion into Vibert boxes increased with downstream distance from the dam. Measurements of intragravel permeability highlight the influence of gravel manipulation by spawning brown trout (*Salmo trutta*) on permeability. Trends in substrate composition are consistent with the combined impacts of hydro-peaking and influx of fine sediment from tributaries that has apparently resulted in a downstream gradient from larger to smaller sized material in the tailwater.

10. Shen, Y., P. Diplas, and C. W. Krause. 2003. A Comparison of the Impact of Low and High Flows on Brown Trout Spawning Habitat Using Spatial Metrics. Virginia Chapter AFS meeting, Virginia Tech, Blacksburg, VA March 20th.

   Abstract: Altered flow regimes appear to have significant influence on the ecological environment of both natural and regulated rivers. Nevertheless, the efforts that have been made to quantify the effect of flow changes occurring in streams on the quality and extent of aquatic habitat are rather limited. To partially remedy this problem, a two-dimensional hydrodynamic model is employed to model the flow behavior in two selected reaches of the Smith River in Virginia, where brown trout redd locations have been observed and monitored. First, velocity measurements collected from these two sites are used to carefully calibrate the hydraulic model. Second, river-specific habitat criteria and recently developed spatial hydraulic metrics are computed throughout the two study sites for a low (~60 cfs) and a moderately high (~700 cfs) flow, respectively. Finally, a comparison is made to evaluate the impact of different flow regimes on brown trout spawning habitat selection. Our numerical simulation results indicate that the potential spawning habitat sites generally decreased as water discharge increased. This might be attributed to the fact that more intricate flow patterns, higher flow complexity, were found surrounding the redd locations at low flows compared to those at high flows. Results are in agreement with established beliefs that a complex hydraulic environment is an important component of some fish habitats and suggest that the new spatially explicit metrics evaluated here have the potential of becoming an important component of HSC (Habitat Suitability Criteria). These results could be useful for, among other things, developing suitable schemes for reservoir releases and implementing appropriate morphological changes in stream rehabilitation projects for the purpose of enhancing the quality and abundance of aquatic habitat.

11. Anderson, M. R., D. J. Orth, and S. M. Smith. 2003 (in review). Historical change in the brown trout fishery in the Smith River Tailwater, Virginia. Southeastern Association of Fish and Wildlife Agencies.

    Abstract: Historical data on brown trout from the Smith River tailwater, Virginia, below Philpott Dam, from 1971-2002 were reviewed to assess changes in the fishery over the last 30 years. Data from citation brown trout and electrofishing data were evaluated for changes in size distribution and fish condition. We observed a decrease in the number of citation brown trout over the last 30 years. Relative stock density has also decreased. Although relative condition of citation brown trout was high in the early 1970s, values decreased and have remained stable for the last 20 years. Possible explanations for the decline in the numbers of large brown are also presented.

12. Krause, C. W., T. J. Newcomb, and D. J. Orth. 2003 (in review). Thermal Habitat Assessment of Alternative Flow Scenarios in a Tailwater Fishery. River Research & Applications.

    Abstract: The Smith River tailwater (Henry County, VA) offers a self-sustaining brown trout fishery managed for trophy trout (≥406 mm), however trophy sized fish are rare. Slow growth and small size are likely caused by any one or a combination of limited food resources, physical habitat, and thermal habitat. To evaluate the potential for thermal habitat improvement, temperature changes resulting from alternative flows released from the hydropeaking Philpott dam were assessed with a one-dimensional hydrodynamic model coupled with a water temperature model. Simulated temperatures from each flow scenario were assessed every 2 river kilometers over a 24 kilometer river section below the dam for occurrence of optimal growth temperatures, as well as compliance with Virginia Department of Environmental Quality hourly temperature change and daily maximum temperature standards. The occurrence of optimal growth temperatures was increased up to 11.8% over existing conditions by releasing water in the morning, decreasing the duration of release, and not increasing base-flow. Occurrence of hourly temperature changes greater than 2°C was reduced from 4% to 0-1.2% by non-peaking releases, increasing base-flow, releasing in the morning, and decreasing the duration of release. Maximum temperature occurrence greater than 21°C decreased from 1.3% to 0-0.1% by releasing every day of the week to prevent elevated temperatures on non-generation days, increasing base-flow, increasing duration of release, and releasing in the morning rather than evening. Despite conflicting adjustments to best improve all thermal criteria concurrently, a 7-day/week, morning, one hour release regime was determined to improve all criteria compared to existing conditions.

13. Krause, C. W., T. J. Newcomb, and D.J. Orth. 2003 (in progress). Applications of Three Temperature Models in Virginia Streams: Approaches and Guidelines. North American Journal of Fisheries Management.

    Abstract: Multiple stream temperature prediction models are available, however a lack of scientific reviews and performance evaluations can make choosing a model capable of answering study objectives challenging. This study evaluated the SNTEMP, QUAL2E, and RQUAL models on predictive ability, parameter sensitivity, and advantages/shortcomings to provide information for informed model selections. All models had high predictive ability with the majority of predictions (80-90%) within 3°C of the measured water temperature. Sensitivity of model input parameters was found to differ among models, stream system, and season. The most sensitive of assessed parameters, dependent on model and stream, were lateral inflow, starting-water, air, and wet-bulb temperature. Choosing the "best" of the assessed models based on predictive ability was not possible due to similar predictive ability. Therefore, model choice can be based on model capabilities such as RQUAL's ability to predict hourly temperature or SNTEMP's ability to assess alternative shade levels.

14. Hunter, A. K., and A. C. Dolloff. 2003 (in progress). Longitudinal Patterns of Stream Fishes in a Virginia Tailwater. Canadian Journal of Fisheries and Aquatic Sciences.

    Abstract: Community structure of a diverse warmwater fish assemblage was examined in a cool tailwater to discern patterns of abundance, diversity, and distribution in relation to longitudinal and environmental gradients below the dam. We evaluated data across 3 seasons and 3 years during which the peaking flows and temperatures varied. Analyses determined that abundance and diversity did not change significantly between time periods (Kruskal Wallis $p>0.05$). Patterns of abundance and diversity increased with distance from the dam and peaked at tributary junctions. Fish composition was persistent during the study despite changing environmental conditions and faunal similarity increased with increasing distance from the dam. Longitudinal patterns of fish reflected a response to a gradient of increasing temperature and attenuating flows. Multiple linear regression identified mean monthly temperature, temperature depressions, and tributary location as the variables which explained a high level of variability in fish abundance. The observed fish assemblage appears to exist in well-developed patterns under the constructs of high environmental variability. Yet, fish populations do not appear to be stabilized because numbers of individual species highly fluctuated during the study.

15. Anderson, M. R., and C. W. Krause. 2003. Historical Changes in the Brown Trout Fishery in the Smith River Tailwater, Virginia; and Effects of a Hydro-Peaking Tailwater on Age-0 Trout and Nongame Abundance. Roanoke Trout Unlimited, May 21st, Roanoke, Virginia.

16. Anderson, M. R., and C. W. Krause. 2003. Macro-invertebrate Mayhem; a Project Wet activity. Bassett River School program, Bassett Middle School, June 19th.

17. M. R. Anderson completed the Coosa Valley Chapter of Trout Unlimited progress report for the research grant on the Smith River brown trout diet study.

18. Rummel, M. 2003. What Lies Beneath… The surface of the Smith River. Virginia Wildlife.

19. Krause, C. W., and Y. Shen. 2003. Measuring Water-Velocity Profiles with Acoustic Doppler Technology in a Virginia Tailwater. Accepted for presentation at the Virginia Water Research Symposium, October 8-10th.

    Abstract: The two-dimensional flow model RMA-2V was developed for two sections of a hydropeaking tailwater in Virginia as part of a Brown trout fisheries research study. We required known water velocity data at multiple flows to calibrate and validate the model. To measure these velocities at flows too high and swift to wade with a flow meter, we utilized a wireless acoustic Doppler profiler. The floating profiler towed across the channel with a cableway system allowed peak flow water velocities to be safely measured by operators on shore.

20. Shen, Y., and P. Diplas. 2003. Fish habitat assessment with one- and two-dimensional ecohydraulic models. Accepted for presentation at the Virginia Water Research Symposium, October 8-10th.

    Abstract: This paper presents a study that evaluates the predictions of brown trout spawning habitat using 1-D (PHABSIM) and 2-D (RMA2) ecohydraulic models at a selected site in the Smith River, Virginia. Both the 1-D and 2-D models are first calibrated to the moderate flow, and then validated at the base and bankfull flows. During the calibration procedure, the roughness coefficients (1-D and 2-D models) and eddy viscosity (2-D model only) are adjusted so that the model output can match closely the field observations. To transfer hydraulic model output to habitat quality indices, a stream-specific HSC (Habitat Suitability Criteria) is developed for spawning brown trout to insure a successful analysis of fish habitat. The integration of hydraulic output with the HSC makes it possible to examine the relationships between redd density and predicted habitat quality through a polynomial regression analysis, and determine whether these two parameters are significantly correlated. In the end, a scenario including multiple flow simulations, ranging from summer base to spring peaking flows, is implemented with both 1-D and 2-D models. The results are used to quantify the effects of flow regulation on physical spawning habitat in the Smith River.

21. Orth, D. J. 2003. Closing Remarks. International IFIM Users Workshop, Fort Collins, Colorado, June 2-5th.

22. Shen, Y., P. Diplas, and D. W. Crowder. 2003 (manuscript in progress for journal submission). Comparison of one- and two-dimensional ecohydraulic models in a regulated river.

23. Shen, Y., P. Diplas, and D. W. Crowder. 2003. Modelling of scour and fill patterns for embryo survival in the Smith River, Virginia. (Preparing for 5th International Conference in Ecohydraulics, Madrid, Spain, Sept. 12-17, 2004).

24. Crowder, D. W. and P. Diplas. 2003 The use of two-dimensional hydrodynamic modeling in evaluating stream habitat. (Preparing for 5th International Conference in Ecohydraulics, Madrid, Spain, Sept. 12-17, 2004).

25. Anderson, M. R., T. J. Newcomb, and D. J. Orth. 2003. Diet Composition of Brown Trout in a Hydropeaking Tailwater. Abstract submission for Midwest Fish and Wildlife Conference, Kansas City, MO Dec 2003.

Abstract: The Smith River tailwater, Virginia, once provided a unique opportunity for anglers to fish for trophy-sized wild brown trout; however, today, the size structure of the population has shifted to one dominated by small brown trout. Information is lacking on what caused the decline in the size structure, so this study was designed to assess the role that diet composition may have in determining brown trout growth rates. During 2002, 320 brown trout were collected from four reaches during February, May, September, and December via backpack electrofishing. Stomach contents were removed and preserved and returned to the lab where food items were identified, enumerated and weighed. Longitudinal trends in diet composition show that trout in the first 5 km had a diet dominated by Isopoda and Diptera larvae, whereas trout in the lower 18 km of the tailwater had diets consisting of Trichoptera, Ephemeroptera, crayfish and fish. Seasonal trends indicated that Ephemeroptera was an important food item in February, terrestrial insect matter was abundant in the trout diets during May and September, and fish were important during December. The lack of fish in the trout diets from the upper portion of the tailwater may be related to the slow growth of brown trout in the Smith River. Temperature manipulation through alternative flow regimes could aid the abundance of warm-water forage fish throughout a greater portion on the tailwater.

Appendix B. Attached are manuscripts currently in progress and/or review for publication covering research on the Smith River study. The following manuscripts are included in this report.

1. Anderson, M. R., D. J. Orth, and S. M. Smith. 2003 (in review). Historical change in the brown trout fishery in the Smith River Tailwater, Virginia. Southeastern Association of Fish and Wildlife Agencies.

2. Krause, C. W., T. J. Newcomb, and D. J. Orth. 2003 (in review). Thermal Habitat Assessment of Alternative Flow Scenarios in a Tailwater Fishery. River Research & Applications.

3. Hunter, A. K., and A. C. Dolloff. 2003 (in progress). Longitudinal Patterns of Stream Fishes in a Virginia Tailwater. Canadian Journal of Fisheries and Aquatic Sciences.

4. Krause, C. W., and Y. Shen. 2003. Measuring Water-Velocity Profiles with Acoustic Doppler Technology in a Virginia Tailwater. Accepted for presentation at the Virginia Water Research Symposium, October 8-10th.

5. Newcomb, T. J., K. M. Hanna, and M. R. Anderson. 2001. Macroinvertebrate forage in the Smith River tailwater. Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies, 55:116-125

Historical Changes in the Brown Trout Fishery in the Smith River Tailwater, Virginia

M. R. Anderson[1], D. J. Orth[1], and S. M. Smith[2]

[1] Virginia Polytechnic Institute and State University, Department of Fisheries and Wildlife Sciences, 100 Cheatham Hall, Blacksburg, VA 24061-0321; [2] Virginia Department of Game and Inland Fisheries, 1132 Thomas Jefferson Road, Forest, VA 24551-9223

Abstract: Historical data on brown trout from the Smith River tailwater, Virginia, below Philpott Dam, from 1971-2002 were reviewed to assess changes in the fishery over the last 30 years. Data from citation brown trout and electrofishing data were evaluated for changes in size distribution and fish condition. We observed a decrease in the number of citation brown trout over the last 30 years. Relative stock density has also decreased. Although relative condition of citation brown trout was high in the early 1970s, values decreased and have remained stable for the last 20 years. Possible explanations for the decline in the numbers of large brown are also presented.

Key words: Brown trout, Salmo trutta, tailwater

INTRODUCTION

The Smith River tailwater below Philpott Dam has been called a top quality trout fishery and has been nationally recognized as one of the top 100 trout streams by Trout Unlimited. In the Southeastern portion of the United States, the numbers of natural coldwater areas are limited; however, many tailwater trout fisheries exist where hypolimnetic dam releases occur. Unlike many other tailwater trout fisheries, the Smith River tailwater contains a self-sustaining population of brown trout Salmo trutta, which provides a unique opportunity for anglers to fish for wild brown trout.

Historical accounts of the Smith River trout fishery indicate that the tailwater produced trophy-sized brown trout. Over a three week time period in 1974, the state brown trout record was broken on three separate occasions in the Smith River, with the largest trout weighing 6.5 kg (Cochran 1975). With the potential to catch a trophy-sized brown trout, the Smith River tailwater gained popularity with regional anglers.

While it was once common for anglers to catch brown trout weighing more than 1.6 kg, the current trout population seldom produces fish of the size desired by anglers. The Smith River tailwater has been a trout fishery since 1954, when trout were first stocked into the tailwater area. Although changes in the reservoir and tailwater fisheries have occurred, these changes have not been well documented. This paper will report historical changes in the tailwater fishery over the last 30 years and discuss possible causes for changes in the fishery.

STUDY AREA

The Smith River is a fifth order tributary of the Dan River located in Henry Country, Virginia (Figure 1). The Smith River became impounded in 1953 with the completion of Philpott Dam, which is owned and operated by the U. S. Army Corps of Engineers. The dam was built for flood control purposes and to serve as an electric hydropeaking facility. After completion of the dam, flows fluctuate between 1.27 cubic meters per second (cms) minimum base flow and 39.65 cms generation flow. Hypolimnetic releases from Philpott Reservoir maintain cold temperatures in the tailwater for approximately 23 km.

The tailwater was stocked with rainbow trout Oncorhynchus mykiss and brown trout following completion of Philpott Dam to create a coldwater fishery. However, beginning in the late 1960s, there was anecdotal evidence of brown trout reproduction, which was documented in 1977. By the mid-1980s, brown trout stocking was discontinued in the Smith River, but 31,000 harvestable sized rainbow trout (250-325 mm) are still stocked annually.

The tailwater fishery is managed by the Virginia Department of Game and Inland Fisheries (VDGIF). There are two trout management strategies in the Smith River tailwater. The section immediately below Philpott Dam to Town Creek (5.3 km below the dam) and the section from the towns of Bassett to Kohler (10.0-24.0 km below the dam) are managed with a 178 mm minimum length limit and 6 trout per day creel limit. In July 1976, the section of the tailwater from Town Creek to the town of Bassett (5.3-10.0 km below the dam) was established as a Special Regulations Section and is managed as a trophy trout area with a 406 mm minimum length limit, two trout per day creel limit, and single hook artificial lure gear restriction.

METHODS

Brown trout length and weight data and citation numbers were collected using citation catch records from 1971-2002 from the Angler Recognition Program of VDGIF. The citation criteria changed over the years with a 907 g minimum weight from 1971-1983. From 1984-1988, minimum citation weight for trout was 1361 g, and from 1989-1991, the minimum weight was 1814 g. Beginning in 1992, the minimum weight requirements for brown trout was raised to 2268 g, and starting in 1995, a 635 mm minimum length could be used in place of the minimum weight requirement.

Electrofishing was used to assess the trout population beginning in 1983. During the 1980s and 1990s a variety of electrofishing gear types were used including barge, boat and backpack electrofishing units. Beginning in 2000, paired electrofishing barges were used to collect brown trout. All trout were measured to the nearest mm (total length) and starting in 1991, fish were weighed to the nearest g.

Relative stock density (RSD) indices were used to assess length-frequency distribution of trout sampled by electrofishing. Relative stock density (Wege and Anderson 1978) was calculated by the formula: RSD = (number of fish$\geq$specified length/number of fish$\geq$minimum stock length) x 100; where the specified lengths were 230 mm (quality length) and 300 mm (preferred length) and minimum stock length is 150 mm (Milewski and Brown 1994). Confidence intervals were calculated for RSD values by the formula presented in Gustafson (1988).

Trout condition was assessed by relative weight (W_r; Wege and Anderson 1978) and was calculated for all citation trout and trout sampled by electrofishing. Relative weight was calculated by the formula: $\underline{W}_r$ = (W/$\underline{W}_s$) x 100; where W is the weight of the individual fish and $\underline{W}_s$ is the length-specific standard weight of the fish. The standard weight ($\underline{W}_s$) equation was proposed by Milewski and Brown (1994) and is: $\log_{10}\underline{W}_s$ (g) = -4.867 + 2.960 $\log_{10}$TL (mm).

RESULTS

The size structure of the brown trout population in the Smith River tailwater has changed since the 1980's. During the early and mid 1980's, RSD values for brown trout collected by electrofishing were high for both the RSD-230 and RSD-300 categories (Figure 2). However, in the late 1980's, RSD values decreased and have remained between 40 and 50 throughout the 1990's. Relative stock densities have significantly declined since 2000.

The size structure of citation brown trout has also changed over time. Due to changes in the citation criteria after 1984, mean total length of brown trout was assessed from 1971-1984 and included all citation brown trout greater than 907 g. Mean total length of citation brown trout during this time period was greatest in the early 1970's (Figure 3). During the mid to late 1970's, mean total length of brown trout decreased. Mean total length of citation brown trout remained stable through the early 1980's.

Number of citation brown trout

The number of citation brown trout larger than 2268 g has varied over the last 30 years, with a large decline in numbers since 1995. The number of citation brown trout larger than 2268 g was consistent until 1985 (Figure 4). After 1985, there were peaks of smaller citation trout greater than 2268 g in 1989 and 1991, but following these peaks, the number of citation brown trout has diminished. From 1973-1977, the number of citation brown trout larger than 3402 g peaked (Figure 4). The peak of citation brown trout weighing between 2268 g and 3402 g was from 1988 to 1991. During the 1970's, 7 citation brown trout larger than 4536 g were caught; however, only 3 have been reported since 1979.

Relative Weight

The condition of citation brown trout has changed markedly since the 1970's. Relative weight of citation brown trout was high in the early 1970's. From 1971-1975, average $\underline{W}_r$ values for citation brown trout were well over 120, with average $\underline{W}_r$ values being as high as 160 in 1971 (Figure 5A). However, in 1975 average $\underline{W}_r$ values for citation brown trout decreased, and the values have remained stable at 100-120 since the late 1970's.

Relative weight values of brown trout collected by electrofishing has remained stable from 1991 until 2002 (Figure 5B). Relative weight values of trout randomly collected in the field have remained stable over the last 10 years, which is consistent with the stabilization of $\underline{W}_r$ values of citation brown trout. Brown trout larger than 300 mm have a higher $\underline{W}_r$ value than trout that are 200-300 mm (Figure 5B). In addition, $\underline{W}_r$ values for citation brown trout are greater than $\underline{W}_r$ values for trout collected by electrofishing (Figure 5).

