

Final Report

Virginia

DJ Grant F-121-R

**Influences of Fluctuating Releases on Stream Fishes and Habitat
in the Smith River, Below Philpott Dam**

September 1, 1999 to June 30, 2004

Final Report

State: Virginia

Grant No: F-121-R

Grant Type: Research

Date: December 2004

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

Grant Period: September 1, 1999 to June 30, 2004

Grant 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. Job 1 **Characteristics of Spawning and Rearing Habitats for Brown Trout.** To characterize the instream habitat conditions in areas where successful spawning and juvenile rearing of brown trout occurs. Job 2 **Determinants of Brown Trout Growth and Abundance.** To collect biological data to quantify abundance of trout and nongame fishes in Smith River from Philpott Dam to Martinsville, quantify temperature limits on fish occurrence, and monitor annual variation in brown trout recruitment success. To evaluate the bioenergetic constraints on growth under existing temperature regimes. Job 3 **Hydraulic Model Development and Application to Smith River tailwater.** 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.

Activities: See attached report submitted by Dr. Donald Orth et al.

Significant Deviations: None

Total Costs:

<u>State</u>	<u>Federal</u>	<u>Total</u>
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* Virginia Tech waived indirect costs as third (3rd) party in-kind match.

Influences of Fluctuating Releases on Stream Fishes in the Smith River below Philpott Dam

Final Report

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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**

Operations of Philpott dam for flood control and peak power generation since 1953 have substantially altered downstream ecosystem conditions in the Smith River from the dam to Martinsville. We have described in intensive detail aspects of habitat and fish populations in the tailwater over the past four years (2000 to 2004), analyzed the limited historical information available to develop a better understanding of the mechanisms behind observed changes in the tailwater, and recommend appropriate actions to improve depressed fish populations and environmental conditions. Our research emphasizes that there are no “silver bullet” solutions and the most successful path toward improving the tailwater will reflect numerous tradeoffs to balance environmental, economic, and recreational goals. However, it is clear that enhancing conditions in the Smith River hinge on mitigating the effects of fluctuating releases from Philpott Dam through a combination of flow management (e.g. characteristics of dam operations during baseflow and peak flow periods) and habitat improvement (e.g. channel restoration, temperature management, enhanced biological productivity). In addition, removal or modification of Martinsville Dam to enhance flow, habitat, and fish and sediment passage would benefit fish populations and environmental conditions in the lower tailwater. Current fishery management strategies which are ineffective for enhancing brown trout should be re-evaluated after habitat and flow changes are instituted. Management actions for improving flow and habitat also should be assessed in light of the presence of the Federally Endangered Roanoke logperch *Percina rex* that also appear to be limited by degraded environmental conditions in the tailwater.

This report is organized into distinct jobs, which were initiated at different schedules. Job 1 is “Characteristics of spawning and rearing habitats for brown trout.” Job 2 has two parts: Part A is “Determinants of brown trout growth and abundance;” Part B is “Longitudinal patterns of community structure for stream fishes in a Virginia tailwater. Job 3 is “Hydraulic model stream development and application to Smith River tailwater.” The final management recommendations are drawn from all job findings.

Coldwater release provides habitat that is now conducive for non-native trout including a wild brown trout population and stocked rainbow trout. These species are the dominant component of the fish fauna. The wild brown trout population has followed a boom and bust cycle since natural reproduction first occurred approximately 30 years ago and special regulations were imposed to sustain the boom 20 years ago. Reservoir productivity has also followed a boom and bust cycle characteristic of new reservoirs. A once plentiful alewife population, which routinely supplemented the food base in Smith River through turbine passage, no longer provides an external source of food to brown trout in the Smith River.

Consequently, the brown trout population now subsists on instream production of depressed populations of aquatic insects. Fish contribute little to the annual diet and subsequent growth of

brown trout and only in the downstream portions of the tailwater where the relative abundance of nongame fishes exceeds 100 fish per 100 meters. Alkalinity (19 mg/L) and phosphorous (0.08 mg/L) concentrations are low and place additional constraints on biological productivity in the Smith River.

Recruitment of brown trout to the fishable size classes is constrained by the daily hydropower peaking operations. The number of young brown trout produced each year was strongly related to the average magnitude of the peak flow and the duration of generation flows. Further, number of young brown trout was unrelated to temperature. Therefore, flow management has the potential to influence the recruitment of brown trout in the Smith River tailwater.

Growth of juvenile brown trout is positively related to water temperature, which indicates that adequate prey exists for small trout. However, growth of yearling and older brown trout is less related to temperature and is depressed by limited prey availability in most seasons and stream reaches.

The section of the Smith River between 3 and 10 river kilometers from the dam is the critical reach for supporting the wild brown trout population and popular wild trout fishery. Here we found the highest redd densities, juvenile abundance, and spawner biomass. Other reaches provide suboptimal habitat for brown trout and lower populations. In the reach nearest the dam, brown trout are limited by cold temperatures and lack of spawning, nursery, and productive feeding habitats. In the lower reach, brown trout are limited by the heavily sedimented streambed and a lack of deep pools and rich prey base. The brown trout do actively remove fine sediment via redd construction and spawning, thereby increasing gravel permeability. However, the fine sediments from tributaries and bank erosion are transported by daily peaking flows and rapidly intrude into the spawning gravel in downstream reaches of the river.

Opportunities for enhancing the environmental conditions via changes in the operating conditions of Philpott dam were evaluated as well as other management actions to enhance the quality of the habitat and fishery and native fish populations. Conditions throughout the tailwater are consistent with the patterns expected after 50 years of dam operation with daily peaking, no ramping restrictions, loss of upstream gravel addition, and the rapid warming of cold water as it travels and equilibrates with the prevailing air and tributary inflows.

Operation of the dam for daily peak power generation has created a wider, rectangular-shaped channel, with steeper banks. Also the base channel elevation in the reach nearest the dam has decreased and caused steepening [headcutting] of tributary channels and bank erosion near the mainstem. Bottom substrate in the dam reach is dominated by bedrock and boulder and cobbles, gravel and sand are limited. As a consequence, spawning and nursery habitat for brown trout is limited and rooted aquatic vegetation is sparse. No native fish are present in the dam reach. Temperatures in the dam reach are too cold to permit adequate growth for a productive wild trout fishery.

In the middle reach, there are sufficient concentrations of cobbles, gravels, and islands to create a riffle-and-pool channel morphology. This middle reach supports the vast majority of the successful brown trout spawning and nursery habitat for juvenile survival and growth.

Temperature in the middle reach is more conducive to acceptable brown trout growth and production. A few species of native fish exist at low densities in this reach. Populations of native fish appear to be enhanced by a tributary influence on temperature, nutrients, and nursery or spawning habitat.

The stream channel in the lower reach is characterized by a high width-to-depth ratio, which is not conducive to bedload transport. Consequently, excessive aggradation of the channel bed is apparent from river kilometer 10 and further downstream. Temperature in the lower reach is warmer and a population of native fish is present, but production appears to be limited by cool temperature, shallow depth, and fine sediment deposition.

Environmental improvements that could mitigate for the effects of dam operations should be evaluated during the U. S. Army Corps of Engineers Section 216 Feasibility Study. The following would be appropriate actions that could be funded by the U.S. Army Corps of Engineers Section 1135 Funding Authority.

Temperature Management -- Temperature is an important driver for many biological processes in the Smith River. During this study we confirmed a number of important processes that were strongly tied to temperature. Brown trout initiate spawning in November or December when water temperatures first fall below 9 C and timing of spawning depends on distance from Philpott dam. Growth of age-0 brown trout was strongly positively influenced by degree-day accumulation between May and October. Therefore, age-0 brown trout grew faster at downstream reaches and in years with lower volume of cold water releases during this time period. Species richness and abundance of native fish were closely tied to distance from the dam and tributaries, both of which affect water temperature. Densities of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) were different among stream reaches of the river and significantly depressed below expectations in the dam reach. Although temperature regime could be improved to increase the length of tailwater with suitable temperature for brown trout and native fishes and invertebrates, we were unable to test our predictions of the increased growth rate due to unknown effects of temperature management on future prey availability. The direction of the response is, however, clearly positive if prey populations also increase.

Dynamic flow and water temperature models were used to predict thermal habitat under alternative flow scenarios. Model output and species thermal criteria enabled assessment of potential benefit or detriment to brown trout and warmwater species. Currently the average release temperature (8°C) is below the optimal brown trout growth range (12-19°C). A 12°C outflow scenario predicted the greatest increase of optimal growth temperatures. Warmer temperatures also increase the area of suitable thermal habitat for warmwater species, including the Roanoke logperch. With changes in flow management we found it is possible to improve the trout fishery without detrimental effects to the warmwater community. However, the trout fishery improvement is dependent on concomitant increases in productivity of the trout prey base and appropriate temperature management. Temperature management actions will have to be closely monitored in order to determine the level of increased growth that is realized by a new thermal regime.

Flow Management -- Hydraulic models were used in predicting physical habitat for brown trout spawning under alternative flow scenarios. Results from the habitat simulation indicates a significantly positive relationship between redd density and habitat quality predicated by the model. Based on our regression analysis, current base flow appears to be below the optimal reservoir release range (9-15 m³/s), while the peak flow is too high to support suitable brown trout spawning environment. A 12 m³/s reservoir release scenario predicted the best suitable habitat availability in our study site. However, because the highly fluctuating flow causes temporal changes in the locations of suitable habitat, an adaptive monitoring program is needed to correctly evaluate the fish habitat under such a flow scenario. Because brown trout spawning predictable based on daily temperatures, we recommend that peaking flows be restricted during the time of peak spawning. Furthermore, we identified a significant response of brown trout to frequency and magnitude of peaking flows during the incubation and emergence, which indicates that peaking operations could be restricted during these times, especially in dry years, in order to enhance brown trout recruitment success.

The rapid increase in flows during generation causes substantial increase in the shear stresses on the channel bed. Our analysis from hydraulic models shows that gravels, which are important to redd construction, may be moved within the initial release period. This fluctuating flow also causes a higher stress to fish, may displace young brown trout, and prohibits successful spawning by minnows. Our recommendation is to use a two-step flow release scenario. It is better to use one turbine to release flow for half an hour and then add another turbine to release additional flow if necessary. Study illustrates the shear stress acting on gravel and drag force exerted on fish may be greatly reduced under such a hypothetical reservoir release scenario, and hence a healthier stream can be maintained without affecting power generation requirement.