DISCUSSION

The Smith River brown trout population has changed over the last 30 years. During the early years of the tailwater fishery, the population included numbers of large trophy-sized brown trout. However, through time, the population has shifted to a distribution dominated by smaller brown trout. This is apparent through shifts in the RSD values and a decline in the number of citation brown trout. In addition, $\underline{W_r}$ values were high for citation brown trout in the 1970's and have decreased over the years.

The declines in the number of trophy-sized trout and smaller size distribution may be related to trophic interactions or overexploitation of trout in the tailwater. During the 1970's, a large number of trophy brown trout were caught in the tailwater, which increased the popularity of the fishery. With the additional fishing pressure during the 1970's, overexploitation of the brown trout may have occurred. However, if the fishery was overexploited, the fishery should have the potential to rebound. In 1995, harvest rates of brown trout in the Smith River tailwater were low (5%), which indicates that overexploitation of brown trout is not the cause of the reduction in trout size and citation numbers (Hartwig 1998).

Because trophy-sized and small trout have both changed over time, the population may be influenced by slower growth rates or higher mortality, which could be related to the trophic levels in the tailwater. Several factors may have contributed to the changes that occurred in the Smith River tailwater, including water quality and reservoir trophic status and forage availability.

One potential cause for the changes to the tailwater is that water quality and reservoir trophic status have changed. Changes in the reservoir can impact the primary production potential in the tailwater area, which can alter forage availability. Below Philpott Dam, constant cold water temperature limits the ability of fish, other than trout, to survive in the area. With the loss of non-salmonid fish species, trout will have to utilize alternative food resources such as drift and other macroinvertebrates, which are also impacted by the changes in flow. In addition, the constant flushing of water during generation periods causes changes in the substrate composition of the tailwater, thus potentially limiting the macroinvertebrate community Research has shown that after the completion of hydroelectric facilities, macroinvertebrate species richness decreases as well as total density and biomass (Trotzky and Gregory 1974; Garcia de Jalon et al. 1994). If food availability for trout is reduced, growth rates of trout may be suppressed

One primary cause for the trophic change in the tailwater trout fishery is that alewives <u>Alosa pseudoharengus</u> no longer pass through the turbines during generation, thus eliminating a forage base for trout near Philpott Dam. Discussions with anglers and biologists indicated that trophy brown trout were present near the dam and alewives were present in the tailwater area. The alewives provided a valuable forage base for trout in the tailwater immediately below the dam. Alewives were stocked into Philpott Reservoir in the late 1960's, and it is speculated that the alewife population exploded in the 1970's. This could account for the increased number of brown trout trophies in the tailwater area during the early 1970's. Since that time, the number of alewives in Philpott Reservoir may have stabilized, eliminating alewife passage through the dam.

Although the number of trophy brown trout in the tailwater has decreased, the overall number of brown trout in the tailwater has not. High numbers of brown trout provide numerous opportunities for anglers, but smaller fish are not highly desired by Smith River anglers (Hartwig 1998). In 1995, the value of the Smith River fishery was estimated at $440,000 per year (36,000 angler hours), and if the number of large brown trout in the system were to increase, the value of the fishery could possibly double (Hartwig 1998). Despite the shift to smaller brown trout, the Smith River is still highly valued for naturally reproducing brown trout.

Further research into the causes for the decline in the Smith River trout population is needed. Because exploitation rates of brown trout in the tailwater are low, the decline in the trout population appears to be the result of changes in growth rates, mortality, or recruitment. While alewives once provided a valuable prey base, they are no longer available as forage below the dam. Food resources in the tailwater need to be evaluated to determine potential forage options for trout. Additional research is also needed to evaluate recruitment and mortality of the brown trout population. If limiting factors could be identified and remedied, the brown trout population could be improved, thus the increased potential for larger, trophy-size brown trout.

ACKNOWLEDGMENTS

Citation data was obtained from the Virginia Angler Recognition Program. Funding for data obtained prior to 2000 was provided by the Federal Aid to Sport Fish Restoration Grants F-35R and F-111R. Funding for data obtained after 2000 was provided by the Federal Aid to Sport Fish Restoration Grant F-124R. Additional thanks to all of the Virginia Department of Game and Inland Fisheries and Virginia Polytechnic Institute and State University personnel who have assisted with data collection.

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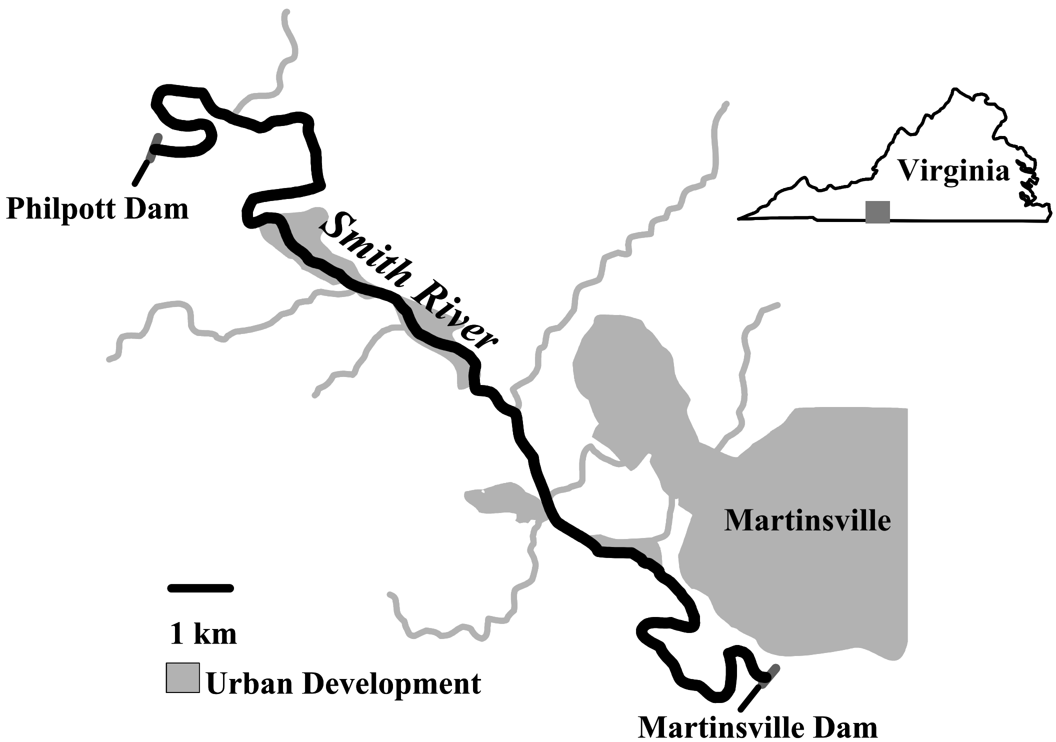

Figure 1. Map of the Smith River, VA, tailwater.

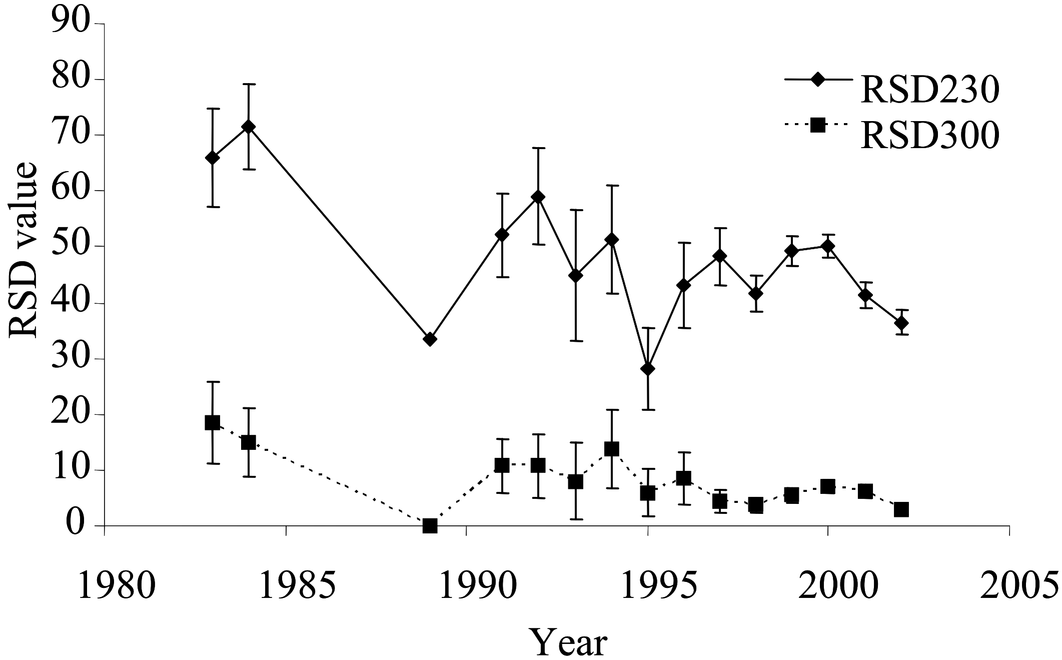

Figure 2. Relative stock density (RSD) indices values and 95% confidence intervals for brown trout collected by electrofishing from the Smith River, Virginia, from 1983 to 2002.

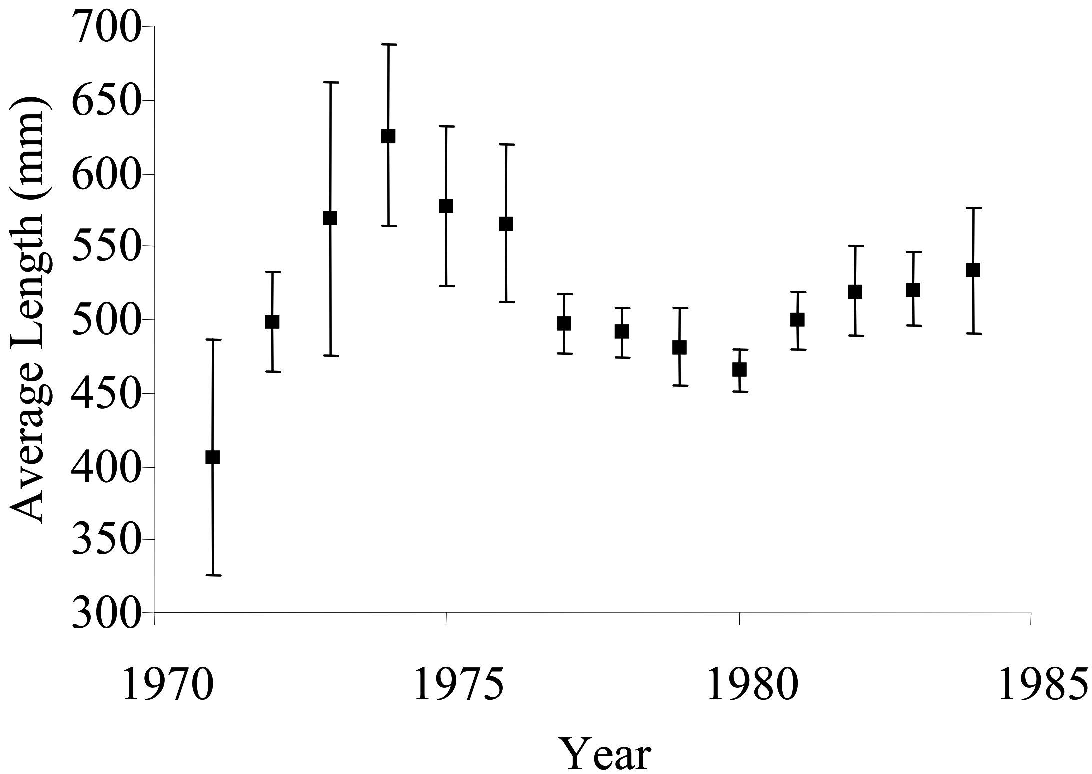

Figure 3. Average length and 95% confidence intervals for citation brown trout from the Smith River, VA, from 1971-1984. All citation brown trout 2 lbs and greater were included.

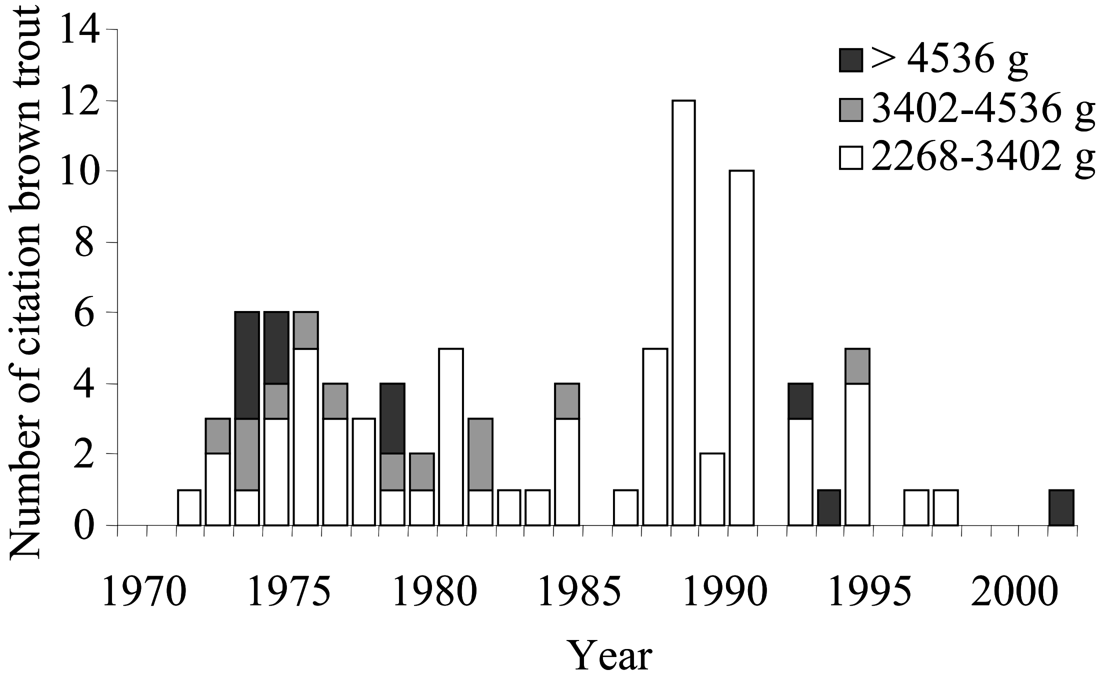

Figure 4. Total number of citation brown trout from the Smith River, VA from 1971-2002.

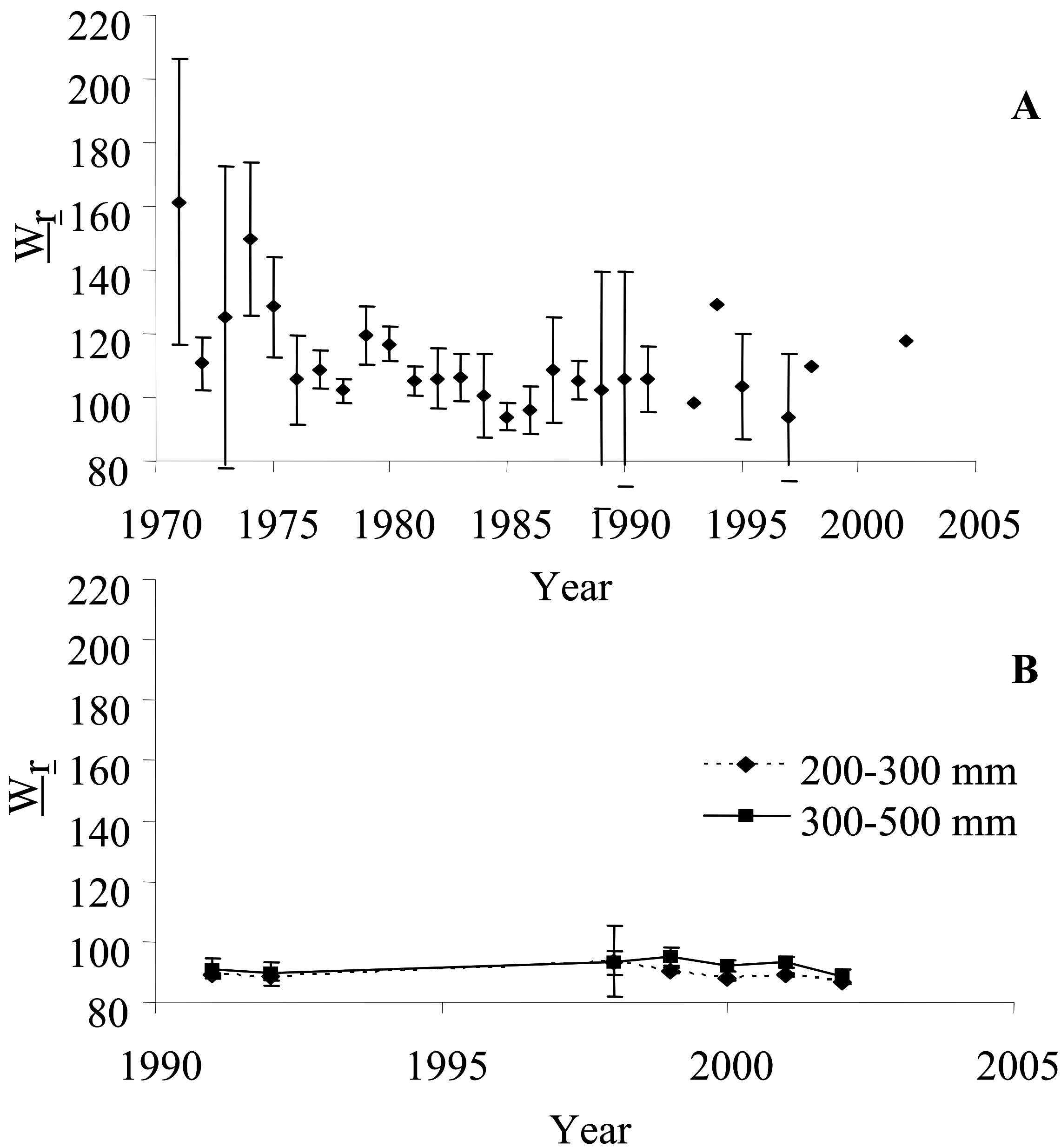

Figure 5. Average relative weight ($\underline{W}_r$) and 95% confidence intervals for citation brown (panel A) and brown trout collected by electrofishing (panel B) from the Smith River, VA.

Thermal Habitat Assessment of Alternative Flow Scenarios in a Tailwater Fishery

Short Title: Thermal Habitat in Tailwater Fishery

COLIN W. KRAUSE*, TAMMY J. NEWCOMB[a], AND DONALD J. ORTH
Virginia Tech, Dept. of Fisheries and Wildlife Sciences,
100 Cheatham Hall, Blacksburg, VA 24061, USA

Grant Sponsors: Virginia Water Resources Research Center (primary); Virginia Department of Game and Inland Fisheries (cooperating)

*Correspondence to: Virginia Tech, Dept of Fisheries and Wildlife Sciences, 100 Cheatham Hall, Blacksburg, VA 24061, USA. 540-231-530, Fax 540-231-7580, cokrause@vt.edu

[a]Current Address: Michigan Dept. of Natural Resources, Fisheries Division, P.O. Box 30446, Lansing, MI 48909, USA

KEY WORDS: temperature modeling, thermal habitat, flow regime, tailwater, ADYN & RQUAL, brown trout, growth

ABSTRACT

The Smith River tailwater (Henry County, VA) offers a self-sustaining brown trout fishery managed for trophy trout (≥406 mm), however trophy sized fish are rare. Slow growth and small size are likely caused by any one or a combination of limited food resources, physical habitat, and thermal habitat. To evaluate the potential for thermal habitat improvement, temperature changes resulting from alternative flows released from the hydropeaking Philpott dam were assessed with a one-dimensional hydrodynamic model coupled with a water temperature model. Simulated temperatures from each flow scenario were assessed every 2 river kilometers over a 24 kilometer river section below the dam for occurrence of optimal growth temperatures, as well as compliance with Virginia Department of Environmental Quality hourly temperature change and daily maximum temperature standards. The occurrence of optimal growth temperatures was increased up to 11.8% over existing conditions by releasing water in the morning, decreasing the duration of release, and not increasing baseflow. Occurrence of hourly temperature changes greater than 2°C was reduced from 4% to 0-1.2% by non-peaking releases, increasing baseflow, releasing in the morning, and decreasing the duration of release. Maximum temperature occurrence greater than 21°C decreased from 1.3% to 0-0.1% by releasing every day of the week to prevent elevated temperatures on non-generation days, increasing baseflow, increasing duration of release, and releasing in the morning rather than evening. Despite conflicting adjustments to best improve all thermal criteria concurrently, a 7-day/week, morning, one hour release regime was determined to improve all criteria compared to existing conditions.

INTRODUCTION

Tailwater fisheries are present in river reaches below dams releasing water from an upstream impoundment. Hypolimnetic release tailwaters are often important fisheries for trout because without the dam the river could not support coldwater fish species. Because water temperature is a critical parameter for the survival, growth, spawning, and embryonic development of fish, the impact tailwaters have on water temperature is important for management of these fisheries. In hydropeaking tailwaters that rapidly fluctuate releases to generate electricity, the wide range and quickly changing temperatures can reduce the survival and growth of the trout in the fishery. Temperature influences survival and growth because it regulates the speed of muscle contractions and metabolic rates, which dictates swimming, prey capture, and food assimilation ability (Chavin, 1973; Reynolds and Casterlin, 1979; Wardle, 1979; Saltveit, 1990). Peaking flow regimes in tailwaters have the ability to cause rapid temperature declines when cold-water flow-pulses mix with downstream water warmed by ambient conditions (Cushman, 1985; Krause, 2002; Orth et al., 2001; 2002). When temperature quickly declines (i.e., cold-shock) the rate of body heat loss is rapid (Reynolds and Casterlin, 1979) and fish can experience a loss of equilibrium or mortality (Chavin, 1973; Smythe and Sawyko, 2000). Downstream displacement may also result from reduced swimming ability due to cold-shock (Ottaway and Forrest, 1983; Saltveit et al., 1995).

Temperatures that are too cold or hot cause stress or mortality to fish. For brown trout (*Salmo trutta*) 0-4⑤C and 19-30⑤C are the lower and upper critical ranges (Elliot, 1981). The range for which temperature is critical is due to acclimation temperature, duration spent at a cold or hot temperature, and the rate at which temperature changes (Chavin, 1973). Between the lower and upper critical temperature ranges lies an optimal temperature growth range, which for brown trout is approximately 12-19⑤C (Brown, 1974; Brungs and Jones, 1977; Raleigh et al., 1986; Smith, 1994; Ojanguren et al., 2001). In addition to optimal temperatures, the thermal regime is also important for growth. A diel temperature cycle causes significantly greater growth rates in brown trout than constant temperature conditions (Spigarelli et al., 1982). In tailwaters, many of these restrictive temperature conditions such as constantly cold release temperatures, warm temperatures downstream due to ambient conditions and low flows, and large hourly temperature changes during hydropeaking may restrict growth of brown trout. Such thermal conditions occur in the Smith River tailwater (SRT) in southwestern Virginia, and are potential causes of the slow growth and small size of brown trout in this tailwater.

One hypothesis based on preliminary temperature data from year 2000 in the SRT is that brown trout growth is limited by lack of optimal growth temperatures, rapid hourly temperature fluctuations, and high temperatures at downstream locations. Near the dam (0.5 rkm) temperature averages 8°C (SD = 1°C) and exhibits very little daily fluctuation. At upstream locations (~0-5 rkm) daily temperature rarely exceeds 12°C and these cold temperatures would extend further downstream during summer if not for inflows from a tributary at 5.3 rkm. Occurrences of brown trout optimal growth temperatures (12-19°C) are greatest from May through September and occurrence increases with downstream distance. During non-generation periods (typically weekends) at downstream locations

(~14-24 rkm) water temperatures up to 25°C were recorded which infringes upon the upper critical range (19-30°C) of brown trout. Additionally, these elevated temperatures exceed the Virginia Department of Environmental Quality's (DEQ) 21°C maximum temperature standard for stockable trout waters (DEQ 1997). The DEQ's hourly temperature change standard of 2°C was also exceeded by temperature declines up to 7°C within an hour caused by the hydropeaking releases.

The SRT flow schedule directly influences thermal regime, therefore, adjustment to the flow schedule could improve thermal habitat. Adjustments could include changes in flow duration, magnitude, time of day, days per week, and outflow temperature of the release. Additional alternatives include increases in baseflow and ramping rather than peaking the release. The most time and cost efficient method to assess temperature under numerous alternative flow scenarios is with a model. Therefore, the objectives of our study were to use a stream temperature and flow model to assess alternative flow scenarios in the SRT to determine which would: (1) increase occurrence of optimal growth temperatures (12-19°C), (2) reduce occurrence and magnitude of hourly temperature fluctuations, and (3) reduce occurrence of 21°C temperature exceedance in the SRT for the improvement of brown trout growth and survival.

STUDY AREA

The SRT is located in southwestern Virginia (Henry County) and was formed when Philpott Dam was completed by the Army Corps of Engineers (USACE) in 1953 to provide flood control, hydropower, and recreational opportunities (Figure 1). The hydropower operation uses a peaking regime in which flow releases vary widely and rapidly (1.4 to 36.8 cms within 30 minutes) to provide electricity during peak demand periods (USACE, 2001; USGS, 2001). The release is hypolimnetic, thus cold water temperatures enable a self-sustaining brown trout population and stocked rainbow trout fishery to exist from Philpott dam to Martinsville dam, a 32 rkm reach of which ~24 rkm are fishable by wading. The SRT, managed for trophy trout (i.e., ≥ 406 mm) from 5.3 to 10.0 rkm below Philpott dam, produced the historic Virginia state record brown trout caught in 1979 weighing 8.48 kg (Mohn, 2001). Presently, brown trout seldom exceed 406 mm (0.63 kg) and possible reasons are limiting thermal habitat, food resources, and/or physical habitat (Newcomb et al., 2001; Orth et al., 2001; 2002).

METHODS

Data Collection and Field Measurements

The hydrodynamic model, ADYN, coupled with the water quality model, RQUAL, of the Tennessee Valley Authority's River Modeling System (Hauser and Walters, 1995) was used to predict hourly temperatures from July 1999 through February 2001 at 2 rkm intervals from 0.6 to 24.3 rkm below Philpott dam. We evaluated thermal habitat from May through September 2000 when SRT brown trout mean absolute growth rates are greatest (Orth et al., 2001). To develop the model for the SRT a suite of input parameters were collected. Hourly meteorological and solar radiation parameters were obtained from the nearest observation stations; Roanoke, VA 74 km away (NCDC, 2001) and Bluefield, WV 144 km away (CONFRRM, 2001). Discharge data was obtained from three gaging stations along the SRT (USGS, 2001). Lateral inflows were estimated by calculating flow differences between gaging stations. Water temperature was recorded every 30 minutes with Onset® data loggers at locations near the dam and downstream (0.6, 2.7, 5.1, 5.6, 10.2, 18.3, and 24.3 rkm) for SRT thermal habitat assessment and model calibration, validation, and predictive ability assessment. Cross-sectional streambed profiles at 37 locations were measured using surveying techniques. Stream width, riparian vegetation offset from stream-bank, and vegetation height (Bartholow, 1989) were measured at 102 random locations along each stream bank from 0.5-24.0 rkm. Elevation, latitude, longitude, river kilometer locations, and azimuth were measured from a topographic map.

Model Calibration and Validation

The RQUAL model was calibrated with one year of data and validated with a second year of data. To calibrate the model, input parameters (typically calibration coefficients) were adjusted until the trend of the predicted and measured water temperature closely matched when viewed graphically at multiple longitudinal river locations. Calibration effectiveness was further assessed by calculating predictive ability, which is the difference between the measured and predicted temperature values. Predictive ability of hourly temperature predictions and daily maximum hourly temperature change was assessed as hourly residuals (i.e., difference between predicted and measured temperature) averaged by month.

Model validation was assessed with a one sided chi-square test for difference ($P \leq 0.05$) between counts of absolute residuals from the calibrated time periods (summer, fall, winter 1999/2000 predictions) to the validation time periods (summer, fall, winter 2000/01 predictions). Predictions for the validation periods used the same calibrations as the calibrated seasons and the predictions tested were from the most downstream modeled site (24.3 rkm) where residual error was greatest. Counts of absolute residuals were separated into a 2x2 contingency table based on a predictive ability category of suitable (0-4°C) versus unsuitable (>4°C) for predicting biological differences (Conover, 1971).

Alternative Flow Scenarios

Fifteen alternative flow scenarios were developed in addition to the existing flow regime from May to September 2000 (Table 1). Scenarios differed from existing conditions by altering the number of days per week of generation, baseflow, time of day of generation release, whether releases were peaked or ramped, generation duration, as well as no generation. A run of river flow regime was developed using daily inflow into Philpott reservoir computed by the USACE. Ramping scenarios increased flow from 1.4 cms to 36.8 cms over three hours and remained at 36.8 cms for one hour. Total quantity of water released over this four hour period (3 hr ramping + 1 hr at 36.8 cms) is equal to that of the two hour generation scenarios. Time at which flow reached 36.8 cms was 7 am or 5 pm.

Hourly predicted temperatures from May through September 2000 at 13 locations (2 rkm intervals) from 0.6-24.3 rkm downstream of Philpott dam were compared between the 15 alternative flow scenarios and existing flow conditions. Scenarios were evaluated for percent time optimal brown trout growth temperatures (12-19°C) occurred, percent time daily maximum hourly temperature change (MHTC) exceeded 2°C, magnitude of MHTC, and percent time 21°C was exceeded. Selection of these temperature criteria were based on the assumption that temperatures outside 12-19°C will restrict food assimilation and metabolic activity, rapid temperature fluctuations will induce stress, and temperatures greater than 21°C will induce stress and/or be lethal.

RESULTS

Model Calibration

Hourly predictions closely followed the diel temperature fluctuation for most days and river locations, however there were occurrences of poor predictive ability (Figure 2). Absolute residuals of hourly predictions averaged from May through September 2000 (average under and over-prediction residuals in parentheses) were 0.9°C (-0.6, +1.1), 1.4°C (-1.2, +1.3), and 1.6°C (-1.7, +1.0) at 5.1, 18.3, and 24.3 rkm respectively. Mean absolute residuals of MHTC were 2.3°C (-0.6, +2.5), 1.8°C (-0.1, +1.9), and 1.0°C (-0.3, +1.0) at 5.1, 18.3, and 24.3 rkm respectively.

Model Validation

Graphically, the trend of the predicted and measured temperature for the independent dataset seasons matched in closeness and similarity to the calibrated seasons. Statistically, the RQUAL model validated (i.e., no statistical difference between residual error counts of the calibration and validation time periods) for all assessed seasons within the suitable predictive ability category (0-4°C) ($P = 0.50$).

Alternative Flow Scenarios

The alternative flow scenarios caused thermal conditions that were better or worse for brown trout than the existing conditions in three different ways; (1) the scenario caused improvement at all river locations (i.e., from the dam at 0 rkm to 24.3 rkm downriver), (2) the scenario caused no improvement at any river location, or (3) the scenario caused improvement at some locations but not others. The range over which scenarios caused better and worse conditions in relation to the existing conditions is shown in Figure 3.

<u>Occurrence of Optimal Growth Temperatures</u>. The morning 1 hr release scenario caused the largest mean increase over existing conditions that optimal growth temperatures occurred from 2.2 to 24.3 rkm (+11.8%, Table 2). The percent occurrence of optimal growth temperatures increased to +18.9% when narrowed to the trophy trout section (5.3-10.0 rkm). Only the steady baseflow scenario caused greater percent occurrence of optimal growth conditions (+27.8%) within the trophy trout section. However, the steady baseflow scenario caused a large decline in percent occurrence (-20.3%) over existing conditions downstream (18.3-24.3 rkm). Morning releases (7 am), shorter duration releases (1 hr), and scenarios with 1.4 cms baseflow, caused greater occurrence of optimal growth temperatures than evening releases (5 pm), longer duration releases (2 hr), and scenarios with increased baseflow (2.8 cms), respectively.

Occurrence of optimal growth temperatures (12-19°C) was greatest with release in the morning, for 1 hr, and/or no increase to baseflow. In this scenario, water temperature released from the dam averages 8°C, therefore as water travels down-river it warms at a rate reliant on ambient conditions to reach the optimal growth range. Morning flow releases allow ambient conditions to warm the water all day following the release, thus water temperatures remain within the optimal growth range for a greater duration of the night. Whereas evening releases halt the day-time warming, temperatures fall below the optimal range, and are unable to warm again until the following day. A shorter duration release (1 hr vs. 2 hr) and no increase to the baseflow allow the river warm more easily because there is less volume of water in the channel than for other scenarios.

<u>Maximum Hourly Temperature Change (MHTC)</u>. Scenarios with no hydropeaking (steady baseflow, increased steady baseflow, and run of river) caused the largest reduction in daily MHTC from existing conditions. Additionally, these scenarios prevented MHTC from exceeding 2°C during more than 99% of May through September 2000 at all river locations (0.6-24.3 rkm) (Table 2). Of the unsteady release scenarios (i.e., hydropeaking), the morning 1 hr release with increased baseflow caused the largest reduction to the average daily MHTC and percent time MHTC exceeded 2°C (2.8% reduction over existing conditions; for reference 4.2% = 1 hour per day 2°C exceeded out of 30 days) (Table 2). Morning releases (7 am), shorter duration releases (1 hr), and scenarios with increased baseflow (2.8 cms), caused less occurrence of MHTC exceeding 2°C than did evening releases (5 pm), longer duration releases (2 hr), and scenarios with 1.4 cms baseflow, respectively.

Daily MHTC was decreased with release in the morning, for 1 hr, increased baseflow, and non-peaking releases. Downstream temperatures are cooler in the morning and temperature of released water is closer to the channel water temperature, which reduced MHTC when mixed. The lesser quantity of water released over 1 versus 2 hrs had reduced ability to change temperature in the channel, and because of attenuation, impacted less distance downstream. An increased baseflow decreased MHTC by dampening the impact of released water and by maintaining cooler temperatures within the channel. Non-peaking scenarios completely eliminate the pulse of cold water traveling rapidly downriver and thus the source of large hourly temperature changes. Ramping flows slightly decreased the magnitude of MHTC by gradually increasing the flow of released water so that released and downstream water mixed more slowly. However, the evening ramped release increased the percent time 2°C MHTC was exceeded due to an extended mixing period.

<u>Exceedance of 21°C</u>. Temperature predictions exceeded 21°C most often under the steady baseflow and run of river scenarios at downstream sites (~12.4-24.3 rkm) (Table 2 and Figure 3). Exceedance of 21°C was prevented during more than 99% of May through September 2000 by seven scenarios (increased steady baseflow, morning 2 hr release, evening 1 hr & 2 hr release with increased baseflow, morning 1 hr & 2 hr release with increased baseflow, and morning ramped release) (Table 2). Predicted temperatures rarely exceeded 21°C (<1%) during the months of May and September for all scenarios at all river locations (0.6-24.3 rkm).

The 5-day/week release scenarios caused temperatures to exceed 21°C during weekends when no hydropeaking occurred at sites from 6.4 to 24.3 rkm (Figure 4). Additionally, temperatures remained elevated throughout the weekend where weekend minimum temperatures are similar to weekday maximum temperatures. A 7-day/week release reduces 21°C temperature exceedances and prevents elevated temperatures occurring for prolonged durations (Figure 4). Elevated weekend temperatures from the 5-day/week release scenario increases the diel temperature flux when flows are peaked at the beginning of a week.

Exceedance of 21°C was reduced by 7-day/week, morning, 2 hr, and/or increased baseflow release. Seven day/week release prevented the occurrence of elevated temperatures during non-generation weekends. Morning release cooled temperatures at the beginning of the day, thus reducing the ability of ambient conditions to raise temperatures above 21°C by the end of the day. The larger quantity of water released over 2 hrs versus 1 hr, as well as with an increased baseflow, cooled downstream (~18.3-24.3 rkm) temperatures thus reducing maximum temperatures.

<u>Non-generation flow scenarios</u>. The run of river flow scenario releases the same flow volume and regime as that incoming into the reservoir. This provided a natural flow regime in terms of water quantity, but not water temperature since the release is hypolimnetic. The run of river and steady baseflow scenarios could be detrimental to the downstream trout fishery because they allow high percentages of June, July, and August to exceed 21°C from 12.4 to 24.3 rkm. The increased steady baseflow scenario prevented 21°C exceedances, but ~~it~~ greatly reduced (-20.2%) the occurrence of optimal temperatures from existing conditions (Table 2). The only clear improvement caused by non-generation scenarios was the elimination of MHTC >2°C.

DISCUSSION

This study demonstrates that modeling exercises empower river resource managers with the ability to determine how flow release regimes will affect thermal habitat and which regimes will improve thermal habitat for the growth of brown trout in the Smith River. We found that the best scenario to improve one temperature criteria was not the best to improve others, therefore managers will be faced with choosing a scenario that improves all criteria at the least compromise. For example, the morning 1 hr release scenario was determined to offer minimal compromise by providing improvement over the existing conditions for all assessed criteria. However, the steady baseflow scenario improved optimal growth conditions more than the morning 1 hr release scenario within the trophy trout section, but worsened conditions downstream. Therefore, if the desire is to improve the fishery in the trophy trout section because of an increased quality of fishing experience, the tradeoff maybe appropriate. Through the use of a stream temperature modeling tool, we were able to assess the complexities and trade-offs regarding alternative releases and river temperatures with measures of certainty of anticipated responses. Furthermore, the results from this study provide for the generation of hypotheses to be used in an adaptive management framework.

Biological Significance of Alternative Flow Scenarios

Alternative flow scenarios considered for implementation that improved thermal conditions must also be evaluated from other physical and biological aspects. A small scale Instream Flow Incremental Methodology (IFIM) study assessed availability of physical habitat under different flow regimes in the SRT (USFWS, 1986). Findings indicated that habitat for all brown trout life stages are limited by the existing flow regime where baseflow (~1.4 cms) is too low and generation flow (~36.8 cms) is too high for optimal amounts of habitat. Maximum available habitat from ~0-12 rkm occurs at flows <17 cms and further downstream optimal flows are unknown. Associating IFIM information with temperature modeling reveals that morning release with increased baseflow would improve physical habitat and thermal habitat. However, the increased steady baseflow scenario, which approximates mean annual flow, may improve physical habitat but would reduce the occurrence of optimal growth temperatures.

Flow induced impairment of Brown trout feeding should be assessed prior to implementing an alternative flow aimed toward improving trout growth. Brown trout in the SRT primarily forage on aquatic invertebrates (Orth et al., 2002). Increased water velocities during peak flows improve the availability of food resources by dislodging invertebrates into the water column (e.g. drift) (Lauters et al., 1996; Lagarrigue et al., 2002). Whether brown trout are able to forage with equal effort during peak and base flows in the SRT is unknown. Lagarrigue et al. (2002) found evidence that brown trout did not feed during peaking flows, rather consumption was highest after hydropeaking. Brown trout longitudinal movement is also less during peak flows (Lauters, 1995 as cited in Lagarrigue et al., 2002) suggesting shorter duration releases would cause less restriction on brown trout forage ability. Thus, hydropeaking may increase the availability of aquatic invertebrate drift, but brown trout feeding is restricted until after peak flows subside (Lagarrigue et al., 2002).

Despite the potential for increased invertebrate drift, the magnitude and duration of hydropeaking may also serve to reduce invertebrate populations. Impaired invertebrate populations occur from their persistent flushing without replenishment from an upstream source (e.g. due to a dam) and from poor substrate diversity due to streambed scouring (Moog, 1993; Lauters et al., 1996). Hydropeaking in the SRT has scoured and reduced substrate diversity in the upstream reaches (<4 km downstream of the dam) to predominately (80% bottom coverage) large rocks (>64 mm) and bedrock (Orth et al., 2001). The poor substrate diversity, as well as instability in depth, velocity, and temperature are hypothesized reasons for the low invertebrate density and family richness in the upstream reaches of the SRT (Newcomb et al., 2001). Throughout the SRT invertebrate densities are 2-5 times lower than those in unregulated Virginia rivers of similar size (Newcomb et al., 2001). Unregulated Virginia rivers typically have 800-1,000 invertebrates/m^2, whereas the majority of sites in the SRT have less than the poor food grade classification number of 538 organisms/m^2 (Newcomb et al., 2001). Brown trout, which are also piscivorous, must rely heavily on invertebrates in the SRT because reaches where brown trout densities are highest (3-9 rkm) do not overlap with areas of high forage fish densities (13-24 rkm) (Orth et al., 2002; Hunter, 2003).