Endangered Species Restoration -- The middle and lower reaches of the Smith River support a population of the Federally listed, endangered Roanoke logperch *Percina rex*. Logperch were present at low abundance but we cannot establish normative population abundance levels from other Roanoke logperch populations. The Roanoke logperch population is disconnected from a population in Town Creek and the main channel population is likely depressed by cold summer temperatures, flow fluctuation during spawning time, and excessive silt and sand in pool habitats. Consequently, the management actions to improve the habitat in the Smith River via channel restoration and flow and temperature management are required and will require Section 7 consultation between the U. S. Fish and Wildlife Service, Department of Game and Inland Fisheries, and the U. S. Army Corps of Engineers.

Habitat Management -- The current stream channel in the lower reach does not adequately function to transport the heavy sediment load under current operations. Flood management has eliminated flood flows in the Smith River such that the highest flow is the peak generation flow, which can occur daily. As a consequence, much of the water surface is shallow and slow moving between generation releases; this causes more rapid warming of the water. We recommend that channel restoration, using natural channel design principles, be implemented in the lower reaches of the Smith River in order to enhance bedload transport, stabilize banks and floodplains, increase habitat suitability, and reduce flood risk. Channel design in the lower reaches should include narrowing and deepening the channel, boulder addition, restoring

floodplain contours, increased sinuosity, and adding nearstream woody vegetation to shade the channel and protect streambanks from erosion.

Channel design in the dam reach should include addition of gravel and cobble materials to improve the conditions for invertebrate production if peaking flows were reduced. The amount of material to be added to mitigate for 50 years of channel degradation will be substantial. The use of limestone materials is recommended to enhance the alkalinity and biological productivity. Special attention needs to be made to address the many steep and eroding stream banks in the main channel and the tributary streams (especially Town Creek) which have down cut due to lowered base elevation of Smith River in dam and middle reaches. Design and implementation of the channel restoration will require modifications in the timing and magnitude of the releases from Philpott dam and cannot be successfully implemented without some change in peaking flows.

Daily peaking will continue to destabilize steep banks, thereby adding fine sediment to the channel. This fine sediment rapidly intrudes into the channel bed further depressing the aquatic life processes in the intergravel habitats. Combination of shallow depth, fine sediment, and mobile fine bed deposits provide very poor habitat for production of insects and plants, and for spawning and feeding habitats for native fish and trout.

Our analysis with two- and three-dimensional hydrodynamic models documents the potential mitigating effect that large boulders have in creating stable downstream wakes behind boulders, even during generation flows. These models can be adapted for use in a newly designed channel to create appropriate boulder placements to increase refuges for fish during generation flows and still permit the transport of bedload.

The Martinsville Dam creates a long backwater effect that traps substantial fine sediment, warms water temperatures, thereby further decreasing the amount and quality of suitable fish habitat in the Smith River. Furthermore, the dam creates a barrier to fish movements and prevents recolonization by native fish downstream from the dam. Species, such as the Roanoke Darter *Percina roanoka* and Roanoke Bass *Ambloplites cavifrons*, could benefit from reconnection with a restored Smith River. Native mussels are also extirpated from the mainstem Smith River between Philpott Dam and Martinsville Dam due to cold temperatures and barrier effect of Martinsville Dam. We recommend that the City of Martinsville and the Virginia Department of Game and Inland Fisheries evaluate the costs and benefits of barrier mitigation options at Martinsville Dam.

Harvest Management -- The current management strategies for the trout fishery include a special regulations section with 16 inch minimum size limit and general trout regulations elsewhere. Trophy regulations are not warranted in this population as the brown trout growth is slow and ceases soon after fish are mature enough to spawn. Trout from 8 to 12 inches are in fair to poor condition and nonfishing mortality is very high. Changes in the harvest regulations in the Smith River should be re-evaluated after major changes in dam operations are in place to alter the habitat and potential for enhanced prey production. We further recommend that separate regulations be established for brown trout and stocked rainbow trout in the Smith River in order to emphasize the distinction between wild and stocked trout among the fishing public. There is no compelling biological reason to protect brown trout less than 16 inches since fewer than 1 in

1,000 of the brown trout sampled during this study exceeded 16 inches. However, the population is unlikely to respond to any regulation change at current limited growth rates.

Timing of implementation -- The recommended actions provide a template for future management actions to be considered by the U.S. Army Corps of Engineers and the Department of Game and Inland Fisheries. Clearly, some of these actions will be more expensive and require more time to implement, whereas others can and should be implemented earlier. We recommend guidelines for the timely implementation of changes.

1. Flow changes
2. Channel restoration and habitat management
3. Temperature management
4. Endangered species restoration
5. Fishing regulation changes

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**

Introduction

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 1.78 cms to 42 cms 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). More specific proposals for reservoir releases will be

needed before U.S. Army Corps of Engineers is 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 Fisheries 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. Descriptions of habitat for 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 14°C annually below the dam, but approach 25°C in reaches more than 20 km downstream. Temperatures between 12-19°C result in optimal growth of juvenile brown trout (Ojanguren et al. 2001) and temperatures > 19°C results in visible thermal stress (e.g. cessation of feeding; Elliott 1981). Lethal temperatures for brown trout range from 25 to 30°C. 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 (10°C 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. Constraints imposed by habitat and limited movements on recruitment of brown trout are unknown. 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 indicates 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). 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 studies 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

Job Objective: 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, and spawning habitat quality are of interest to fisheries managers. We measured and monitored these conditions, as well as discharge, water temperature, and water quality over 5 years (2000-2004).

The tailwater maintains cool water conditions for salmonids due to hypolimnetic release from Philpott dam. Philpott dam (lat. 36 46'50", lon. 80 02'40") located 12 km upstream of Bassett, VA (Henry county) was completed in 1953 by the Army Corps of Engineers (USACE) for flood control, hydropower, and recreational opportunities. The dam forms Philpott reservoir (1,165 hectares at 297 m elevation) and the Smith River tailwater (32 km in length from Philpott dam to Martinsville dam). Philpott dam (67 m high and 280 m long) is a concrete gravity dam with an ogee spillway, a powerhouse, and switchyard (USACE, 2004). Within the powerhouse are two 6,700 kw and one 600 kw vertical shaft Francis turbines. The 600 kw turbine releases baseflow (typically 1.6 cms) and the two 6,700 kw turbines release peakflow (19.8 cms per turbine for a total of 39.6 cms). Peak flows for hydropower are typically released daily Monday through Friday. Baseflow and peakflow are drawn from the reservoir at a depth of 26.2 m (at a normal reservoir elevation of 296.1 m). If the 600 kw turbine is non-operational then two low-flow outlets (41.9 m reservoir intake depth) enable discharge to maintain water in the tailwater.

In the Smith River, peaking flow, water temperature, and sedimentation influence spawning and recruitment success. Peakflows during the incubation period may scour spawning gravels causing egg mortality, and during the emergence period can inhibit survival. Water temperature, which is directly affected by the hypolimnetic release from Philpott dam, is a key factor to cue spawning, control incubation of eggs, and determine age-0 growth. Sedimentation, which causes fine sediment intrusion into spawning nests, restricts intragravel permeability and thus the replenishment of dissolved oxygen to developing embryos. Over the long-term, peaking flows have concentrated and removed spawning gravels intermittently throughout the tailwater, thus limiting potential spawning habitat. Tributaries have delivered fine sediment leading to highly impacted substrates downstream of their confluence with the mainstem. In the 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 flow operations may therefore lead to rapid deterioration and/or failure of redds and displacement of newly-emerged age-0.

Sampling of age-0 emergence and growth was conducted at five successful spawning locations distributed throughout the tailwater. Other factors such as water temperature, dissolved oxygen, substrate composition, and intragravel permeability were measured at additional locations and assessed within four reaches of the tailwater differentiated based on management regulations and gross morphometric habitat variations. Those four reaches are the Dam (0.0-5.3 km), Special Regulations (5.3-10.0 km), Bassett (10.0-15.9 km), and Koehler (15.9-24.0 km) reaches. The Special Regulations reach is managed by the Virginia Department of Fish and Game for trophy trout (i.e. = 406 mm).

Procedures

Discharge Analysis

Discharge data (15-min interval data, 1991-2004) recorded at the USGS gage station (#02072000) located 0.5 km below Philpott dam was analyzed for occurrence, magnitude, and duration of peaking flow. Peakflow was classified as discharge >2.8 cms (>100 cfs) to conservatively designate it as any discharge greater than the typical 1.3-1.8 cms (45-65 cfs) baseflow, which occurs when either one or two turbines are operating. Additionally, max-peakflow was also assessed as discharge >28.3 cms ($>1,000$ cfs) to separately account for times when both 6,700 kw generators were operating at full capacity. Occurrence is the percent time that flows occurred, magnitude is the average discharge of flows, and duration is the average time in hours per day that flow was released.

A representative 'typical' flow regime released by the USACE, per season, was developed using 13 years (1991-2003) of 15 min interval discharge data to assess disparity of the flow regimes occurring during this study (USGS does not have 15 minute data prior to 1991) (USGS, 2004). Typical conditions for baseflow discharge, drawdown discharge prior to peaking flow, and time-of-day of peaking flow release were determined with histograms as those most frequently occurring. The typical magnitude and duration of peakflows were determined as the average of what occurred. The combination of these variables enabled the development of a representative flow regime released by the USACE each season. The same methods were performed on discharge data from each year of this study to characterize the 2000, 2001, 2002, and 2003 flow regimes.

Hydrologic regime characteristics were analyzed over time with the Indicators of Hydrologic Alteration software (IHA, version 5) developed by the Nature Conservancy with Smythe Scientific Software (Nature Conservancy, 2004). We analyzed Smith River daily flow data from the Philpott gage station with the IHA model from 1946-2002. The model accounts for impacts such as the construction of a dam; therefore we compared flows pre-impact (1946-1952, 7 yrs) and post-impact (1954-2002, 49 yrs) of the construction of Philpott dam (1953).