Enhancing brown trout growth through thermal habitat improvement assumes water temperature is the primary growth-limiting factor. Alternative flow scenarios that improve thermal habitat may not enable increased growth due to the potentially limiting food resources in the SRT. If improved thermal habitat causes growth to the invertebrate forage-base and/or increased habitat overlap between forage fish and brown trout, then there is greater potential for noticeable results from an alternative flow scenario. Understanding which factors are limiting, as well as the interaction between factors such as water temperature, food resource availability, and habitat availability, will enable managers to determine the potential for flow regime management to improve the SRT brown trout fishery. Determining growth-limiting factors and evaluating their effect on growth can be predicted with bioenergetics modeling, which links growth to energy intake and losses.

Control Rules

The alternative flow regimes assessed by this study were consistent from one day to the next, however, another option is to alter the flow regime depending on daily conditions. Changing the flow regime from one day to the next could be based on a control rule that if met by conditions one day, would determine the flow regime used later that day or the next day (Schreiner, 1997; 2001). Control rules are typically seasonally specific and must address selected criteria. For example, if a very warm summer day occurs surpassing a set of meteorological conditions (i.e., control rule) known to cause downstream SRT temperatures to exceed 21°C, the flow regime would be changed (e.g. increased baseflow or release duration) to cool downstream temperatures (Schreiner, 2001). The implementation of day to day temperature management via flow alteration requires real-time monitoring of water temperatures, meteorological conditions, and flow. Implementation of control rules may be viewed as disadvantageous by recreationists and anglers in the river if flow schedules are subject to daily changes and thus present a lack of predictability for recreation and fishing.

Cost-Benefit Analysis

The alternative flow regimes were assessed from a thermal habitat perspective, yet it is unknown how they differ economically. It is recommended that the USACE consider integrating the results of this habitat assessment with hydropower operations via cost-benefit analysis. This may determine if the value of the fishery to the local community would surpass the value of the power created by Philpott dam.

Conclusion

Our temperature modeling approach demonstrates that thermal conditions in the SRT can be modified to benefit brown trout compared to existing conditions. Other implications of flow scenarios on factors other than temperature (e.g. food and habitat resources) requires further evaluation. This study offers a basis toward achieving improved growth via thermal habitat enhancement by providing an understanding of flow effects on temperature. In summary, those effects are: 1) increased occurrence of optimal growth temperatures (12-19°C) by releasing in the morning, for shorter durations, and/or not increasing baseflow; 2) decreased MHTC by releasing in the morning, for shorter durations, increasing baseflow, and/or ramping flow; and 3) decreased 21°C exceedance by releasing every day of the week, in the morning, for longer durations, and/or increasing baseflow. Maximum enhancement of the tailwater fishery will result from a combination of careful selection and adherence to the appropriate flow scenario(s) and adaptive management that provides the greatest benefit for the least compromise.

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Table 1. Flow scenarios assessed with the ADYN & RQUAL model on the Smith River from May 1 to September 31, 2000.

Scenario Name	Days per Week of Generation	Baseflow (cms)	Peak Flow (cms)	Release Time	Ramping Duration (hours)	Generation Duration (hours)
Existing conditions	~7	~1.4	~36.8	~5pm		~1
5 day 2hr release	5	1.4	36.8	5pm		2
5 day 5hr release	5	1.4	36.8	5pm		5
Steady baseflow	0	1.4	1.4			
Increased steady baseflow	0	8.5	8.5			
Run of river	0	------------Daily Inflow to Philpott Reservior------------				
Evening 1hr release	7	1.4	36.8	5pm		1
Evening 2hr release	7	1.4	36.8	5pm		2
Morning 1hr release	7	1.4	36.8	7am		1
Morning 2hr release	7	1.4	36.8	7am		2
Evening 1hr release with increased baseflow	7	2.8	36.8	5pm		1
Evening 2hr release with increased baseflow	7	2.8	36.8	5pm		2
Morning 1hr release with increased baseflow	7	2.8	36.8	7am		1
Morning 2hr release with increased baseflow	7	2.8	36.8	7am		2
Evening ramped release	7	1.4	36.8	5pm	3	1
Morning ramped release	7	1.4	36.8	7am	3	1

Table 2. The RQUAL model temperature predictions for existing conditions and alternative scenarios (averaged from 2.2 to 24.3 rkm from May 1 to September 31, 2000) shown as percent time 12-19°C temperatures occur, percent time the maximum hourly temperature change (MHTC) exceeds 2°C, the average daily MHTC in degrees (°C), and the percent time 21°C is exceeded. Superscripts rank the scenarios based on their ability to cause improvement over the existing conditions (e.g. 1 representing the greatest improvement).

Scenario	% time 12-19°C	% time MHTC >2°C	daily MHTC (°C)	% time >21°C
Existing Conditions	59.8	4.0	4.4	1.3
5 day 2hr release	57.3	3.2 [7]	4.2 [11]	2.2
5 day 5hr release	46.9	3.1 [6]	4.8	1.9
Steady baseflow	63.9 [5]	0.0 [1]	1.0 [2]	5.3
Increased steady baseflow	39.6	0.0 [1]	0.9 [1]	0.0 [1]
Run of river	60.0 [8]	0.0 [1]	1.0 [2]	3.8
Evening 1hr release	63.1 [6]	3.6 [9]	4.0 [10]	1.6
Evening 2hr release	54.3	4.4	5.4	1.0 [5]
Morning 1hr release	71.6 [1]	2.2 [3]	2.5 [5]	0.4 [3]
Morning 2hr release	69.9 [2]	3.3 [8]	3.7 [8]	0.1 [2]
Evening 1hr release with increased baseflow	57.6	2.9 [5]	3.0 [6]	0.1 [2]
Evening 2hr release with increased baseflow	48.1	3.9 [10]	4.4	0.0 [1]
Morning 1hr release with increased baseflow	66.1 [4]	1.2 [2]	1.6 [3]	0.0 [1]
Morning 2hr release with increased baseflow	62.9 [7]	2.3 [4]	2.3 [4]	0.0 [1]
Evening ramped release	57.6	4.5	3.9 [9]	0.9 [4]
Morning ramped release	68.5 [3]	3.1 [6]	3.6 [7]	0.1 [2]

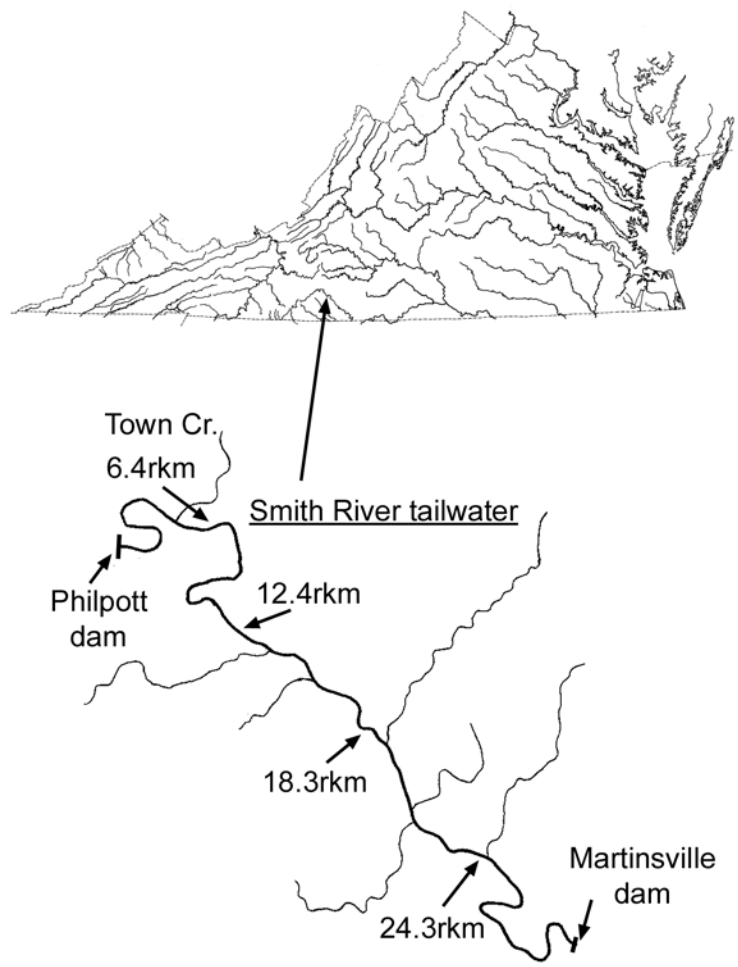

Figure 1. Location of the Smith River tailwater in southwestern Virginia and selected river kilometer (rkm) locations where temperatures were modeled.

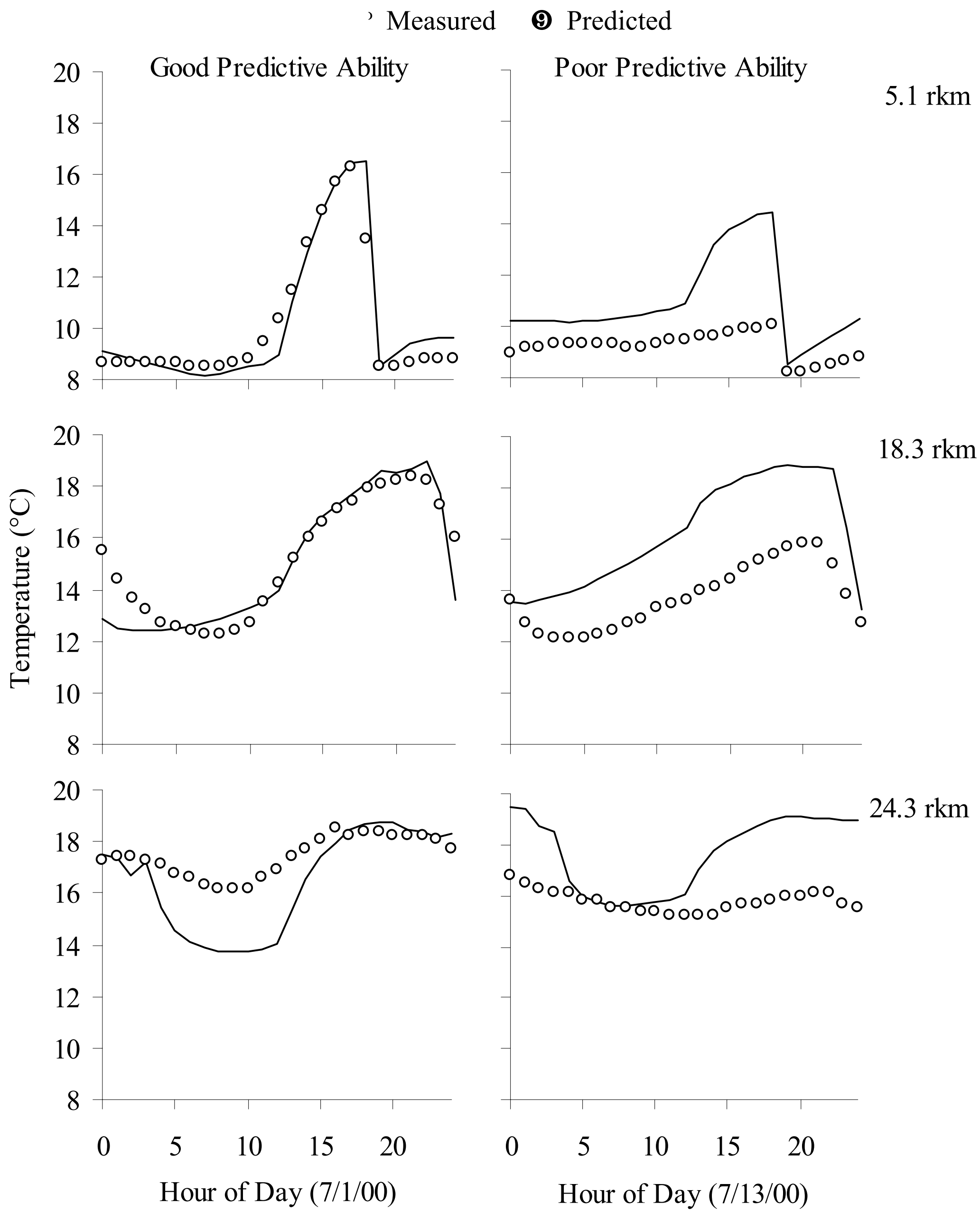

Figure 2. Examples of good (July 1, 2000) and poor (July 13, 2000) predictive ability over a 24-hour period. Graphs display hourly RQUAL predicted temperatures and data logger measured temperatures at 5.1, 18.3, and 24.3 rkm below Philpott dam.

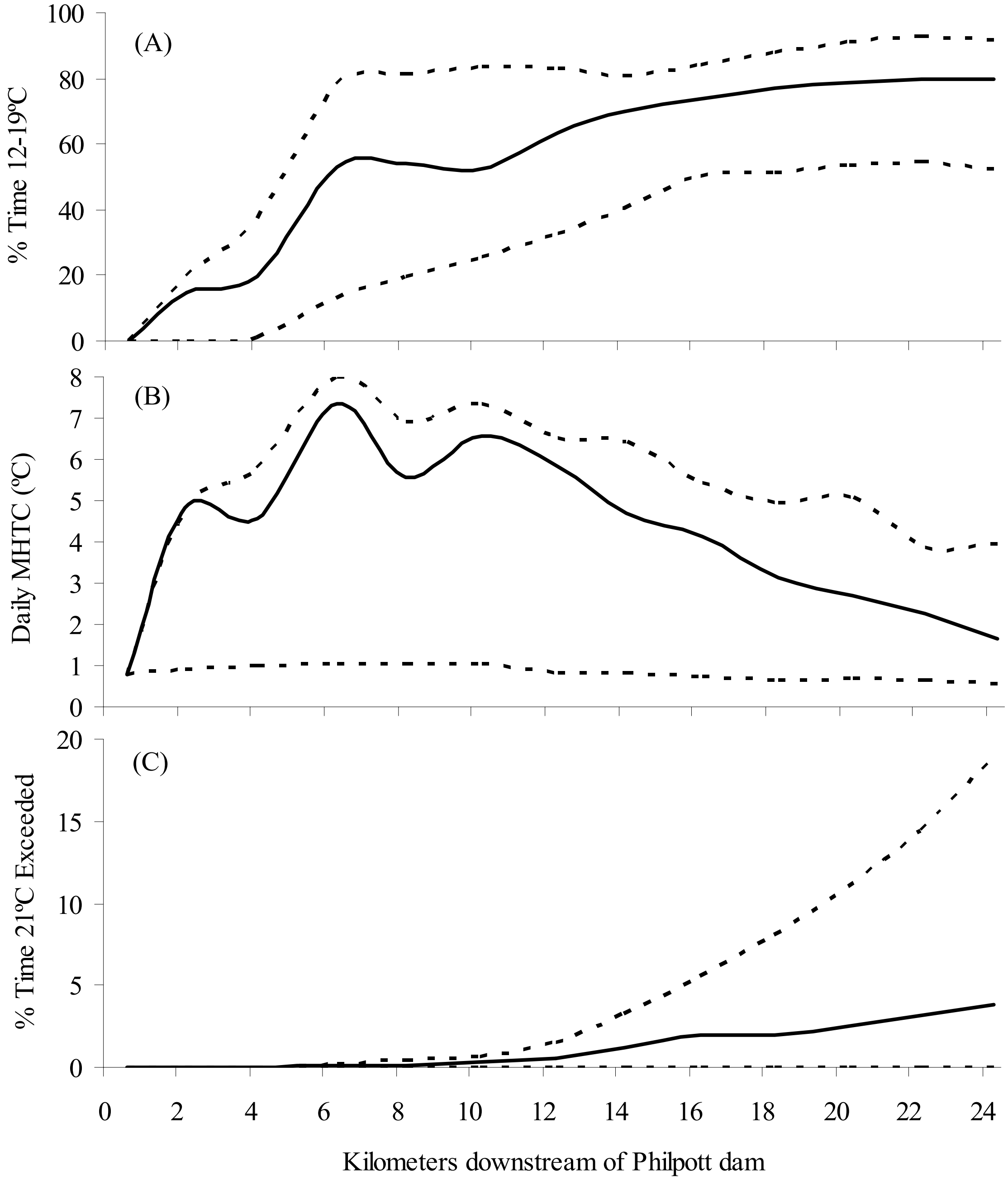

Figure 3. (A) Percent time (of May through September 2000) that water temperature is within 12 to 19ºC (i.e., optimal growth range for brown trout), (B) average daily maximum hourly temperature change (MHTC), and (C) percent time that 21ºC is exceeded under existing conditions (solid line) and the range over which alternative flow scenarios fell (dotted lines).

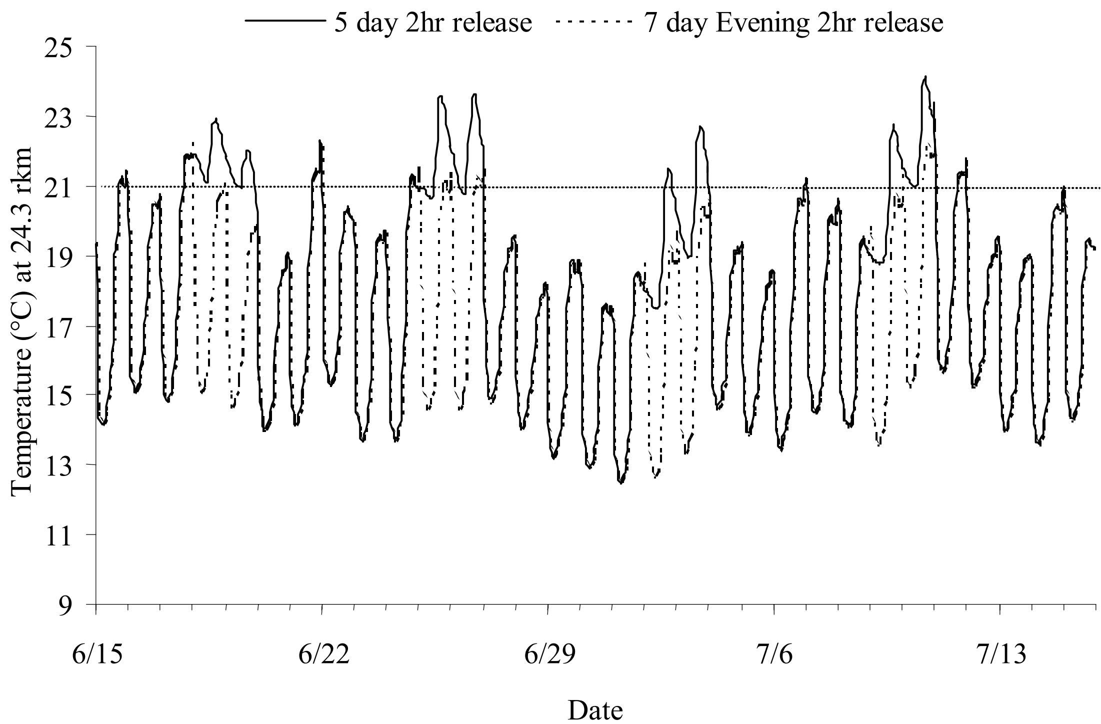

Figure 4. Hourly stream temperature at 24.3 rkm below Philpott dam from June 15 – July 15, 2000 under a 5 versus 7-day/week generation scenario. Horizontal line indicates the Virginia Department of Environmental Quality's 21°C maximum temperature standard for stockable trout waters

Longitudinal Patterns of Community Structure for Stream Fishes in a Virginia Tailwater

Anne K. Hunter [1] and C. Andrew Dolloff [2]

[1] 360 Albemarle Ave SW Apt. 4, Roanoke, Virginia, 24016, 540.985.3795, ahollowa@vt.edu

[2] Department of Fisheries and Wildlife Sciences, Virginia Polytechnic University, 100 Cheatham Hall, Blacksburg, Virginia 24061, 540.231.4864, adoll@vt.edu

ABSTRACT

Community structure of a diverse warmwater fish assemblage was examined in a cool tailwater to discern patterns of abundance, diversity, and distribution in relation to longitudinal and environmental gradients below the dam. We evaluated data across 3 seasons and 3 years during which the peaking flows and temperatures varied. Analyses determined that abundance and diversity did not change significantly between time periods (Kruskal Wallis $p>0.05$). Patterns of abundance and diversity increased with distance from the dam and peaked at tributary junctions. Fish composition was persistent during the study despite changing environmental conditions and faunal similarity increased with increasing distance from the dam. Longitudinal patterns of fish reflected a response to a gradient of increasing temperature and attenuating flows. Multiple linear regression identified mean monthly temperature, temperature depressions, and tributary location as the variables which explained a high level of variability in fish abundance. The observed fish assemblage appears to exist in well-developed patterns under the constructs of high environmental variability. Yet, fish populations do not appear to be stabilized because numbers of individual species highly fluctuated during the study.

INTRODUCTION

The alteration of flow by hydroelectric dams creates disturbance outside the natural range experienced by stream fishes (Bain et al. 1988). Peaking flow regimes associated with hydroelectric facilities pose unnatural conditions through the frequency of high magnitude flows and the rate of change in flow. Few aquatic organisms are adapted to thrive in this type of environment though some fish species are more resistant to habitat variability than others such as macrohabitat generalists (Bain et al. 1988). A peaking flow environment is associated with changes in critical habitat variables during water release including changes in depth, width, velocity, temperature, and water quality (Cushman 1985). Thus, the range in physical habitat parameters is much greater in a regulated river than an unregulated river over a shorter time interval than what might occur naturally.

It is increasingly recognized that the effects of flow variability limit the distribution and abundance of riverine species (Bain et al. 1988, Marchetti and Moyle 2001). A strong correlation exists between stream flow and a river’s physicochemical characteristics such as water temperature and habitat diversity (Poff et al. 1997). Research in the distributional ecology of fishes suggests that fish assemblages form in response to the physicochemical factors of the environment (Matthews 1987). Change in the assemblage structure of stream fishes or species composition is imposed by temporal variation in stream flow (Grossman et al. 1982, Schlosser 1982, 1985). Studies show that flow variability affects use of spatial resources or patterns of microhabitat use (Grossman et al. 1998). Indeed, the quantity and timing of flow are crucial components of ecological function in river systems.

The effects of flow regulation operate as a main structuring agent for fish abundance, diversity, and distribution in tailwater environments. Thus, understanding how a fish community is structured in flow-regulated rivers has management implications for conservation of aquatic fauna. It is often impractical to reveal the underlying mechanisms behind community structure, because it requires experimental study of multiple cause-and-effect relationships. However, community patterns can be discerned along longitudinal and environmental gradients providing information about what factors most influence the fish community. Knowledge about the factors that most affect fish community characteristics can be incorporated into decisions to protect and enhance fisheries below dams.

We evaluated patterns of fish community structure in a tailwater (Smith River, Henry Co., VA) with a hydropeaking flow regime and a hypolimnetic release that dramatically depresses the temperature of the river. We examined patterns in nonsalmonid abundance, diversity, and distribution in relation to longitudinal and environmental gradients in an effort to answer the following questions:

1) What are the longitudinal patterns in fish abundance, species richness, and distribution?
2) How does fish composition change spatially or temporally?
3) How do environmental variables relate to relative abundance?

METHODS

Study Area, Discharge, and Temperature

The Smith River is a sixth order, regulated tributary of the Dan River, located in the Roanoke drainage where forestry and agriculture are the dominant land uses in the upper sections, and urban and industrial development are predominant in the downstream sections. The Smith River tailwater was created by Philpott Dam, constructed in 1952 by the U.S. Army Corps of Engineers and used for hydroelectric generation, flood control, and recreation. The dam is a peaking production facility that operates by energy demands and water availability. The hydrology of the Smith River is dominated by regulated flows 12 months a year.

The flow regime can fluctuate from 0.7 to 39.6 cms. The dam operated under different flow regimes during the sample years of 2000, 2001, and 2002 (USGS Philpott gage # 02072000). From January 2000 to May 2001, peaking flows were 39.6 cms, 7 days/wk, for 1-hour duration. From June 2001 to October 2001, peaking generation was at 19.8 cms, 5 days/wk, for a duration of 2-10 hours. The peaking flows returned to 39.6 cms from November 2001 to February 2002, 5 days/wk, for a duration of 3-4 hours. Finally, generation releases were 19.8 cms for a 1-hour duration from March 2002 to the end of the study in October 2002.

Krause (2002) continuously monitored temperature in the tailwater using data loggers and modeled temperatures using the RQUAL (Tennessee Valley Authority) stream temperature prediction model. We used this information to evaluate the relationship between fish abundance and temperature. Mean monthly temperature directly below the dam is less than 10^{o}C and increases with distance from the dam. Maximum temperatures occur at the furthest downstream sites in the study area during spring, summer, and fall because of warmer, seasonal air temperatures. Maximum mean monthly temperature ranges from 12 to 18^{o}C. Daily maximum hourly temperature depression ranges from 0.2 to 8.4^{o}C over all seasons with the greatest flux occurring in the summer. Temperature flux depends on how greatly the ambient air temperature has warmed the water before the coldwater release.

Fish Sampling Procedures

We electrofished 12 sites located directly below Philpott Dam to 24 km downstream during the spring, summer, and fall (Figure 1). Paired multiple anode, pulsed DC barge electrofishers were used to capture fish. Single-pass depletion sampling was performed in the spring and fall while three-pass depletion sampling was performed during the summer. All nonsalmonid fish were identified to species, counted, and released. During June of 2000, 2001, and 2002, three-pass depletion electrofishing was conducted in 100-m sections enclosed by block nets at each location. In April of 2001, 2002 and October of 2000, 2001, 2002 single-pass depletion electrofishing was performed in 100-400 m reaches without block nets.

Data Analyses

We calculated the relative abundance of fish per 100 m for each site in each sampling period. For all June samples, first-pass numbers of fish were used in analyses. To qualitatively discern the longitudinal pattern below the dam to 24 km downstream, relative abundance and species richness were plotted for each site across sampling periods. To determine if a significant difference in relative abundance or species richness was present among the eight sampling periods, we used a Kruskal-Wallis test (Zar 1996). To classify population variability for relative abundance and species richness, a coefficient of variation (CV) was calculated at each site across sampling periods and is reported as a percentage. The arbitrary classification scheme proposed by Freeman et al. (1988) using CV values was followed: (1) CV ❣ 25% = stable population; (2) 26% < CV ❣ 50% = moderately stable population; (3) 51% < CV ❣ 75% = moderately fluctuating population; (4) ⑧ 76% = fluctuating population.

Similarity of fish assemblages was estimated using Morisita's Index (Morisita 1959). The index was used to compare consistency of fish composition between time periods at a specific site as well as across sites in one time period. An index value was calculated for each site across successive surveys (i.e. June 2000 to October 2000) and like seasons (i.e. June 2000 to June 2001). The original measure of Morisita's index (I_m) was used because it is found to be independent of sample size and diversity (Wolda 1981). Smith and Zaret (1982) measured bias of such indices in terms of sample size, diversity, and evenness and found that the original measure of Morisita's index gives the most accurate results. Values calculated from Morisita's Index range from zero, suggesting no assemblage similarity, to approximately one, suggesting identical assemblages.

The Spearman rank correlation coefficient (r_s) was used to compare significant changes in fish assemblage structure across space and time (Siegel 1956). Relative abundance of all species combined was used to compare assemblages across sites between successive samples (i.e. June 2000 and October 2000) and like seasons (i.e. June 2000 and June 2001). Relative abundance of each of the 13 most common or numerically dominant species was used to compare assemblages between successive samples and like seasons.

Multiple comparisons were performed using Kendall's coefficient of concordance (W), after correcting for ties (Siegel 1956). Multiple comparisons across all sampling periods for each of nine sites were made using the relative abundance of the 13 most common species. The first three sites were omitted from the concordance analysis because of the number of zeros present in the data. Significance of r_s and W was tested by using the large-sample method and χ^2 values respectively, after Siegel (1956). Because rank correlation is susceptible to Type I error (Grossman et al. 1982), or rejection of a null hypothesis that is true, a conservative critical value was set at $p = 0.01$ following the approach of Schlosser (1987).

We used information from Jenkins and Burkhead (1993) to classify each of the 13 most common species as fluvial specialists or habitat generalists. A fluvial specialist (FS) was considered to be the type of fish that is obligate to a river, sensitive to stress, and a microhabitat specialist. A habitat generalist (G) was considered to be a fish with broad habitat requirements or a high stress tolerance. These definitions were derived from Bain and Boltz (1989) and follow the theoretical framework of Kinsolving and Bain (1993) and Travnichek and Maceina (1994). Because fluvial specialists are sensitive to changes in flow, measures of their relative abundance are practical for assessing the effects of flow on community structure (Travnichek et al. 1995). Using relative abundance (fish per 100 m) of each species by site, we averaged the number of fluvial specialists and habitat generalists by site within each season and plotted the longitudinal patterns with distance from the dam.

Multiple linear regression was used to discern the relationship between nonsalmonid abundance and other biotic and abiotic variables in the Smith River. Relative abundance of all nonsalmonid fish for each of the 12 sites was estimated for time periods June 2000, October 2000, and April 2001 and served as the dependent variable for the regression model. Data for brown trout abundance, density of macroinvertebrates (Newcomb et al. 2001), chlorophyll *a* content, temperature (Krause 2002), and substrate composition (Orth et al. 2001) were input into the model as the independent, predictor variables. Each independent variable had site specific values coinciding with the 12 fish sampling sites. Specific variables used in the full model included: brown trout relative abundance (number of fish per 100 m), mean monthly temperature, maximum hourly temperature flux, tributary junction (binary), chlorophyll *a* in mg/m^2, mean number of macroinvertebrates per 0.1 m, percent composition of sand and silt (< 2 mm), percent composition of aquatic macrophytes.

A correlation matrix of all the variables in the full model was used to reveal a collinearity problem if one of the pairwise correlations exceeded 0.9 or several exceeded 0.7. Collinearity was also assessed by running a variance inflation factor (VIF) test in SAS for the full model. If the calculated VIF is 10, then the variable is most likely collinear with another variable. No variables were determined to be collinear. A stepwise regression procedure was used to obtain the final model.

RESULTS

Longitudinal Patterns

During this study, a total of 14,245 nonsalmonid fish were caught in the Smith River tailwater, representing 5 families and 34 species (Table 1). The longitudinal distribution of fishes exhibited much spatial variation and minimal temporal variation in terms of abundance and species richness. A general trend was present of increasing abundance and numbers of species as distance increased from the dam with marked increases or peaks in relative abundance and species richness occurring at four tributary junctions in the mainstem (Figure 2). Longitudinal patterns of relative abundance and species richness occurred consistently over time across the 12 sampling sites. Neither relative abundance nor number of species changed significantly between time periods (Kruskal Wallis, $p > 0.05$)**.** Though relative abundance was not statistically different between time periods, the numbers of fish caught in October 2002 were markedly higher than all other sampling periods (Table 2). During October 2002, abundance was higher for 10 of the 13 numerically dominant species (Table 3). Higher numbers of fish caught during October 2002 coincided with a discharge schedule that had lower magnitude and duration releases compared to time periods in 2000 and 2001.

Across sampling periods, we found relative abundance to be moderately fluctuating to fluctuating for sites 1-4 and 8-12 (Table 2). However, sites 5, 6, and 7 were moderately stable. CV values for relative abundance of the 13 most common species (Table 3) were classified as moderately fluctuating to fluctuating for all species except *N. leptocephalus*, a moderately stable species. The number of species per sampling period ranged from a high of 29 in June 2001 and October 2002 to a low of 24 in October 2000. The highest number of species occurred in site 11 with 26 species during October 2001, and the lowest with 0 species in site 2 during April 2001. The greatest variation in species richness occurred at site 2, followed by sites 1,3, and 5 with CVs at 76, 47, 42, and 43 percent, respectively. Variability in species richness was moderately stable to stable for all sites across sampling periods with the exception of site 2, which highly fluctuated.

Morisita's Index of Similarity

The fish assemblages of successive surveys at each site ranged from no similarity to almost identical assemblages with I_m values ranging from 0 to >1.00 (Table 4). However, similarities between sampling periods did not differ significantly across time (single-factor ANOVA, $p = 0.99$). Thus, no interval occurred between sampling periods where there was a complete change in the fish assemblage, despite the change in magnitude and duration of water releases during the interim of the study. The most variable fish assemblages were found within sites 1, 2, and 3, nearest the dam. These sites had I_m values ranging from 0 to >1.00 across sampling periods. I_m values of 0 resulted when no fish were caught, none of the same species were caught, or no more than 1 individual of each species was caught. I_m values > 1 resulted from very low sample sizes in each sample. We omitted comparisons from analyses with I_m values over one.

Fish assemblages were significantly different among sites (single-factor ANOVA, $p < 0.001$). The lowest similarity in ichthyofauna was found near the dam in contrast to high similarity farther away from the dam, producing a longitudinal gradient of increased consistency in composition downstream of the dam with the exception of site 11. Though site 11 is one of the furthest downstream, it has a low mean I_m value of 0.43. In the June 2002 to October 2002 comparison, site 9 is unique in that it has an I_m value of 0.17 or very low similarity between sampling periods. The most obvious difference between the two sampling periods was that October had 9 more species present and had far greater numbers of individuals present than the June sample (i.e. 1347 *N. hudsonius* in October 2002 vs. 5 *N. hudsonius* in June 2002).

The comparison of fish assemblages at each site across like seasons provided information on the annual variability of fish assemblages (Table 5). Across all comparisons, 61% of the I_m values were > 0.70 such that more than half of all sites had high annual similarity. Those comparisons that had low I_m values were within the first 3 sites below the dam. Additionally, several comparisons across site 9 and 11 had low I_m values including the lowest at 0.14 for the site 9 comparison of October 2000 and October 2002. Annual variation was highest between the June 2000 and June 2001 sampling periods with low I_m values for 5 sites.

Spearman Rank Correlation Coefficient

The relative abundance of all nonsalmonid fish during successive surveys and like seasons showed significant associations in site ranks across time (Table 6). The significance of site ranks over all comparisons illustrates consistency of a longitudinal pattern in fish abundance. Using relative abundance to compare species ranks for the 13 most common species demonstrated significant correlations between successive samples and like seasons except for the comparisons between June 2000 and October 2000, June 2000 and June 2001, and June 2000 and June 2002 (Table 6). It is likely that this analyses was driven by the absence of *N. hudsonius* in the June 2000 sample, which differs greatly from all other sampling periods. The comparison between relative abundance of specific species demonstrates that the fish assemblages were not always consistent across time on a scale of all sites combined.

Kendall's Coefficient of Concordance

Within individual sites, multiple comparisons of species ranks were made across all sampling periods, showing significant correlations of fish assemblage over time (Table 7). The previous species rank tests between successive surveys and like seasons masked the spatial structure of the data, yielding inconsistency of fish assemblages among three comparisons. The multiple comparisons analysis includes both the spatial and temporal elements of the data and indicates no overall change in fish composition within a site over all sampling periods. Though the concordance analysis was not performed for the first three sites, the same species were often present in low numbers. Thus, there exists a consistent grouping of assemblages on a longitudinal basis over time.

Fluvial Specialist and Macrohabitat Generalists

Of the 13 most common species, 6 were classified as fluvial specialists and 7 as habitat generalists (Table 3). Averaging relative abundance (number of fish per 100 m) of each macrohabitat class within each season, April and October were evenly split with 50% of the fish being fluvial specialists and 50% being habitat generalists. The June sample mean indicated 69% of the fish were fluvial specialists while 31% were habitat generalists. The raw data indicates that June 2002 had the highest number of fluvial specialists at 3 sites compared to all other time periods. October 2002 had the highest number of habitat generalists at 5 sites compared to all other time periods. By plotting fluvial specialists and habitat generalists separately, higher numbers of fluvial specialists were seen at tributary junctions, and increasing numbers of both classes were seen with increasing distance from the dam (Figure 3).

Regression

Specific variables used in the full regression model included: brown trout relative abundance (number of fish per 100 m), mean monthly temperature, maximum hourly temperature flux, tributary junction (binary), chlorophyll *a* in mg/m^2, mean number of macroinvertebrates per 0.1 m, percent composition of sand and silt (< 2 mm), percent composition of aquatic macrophytes. A correlation matrix was run for the eight variables (Table 8) in the regression model to reveal the linear relationship between the variables as well as the strength of that relationship. The relationships marked by the strongest correlations with nonsalmonid fish abundance were those of tributary junction, mean monthly temperature, macroinvertebrate density, and percent composition of sand and silt. Strong correlations also existed between the following variables: percent composition of sand and silt and mean monthly temperature, percent composition of sand and silt and brown trout abundance, percent composition of aquatic macrophytes and macroinvertebrate density. The correlation coefficients were as high as 0.7 for aquatic macrophytes and macroinvertebrate density as well as macroinvertebrate density and tributary junction. These variables were not dropped from the full regression model based on the correlation matrix, but were further tested. The VIF test in SAS gave values for each variable in the full model much lower than 10, so no variables were dropped from the original full regression model.

Model reduction was necessary to achieve a model with the fewest variables explaining the highest amount of variability in nonsalmonid abundance. The stepwise regression procedure produced a three-regressor model including: tributary junction, maximum hourly temperature flux, and percent composition of sand and silt. This model explained 78% of the variance in fish abundance at a significant level ($p < 0.0001$). The parameter estimates from the model were used to predict abundance using the following equation:

$$\text{Nonsalmonid abundance} = -8.77 + \text{tributary junction } (94.30) + \text{temperature flux } (-5.97) + \%\text{ sand/silt } (2.75)$$

The relationship of fish abundance and sand and silt is strong because both variables increase with distance from the dam. Thus, the level of sand and silt is likely not a better predictor for fish abundance than distance from the dam, and does not represent a strong biological relationship. Additional two-regressor models were compared with the final stepwise regression model to evaluate the ability of simpler models to predict fish abundance. Variables were chosen which correlated highly with nonsalmonid abundance, represented an inherent biological relationship with nonsalmonid abundance, and could be easily measured in the field. Table 9 shows each predictive model with its associated statistics and confidence intervals. Mean monthly temperature and tributary junction had the lowest mean-square error (MSE) compared to the other 2-regressor models and explained 62% of the variability in fish abundance.

DISCUSSION

Longitudinal Patterns in Abundance, Diversity, and Distribution

The progressive pattern of additions of species from upstream to downstream, termed "longitudinal succession," has been observed in headwater streams (Sheldon 1968). This concept of an upstream to downstream gradient change in the fish community has been hypothesized to exist below hydroelectric dams, based on the premise that disturbance diminishes as flow fluctuation attenuates downstream (Bain and Boltz 1989). In this study, we found results consistent with this hypothesis in that non-salmonid fish abundance and species richness increased with increasing distance from the dam. The most upstream fish community was greatly reduced compared to the most downstream fish community. However, peaks in abundance and species richness were consistently found at tributary junctions, re-defining the longitudinal gradient to fluctuate in the vicinity of major tributaries. Tributaries could provide sites of refuge from peaking flows and predation, or could represent areas with less restrictive physiological features such as more favorable temperature conditions as well as areas of greater food availability. For instance, macroinvertebrate data in the Smith River shows peaks in abundance that coincide with tributary location (Newcomb et al. 2001). The synchronized nature of high and low abundances around tributary junctions for both fish and macroinvertebrate data indicates that patterns of these taxa are not random but highly structured, suggesting a major tributary effect. Further, the dominant presence of fluvial specialists at tributary junctions suggests that tributaries moderate the mainstem conditions for sensitive species. In a study on the Tallapoosa River in Alabama, Kinsolving and Bain (1993) also noted synchronized patterns of high and low abundance of several fish species around tributaries.

The most consistent peak in abundance and species richness at the most upstream tributary junction, 6.2 km below the dam, could have been driven by a tributary effect on mainstem temperature. For example, temperature in June of 2000 at this site increased to almost 17^{o}C, comparable to the most downstream site at 23 km below the dam. After generation, the temperature increased at a faster rate at this tributary junction than at other sites. The water from the Town Creek tributary has a strong warming effect on the mainstem producing more suitable conditions for warmwater species. However, temperature flux is also great at this site because the drop in degrees during the coldwater release is more precipitous than at non-tributary sites.