Water Temperature

To obtain accurate half-hourly water temperature data, an Onset[®] submersible temperature logger was secured with steel cable at 0.7, 2.7, 5.1, 5.6, 10.2, 18.3, and 24.3 rkm. Additionally, an Onset[®] temperature logger was also deployed in Town Creek (0.5 rkm from confluence) and Reed Creek (0.3 rkm from confluence). Temperature loggers were placed in locations of flowing water 0.3-0.75 meters deep and in shaded areas where possible. Temperature data was collected from 1999 to 2004. Loggers were downloaded monthly to minimize potential data loss and at that time the stream's water temperature was measured with a hand-held thermometer to verify proper functioning of the logger. Multiple linear regressions were used to fill water temperature data lost due to data logger failure (Krause, 2002).

Water Quality

During November and December 2002 dissolved oxygen (DO) content of intragravel water was measured in the egg-pocket region of redds and 1 m beside those redds ($n = 30$), as well as in the free-flowing channel water above the redd, with a MARK VI standpipe and YSI meter model 95 (Barnard and McBain, 1994; McBain and Trush, 2000). Dissolved oxygen was measured at 2-4 redds at sampling sites 1, 2, 3, 4, 5, 7, 8, 9, 10, and 12 (Figure 1.1). To evaluate DO content of free-flowing channel water during baseflow and peakflow, DO was measured

with a YSI 95 meter ($n = 27$) at 12 locations from 0.2-13.1 km below Philpott dam during August and September 2003.

We further evaluated DO conditions in the reservoir and tailwater with Virginia Department of Environmental Quality (DEQ) data measured at 5.1 rkm below Philpott dam (1992-2003) and DO & water temperature profiles within Philpott reservoir near Philpott dam (DEQ personal communication). The reservoir profile data, measured every meter from the lake surface to the bottom, was available for up to 6 months out of the year for years 1995-2002. To determine the DO and temperature of water passing through the dam, the depth of the hydropower intake was determined for each month of each year based on USACE lake surface elevation records and the known intake elevation. The DO data at 5.1 km below Philpott dam was delineated between baseflow and peakflow using the DEQ time of measurement compared to USGS discharge data (travel time was accounted for).

Data for additional water chemistry parameters including conductivity, pH, total alkalinity, and total phosphorus was obtained from the Virginia Department of Environmental Quality (DEQ, 2004). The DEQ sampled ($n = 127$ -187) 5 km and 22 km below Philpott dam during 1967-2003.

Tailwater Substrate Composition

Beginning at Philpott Dam, we used canoes to survey macrohabitat characteristics of the Smith River at baseflow to 24 km downstream. We partitioned the river into either pool/run or riffle habitat units. At the beginning and end of a habitat unit, we used a handheld GPS unit to obtain 2-Dimensional spatial coordinates and determined wetted width of the river with a laser rangefinder. As we canoed the length of a habitat unit we estimated percent coverage of substrates by size (modified Wentworth scale; Cummins, 1962). This approach allowed us to document longitudinal patterns in habitat at a relatively large spatial scale.

Substrate Permeability

Substrate permeability (i.e. intragravel flow) was measured to evaluate whether redd construction improves egg incubation habitat, whether redd condition changes longitudinally throughout the tailwater, to compare Smith River permeability quality with values in the 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 reduce measurement variability. Confidence intervals (95%) around mean permeability values were calculated (McBain and Trush, 2000) to discern difference in and beside redds, and ANOVA and Tukey-Kramer HSD ($\alpha = 0.05$, JMP 4.0.4 statistical software) tested for differences in permeability among reaches. In addition to permeability, DO and temperature of intragravel water was measured in the egg-pocket region of redds and beside redds by lowering a YSI meter model 95

probe down the standpipe. Temperature and DO were also measured in the free-flowing channel water above the redd.

Fine Sediment Intrusion

Fines < 2 mm can inhibit egg incubation 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. Vibert boxes (142 x 88 x 60 mm), made of polypropylene with rectangular openings, are typically used to hold gravel and salmonid embryos while incubating in a streambed (Garrett and Bennett, 1996). Substrate compositions, which were representative of redds at five spawning areas (4.2, 6.2, 13.1, 14.9, and 22.8 km), were placed in the Vibert boxes. Redd substrate compositions were collected from five redds in each of the five spawning areas in January 2001 using a McNeil bulk core sampler. From these compositions, fines < 2 mm were removed, thus intruded fines (< 2 mm) 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 placed at two corners of the grid. 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°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. Difference in fine sediment intrusion between sites and over time was assessed with ANOVA and Tukey-Kramer HSD (alpha = 0.05, JMP 4.0.4 statistical software).

Spawning Patterns and Characteristics

Redd surveys were conducted during the 2000, 2001, 2002, and 2003 spawning season (November-December) to monitor spawning activity. Because we did not observe spawning on most redds, redd surveys are more an index of spawning activity than a measure of actual spawning. Two observers noted the presence of redds (i.e. spawning nests) in reaches typically 200-400 m in length at locations throughout tailwater. This amounted to a total search distance of 3-15 km or 13-62% (depending of year of study) of the 24 km stretch of tailwater under study. Searches were conducted by wading during daylight hours at baseflow conditions. Detailed maps of the survey reaches were drawn and redd locations marked; river kilometer locations of redds could then be determined with DeLorme[®] topoquad software. In 2003, surveys were not possible during the latter part of November and the middle of December due to rain, as well as

snowmelt causing poor water clarity and frequent, long-duration peakflow releases from Philpott dam.

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) at 2 cm above the bottom at locations measured for depth, and mean column velocity at 60% depth over the middle of the pit. Redd characteristics were measured for 46-148 (depending on the characteristic), 3, and 44 redds in 2000, 2001, and 2002 respectively. Redd characteristics were not recorded in 2003 due to frequent peaking flows and poor water clarity. Significant differences of redd characteristics between 2000 and 2002, and among reaches were evaluated with ANOVA and Tukey-Kramer HSD (alpha = 0.05, JMP 4.0.4 statistical software).

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 monitored five known spawning sites. Spawning sites were selected based on high redd densities (relative to other nearby sites) observed during fall 2000 surveys. The five sites, 4.2, 6.0, 12.9, 14.1, and 22.8 km downstream of Philpott dam, were sampled in March 2001 and 2002 and May 2001-2004. Timing of emergence was determined by visually searching (on foot) in 2001 and 2002, and by electrofishing in 2003. Once age-0 trout were present at all five sites, we allowed two weeks for more age-0 to emerge before commencing with population density sampling. At each site we conducted 3-pass depletion samples with a backpack electrofisher (Smith-Root Model 15-D) within 3 m of the riverbank in three 25 m sections per site (75 m² per section). The first section was located at the downstream boundary of the site's spawning area (0-25 m), the second 75 m downstream (75-100 m), and the third 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. Additionally, in 2003 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. Depletion data enabled calculation of age-0 abundance (i.e. population estimate) within each of the 25 m sections using MicroFish software (Van Deventer and Platts, 1985). Abundance within the three 25 m sections per site were averaged to obtain a site abundance. Confidence intervals (95%) around the site abundance were calculated as

$$95\% \text{ CI} = 1.96 * (\text{Square Root } ((SDa^2 + SDb^2 + SDc^2)/3^2))$$

where SDA is the population estimate standard error (calculated by MicroFish) in the 0-25 m section, SDb is the 75-100 m section, and SDc is the 150-175 m section. Statistical difference of mean age-0 length among sites was tested with ANOVA and Tukey-Kramer HSD, and difference in abundance among sites and years was assessed with 95% confidence intervals.

The growth rate of age-0 brown trout was calculated two ways: (1) the slope from a regression through length versus sample-date data (Bettoli et al., 1999) and (2) absolute growth per unit time (Kohler and Hubert, 1999). Growth rates were nearly identical among the two methods (maximum difference = 0.01 mm/day) and not statistically different among reaches when years were combined (P = 0.58-0.99, ANOVA, JMP 4.0.4 statistical software). Therefore

only the regression slope values are presented. Growth was calculated for years 2000, 2001, 2002, and 2003 with average total length (mm) data from May, June/July, and October in the Dam, Special Regulations, Bassett, and Koehler reaches. To obtain growth via regression, the average total length versus calendar sampling date was plotted and the slope from the resulting best fit line is the growth rate (mm/day) (Bettoli et al., 1999).

Habitat characteristics of the three 25 m reaches within the five spawning sites were measured in September 2003 to characterize the sampling areas. We measured meso-habitat type (pool, run, riffle), embeddedness, water depth, water velocity (demersal and mean column), substrate (e.g. D50), overhead canopy cover, and bank angle. The GPS location at the upstream end of each 25 m sampling reach was recorded. Meso-habitat type was visually assessed for each reach (Platts et al., 1983). Habitat characteristics were measured at four locations, at 5 m intervals, within each 25 m reach. A sampling grid (60 x 60 cm with cords spaced 10 cm creating 49 intersections) was placed ~ 1 m from the water's edge at the 5 m interval locations (Bunte and Abt, 2001a). Embeddedness was visually estimated within the sampling grid area as the percent bottom covered with fines based on 4 quartiles; 0-25, 25-50, 50-75, and 75-100 percent (Bovee, 1986; Persinger, 2003). Depth (cm), demersal water velocity (2 cm from channel bottom) and mean column (60% depth) water velocity (m/s) were recorded at the center of the grid with a wading rod and Marsh-McBirney model 2000 flow meter. Substrate particles at each of the 49 grid intersections (n = 196 per reach) were measured with a gravelometer template (Stancil, 2000; Bunte and Abt, 2001a, 2001b). A spherical densiometer was used to measure percent overhead area occupied with canopy; measured from the middle of each 25 m reach 4 times: facing upstream, downstream, left-bank and right-bank (Lemmon, 1957). Bank angle was recorded by laying a straight surface (stadia rod) from the water's edge to the peakflow high-water mark and measuring the angle with clinometer at the 5 m interval locations (Platts et al., 1983).

Macroinvertebrates

Benthic macroinvertebrates at the 12 sampling sites (Figure 1.1) were sampled in July 2000 by Newcomb et al (2001). This dataset was supplemented by additional collections in July 2001, 2002, and 2003. We followed the same methods as presented in Newcomb et al (2001). Those methods in summary are: Surber samples were collected at two randomly selected locations along an upstream, middle, and downstream transect within a riffle; totaling six samples per site. Macroinvertebrates were identified to family and samples were measured for wet weight.