Based on regression results, tributary location plays an influential role in fish numbers as does maximum hourly temperature flux. Longitudinal fish abundance also related significantly to gradients in mean monthly temperature. Certainly, the longitudinal distribution of fish provides evidence of response to temperature variation in the tailwater. The upper sites of the 24 km study area above Town Creek are predominantly fishless except for *L. cyanellus* and *C. commersoni*, two species that are capable of withstanding conditions near the dam. The second most numerous species in the tailwater, *N. hudsonius,* showed a distinct distributional pattern over time suggesting thermal selectivity. *Notropis* species are known to exhibit sharp range boundaries related to fixed thermal limits that regulate their distributional patterns (Matthews 1987). Out of 5,526 *N. hudsonius,* we caught only 12 individuals in the first 12.6 km below the dam, and most of these individuals were caught at the Town Creek tributary junction.

E. flabellare was the most numerous fish over all sampling periods and presumably capable to withstand the high environmental variability imposed by the flow regime. This species was ubiquitous throughout the study area and produced the highest numbers of individuals at the upper distributional range for nonsalmonid species, the Town Creek tributary. Matthews and Styron (1981) found that *E. flabellare* was very tolerant of temperature fluctuations. Hlohowskyj and Wissing (1987) suggested that *E. flabellare* be considered a "thermal generalist," a fish less sensitive to temperature change. It is likely that the "thermal generalist" nature of *E. flabellare* explains its wide distribution in the Smith River.

The environmental factors most influential to the structure of the fish community in terms of abundance proved to be mean monthly temperature, maximum hourly temperature flux, and tributary location based on regression results. Because the two temperature variables are directly related to dam operation, flow management may be a viable tool to increase nonsalmonid productivity in the tailwater. In general, warmer waters seem to benefit fish abundance and tributaries seem to favor higher numbers of individuals.

Comparison of Fish Assemblages

Morisita's index and rank correlation tests resulted in approximately equal temporal patterns in fish assemblages across all sampling periods despite environmental variability. Yet, faunal similarity was highly variable among sites, indicating a gradient of increasing consistency of composition downstream of the dam, the source of disturbance. Annual variation of ichthyofauna was greatest between June 2000 and June 2001, suggesting a possible seasonal response to the change in flow regime from the high magnitude, short duration release of 2000 to the lower magnitude, longer duration release of 2001. This variation was not observed between October 2000 and October 2001. Overall, minimal change occurred in the fish assemblages at each site through time both within and between years. If faunal "persistence" is a qualitative measure of continued presence of taxa, as considered by several authors (Connell and Sousa 1983, Ross et al. 1985, and Matthews 1986), then stream fishes of the Smith River demonstrate persistence across several years under abruptly changing, harsh conditions.

Moreover, Ross et al. (1985) found that pooling sampling stations masked variation of species ranks within individual stations. Thus, concordance of species abundance ranks was tested within each site across all sampling periods to detect both spatial variation and temporal variation. Concordant species ranks further substantiated the persistence of species and the consistency in their longitudinal distribution. It would seem that variability in the Smith River fauna is not precipitated by high environmental variability except for the much higher numbers of individuals present during the October 2002 sampling period. During this time period, abundance was higher for 10 of the 13 most common species compared to all other time periods. Higher numbers of fish seem to be a result of less flow variability during 2002 compared to study years 2000 and 2001.

Though numbers of individuals were higher in this time period, faunal similarity, species ranks, and site ranks remained high among all time periods.

Angermeier and Schlosser (1989) suggest that in a system that frequently oscillates between physically harsh and benign conditions, species composition and abundance may remain in continual flux due to immigration/emigration dynamics. Though the Smith River has great environmental oscillation, faunal persistence suggests that these dynamics are not the crux of community organization. Tributaries are the only venues for fish movement into and out of the study area with downstream and upstream immigration/emigration blocked by dams. Because the 13 most common species of the mainstem are found in the tributaries, and consistent peaks of abundance and diversity occur at tributary junctions, mainstem fish assemblages could be influenced at some level by movement of fish into and out of the tributaries.

Population and Environmental Variability

Numerous queries have been made into fish community ecology, but three key hypotheses exist as to what mechanisms act as structuring agents in a fish community. As proposed by Grossman et al. (1982), the stochastic hypothesis suggests that the relative abundance of fish is determined through the differential response of species to change in the physicochemical environment. Alternatively, the deterministic hypothesis states that biological interactions such as competition and predation regulate fish assemblages, creating highly structured communities. Finally, Strange et al. (1992) performed a 10-year study in which they found that community structure depends on how stochastic and deterministic processes combine to influence change in the fish assemblage. The mechanisms by which fish communities develop and stabilize are controversial, and particularly hard to determine due to contrasting life histories of fish species.

This research suggests that the fish assemblage of the Smith River should be placed more on the deterministic end of the deterministic-stochastic continuum because the assemblage characteristics are those of a highly structured community. The constant environmental variability of the Smith River would predictably create high variability in community structure. Yet, Moyle and Vondracek (1985) found well-defined structure in fish communities subjected to extreme flooding in a Californian stream. The longitudinal patterns of abundance, diversity, and distribution in the Smith River appear to be driven by the dynamics of flow and temperature, but the fish community persists in well-developed patterns under the constructs of this environmental variability.

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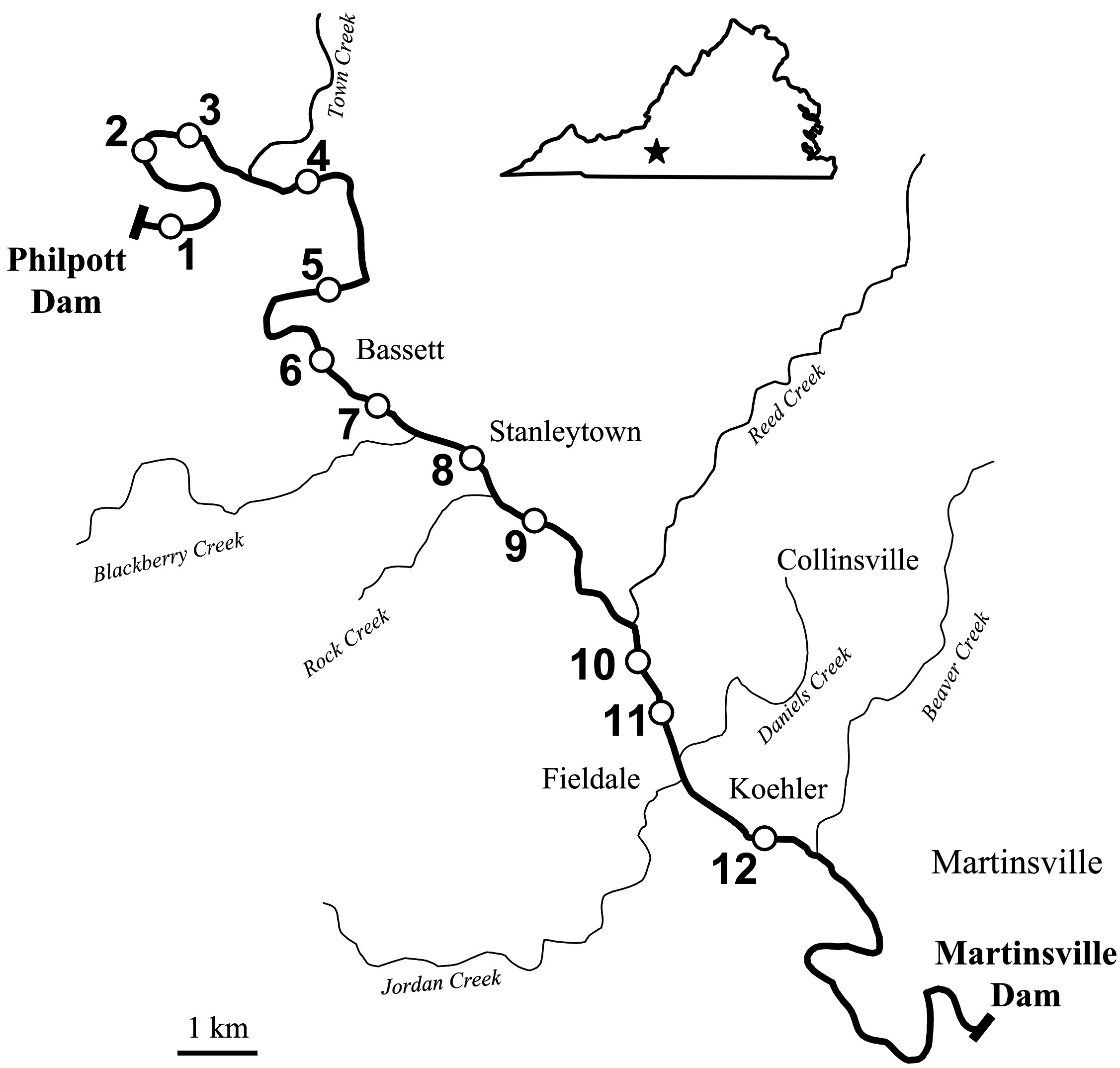

Site #	1	2	3	4	5	6	7	8	9	10	11	12
km below Philpott Dam	0.5	3.4	4.2	6.2	8.9	11.3	12.6	14.3	15.9	18.9	20.1	23.0

Figure 1. Map of the Smith River tailwater between Philpott Dam and Martinsville Dam with sampling sites numbered upstream to downstream (Krause 2002).

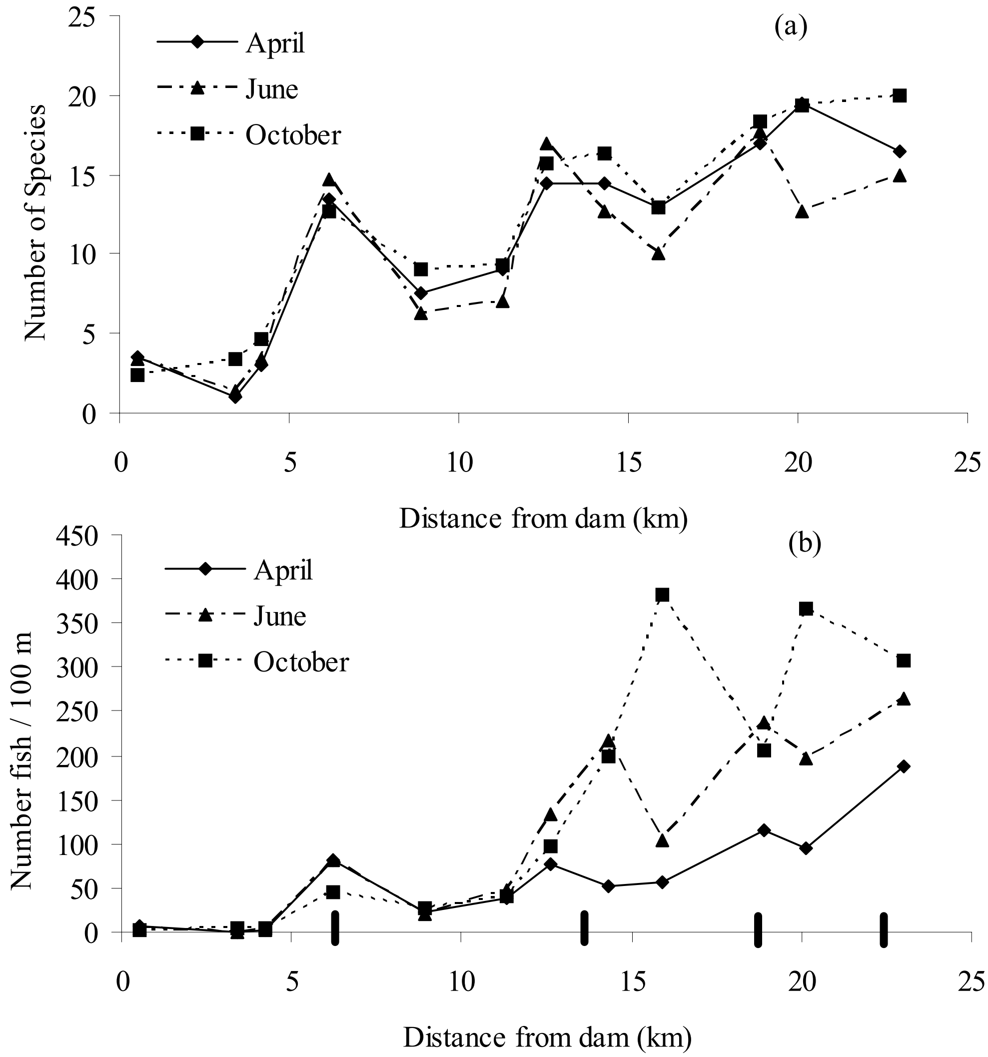

Figure 2. Species richness (a) and relative abundance (b) for all species averaged within each season as distance increases from the dam for sampling periods in 2000, 2001, and 2002. Vertical bars represent tributary junctions.

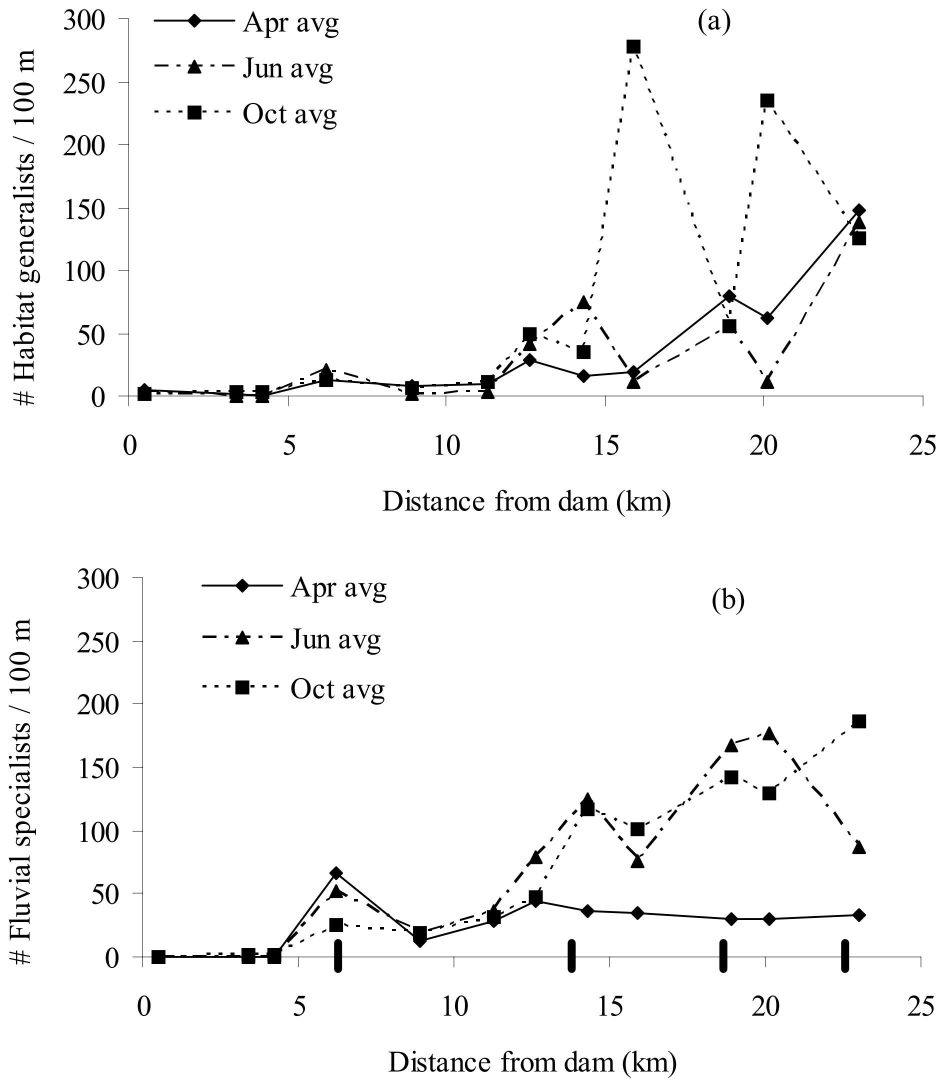

Figure 3. Number of habitat generalists (a) and fluvial specialists (b) per 100 m for the 13 most common species averaged within each season across all sampling periods. Vertical bars represent tributary junctions.

Table 1. Scientific and common name given with accompanying acronym for each species.

Species		Acronym
<u>*Catostomidae*</u>		
Catostomus commersoni	White sucker	WHS
Hypentelium nigricans	Northern hogsucker	NHS
H. roanokense	Roanoke hogsucker	RHS
Moxostoma erythrurum	Golden redhorse	GOR
M. pappilosum	V-lip redhorse	VLR
Scartomyzon cervinus	Black jumprock	BLJ
<u>*Cyprinidae*</u>		
Nocomis leptocephalus	Bluehead chub	BHC
Semotilus atromaculatus	Creek chub	CRC
Campostoma anomalum	Central stoneroller	CES
Exoglossum maxillingua	Cutlips minnow	CUT
Clinostomus funduloides	Rosyside dace	RSD
Luxilus cerasinus	Crescent shiner	CRS
Phoxinus oreas	Mountain redbelly dace	MRD
Notropis hudsonius	Spottail shiner	SPS
N. procne	Swallowtail shiner	SWS
N. chiliticus	Redlips shiner	RES
Notemigonus chrysoleucas	Golden shiner	GOS
Luxilus albeolus	White shiner	WS
Cyprinella galactura	Whitetail shiner	WTS
Lythrurus ardens	Rosefin shiner	ROS
<u>*Centrarchidae*</u>		
Micropterus salmoides	Largemouth bass	LMB
M. dolomieu	Smallmouth bass	SMB
Lepomis auritus	Redbreast sunfish	RBS
L. cyanellus	Green sunfish	GSF
L. macrochirus	Bluegill	BLG
Ambloplites cavifrons	Roanoke bass	ROB
Pomoxis nigromaculatus	Black crappie	BLC
<u>*Percidae*</u>		
Etheostoma flabellare	Fantail darter	FND
E. vitreum	Glassy darter	GLD
E. podostemone	Riverweed darter	RWD
Percina roanoka	Roanoke darter	RND
P. rex	Roanoke logperch	ROL
<u>*Ictaluridae*</u>		
Ameirus nebulosus	Brown bullhead	BRB
Noturus insignis	Margined madtom	MAM

Table 2. Abundance data for nonsalmonid fish in the Smith River, VA over eight sampling periods where RA = relative abundance or number of individuals per 100 m and CV is the coefficient of variation for relative abundance across time periods within each site.

Site	Distance from dam (km)	June 2000	October 2000	April 2001	June 2001	October 2001	April 2002	June 2002	October 2002	CV
		RA	RA	RA	RA	RA	RA	RA	RA	
1	0.5	6	2	6	1	1	6	4	2	0.66
2	3.4	0	7	0	1	3	1	1	1	1.32
3	4.2	0	9	2	3	1	1	4	3	0.99
4	6.2	128	57	117	28	58	46	90	18	0.59
5	8.9	12	25	17	24	21	27	25	37	0.31
6	11.3	64	32	44	40	31	33	36	62	0.31
7	12.6	149	59	95	129	67	60	124	168	0.40
8	14.3	197	113	74	243	72	32	209	413	0.73
9	15.9	64	92	51	69	187	60	181	868	1.41
10	18.9	193	111	126	126	107	103	391	399	0.65
11	20.1	153	113	91	171	227	99	264	761	0.94
12	23	146	277	192	387	115	182	263	528	0.53

Table 3. Abundance data summed over all sites for the 13 most common or numerically dominant nonsalmonid fish in the Smith River, VA over eight sampling periods where FS = fluvial specialist, G = habitat generalist, RA = relative abundance or number of individuals per 100 m, and CV is the coefficient of variation for relative abundance across time periods within each site. (* denotes highest abundance over time)

13 Most Common Species	FS or G	June 2000	October 2000	April 2001	June 2001	October 2001	April 2002	June 2002	October 2002	CV
		RA	RA	RA	RA	RA	RA	RA	RA	
N.leptocephalus	FS	266	224	151	230	155	63	278	309*	0.39
C. anomalum	FS	32	21	2	2	12	3	2	35*	1.02
L. cerasinus	G	21	17	5	12	32*	3	5	25	0.71
P. oreas	G	9	9	5	9	13	2	13	40*	0.91
C. funduloides	G	37	49	24	85	51	18	62	92*	0.51
N. hudsonius	G	0	48	174	296	167	216	108	1132*	1.35
C. commersoni	G	209	228	150	76	49	131	234	386*	0.58
H. nigricans	G	8	27	10	7	11	7	18	33*	0.65
M. erythrurum	G	19	21	22	0	9	14	27*	1	0.70
E.flabellare	FS	332	117	228	347	268	132	649	466*	0.56
E. podostemone	FS	17	59	9	48	81	21	47	280*	1.25
P. roanoka	FS	53	19	9	42	15	6	74	121*	0.94
N. insignis	FS	21*	1	5	7	3	2	8	6	0.95

Table 4. Morisita's index of similarity (I_m) of fish assemblages within 12 sites of the Smith River, VA across successive sampling periods where SD = standard deviation.