Results and Discussion

Discharge Analysis

The flow regime during the years of study (2000-2003), compared to previous years (1991-1999), deviated from the norm (Table 1.1). Max-peakflows (>28.3 cms) occurred an average of 14% less during the years 2000-2002 than 1991-1999 (Table 1.2). Max-peakflows occurred an average of 14% more during the 2003 annual year than 1991-1999 (Table 1.2). Additionally, the percent time during 2003 that max-peakflows were released from Philpott dam was over twice that of the average for the past 12 years (1991-2002). The duration (hrs) of peakflow release was an average of 3 hours less during 2000-2002 and 2 hours longer during 2003 compared to previous years (1991-1999) (Table 1.2). Lessened occurrence and duration of peakflows during 2000-2002 were due to drought conditions which were most severe in 2002.

The following year, 2003, experienced substantial rainfall which resulted in frequent, long duration releases to maintain normal reservoir levels for flood control (Figure 1.2). Reduced release magnitude (cms) in spring 2002 and summer & fall of 2001-2002 was due to only one of the two 6,700 kw turbines being operational (Table 1.1 and Figure 1.2).

Multiple study years experienced flow regimes atypical of the 1991-1999 norm, 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. 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.

Non-parametric analysis of daily flow regime characteristics pre and post construction of Philpott dam with the IHA model found several distinct changes. Mean annual flow declined from 8.30 cms (293.1 cfs) pre-dam to 7.81 cms (275.7 cfs) post-dam (Table 1.3). The dam operations caused mean monthly flow to decline an average of 1.36 cms for all months except January, July, and August which had an average increase of 0.61 cms. The one and three day minimum and maximum daily flows were lower post-dam (Table 1.3). The daily count of low and high pulses increased and the daily duration of low and high pulses decreased. Overall, the dam changed the daily flow regime to one with baseflows and peakflows that are now of less magnitude, shorter duration, and more frequent than pre-dam flows. The developers of the IHA model recommend a 20 year length of record to obtain reliable pre- vs. post-impact comparisons. Our pre-impact dataset is 7 years, thus the model output should be assessed with caution.

Water temperature

Water temperature in the Smith River tailwater is influenced by hypolimnetic release from Philpott dam (~ 26 m intake depth within reservoir), peaking flow regime, and ambient conditions. Near the dam (0.7 rkm) temperature averages 8°C (SD = 3°C) and exhibits very little daily fluctuation (Figure 1.3 and 1.4). At upstream locations (~ 0-5 rkm) daily temperature rarely exceeds 12°C and these cold temperatures would extend further downstream during summer (June, July, August) if not for inflows from Town Creek at 5.3 rkm which raises temperature an average of 2°C. Occurrences of brown trout optimal growth temperatures (12-19°C; Brown, 1974; Brungs and Jones, 1977; Raleigh et al., 1986; Smith, 1994; Ojanguren et al., 2001) are greatest from May through September and occurrence increases with downstream distance (Figure 1.4). The occurrence of optimal growth temperatures did not exceed 26.3% of any one month (May-Sep) upstream at 5.1 km in 2000, 2001, and 2002 (Figure 1.4). 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 (22-26°C; Brungs and Jones, 1977; Elliot, 1981) of brown trout. These elevated temperatures exceed the Virginia Department of Environmental Quality's (DEQ) 21°C maximum temperature standard for stockable trout waters (DEQ, 2004). However, monthly maximum temperatures only exceeded 21°C at 24.3 km twice (June and July 2002) during the study (Figure 1.3). Daily maximum temperatures =21°C occurred at sites 5.1, 5.6, 10.2, 18.3 and 24.3 km during the study. The largest occurrence of temperatures =21°C was 25 and 23 days of June and July 2002 at 24.3 km, whereas an average of 2 days experienced temperatures =21°C during June and July of the other study years at this location. Elevated temperatures in summer 2002 occurred from

lessened peakflow occurrence, magnitude, and duration (Table 1.2) due to drought conditions. 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. Hourly declines in water temperature were greatest in summer when warm ambient air conditions and tributary water temperatures increased the temperature difference between baseflow and peakflow water temperature. During baseflow, water warms as it travels downstream and it is rapidly cooled by peakflow, which travels downstream faster and in greater volume, restricting its ability to be warmed by ambient conditions (Figure 1.5). The largest hourly temperature declines were measured by data loggers from 2.7-10.2 km, which is where peakflow rises rapidly and is near its original temperature prior to effects of cumulative attenuation and baseflow-mixing, respectively (Figures 1.4 and 1.5).

Water temperature data revealed an interesting trend of declining temperature during baseflow on particular dates in the second half of 2003 (Figure 1.6). On July 12th, August 3rd, and 10-11th, September 13th, October 2-6th and 24th, November 17-21st, and 30th, and December 1-3rd water temperature declined an average of 5°C when baseflow was released (a maximum decline of 7°C was recorded). On these dates Philpott Dam released baseflow through the low-flow outlets; confirmed by Mary Gardener (Philpott Dam Conservation Biologist) who contacted the Kerr reservoir operator (who maintains Kerr and Philpott operation logs). Typically baseflow is released through the smaller 600 kw turbine (peakflow is released through two 6,700 kw turbines). However, when the smaller turbine is not operational the two low-flow outlets enable baseflow release to maintain water in the tailwater. Both the 600 kw and 6,700 kw turbines intake water from the reservoir at an elevation of 269.9 m whereas the low-flow outlets intake water at an elevation of 254.2 m. With a normal reservoir elevation of 296.1 m the intake depths are 26.2 m (86 ft) and 41.9 m (137.5 ft). It is this deeper intake depth of the low-flow outlets where water temperatures are colder that caused the temperature declines during baseflow.

Temperature declines of this nature have not been observed during the other years of this study (1999-2002). This could be associated to no low-flow outlet use or different reservoir temperature profiles than in 2003. There would certainly have been a different temperature profile in the reservoir in 2003 because water temperature released from the dam was twice as warm compared to 1999-2002 (Figure 1.7). This profile would have had a large temperature difference between the two intake depths. DEQ reservoir temperature profiles show that temperatures at the 26.2 m versus 41.9 m depth are typically similar (Figure 1.8a and 1.8b). The exception is the fall season prior to the lake turnover when the epilimnion descends pushing warmer temperatures into the 26.2 m intake range.

Rather than only using the low flow outlets during non-operational periods of the 600 kw turbine, the low flow outlets could potentially be used for temperature control via flow management. If temperature and dissolved oxygen conditions at the 26.2 m and 41.9 m intake depths were monitored, flows could be selectively released from either depth to cool water temperatures in the tailwater. This could be useful if tailwater temperature were exceeding brown trout thermal tolerances downstream. However, the tailwater temperatures are typically below the brown trout optimal growth range and therefore, flow management to warm temperatures is more likely to be desired.

Water Quality

Dissolved Oxygen - Dissolved oxygen (DO) is a critical water quality parameter for growth and survival of brown trout. Optimal DO content for the incubation of brown trout eggs is >7 mg/l (Raleigh et al., 1986; Chapman, 1988; Maret et al., 1993, Ingendahl, 2001). During the 2002 spawning season (November and December), DO of intragravel water within the egg pocket region of redds averaged 11 mg/l (SD = 0.52), which is not limiting to embryo survival. The DO concentration followed a consistent trend, being greatest in the free-flowing channel-water (Mean = 11.09 mg/l; SD = 0.49), followed by the egg pocket (10.9 mg/l; 0.52), with intragravel water beside the redd having the least DO (10.32 mg/l; 0.85). Water temperature was very similar among the in-redd, beside-redd, and in-channel locations; among these locations temperature never differed more than 0.2°C. Porter (1985) and Maret et al. (1993) also found temperature between surface and intragravel water to be highly correlated. The DO of free-flowing channel water during peakflow (Aug-Sep 2003) was lower than that of baseflow, and DO was lowest upriver near the dam (Figure 1.9). Lower DO upriver and during peakflow is likely due to the greater volume of water traveling rapidly downriver, thus limiting the ability for re-aeration. Within the first 5 rkm of the tailwater, peakflow DO averaged 3.6 mg/l (Aug-Sep 2003) which is lower than the ≈ 7 mg/l brown trout optimum (Figure 1.9). During September 2003 at 0.2 rkm below the dam, the DO concentration of baseflow (4.9-5.5 mg/l) dropped during peakflow to 1.6-1.8 mg/l. Further downstream (4.2 rkm) the DO of baseflow (7.4-9.7 mg/l) dropped during peakflow to 3.7-4.4 mg/l. Dissolved oxygen content between base and peakflow is similar during spring, summer, and winter (DEQ, 2004) (Figure 1.10). However, during fall the DO of peakflow is lower (avg. 6.6 mg/l) (Figure 1.10).

In Philpott reservoir, DO levels decline from spring to fall and they decline sooner in the upper reservoir (i.e. upstream end) than the lower reservoir near the dam. Thus, DO levels are less in the upper reservoir by the time the reservoir de-stratifies and turns-over in November. The cause of declining DO is from consumptive processes not compensated by photosynthetic production of oxygen, especially at deeper depths where light penetration is minimal. In the upper reservoir where the majority of the inflow occurs, concentrations of dissolved or suspended solids, nutrients, and organic wastes will be highest. Increased levels of solids limit the ability of oxygen to dissolve into the water, organic waste is decomposed by bacteria that remove DO, and nutrients promote algae growth which then become organic waste. As water progresses from the reservoir mouth to the outlet, bacteria decompose organic waste which removes DO. If the transition time for water to flow completely through the reservoir is quick enough, then anoxic levels present in the upper reservoir can reach the lower reservoir and subsequently be released into the tailwater. Based on DEQ reservoir DO profiles, the low DO levels recorded in the tailwater in 2003 are normal (Figure 1.8b). It is possible that the DO content was a little lower than is typical due to the heavy precipitation regime in 2003. The large amount of water entering the reservoir would have increased solids and organics, and it increased the hydropower release frequency and volume, possibly shortening the transition time of water through the reservoir.

The DEQ has never collected DO profile data from November through February so actual DO content in the reservoir during this timeframe is not known. The baseflow DO data collected during November and December 2002 in the intragravel and free-flowing channel water was not limiting nor is the DEQ DO data at 5.1 rkm (Figure 1.9 and 1.10). Reservoir turn-over, typically in November, is what increases the DO in the deeper water by mixing it with surface water rich in DO.