Site	Jun 2000 vs. Oct 2000	Oct 2000 vs. Apr 2001	Apr 2001 vs. Jun 2001	Jun 2001 vs. Oct 2001	Oct 2001 vs. Apr 2002	Apr2002 vs. Jun 2002	Jun 2002 vs. Oct 2002	Mean across time	SD
1	0.53	0.96	0.00	0.00	0.14	0.71	>1.00	0.39	0.40
2	>1.00	0.00	0.00	0.26	>1.00	0.54	0.54	0.27	0.27
3	0.20	0.19	0.21	>1.00	0.00	0.00	>1.00	0.12	0.11
4	0.68	0.59	0.85	0.91	0.90	0.93	0.91	0.83	0.13
5	0.88	0.84	0.66	0.76	0.77	0.68	0.75	0.76	0.08
6	0.82	0.71	0.97	1.00	0.86	0.87	0.78	0.86	0.10
7	0.59	0.81	0.98	0.78	0.74	0.91	0.78	0.80	0.12
8	0.72	0.66	0.72	0.87	0.75	0.88	0.95	0.79	0.11
9	0.93	0.89	0.82	0.91	0.84	0.94	0.17	0.79	0.27
10	0.74	0.37	0.68	0.73	0.62	0.61	0.56	0.62	0.13
11	0.12	0.58	0.51	0.60	0.86	0.24	0.14	0.43	0.28
12	0.64	0.71	0.96	0.86	0.58	0.83	0.76	0.76	0.13
Mean across sites	0.62	0.61	0.61	0.70	0.64	0.68	0.64		
SD	0.26	0.29	0.36	0.31	0.30	0.30	0.28		

Single-Factor ANOVA across sampling periods:
$F = 0.17$ $p = 0.99$

Single-Factor ANOVA across sites:
$F = 9.23$ $*p < 0.001$

Table 5. Morisita's index of similarity (I_m) of fish assemblages within 12 sites of the Smith River, VA across like seasons where SD = standard deviation.

Site	Jun 2000 vs. Jun 2001	Jun 2001 vs. Jun 2002	Jun 2000 vs. Jun 2002	Apr 2001 vs. Apr 2002	Oct 2000 vs. Oct 2001	Oct 2001 vs. Oct 2002	Oct 2000 vs. Oct 2002	Mean across time	SD
1	>1.00	>1.00	>1.00	1.01	0.00	0.42	0.53	0.49	0.41
2	0.00	0.57	0.00	0.00	0.11	0.33	0.00	0.14	0.22
3	0.00	>1.00	>1.00	0.00	0.80	>1.00	0.53	0.33	0.40
4	0.93	0.79	0.91	0.97	0.71	0.85	0.63	0.83	0.12
5	1.00	0.99	0.98	1.03	0.94	0.93	0.83	0.96	0.07
6	0.96	0.99	1.00	0.88	0.71	0.76	0.89	0.88	0.11
7	0.90	0.84	0.68	0.93	0.77	0.82	0.59	0.79	0.12
8	0.78	0.72	0.95	0.88	0.74	0.86	0.54	0.78	0.13
9	0.44	0.95	0.43	0.95	0.63	0.55	0.14	0.58	0.29
10	>1.00	0.98	0.91	0.99	0.58	0.75	0.67	0.81	0.17
11	0.32	0.85	0.96	0.67	0.32	0.83	0.55	0.64	0.26
12	0.35	0.90	0.65	0.94	0.79	0.83	0.91	0.77	0.21
Mean across sites	0.57	0.86	0.75	0.77	0.59	0.72	0.57		
SD	0.39	0.14	0.32	0.37	0.29	0.20	0.27		

Table 6. Spearman rank correlation tests between species ranks for the 13 most common species and site ranks (* $p < 0.05$).

Successive samples compared	Site Ranks (r_s)	t	Successive samples compared	Species Ranks (r_s)	t
Jun 2000 & Oct 2000	0.86*	5.29	Jun 2000 & Oct 2000	0.44	1.62
Oct 2000 & Apr 2001	0.82*	4.47	Oct 2000 & Apr 2001	0.77*	3.99
Apr 2001 & Jun 2001	0.79*	4.10	Apr 2001 & Jun 2001	0.76*	3.92
Jun 2001 & Oct 2001	0.84*	4.98	Jun 2001 & Oct 2001	0.95*	9.72
Oct 2001 & Apr 2002	0.87*	5.69	Oct 2001 & Apr 2002	0.80*	4.37
Apr 2002 & Jun 2002	0.90*	6.61	Apr 2002 & Jun 2002	0.85*	5.45
Jun 2002 & Oct 2002	0.87*	5.46	Jun 2002 & Oct 2002	0.81*	4.63
Like seasons compared			Like seasons compared		
Jun 2000 & Jun 2001	0.87*	5.60	Jun 2000 & Jun 2001	0.40	1.43
Jun 2000 & Jun 2002	0.90*	6.61	Jun 2000 & Jun 2002	0.49	1.88
Jun 2001 & Jun 2002	0.88*	5.94	Jun 2001 & Jun 2002	0.83*	4.85
Apr 2001 & Apr 2002	0.91*	7.06	Apr 2001 & Apr 2002	0.92*	7.58
Oct 2000 & Oct 2001	0.91*	7.06	Oct 2000 & Oct 2001	0.71*	3.35
Oct 2000 & Oct 2002	0.89*	6.17	Oct 2000 & Oct 2002	0.72*	3.45

Table 7. Kendall's coefficient of concordance (W) for the 13 most common species across all sampling periods (** $p < 0.001$).

Site	Distance from dam (km)	W	χ^2
4	6.2	0.49**	47.46
5	8.9	0.44**	42.11
6	11.3	0.69**	66.43
7	12.6	0.44**	42.00
8	14.3	0.71**	67.85
9	15.9	0.62**	59.61
10	18.9	0.61**	58.51
11	20.1	0.70**	66.87
12	23	0.67**	64.53

Table 8. Pearson r correlations and p-values for 9 variables. Data for fish and temperature represent data from June 2000, October 2000, and April 2001.

	Non-salmonid abundance	Tributary Junction	Temperature Flux	Mean Temperature	Trout Abundance	Invertebrate Abundance	Chlorophyll *a*	% Sand/ Silt	% Aquatic Vegetation
Nonsalmonid abundance	-	0.67 < 0.0001	0.01 0.966	0.63 < 0.0001	-0.48 0.003	0.60 < 0.0001	-0.30 0.08	0.65 < 0.0001	0.48 0.0003
Tributary Junction		-	0.24 0.158	0.31 0.068	-0.15 0.367	0.74 < 0.0001	-0.12 0.477	0.13 0.463	0.54 0.0006
Temperature Flux			-	-0.42 0.012	-0.04 0.824	- 0.21 0.21	0.27 0.11	0.07 0.705	-0.03 0.865
Mean Temperature				-	-0.42 0.010	0.27 0.106	-0.36 0.031	0.58 0.0002	0.23 0.183
Trout Abundance					-	-0.27 0.118	-0.17 0.322	-0.56 0.0004	-0.28 0.096
Invertebrate Abundance						-	-0.10 0.546	0.21 0.212	0.71 < 0.0001
Chlorophyll *a*							-	-0.35 0.034	-0.20 0.231
% Sand/ Silt								-	0.12 0.478
% Aquatic Vegetation									-

Table 9. Multiple linear regression models with respective statistics. 2-regressor model results are given to compare with the final stepwise 3-regressor model.

Multiple linear regression predictive models for nonsalmonid abundance	N	df	R^2	MSE	Regressor	Confidence interval for each regressor (±)
Final Stepwise Model						
(1) -8.77 + trib (94.30) + flux (-5.97) + ssilt (2.75)	36	32	0.78	1057.09	Tributary junction	94.29
					Temperature flux	5.98
					% Sand/silt	2.77
2-Regressor Models						
(2) -96.12 + mtemp (12.74) + trib (77.61)	36	33	0.62	1843.15	Mean Monthly Temp	12.75
					Tributary junction	77.68
(3) 93.38 + bnt (-0.68) + trib (86.98)	36	33	0.58	2016.96	Brown trout abundance	0.69
					Tributary junction	86.94
(4) -139.58 + mtemp (20.95) + flux (10.48)	36	33	0.45	2712.48	Mean Monthly Temp	16.50
					Temperature flux	37.51
(5) 30.38 + trib (104.21) + flux (-5.28)	36	33	0.45	2721.22	Tributary junction	104.12
					Temperature flux	5.28

MEASURING WATER-VELOCITY PROFILES WITH ACOUSTIC DOPPLER TECHNOLOGY IN A VIRGINIA TAILWATER

Colin Krause*
Dept of Fisheries and Wildlife Sciences
Virginia Tech, 125 Cheatham Hall
Blacksburg, VA 24061-0321
cokrause@vt.edu

Yi Shen
Dept of Civil and Environmental Engineering
Virginia Tech, Patton Hall
Blacksburg, VA 24061
yishen@vt.edu

KEYWORDS: acoustic Doppler profiler, RiverCat, water velocity, discharge, hydropeaking

ABSTRACT

The two-dimensional flow model RMA-2V was developed for two sections of a hydropeaking tailwater in Virginia as part of a Brown trout fisheries research study. We required known water velocity data at multiple flows to calibrate and validate the model. To measure these velocities at flows too high and swift to wade with a flow meter, we utilized a wireless acoustic Doppler profiler. The floating profiler towed across the channel with a cableway system allowed peak flow water velocities to be safely measured by operators on shore.

INTRODUCTION

The trout fishery in the Smith River tailwater below Philpott dam in southwestern Virginia supports a naturally reproducing brown trout population as well as stocked rainbow trout. Despite rapidly varying flows (1 to 40 cms in 15 min) from the Philpott dam hydropeaking electric facility, the tailwater supports trout due to a hypolimnetic release (i.e., cold-water). In the 1970's brown trout in the Smith River reached trophy sizes up to 8.2 kg with fish of 4.5 kg not being uncommon (Mohn, 2001). However, over the last twenty years the fishery has rarely produced >4.5 kg trout and in the last ten years there are very few catches of trophy size trout (635 mm, 2.3 kg) (Anderson et al., 2003). This is of concern to the Virginia Department of Game and Inland Fisheries (VDGIF) who manage the river from 5.3 to 10 river kilometers (rkm) below the dam for trophy brown trout. Therefore, VDGIF and the Virginia Tech Department of Fisheries and Wildlife Sciences began a 5-year study in 1999 to develop a scientific foundation for supporting alternative flow scenarios that could improve the fishery (Orth et al., 2002).

A component of this study developed a two-dimensional hydraulic model to measure the effects of varying flow on shear stress, mobilization of streambed gravels, and discharge relationship to the amount of spawning-nest scouring or brown trout fry displacement. This information coupled with flow records can predict catastrophic year-class failures and flow ranges that provide acceptable reproduction. The 2-D hydraulic model (RMA-2V) developed for sections of the Smith River required water velocity data at multiple discharges for calibration and validation purposes. Water velocity is typically measured with flow meters (e.g. models such as Marsh McBirny and Price AA) when flows are shallow (<1m) and slow (<1 m/s) enough for the operator wade in the river. However, in cases such as the Smith River during peak flow, the river is too deep and swift. The SonTek® RiverCat®, a wireless Acoustic Doppler Profiler (ADP®) with integrated Global Positioning System (GPS) mounted on floating pontoons, enables data collection in unwadeable situations when towed across the river on a portable cableway. Additionally, the RiverCat enables discharge and velocity data collection where there is no bridge or when the river is too small and shallow for a motorized boat.

The ADP measures water velocity based on a physical principle called the Doppler effect. When a source of sound is moving relative to the receiver, the frequency of the sound at the receiver is shifted from the transmit frequency; this is the Doppler effect. The ADP emits sound waves and tracks the reflection of the sound waves off particles in the water. The change in frequency of the reflected sound is proportional to the velocity of the water. This technology enables water velocity measurement at multiple profiles across the river channel and at multiple cells from near the water surface to the channel bottom.

METHODS

The RMA-2V hydraulic model was developed for a 165 m section at 4.2 rkm and a 200 m section at 12.6 rkm below Philpott dam which corresponded with trout spawning areas. Water velocity data for model calibration and validation was measured at baseflow (~1.8 cms), moderate flow (~19 cms), and peak flow (~40 cms). Mean column depth velocity at baseflow was measured while wading in the river with a Marsh McBirny® 2000 model flow meter. With the flow meter we measured velocity at 186 locations over 9 transects at the 4.2 rkm site and 194 locations over 10 transects at the 12.6 rkm site. During moderate and peak flows, velocity was measured with the RiverCat. At the 4.2 rkm and 12.6 rkm site, 7 transects (3 at moderate and 4 at peak flow) and 11 transects (4 at peak and 7 at moderate) were measured, respectively. At each transect location the RiverCat was pulled back and forth across the channel multiple times to provide between 2 to 10 replicates for averaging. The RiverCat was set to record velocity in 15 cm cells every 10 seconds. The RiverCat (powered by 16 C-cell batteries) communicates wirelessly with a laptop computer and we used a Honda® EX350 generator to power the laptop and wireless transceiver.

To deploy the RiverCat across the river channel along transects perpendicular to the flow, a portable cableway was built. The cableway consisted of anchoring a CMI® micro pulley on each bank at least 1.5 m above the highwater level to either a tree or tripod. Pulleys were connected with a carabiner to a NRS® tie-down strap looped around either a tree or a Topcon® aluminum tripod secured to the ground with two tie-down straps using screw-type earth anchors. A perlon rope (7 mm diameter) was strung through both pulleys forming a circular loop over the river. The two ends of the rope were connected with a Petzl® basic ascender which allowed the rope to be pulled taught. Different lengths of tie-down straps were used to connect the RiverCat to the rope to facilitate measuring multiple transects from one cableway location. The RiverCat could then be moved back and forth across the channel by pulling the rope through the pulleys. Ideally we setup the cableway shortly before the peak flow arrived and water level rose. However, we encountered periods of 24-hr continuous peak flow release which required us to paddle across the river using a Hobie® Float Cat boat to setup the cableway.

RESULTS AND DISCUSSION

The RiverCat and cableway system enabled us to measure water velocities at high flows when wading with a flow meter is not possible. At baseflow we used a flow meter because the river depths were less than the 0.55 to 6 m range that the 3.0 MHz frequency ADP requires to function. Other SonTek ADP's operate at different frequencies and thus can measure different depth ranges (5.0 MHz for 0.3-2.5m and 1.5 MHz for 0.8-25m).

Faster water velocity occurred at higher flows and in the middle of the channel where depth was greatest (Figure 1). Friction caused by the channel bottom in shallow water near the banks lessened the velocity such that moderate and peak flow velocities were similar. Based on depth measurements averaged across the channel, the water surface elevation increased 1.2 and 1.0 m between base and peak flow at sites 4.2 rkm and 12.6 rkm, respectively. Thus there is a large fluctuation in water level as well as water velocity on a daily basis due to the hydropeaking operation. Measuring each transect multiple times and then plotting a polynomial regression through the data provides a more complete and accurate velocity profile across the channel (Figure 1).

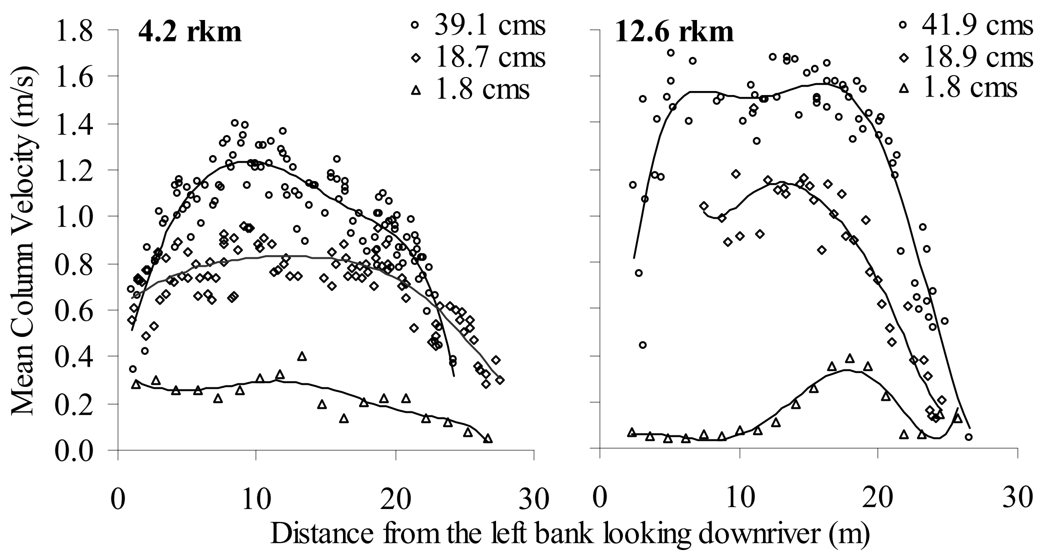

Figure 1. Mean column velocity at sites 4.2 and 12.6 rkm below Philpot dam at baseflow (1.8 cms), moderate flow (18.7 & 18.9 cms), and peakflow (39.1 & 41.9 cms). Trendlines are 6th order polynomial regressions. Data shown for 4.2 rkm at moderate flow is comprised of 2 of 4 replicates and at peak flow 5 of 10 replicates. Data shown for 12.6 rkm at moderate flow is comprised of 2 of 4 replicates and at peak flow 4 of 10 replicates. Data is from selected transects close to one another, however data at base, moderate, and peak flow is not along the exact same transect.

The velocity of the RiverCat as it travels across the channel is determined by either the integrated differential global positioning system (GPS) with <1 m error or by bottom tracking. The smaller the width of the river, the greater the relative error of the GPS position becomes. Therefore, we found bottom tracking more accurate than GPS. The GPS locations also allow the overlay of velocity data on the modeled sites in a geographic information system (GIS). Bottom tracking is only accurate when the channel bottom is stationary, thus GPS can be more accurate in cases where the high velocities cause bedload movement. Data comparison between GPS versus bottom tracking revealed bed movement occurred at the 12.6 rkm site during peak flow along some transects. The concept that bedload movement can be observed with an ADP has lead to other uses for this technology such as measurement of bed load velocity (Rennie et al., 2002). Additionally, suspended sediment concentrations can be measured with an ADP because there is a relationship between ADP signal strength and sediment size, type, and concentration (Sontek, 2003).

CONCLUSIONS

The growth and survival of fish is affected by their surrounding physical habitat and one important factor is flow and the resulting water velocity. The force created by high velocities can scour eggs from spawning nests, displace juvenile fish downriver, and restrict forage ability. With the RiverCat we were able to measure velocities occurring during hydropeaking flows as well as velocity profiles and discharge. This data enabled an accurate predictive model which will enable the prediction of alternative flows to benefit the Smith River trout fishery.

ACKNOWLEDGEMENTS

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Macroinvertebrate Forage in the Smith River Tailwater

T. J. Newcomb, *Virginia Polytechnic Institute and State University, Department of Fisheries and Wildlife Sciences, 100 Cheatham Hall, Blacksburg, VA 24061-0321*

K. M. Hanna, *Virginia Polytechnic Institute and State University, Department of Fisheries and Wildlife Sciences, 100 Cheatham Hall, Blacksburg, VA 24061-0321*

M. R. Anderson, *Virginia Polytechnic Institute and State University, Department of Fisheries and Wildlife Sciences, 100 Cheatham Hall, Blacksburg, VA 24061-0321*

Abstract: Benthic macroinvertebrates were sampled in July 2000 and April 2001 at 12 sites in the Smith River below Philpott Dam in southwestern Virginia. One riffle in each site was stratified into upstream, middle, and downstream transects and Surber samples were collected at 2 randomly-selected locations on each transect. Macroinvertebrates were identified to family and each sample was measured for wet weight. Family richness was calculated and simple linear regression was used to evaluate longitudinal trends in mean abundance and wet weight with increasing distance from the dam. We found low values of family richness near the dam but richness more than doubled by 4.2 km downstream. Mean wet weight and abundance of macroinvertebrates were higher in April than in July and Ephemerellidae proportionately dominated the samples in April. Overall, abundance of aquatic invertebrates in this tailwater was lower than expected for a stream of this size in Virginia. No strong pattern was found between distance from the dam and macroinvertebrate abundance. However, isolated peaks in abundance of macroinvertebrates at spatially discrete locations suggest that localized channel characteristics improved some areas for macroinvertebrate colonization downstream of Philpott Dam.