The average DO and water temperature in the reservoir at the approximate depth released through the dam from April through August is 8 mg/l and 8°C, and during October is 3 mg/l and 14°C (Figures 1.8a and 1.8b). Therefore, limiting DO levels in the tailwater will only occur in the fall season prior to lake turn-over, primarily during peakflow, and in the upriver reach (~ 0-5 rkm below dam).

Water Chemistry - Measurements of water quality provided by the DEQ illustrate the extremely low levels of dissolved or suspended material present during baseflow conditions in the Smith River. This is consistent with other tailwaters which owe their source to hypolimnetic releases (Brooker, 1981; Cushman, 1985). Conductivity was less than 60 $\mu\text{mhos cm}^{-1}$ in all but the most downstream reaches of the tailwater. Downstream of the Henry County Upper Smith River wastewater treatment facility (22 km), conductivity increases to > 100 $\mu\text{mhos cm}^{-1}$. The pH in the Smith River has averaged 6.7 (ranging from 5.5-8.2) over the last 34 years at both an upstream (5 km) and downstream (22 km) site. These pH levels within the tailwater are within the range that brown trout occur (5.0-9.5) as well as the optimal pH for growth (6.8-7.8) (Raleigh et al., 1986). A monotonic decline in pH is evident in the recent record, equivalent to a one unit change. Alkalinity and phosphorous are in low concentrations particularly upstream near the dam. Alkalinity (mg/l as CaCO_3) in the Smith River has averaged 19.2 mg/l (SE 0.2) at 5 km and 21.7 mg/l (SE 0.3) at 22 km over the last 34 and 31 years, respectively. Phosphorous has averaged 0.08 mg/l (SE=0.00) at 5 and 22 km over 15 and 24 years respectively. Alkalinity has been tied closely with trout production, (Kwak and Waters, 1997; Clarke and Scruton, 1999), and combined with low concentrations of nutrients likely sets a low upper limit on production of the forage base for brown trout and other fishes in the Smith River (see Job 2 for additional study on trout and nongame production). Low alkalinity also results in fewer and smaller eggs produced by trout (McFadden et al., 1965). Alkalinity levels preferred for trout culture are 80-200 mg/l (Stickney, 1991) and for optimal growth are 120-400 mg/l (Piper et al., 1982), which are well above concentrations in the Smith River. Alkalinity concentrations in the Smith River are also below those in the Beaver tailwater, Arkansas (63 mg/l), which ranks among the lowest concentrations reported for North American trout streams and is likely the definitive cause of poor brown trout reproductive success in that tailwater (Pender and Kwak, 2002). Low alkalinity in the Smith River may impose limitations on brown trout reproductive success and growth; however water fertility is not easily mitigated, therefore management of habitat, discharge, and/or water temperature are more plausible for enhancing productivity.

Tailwater Substrate Composition

The tailwater habitat survey enabled us to document several important larger-scale, longitudinal patterns in macrohabitat. Upstream, the frequency and length of riffle habitat was greater, whereas downstream, pools tended to be longer and more frequent. In total we identified 84 pool/run and riffle complexes. Width of the tailwater varied little throughout the 24 km (mean = 28.0 m, SE = 0.4, n = 232) with the exception of a small number of island braids (< 10 m wide) and a few wider riffles and pools (up to 52 m wide). Maximum depth in pools and riffles averaged 1.1 m and 0.6 m, while ranging from 0.5 to 2.9 m and 0.2 to 1.3 m respectively (n = 84 and 82).

Substrate composition was correlated (P = 0.00) with distance from the dam (Figure 1.11 and Table 1.4). Upstream reaches < 3 km downstream from the dam were composed predominantly of cobble, boulder, and bedrock (up to 80% bottom coverage), whereas pebble and gravel substrates covered a higher percentage of the river bottom at 3 - 17 km (40%) (Figure

1.11). Sand and silt (< 2 mm) comprised an increasingly significant percentage of substrates > 12 km downstream of Philpott dam (20-50%). Free standing boulders and wood debris were infrequent within 4 km of the dam. In general, boulders and wood were most abundant in middle reaches of the tailwater (4 - 17 km downstream), with a peak downstream in the vicinity of an island complex (22 km). Refuse (tires and other trash) was clearly higher in and downstream of Bassett.

We emphasize that trends in macrohabitat configuration, substrate composition, and structure were summarized across 24 km of river, and there is substantial variation at lower spatial scales. However, trends in substrate composition are consistent with the combined impacts of hydropeaking and influx of fine sediment from tributaries that has apparently resulted in a downstream gradient from larger to smaller sized material in the tailwater.

Substrate Permeability

Mean substrate permeability was greater within the redd egg-pocket than the undisturbed substrate beside the redd for 29 of 30 redds and significantly greater (based on 95% confidence intervals) for 26 of 30 redds (Figure 1.12). For the tailwater, permeability was significantly different between the redd pocket versus beside the redd ($P = 0.00$). Likewise, Porter (1985) and Chapman (1988) found that permeability was significantly greater in redds than at non-redd sites. Substrate permeability in the Dam, Special Regulations, Bassett, and Koehler reach averaged 5,203, 3,851, 4,642, and 4,042 cm/hr in the redd egg pocket, and 2,795, 2,389, 1,397, and 2,012 cm/hr ($n = 9, 5, 7, 9$) beside the redd respectively (Figure 1.12). Permeability in the redd pocket as well as beside the redd was not significantly different among reaches ($P = 0.59$ & 0.49 respectively). These results detected no longitudinal tailwater effect on substrate permeability; likely a result of the trout selecting suitable spawning areas and our sampling of only those redd areas. Additionally, these results demonstrate that the redd modifies the substrate, in-turn enhancing permeability for increased egg survival.

Other studies have found higher mean permeability than we measured in the Smith River. Porter (1985) found permeability within brown trout redds averaged 9,100 cm/hr and in non-redd sites 8,100 cm/hr in two 3rd order North Carolina streams. Chapman (1988) measured a median permeability of 10,500 cm/hr in chinook salmon redds in the Columbia River. In pool/tail sites of 11 tributaries in the Garcia river watershed of California, substrate unmodified by spawning had an average permeability of 2,919 cm/hr (McBain and Trush, 2000). An estimate of egg survival to emergence resulting from redd permeability can be made based on chinook and coho salmon survival versus permeability (McBain and Trush, 2000). Using the predictive model presented by McBain and Trush (2000);

Percent Survival = $14.615\text{Ln}(\text{Permeability}) - 81.132$
results in 44%, 40%, 42%, and 40% survival of age-0 salmonids based on mean permeability in redds within the Dam, Special Regulations, Bassett, and Koehler reaches of the Smith River, respectively. A positive correlation between salmonid survival and permeability was also observed by Kondou et al. (2001).

Fine Sediment Intrusion

Fine sediment intrusion within Vibert boxes increased significantly ($P = 0.01$) with distance downstream of Philpott dam (Table 1.5), which parallels the trend observed in bottom coverage of sand and silt < 2 mm (Figure 1.11). This trend is also related to substrate permeability, which was measured at 3 locations (10 replicates at each location) within the area

where Vibert boxes were buried at the 4.2, 13.1, and 22.8 km site. The mean permeability at those sites was 4,453, 790, and 794 cm/hr, which translates to an estimated 42%, 16%, and 16% survival of age-0 salmonids respectively (McBain and Trush, 2000).

The trends of fine sediment intrusion, bottom coverage, and substrate permeability also correspond with age-0 trout recruitment, which is greatest upstream where there is the least fine sediment and greatest permeability. Interestingly, there was no significant trend of increased intrusion over time ($P = 0.53$, Two-way ANOVA). The lack of increased intrusion over time 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). 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.

Spawning Patterns and Characteristics

Spawning activity was observed via the development of redds (i.e. spawning nests). Redds were constructed in pebble and gravel material, approximately 15-32 mm in the 5 cm strata and 9-15 mm in the 5-10 cm strata. This size range falls within the suitable range (2-64 mm) for spawning brown trout when not highly embedded with fine particles (Kondolf, 2000). Redds were present throughout the tailwater from 0.5-23.7 km, however the majority of redds occurred 3-8 km and 12-17 km below Philpott dam (Figure 1.13). This portion of the tailwater (3-17 km) has higher average percent bottom coverage of gravel and pebble (9% and 25%), while having lower boulder/bedrock (16%) and sand/silt coverage (11%) (Figure 1.11). Substrate composition is correlated with distance from the dam; upstream the Dam reach is dominated by boulder/bedrock (41%) and cobble (36%) and downstream the Koehler reach by sand/silt (32%) (Table 1.4). Such trends are consistent with the combined impacts of hydropeaking and influx of fine sediment from tributaries that has apparently resulted in a downstream gradient from larger to smaller sized material in the tailwater.

Throughout the tailwater, redds were observed in limited and isolated areas of suitable substrate. In 2002 and 2003, the same person surveyed several of the same sites allowing the observation that redds were constructed in many of the exact same locations within the channel each year. Areas of suitable spawning gravel were often present at pool/riffle transitions, side channels behind islands, and gravel deposition areas near boulders. Such areas, especially pool/riffle transitions, presumably provide upwelling and downwelling currents that are beneficial to egg and alevin survival (Kondolf, 2000). Five areas where spawning was observed were selected for annual monitoring; 4.2, 6.2, 13.1, 14.9, and 22.8 km. These spawning sites are also the locations of additional study; redd substrate size distribution, Vibert box sediment intrusion, intragravel substrate permeability, and age-0 emergence/growth sampling.

During the November and December spawning season we observed a total of 189, 3, 113, and 66 redds in 2000, 2001, 2002, and 2003 respectively. For these years we surveyed 62% (15 km), 13% (3 km), 33% (8 km), and 13% (3 km) of the tailwater. The total number of redds observed corresponds to survey effort and timing. For example, in year 2000 when 189 redds were observed, the greatest distance was surveyed (15 km). The number of days spent surveying was similar between 2000 and 2002 (12 and 14 days) (Table 1.6), however a portion of 8 survey days in 2002 were spent conducting redd permeability sampling which reduced the survey distance. Surveys in 2000, which determined the five successful spawning areas that enabled

redd surveys in subsequent years to focus on those areas, further reduced the sampling distance during 2001-2003. For 2001 and 2002, five and six days were spent surveying for redds and 13% of the tailwater was surveyed in both years. However, in 2001 far fewer redds were observed (3 versus 66) most likely as a result of initiating surveys too late in the spawning season. In 2001 redd surveys were initiated based on dates of peak spawning occurrence the previous year. Different weather conditions in 2001, which affected water temperatures that cue spawning, likely resulted in an earlier peak spawning period causing our surveys to miss the majority of the spawning (Figure 1.14). The hypothesis that construction of redds occurred before the surveys is based on angler observations, emergence model predictions, and successful recruitment the following spring.

Peak redd development was observed when water temperatures =9°C occurred in November/December of 2000, 2002, and 2003. The =9°C water temperature at which spawning appears to be cued in the Smith River, is within the temperature range occurring during brown trout spawning in other rivers (Raleigh, 1986; Amour, 1994; Bettoli et al., 1999). As time progressed, spawning was detected progressively further upstream in accordance with declining water temperatures. During the spawning season (November and December) water temperature released from the dam is typically warmer than ambient conditions, thus temperatures cool as water travels downstream. Therefore, cooler water temperatures occur downstream and in some years, temperatures =9°C occur downstream before they do upstream (Figure 1.14). Whether water temperatures reach 9°C in early or late November varied between study years and depends on meteorological conditions (Figure 1.14). The trend of redd development downstream prior to upstream as a result of cooling water temperatures, was observed in 2000 and 2002 when a greater number of surveys and redds occurred. It is probable this trend occurred in 2003 based on temperature data (Figure 1.14), however frequent peakflows and turbid conditions prevented surveying during the 3rd and 4th week of November. Thus, the ability to conduct enough surveys to observe trends is not always possible given meteorological conditions which influence dam release regimes.

At two of the spawning sites (6.2 km and 14.9 km), numbers of redds varied greatly in 2003 from those in 2000 and 2002 due to localized changes to the substrate (Table 1.6). Each of these sites was selected (with fall 2000 surveys) due to their high redd densities (relative to nearby sites) (Figure 1.13), which is influenced by suitable pebble and gravel substrate. In 2003 we observed the quality of the spawning substrate had declined at 6.2 km due to riverweed growth, potentially resulting from increased dam release water temperature (Figure 1.7), and at 14.9 km due to sedimentation.

Redd dimensions, water depths, and water velocities were measured during the 2000 (n = 136) and 2002 (n = 44) surveys. No redd characteristic data from 2001 was analyzed due to only 3 observed redds. In 2000 and 2002, the average redd had a pit with length and width dimensions of 56 x 54 cm and 32 x 34 cm and tailspill dimensions of 74 x 43 cm and 72 x 40 cm respectively. Mean depth at the front edge of the pit averaged 27 and 30 cm, middle of the pit 32 and 34 cm, top of the tailspill 24 and 27 cm, and downstream edge of the tailspill 24 and 24 cm in 2000 and 2002 respectively. Mean velocities on the river bottom in 2000 and 2002, at the front of the pit were 25 and 29 cm/s, middle of the pit 17 and 19 cm/s, top of the tailspill 38 and 38 cm/s, and downstream edge of the tailspill 35 and 41 cm/s respectively. Average mean column velocity measured over the middle of the pit was 42 and 43 cm/s in 2000 and 2002 respectively. The following redd characteristics (data throughout the tailwater combined) were statistically different between 2000 and 2002: pit length and width, velocity at the front of the

pit, and velocity at the downstream edge of the tailspill ($P = 0.00-0.01$). Different observers collected redd characteristic data in 2000 and 2002, and thus is a possible cause of the differences between years.

Redd dimensions, depths, and velocity data measured in 2000 had significant differences between reaches (i.e., Dam, Special Regulations, Bassett, Koehler) ($P = 0.00-0.02$). The Koehler and/or Bassett reach had larger redd dimensions and greater depths and velocities than the Special Regulations and/or Dam reach. This positive trend between dimension, depth, and velocity with distance downstream of Philpott dam, suggests that redds in downstream reaches tended to be larger and constructed in deeper, faster water. This trend is in correspondence with the larger brown trout present downstream compared to upstream (see Job 2). Curiously, there were no significant longitudinal trends in the redd dimension and depth data measured in 2002 ($P = 0.12-0.92$). There were significant longitudinal trends in the 2002 velocity data ($P = 0.00-0.01$), with the exception of velocity in the middle of the pit ($P = 0.91$), where, typically, the Koehler reach had greater baseflow velocities than the dam reach.

Age-0 Emergence and Growth

Emergence Trends - Emergence of age-0 brown trout in the Smith River tailwater occurs from February to April. In 2001, emergence was first observed in mid March (12-26th). In 2002, emergence was earlier; first observed on February 26th. And in 2003, emergence was later, April 2-29th. The timing of emergence is linked to the water temperature during the incubation period (Crisp, 1981; 1988). Hatching occurs after 444 degree days (i.e. sum of daily water temperatures) and emergence after another 408 degree days (Elliot, 1994). Degree days calculated for the Smith River demonstrate that 2002 had the most degree days causing the early emergence, 2003 had the least degree days causing late emergence, and 2001 was in between (Figure 1.15). Emergence occurring as early as February has also been observed in other tailwaters (Pender and Kwak, 2002).

A longitudinal trend of emergence occurring upstream shortly before emergence downstream was observed during all years (2001, 2002, and 2003) that temporal surveys were conducted. A visual survey first detected age-0 in 2001 at 4.1 and 13.1 km (March 12-16th), in 2002 at 4.2, 13.1, and 14.9 km (February 26th), and in 2003 an electrofishing survey first detected age-0 at 4.2 and 6.2 km (April 2nd) (assessed sites were 4.2, 6.2, 13.1, 14.9, and 22.8 km). Age-0 were detected at the remaining sites on subsequent surveys. Despite spawning (i.e. redd development) occurring first downstream, the opposite trend was apparent for emergence. This is likely a result of higher winter water temperatures released from Philpott dam resulting in greater degree days upstream compared to those downstream during the incubation time period (Brooker, 1981) (Figure 1.15). Despite earlier emergence upstream, age-0 downstream had higher growth rates, thus achieving greater total lengths.

Age-0 Length - Mean total lengths of age-0 were statistically different among sites ($P = 0.00$) and sample years ($P = 0.00$), and there was no interaction between sites and years ($P = 0.13$, Two-way ANOVA). Age-0 surveyed in May of 2000 through 2004 had significantly smaller total lengths at the upstream site 4.2 km than downstream sites 6.2, 13.1, 14.9, and 22.8 km (with the exception of 6.2 and 22.8 km in 2003) (Table 1.7). Site 6.2 km also had significantly smaller lengths than sites 13.1, 14.9, and 22.8 km in 2000 and 2001, than sites 14.9 and 22.8 km in 2002, and than sites 13.1 and 14.9 km in 2003 (Table 1.7). Mean lengths in May 2003 were statistically smaller than those in all other study years at all sites with the exception of 22.8 km in 2001 (Table 1.8); a result of later emergence (April) in 2003 than in 2001 and 2002.

Greater growth downstream is likely due to warmer water temperatures and thus, more degree days during the post emergence time period (Figures 1.3 and 1.15). Conversely, abundance of age-0 was greater upstream than downstream.

Age-0 Abundance - Age-0 abundance was significantly greater upstream than downstream, with the exception of the first ~ 3 km below Philpott dam where boulder and bedrock substrate dominate. Abundance of age-0 in May was significantly higher at 4.2 km than all other sites (6.2, 13.1, 14.9, and 22.8 km) during 2001-2004 (Figure 1.16). Sites 6.2 and 13.1 km also had significantly higher abundance than the downstream sites 14.3 and 22.8 km in 2001, 2002, and 2004, and the 22.8 km site in 2003. Higher abundance upstream is likely a result of higher survival while eggs develop in the gravel due to less fine sediment intrusion and higher intragravel permeability. Additionally, abundance differed with year as a result of flow conditions. Abundance was significantly lower in 2003 and 2004, than 2001 and 2002, at all sites except 4.2 km in 2003 (Figure 1.16). Overall, abundance declined with increases in peakflow occurrence and magnitude (Figure 1.17) (Anderson and Nehring, 1985).

The occurrence, magnitude, and duration of peakflow releases was greatest in 2003 and 2004 (Table 1.2) 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 period of February through April the average occurrence of peakflow in 2003 and 2004 was 39% and 21% respectively, compared to 7% in 2000, 6% in 2001, and 5% in 2002 (Table 1.2). The mean peakflow magnitude was similar for 2000, 2001, 2003, and 2004 at 29.55, 28.82, 36.28, and 34.55 cms respectively, but lower in 2002 at 16.18 cms. In 2001 and 2002, successful recruitment (i.e. high age-0 abundance) corresponded with a low occurrence and magnitude of peakflow (Figures 1.17 and 1.18). The opposite occurred in 2003-2004 when poor recruitment corresponded with high occurrence and magnitude of peakflow.

Lower levels of age-0 recruitment occurring in 2003 and 2004 were due primarily to flow rather than water temperature, fine sediment intrusion, or DO which can affect egg survival (Chapman, 1988; Pender, 1998; McBain and Trush, 2000). Water temperature can be ruled out because degree days during the egg incubation period (~ November-March) were not uniformly low for poor recruitment and high for successful recruitment. The 2000/2001 and 2002/2003 incubation periods, which were years with high and low recruitment respectively, had lower degrees days than 2001/2002 which had high recruitment (Figure 1.15). Ruling out fine sediment intrusion and DO is less certain because these parameters were only measured in 2002. However, there were no observable large scale changes to the river substrate composition that would have lessened fine sediment intrusion or lowered DO.

Spawner Biomass - Differences in age-0 abundance between years is unlikely to have been affected by the population of spawning brown trout due to its constancy. The population of spawning brown trout in October, estimated as spawner biomass (g/m) where a spawning trout in the Smith River is characterized as >203 mm (see Job 2), was relatively consistent between years (Figure 1.19). The number of spawning size trout also followed the same pattern as that of biomass. Biomass was not statistically different among study years for the Dam, Special Regulations, Bassett, and Koehler reach ($P = 0.14-0.89$). Among reaches, biomass in the Special Regulations reach was statistically greater than the Koehler reach in 2000 and 2002 ($P = 0.02, 0.03$) and greater than all the reaches in 2001 ($P = 0.01$). Due to sampling only six of the 12 sites in 2003 this statistical analysis could not be performed for this study year.