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Construction of a dam in a river creates 3 functionally different systems that offer unique challenges to aquatic resource managers, sometimes over a small scale. Upstream of the impoundment, the river maintains its free flowing characteristics with the exception of the transitional zones adjacent to the reservoir. The channel area inundated by the reservoir becomes a lentic environment, entirely different in both trophic status and habitat availability. Downstream of the dam, the river is either warmer as a result of an epilimnetic release or colder with a hypolimnetic release. Epilimnetic releases may result in higher levels of productivity from reservoir contributions of phytoplankton and zooplankton; however, higher temperatures may subsequently limit or restructure the downstream fish community. Hypolimnetic releases can result in water temperatures suitable for salmonids but may decrease productivity levels of the macroinvertebrate community. The combined effects of coldwater release and decreased nutrients may, in effect, abiotically "re-set" the channel to headwater conditions (Vannote et al. 1980) but with greater flows than in a headwater stream. The new coldwater thermal regime creates conditions suitable for development of coldwater fisheries and thus increased recreational opportunities. However, the pattern of flow releases combined with the "re-set" channel conditions can limit downstream productivity for fish populations and their forage base (Cummins 1979). Stream temperature is known to influence macroinvertebrate abundance and composition by influencing development rates and excluding taxa that are intolerant to minimum, maximum, or fluxing thermal conditions (Vannote and Sweeney 1980, Ward and Stanford 1982, Sweeney and Vannote 1986, Hawkins et al. 1997).

The Smith River tailwater, created by the construction of Philpott Dam in 1953, provides a highly valuable and desired coldwater fishery in southwestern Virginia (Hartwig 1998). In the 1970s and early 1980s, this fifth order tributary to the Dan River produced large trout, which included the state record brown trout (*Salmo trutta*). Today, however, in spite of special trophy regulations, few large brown trout are present in the Smith River below Philpott Dam (Orth et al. 2001). Hypotheses for the change in the brown trout fishery include growth limitations due to a lack of adequate forage, the metabolic challenges posed by a highly variable flow regime, and in some locations, a thermal regime that fluctuates greatly over a short period of time (Orth et al. 2001, Krause 2002). Philpott Dam currently operates by hydropeaking with hypolimnetic releases and flows increase from 1.1 m^3/second to 37 m^3/second in less than 30 minutes daily. Peaking flows released from Philpott Dam are likely to armor the substrate close to the dam, erode stream banks, and deposit finer sediment downstream and thus limit the macroinvertebrate community. This reduction of substrate diversity, combined with catastrophic macroinvertebrate drift during high flows (Anderson and Wallace 1984), could have a significant negative impact on the macroinvertebrate community particularly near the dam (Cushman 1985). Currently, an ongoing Smith River research project is focusing on the population dynamics and patterns of trout and nongame fish communities but information is lacking on the status of the macroinvertebrate forage base for both trout and other fish species present.

The goal of this study was to conduct a preliminary investigation of the benthic invertebrate community below Philpott Dam in the Smith River. Our specific objectives were to quantify abundance and biomass of the invertebrate community, determine benthic community composition in its relation to potential forage for trout, and identify if a longitudinal pattern of recovery exists in the invertebrate community with increasing distance from the dam.

The authors thank A. Black, M. Chan, and C. Krause for their assistance in collecting macroinvertebrate samples; the Biological Sciences Initiative and the Minority Academic Opportunities Program, both at Virginia Tech, for funding this work; the Department of Fisheries and Wildlife Sciences at Virginia Tech for providing lab-

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Methods

We sampled aquatic macroinvertebrates at 12 riffles in sites below Philpott dam that coincided with stations established by the Virginia Department of Game and Inland Fisheries for estimating brown trout abundance (Fig. 1). All sites were sampled in summer (July 2000) and spring (April 2001) to obtain gross estimates of abundance and to obtain a contrast of abundance at times when most taxa were expected to be at greatest (spring) and lowest (summer) abundances (Anderson and Wallace 1984, Cada et al. 1987). At each riffle, transects were established across the river at upstream, middle, and downstream sections of the riffle. Two sampling locations for each transect were randomly selected resulting in 6 samples per riffle at all 12 sites. A Surber sampler (0.1 m^2, 1000 μm mesh) was placed on the substrate of the sampling location and rocks were scrubbed while disturbing the benthos down to 7 cm. Samples were rinsed into labeled jars and preserved with 70% ethanol. In the laboratory, samples were rinsed with water and the sugar flotation method was used to pick macroinvertebrates from the sample. Macroinvertebrates were identified to family (Merritt and Cummins 1996) under a compound dissecting scope. Wet weight (g) was measured by draining the ethanol off from each sample and allowing it to air dry for 5 minutes before weighing on a microbalance.

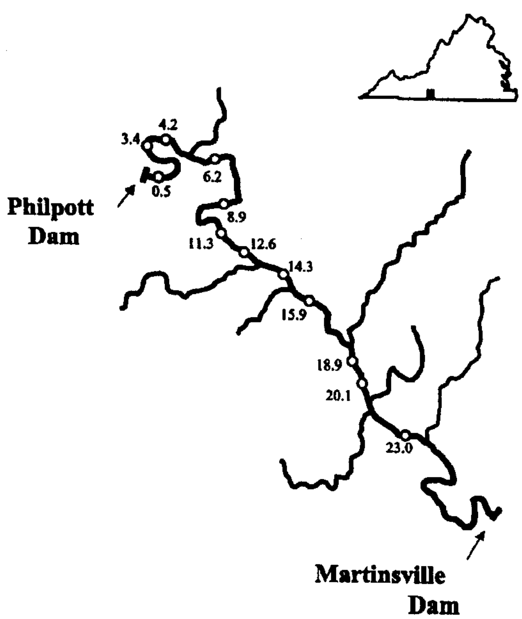

Figure 1. Location of sites (km below dam) for sampling aquatic macroinvertebrates below Philpott Dam in the Smith River, Virginia, in July 2000 and April 2001.

Data from all 6 samples were pooled to calculate Margalef's Index of Richness using family information:

$$D = (S - 1)/\ln N$$

where D = richness, S = the number of families represented, and N = the total number of individuals collected (Brower and Zar 1977). Mean abundance and standard errors for all invertebrates were calculated for each site. The total abundance of Ephemeroptera, Plecoptera, and Trichoptera (EPT) was also analyzed as an indicator of insects that are sensitive to water quality and habitat conditions (Barbour et al. 1999) and are valuable sources of forage for trout. Due to the large differences in site abundances, both count data and wet weight data were log transformed before simple linear regression analyses to examine longitudinal trends with increasing distance from the dam (Gislason 1985).

Results

Family richness more than doubled between 0.5 and 6.2 km from the dam in both July 2000 and April 2001. Beyond 6.2 km, richness values fluctuated between 5.5 and 8.0 but remained higher than those near the dam (Fig. 2). In April, Chironomidae and Ephemerellidae were common to all sites whereas in July, Ephemerellidae, Baetidae, Tipulidae, and Chironomidae were common to most sites. The April

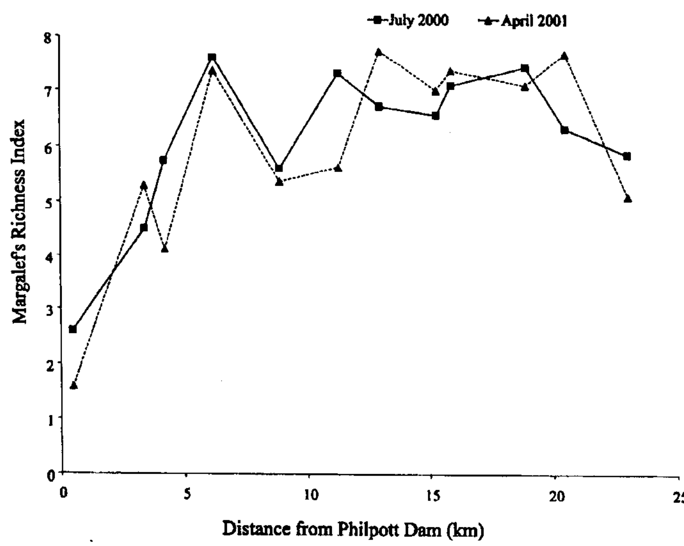

Figure 2. Margalef's Richness Index (based on family) of invertebrate taxa in the Smith River, Virginia, July 2000 and April 2001.

Table 1. Comparison of the top 3 abundant macroinvertebrate taxa between sites and years in the Smith River below Philpott Dam in Virginia (N = total number of macroinvertebrates at the site).

Site (km)	(N)	Family	Proportion of total (%)	(N)	Family	Proportion of total (%)
		July 2000			*April 2001*	
0.5	201	Chironomidae	44	334	Chironomidae	68
		Isopoda	40		Isopoda	17
		Simulidae	5		Ephemerellidae	1
3.4	165	Isopoda	24	48	Ephemerellidae	60
		Chloroperlidae	19		Isopoda	10
		Chironomidae	12		Chironomidae	4
4.2	80	Chloroperlidae	12	85	Ephemerellidae	71
		Chironomidae	12		Isopoda	12
		Simulidae	5		Baetidae	6
6.2	339	Baetidae	30	531	Ephemerellidae	72
		Isonychidae	24		Hydropsychidae	6
		Chloroperlidae	21		Chironomidae	4
8.9	206	Chloroperlidae	50	255	Ephemerellidae	54
		Baetidae	11		Heptageniidae	17
		Hydropsychidae	11		Baetidae	10
11.3	155	Chloroperlidae	26	202	Ephemerellidae	54
		Chironomidae	10		Heptageniidae	13
		Heptageniidae	7		Hydropsychidae	5
12.6	93	Baetidae	8	210	Ephemerellidae	64
		Chloroperlidae	8		Heptageniidae	8
		Chironomidae	7		Hydropsychidae	4
14.3	137	Baetidae	22	509	Ephemerellidae	85
		Chironomidae	10		Hydropsychidae	4
		Hydropsychidae	2		Baetidae	2
15.9	254	Baetidae	41	138	Ephemerellidae	65
		Limniphilidae	14		Tipulidae	9
		Chironomidae	10		Hydropsychidae	5
18.9	197	Hydropsychidae	30	240	Ephemerellidae	58
		Baetidae	13		Hydropsychidae	13
		Heptageniidae	9		Tipulidae	8
20.1	116	Baetidae	12	543	Ephemerellidae	54
		Limnephilidae	8		Hydropsychidae	17
		Hydropsychidae	4		Tipulidae	7
23.0	112	Baetidae	20	542	Ephemerellidae	51
		Limnephilidae	7		Hydropsychidae	23
		Chironomidae	4		Tipulidae	6

samples were dominated by the presence of Ephemerellidae (Table 1). Small chironomids were predominant in the benthic community closest to the dam, with isopods and small mayflies contributing to the community up to 4.2 km downstream.

As expected, mean wet weight (± 1SE) for all sites combined was greater in April 2001 (0.52 ± 0.14 g) than July 2000 (0.17 ± 0.03 g). Furthermore, in July 2000, we found no significant relationship ($r^2 = 0.003$, $P = 0.87$) between the log of wet weight of invertebrates at each sight and increasing distance from the dam (Fig. 3). Although there was a significant positive relationship ($r^2 = 0.72$, $P < 0.01$) in April 2001 with increasing distance from the dam, this relationship was largely driven by the last site in which one of the samples contained a very large tipulid. With the exception of the last site, wet weight appeared to be consistent after 6.2 km from the dam ranging between 4.7 and 9.7 g/m^2.

Mean densities of all macroinvertebrates were also greater in April 2001 (469.72 ± 80.01/m^2) than in July 2000 (288.31 ± 36.78/m^2). However, there was no relationship between invertebrate densities and distance from the dam in either July 2000 ($r^2 = 0.11$, $P = 0.30$, $N = 12$) or April 2001 ($r^2 = 0.27$, $P = 0.08$, $N = 12$) (Fig. 4). There was a significant relationship between mean number of EPT with increasing distance from the dam in April 2001($r^2 = 0.47$; $P = 0.02$, $N = 12$) but not in July 2000 ($r^2 = 0.15$, $P = 0.22$, $N = 12$) (Fig. 5). Although a strong pattern of increasing abundance is not present for macroinvertebrates with increasing distance from the dam, spatially discrete peaks of high abundance in April are present at sites 6.2, 14.3, and 23.0 km from the dam.

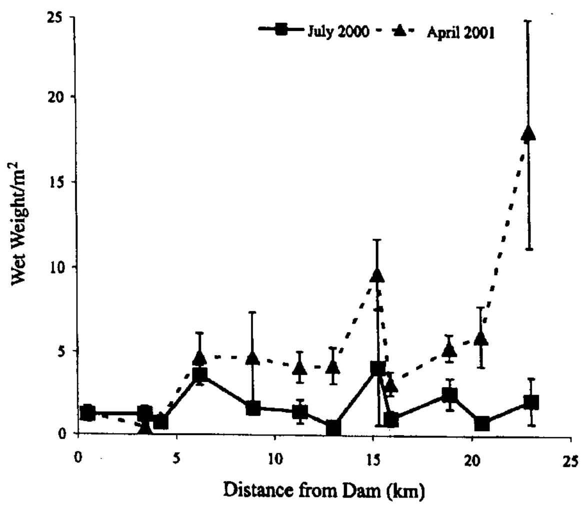

Figure 3. Biomass of aquatic macroinvertebrates (mean wet weight (g)/m^2 ± 1 SE) and distance from Philpott Dam at 12 sites in the Smith River, Virginia, in July 2000 and April 2001.

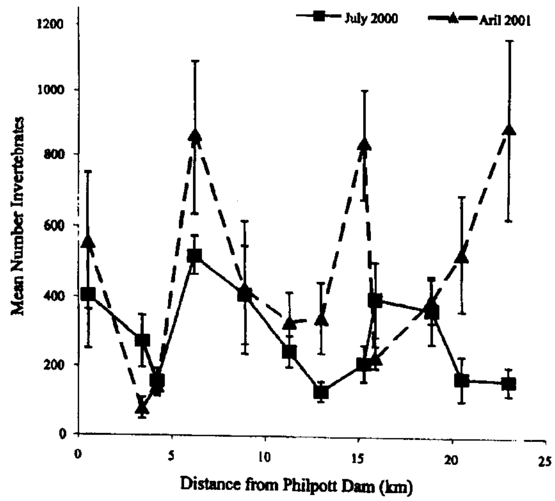

Figure 4. Mean number of invertebrates /m^2 (±1 SE) sampled at 12 sites below Philpott Dam, Smith River, Virginia, in July 2000 and April 2001.

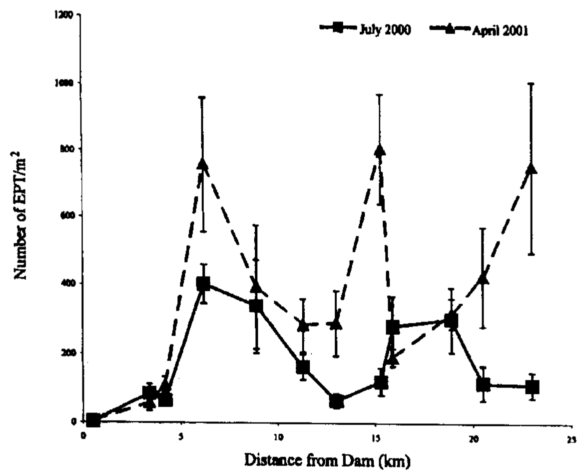

Figure 5. Mean number of Ephemeroptera/Plecoptera/Trichoptera/m^2 (± 1 SE) sampled at 12 sites below Philpott Dam, Smith River, Virginia, in July 2000 and April 2001.

Discussion

The findings of this preliminary investigation suggest that the invertebrate population in the Smith River exists in lower abundances than what would be found in a free-flowing stream of this size. We found the areas closest to the dam to be the most depauperate which corresponds with other investigations that document the greatest reductions and changes in faunal composition within the first 2.0 km below an impoundment where persistently fluctuating flows allow only for a community of flow resistant species (Cushman 1985, Voelz and Ward 1990). In one tailwater study, a 75%–95% reduction in biomass was observed within the first few kilometers of a dam, and a 40%–60% reduction was found as far as 20–40 km downstream (Moog 1993). Additionally, daily hydropeaking flows could repeatedly "flush" upstream portions of the channel by repeated events of increased drift associated with hydropeaking releases (Lauters et al. 1996); however, the reservoir and dam block macroinvertebrates that drift from upstream locations that would potentially recolonize the area.

One of the goals of our study was to evaluate macroinvertebrate abundance as a potential forage limitation for the brown trout population. Both macroinvertebrate biomass and densities measured in this study were similar to those collected in smaller third and fourth order trout streams in the southern Appalachians (Cada et al. 1987). Low macroinvertebrate densities ranging from 241 to 724 / m^2 appeared to result in lower condition and growth rates for trout in these streams (Cada et al. 1987). In contrast, numbers of invertebrates sampled in unregulated streams in Virginia, similar in size to the Smith River, typically range from 800–1,000 macrinvertebrates/m^2 (F. Benfield, pers. commun.). When compared to food grade categories for trout, all sites in the Smith River in 2000 and all except 3 in April 2001 fall into poor food grade classification with fewer than 538 organisms/m^2 (Lagler 1956). Rainbow trout are also stocked in the Smith River tailwater in spring and fall although a naturalized population does not seem to persist. The effects of stocking rainbow trout, in combination with the naturalized brown trout population, could also contribute to competition and further depress forage availability (Weiland and Hayward 1997). Although brown trout are believed to be more piscivorous than rainbow trout, during the summer months the thermal habitat that is the most suitable for trout is located closer to the dam where forage fish (e.g. cyprinids) densities are the lowest (Orth et al. 2001). This may lead to increased competition and reduced growth for brown trout in these areas.

Daily hydropeaking operations pose frequent and intense challenges to benthic macroinvertebrates. In the Smith River, flow is reduced to approximately 0.7 m^3/second approximately 1–2 hours before generation, while baseflows are usually 1.3–1.4 m^3/second near the dam. Dewatering the channel to half of its baseflow could also physically limit the wetted channel area that could be successfully occupied by macroinvertebrates near the dam. The margin areas are known to produce large numbers of invertebrates and therefore the daily dewatering of these areas could significantly decrease macroinvertebrate production (Gislason 1985). For example, Blinn et al. (1995) documented that the permanently submerged part of the channel below Glen Canyon Dam supported 4 times the number of macroinvertebrate biomass than the zone that was submerged and dewatered daily. In the Smith River, both highly variable flows and thermal instability may be limiting production of aquatic invertebrates. Near the dam, thermal stability is high, but water temperature is very cold (mean 8.4 ± 0.02 C in July), and the effects of flow instability are the greatest. Further downstream, as the river warms to ambient conditions, the daily thermal flux is large with up to 10 C cooling in less than an hour with the peaking flows (Krause 2002). However, at the downstream locations, the physical effects of increased flow are moderated in a larger channel with a greater baseflow than what is present near the dam.

Nutrients, and thus primary production, may also limit aquatic macroinvertebrates in the Smith River. Without sufficient forage, many taxa of aquatic insects may fail to thrive. Low temperatures can decrease the growth of periphyton (Blinn et al. 1989) and high rates of stream instability correlate with low levels of primary productivity (Death and Winterbourn 1995). Rates of instability include depth variation, changes in current velocity, substrate stability, and temperature range. All of these parameters change rapidly on a daily basis in the Smith River, making this a likely hypothesis for limitations on invertebrates as well. Additionally, as the reservoir ages, it may be experiencing oligotrophication (Ney 1986) and thereby contributing fewer nutrients (nitrogen and phosphorous) to downstream locations.

In conclusion, macroinvertebrate abundance in the Smith River appears to be lower than abundances observed in unregulated streams in Virginia and could potentially act to limit brown trout growth. Further research on quantifying the disturbance variables within this system, such as thermal flux, tractive force, bedload movement, and nutrient inputs, could be valuable for explaining the patterns of invertebrate abundance and composition observed in this study and to suggest recommendations for improving forage for brown trout in the stream.

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