The substantially lower biomass (62.3 g/m) of spawning trout in October 2003 at 6.2 km, in comparison to 2000-2002 (Figure 1.19), was followed by a mean abundance of 8.7 age-0/75

m² in May 2004 (Figure 1.16). Whereas the highest spawner biomass (148.9 g/m) at 6.2 km, occurring in 2002 (Figure 1.19), was followed by a mean abundance of 13.3 age-0/75 m² in May 2003 (Figure 1.16) which is similar and not statistically different (95% CI) to the 8.7 age-0/75 m² abundance in 2004. This one occurrence of a large variation in spawner biomass, but relatively small change in age-0 abundance after the spawning season, indicates the importance that other factors (primarily flow, but also water temperature, fine sediment intrusion, dissolved oxygen, and density dependence) have on successful recruitment in the Smith River.

The longitudinal trend of biomass and number of spawning size trout throughout the tailwater corresponds with the trend of age-0 abundance and redd development. The highest redd density, age-0 abundance, and spawner biomass occurred from ~ 3-10 km within the Dam and Special Regulations reaches (Figures 1.13, 1.16, and 1.19).

Age-0 Growth– Growth rates of age-0 brown trout were slower upstream and faster downstream (Figure 1.20), thus larger age-0 are present downstream (Table 1.7). Between May and October of 2000-2003 age-0 grew at an average rate of 0.35 mm/day (range 0.30-0.45) upstream (0.0-5.3 km) and 0.44 mm/day (range 0.36-0.50) downstream (5.3-24.3 km). However, in 2003, growth upstream (Dam reach, 0.0-5.3 km) was 0.12, 0.13, and 0.15 mm/day greater than it had been in 2000, 2001, and 2002 respectively (Figure 1.20). Additionally, the 2003 growth rates upstream (Dam and Special Regulations reaches, 0.0-10.0 km) were similar or greater than growth rates downstream (Bassett and Koehler reaches, 15.9-24.3 km). This reversal in the trend was likely due to warmer water temperatures upstream in May-October 2003 (Figure 1.7 and 1.15). A positive trend between age-0 growth rate and degree days in the Dam and Special Regulations reaches (negative trend in the Bassett or Koehler reaches) depicts that warmer temperatures increase growth in the upstream reaches (Figure 1.21). The greater growth rate of age-0 in 2003 in the Dam reach is seen by age-0 starting out smaller (May) due to later emergence, yet ending up larger by October compared to 2000-2002 (Figure 1.22). The 2003 abundance of age-0 upstream (4.2 km) did not change from that in 2001 and 2002, which limits density dependence as the primary cause (Figure 1.16). Yet May-October mean water temperature released from Philpott dam in 2003 was 4.4, 4.8, and 4.9°C greater than that released in 2000, 2001, and 2002 respectively. This resulted in May-October 2003 mean water temperatures (12.8°C at 0.7km and 13.7°C at 5.1 km) within the brown trout optimal growth range (12-19°C), whereas mean temperature from May-October 2000-2002 never exceeded 7.9°C and 10.2°C at 0.7 and 5.1 km respectively. These warmer release temperatures most likely resulted from a lack of strong stratification in Philpott reservoir caused by 59%, 37%, and 36% more rain during 2003 than in 2000, 2001, and 2002 respectively (percentage calculated for months May through October from rain gage data at Philpott dam). The large amount of rain in 2003 also resulted in a high occurrence of long duration peakflow releases (Table 1.2). The greatest abundance and slowest growth rates of age-0 occurred in 2002 (Figures 1.16 and 1.20). The decline in growth rates in 2002 are likely a result of density dependence, which appears to occur throughout the tailwater as indicated by negative trends in growth rate with increased abundance (Figure 1.23). Also in 2002, discharge was less in occurrence, duration, and particularly magnitude than the other study years (Table 1.2). Given that the flow conditions should have caused less restriction on growth in 2002, which likely resulted in the increased abundance, leaves density dependence as a possible cause of the slightly lower growth rates.

River-Edge Habitat Preference - To verify that age-0 prefer river-edge habitat and confirm the legitimacy of electrofishing only within 3 m of the riverbank, we electrofished the middle of the channel parallel to the three 25 m edge sections sampled at site 4.2 km in 2003.

Site 4.2 km was chosen because it consistently produces the most age-0. Only four age-0 trout were found in the middle of the channel and all were under the same rock; compared to 47 age-0 found in the paralleling edge habitat. In the edge habitat for the other two 25 m sections, 109 and 44 age-0 were found, 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, thus we electrofished over known mid-channel redd locations at this site, but saw no other age-0.

Habitat Characteristics - Habitat characteristics measured within the reaches sampled for age-0 brown trout at the five routinely monitored spawning sites provide a descriptive site characterization (Table 1.9). Measured parameters, including meso-habitat type, were variable among reaches within a site. Site averaged embeddedness was least at 4.2 km and 14.1 km (25-50%), and greatest at 22.8 km (75-100%). Longitudinal trends were minimal to nonexistent for depth, velocity, bank angle, and overstory density. Site averaged water depths and mean column velocity ranged from 15-26 cm and 0.05-0.21 m/s respectively. Bank angles and overstory density averaged by site ranged from 121-159° and 47-91% respectively. Site averaged D50 and D90 values were largest upstream (4.2 km) and smallest downstream (22.8 km) (Table 1.9).

Macroinvertebrates

Trends in macroinvertebrate wet weight, density, and Ephemeroptera-Plecoptera-Trichoptera (EPT) density were not common among the sample years (2000-2003) or longitudinally among reaches (Newcomb et al., 2001). Wet weight was significantly greater in July 2002 than in 2000 and 2001 ($P = 0.02$) for the tailwater (i.e. all sites combined). The 2002 increase in wet weight was predominantly in the Koehler reach, where wet weight was significantly greater ($P = 0.01$) than in 2000, 2001, and 2003 (Table 1.10). The only other statistical difference among years was for density within the tailwater ($P = 0.04$), however a Tukey-Kramer HSD test revealed no significant differences between years. Significant longitudinal differences among reaches (i.e. Dam reach downstream to the Koehler reach) were only present for EPT density in 2000 ($P = 0.01$) and 2002 ($P = 0.00$). More specifically, in 2000 EPT density in the Special Regulations reach was different from the Dam and Bassett reach, and in 2002 the Bassett reach was different from all three other reaches. Isolated peaks in density at specific sites suggest localized channel characteristics are of higher quality for invertebrate colonization (Newcomb et al., 2001).

Despite minimal patterns of macroinvertebrate weight and density within the tailwater and among years, the overall abundance in the Smith River tailwater is lower than typical for a stream of this size in Virginia. 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 (Newcomb et al., 2001). Throughout the tailwater, invertebrate densities are 2-3 times lower than those in unregulated Virginia rivers of similar size. Unregulated rivers typically have 800-1,000 invertebrates/m², whereas the majority of sites in the Smith River have less than the poor food grade classification number of 538 organisms/m² (Newcomb et al., 2001).

Conclusions

Discharge Analysis

- The flow regimes during the years of study (2000-2003) deviated from that in previous years (1991-1999). In 2003, peakflows occurred twice that of the 1991-2002 average. In 2002, discharge was less in occurrence, duration, and particularly magnitude than the other study years. The wide range of flow conditions experienced during the study have provided us with a natural experiment for comparing measurements under different flow regimes. Overall, the dam changed the daily flow regime to one with baseflows and peakflows that are now of less magnitude, shorter duration, and more frequent than pre-dam flows.

Water temperature

- Water temperature in the Smith River is directly affected by the hypolimnetic release from Philpott dam and is a key factor to cue spawning, control incubation of eggs, and determine age-0 growth. In the upstream reaches temperature averages 8°C and rarely exceeds 12°C where 12-19°C is the optimal temperature range for brown trout growth. In summer, peakflows cause temperatures to decline rapidly (up to 7°C/hr) from 2.7-10.2 km below the dam.

Water Quality

- Water quality parameters of interest in the tailwater are dissolved oxygen (DO) and alkalinity. Dissolved oxygen varied seasonally where the lowest DO (as low as 1.6 mg/l) occurred upstream in fall during peakflow prior to lake turn-over typically in November. However, the DO content in spawning nests was not limiting (11 mg/l) when measured in November.
- Alkalinity has been tied closely to trout production and the low alkalinity in the Smith River may impose limitations on brown trout reproductive success and growth. However, water fertility is not easily mitigated, therefore management of habitat, discharge, and/or water temperature are more plausible for enhancing productivity.

Tailwater Substrate Composition

- Bottom coverage in the upstream reaches is predominantly larger rocks and bedrock (80%), whereas pebble and gravel substrates cover a higher percentage in the middle sections of the tailwater (40%). Downstream, sand and silt make-up an increasingly significant percentage of substrates (20-50%). Trends in substrate composition are consistent with the combined impacts of hydropeaking and influx of fine sediment from tributaries that has resulted in a downstream gradient from larger to smaller sized material in the tailwater.

Substrate Permeability

- Permeability in redds was significantly greater than in surrounding gravels due to substrate modifications via redd construction. Permeability in redds and surrounding substrate did not differ significantly from upstream to downstream, likely from trout selecting suitable spawning areas. In the Smith River, permeability was lower than that reported in studies of other rivers.

Fine Sediment Intrusion

- Fine sediment intrusion within Vibert boxes increased significantly with distance downstream of Philpott dam, which parallels the trend observed in bottom coverage of sand and silt. Survival of salmonid embryos declines as the percentage of fines increases above 10-20% and downstream sites had samples with >10 % fines. The trends of fine sediment intrusion, bottom coverage, and substrate permeability also correspond with age-0 trout abundance, which was greatest upstream where there is the least fine sediment and greatest permeability.

Spawning Patterns and Characteristics

- The development of spawning nests (i.e. redds) occurred when water temperatures fell below 9°C, typically in November. The majority of redds occurred 3-8 km and 12-17 km below Philpott dam where there is a higher percent bottom coverage of gravel and pebble substrates (9% and 25%), while having lower boulder/bedrock (16%) and sand/silt coverage (11%). Emergence of age-0 brown trout occurred from February to April and the timing of emergence depended on the water temperature during the incubation period.

Age-0 Emergence and Growth

- Spawner biomass did not differ significantly among study years by reach, indicating the importance that other factors (primarily flow) have on successful recruitment in the Smith River.
- The longitudinal trends of spawner biomass in the tailwater matches the trends of age-0 abundance and redd development.
- The highest redd density, age-0 abundance, and spawner biomass all occurred from ~ 3-10 km within the Dam and Special Regulations reaches.
- Age-0 abundance was significantly greater upstream probably resulting from higher survival while eggs developed in the gravel due to less fine sediment intrusion and higher intragravel permeability. Additionally, abundance differed significantly with year as a result of flow conditions where abundance declined with increases in peakflow occurrence and magnitude. In 2001 and 2002, successful recruitment (i.e. high age-0 abundance) corresponded with a low occurrence and magnitude of peakflow. The opposite occurred in 2003 and 2004 when poor recruitment corresponded with high occurrence and magnitude of peakflow.
- Age-0 were significantly smaller in length upstream compared to downstream where age-0 had faster growth rates likely due to warmer water temperatures.
- A positive trend between age-0 growth rate and degree days in the Dam and Special Regulations reaches depicts that warmer temperatures increase growth in the upstream reaches.
- The decline in growth rates in 2002 are likely a result of density dependence, which appears to occur throughout the tailwater as indicated by negative trends in growth rate with increased abundance.

Summary

In the Smith River, peaking flow, water temperature, sedimentation, and permeability are the primary factors influencing spawning and recruitment success. We predict that an improvement in age-0 growth and survival could be achieved via warmer water temperatures upstream, peakflows of less magnitude and occurrence, and a reduction in fine sediments downstream. Age-0 were significantly smaller with slower growth rates upstream where water temperatures rarely reach the optimal growth range and the effects of peakflows are greatest. Age-0 abundance was significantly different with year corresponding to flow conditions where abundance declined with increases in peakflow occurrence and magnitude. Age-0 abundance was significantly greater upstream where fine sediment intrusion was significantly less. The combination of cold water temperatures, rapidly changing flow conditions, and fine sediment deposition are the primary factors affecting rearing habitat for brown trout in the Smith River.

Table 1.1. Characteristics of the representative 'typical' flow regime (based on 1991-2003 discharge data) compared to the flow regimes occurring each year (2000-2003) of this study by season (spring: Mar, Apr, May; summer: Jun, Jul, Aug; fall: Sep, Oct, Nov; winter: Dec, Jan, Feb).

Season	Year	Peakflow (cms)	Baseflow (cms)	Release Duration (hrs)	Release Time (hr)
Spring	Typical	31.3	1.4	6	7 or 21
	2000	26.1	1.4	2	7 or 21
	2001	26.1	1.8	3	7 or 21
	2002	15.0	1.7	1	8 or 21
	2003	36.4	1.8	10	6 or 18
Summer	Typical	31.7	1.5	5	14
	2000	29.6	1.6	2	18
	2001	18.3	1.4	6	12
	2002	14.6	1.6	1	18
	2003	38.1	1.9	11	13
Fall	Typical	27.5	1.5	4	8
	2000	29.4	1.5	1	18
	2001	18.5	1.5	4	7 or 18
	2002	14.9	1.5	1	16
	2003	32.9	2.0	6	18
Winter	Typical	31.2	1.5	5	7
	2000	31.8	1.5	2	7
	2001	29.1	1.8	2	7
	2002	25.1	1.5	3	6 or 18
	2003	24.6	1.5	5	6 or 18

Table 1.2. Discharge statistics evaluating 13 years 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, peakflow was classified as >2.83 cms). Occurrence is the percent time that peakflows occurred (an additional assessment for max-peakflows greater than 28.32 cms is also shown). Magnitude is the average discharge in cubic meters per second (cms) of the peakflow. Duration is the average time in hours per day that peakflow 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	2004
Occurrence (% time > 2.83 cms)														
January	40	27	37	22	30	32	53	11	7	7	6	14	32	22
February	16	10	27	31	17	26	57	49	6	8	6	7	25	23
March	30	20	58	42	19	22	78	39	7	6	6	4	35	13
April	32	32	53	35	9	23	56	38	10	6	6	5	58	31
May	34	28	31	16	15	32	52	28	12	9	20	5	35	25
June	23	55	18	14	16	28	31	18	13	6	20	4	63	19
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	39	
September	18	23	18	21	21	40	19	20	18	6	17	5	22	
October	7	13	14	9	7	40	12	11	7	6	17	5	23	
November	3	25	6	11	4	53	8	10	7	6	19	5	24	
December	4	24	11	10	4	69	7	10	7	6	13	8	31	
Annual Avg.	20	24	26	22	15	34	34	22	12	7	15	6	35	
Water-Yr Avg.		20	28	22	16	22	45	22	12	7	12	9	30	
Occurrence (% time > 28.32 cms)														
January	39	20	36	17	29	31	0	9	6	6	4	9	0	19
February	15	8	26	27	16	25	0	47	6	5	4	0	23	17
March	29	20	57	36	18	16	0	38	6	4	4	0	32	11
April	30	31	52	35	8	22	0	37	9	4	4	0	56	28
May	32	27	30	15	15	24	0	26	2	2	3	0	33	4
June	22	54	17	12	15	27	6	17	12	5	0	0	61	17
July	15	18	17	15	16	15	16	15	15	4	0	0	37	
August	15	17	14	35	15	28	14	14	16	5	0	0	38	
September	6	18	10	5	3	39	6	5	6	4	0	0	20	
October	6	6	4	8	6	0	9	8	4	4	0	0	14	
November	3	25	4	8	3	0	6	9	6	4	4	0	22	
December	3	24	10	9	3	7	6	9	6	4	8	1	29	
Annual Avg.	18	22	23	18	12	19	5	20	8	4	3	1	30	
Water-Yr Avg.		19	26	18	14	20	4	19	9	5	3	2	25	

Table 1.2 continued.

	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004
Magnitude (cms)														
January	35.38	32.42	34.78	31.97	38.25	40.02	18.56	32.91	32.66	31.66	29.25	28.79	18.06	35.48
February	34.06	32.60	34.86	33.37	35.44	34.69	18.41	34.55	34.13	29.85	29.46	18.16	36.24	32.81
March	34.64	34.51	45.32	38.24	35.43	30.32	18.62	35.08	33.09	29.11	28.49	15.81	35.68	34.61
April	34.32	61.04	35.81	35.93	34.30	34.64	18.49	34.88	36.28	29.69	28.51	14.58	36.92	36.25
May	33.87	34.62	35.32	34.73	34.84	30.23	18.64	34.60	21.20	19.61	21.32	14.71	36.64	21.51
June	34.84	40.70	35.16	32.23	35.11	47.55	21.42	34.61	36.45	29.31	18.32	14.67	38.46	36.18
July	33.82	34.42	35.54	33.70	32.01	33.85	33.61	34.77	35.53	29.56	18.42	15.18	38.22	
August	34.74	34.73	35.06	35.71	35.78	35.55	34.52	34.76	29.99	30.01	18.06	13.86	37.48	
September	24.23	31.54	28.98	22.69	21.73	35.73	24.56	23.49	25.31	29.62	17.17	15.41	36.84	
October	32.58	26.36	23.76	34.54	33.43	18.59	31.34	31.70	30.45	29.87	17.60	14.40	25.40	
November	31.62	35.01	29.78	32.44	31.71	18.67	31.48	37.02	34.93	28.72	20.79	14.88	36.40	
December	31.02	35.05	34.64	33.75	31.17	23.40	33.03	36.22	33.79	28.48	28.34	19.36	36.80	
Annual Avg.	33.61	37.74	36.16	33.91	33.57	29.95	21.18	33.84	31.24	28.50	20.86	18.37	35.02	
Water-Yr Avg.		35.98	34.77	32.23	33.64	34.91	22.29	32.96	32.47	29.80	24.67	18.16	30.26	
Duration (hrs)														
January	10	6	9	5	7	8	13	3	2	2	2	3	8	5
February	4	2	7	8	4	6	14	12	2	2	1	2	6	6
March	7	5	14	10	5	5	19	9	2	2	2	1	9	3
April	8	8	13	9	2	5	14	9	2	2	2	1	14	6
May	8	7	7	4	4	8	12	7	3	2	5	1	8	1
June	5	13	4	3	4	7	7	4	3	2	5	1	15	4
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	9	
September	4	6	4	5	5	10	4	5	4	2	4	1	5	
October	2	3	3	2	2	10	3	3	2	1	4	1	6	
November	1	6	1	3	1	13	2	2	2	2	5	1	6	
December	1	6	3	2	1	17	2	2	2	2	3	2	7	
Annual Avg.	5	6	6	5	4	8	8	5	3	2	4	1	8	
Water-Yr Avg.		5	7	5	4	5	11	5	3	2	3	2	7	

Table 1.3. Selected non-parametric output from the Indicators of Hydrologic Alteration model assessment of pre-dam (1946-1952) and post-dam (1954-2002) discharge. Discharge (cms) parameters include annual, monthly, and one and three day minimums and maximums. Low and high pulses are presented as counts (days) and duration (days). Rise and fall rates* are the means of all positive and negative differences between consecutive daily values, respectively. Number of reversals (days) is a count of hydrologic reversals.

Parameter	Pre-dam	Post-dam	Positive or Negative Change
Discharge (cms)			
Mean Annual Flow	8.30	7.81	-
October	6.59	4.74	-
November	5.73	4.72	-
December	6.21	6.01	-
January	6.75	7.62	+
February	11.36	8.85	-
March	9.77	8.09	-
April	10.42	8.99	-
May	9.60	8.32	-
June	8.23	7.24	-
July	6.33	6.84	+
August	6.17	6.63	+
September	7.13	5.84	-
1-day minimum	2.63	1.10	-
3-day minimum	2.67	1.46	-
1-day maximum	97.41	41.91	-
3-day maximum	61.35	39.64	-
Pulse count (days)			
Low pulse count	14	38	+
High pulse count	17	38	+
Pulse duration (days)			
Low pulse duration	10	4	-
High pulse duration	11	5	-
Rate* and Reversals (days)			
Rise rate	149	140	-
Fall rate	-73	-143	-
Number of reversals	113	174	+

Table 1.4. Percent bottom coverage of boulder/bedrock, cobble, pebble, gravel, and sand/silt within the Dam (n = 31), Special Regulations (n = 37), Bassett (n = 54), and Koehler (n = 46) reaches. Significant differences (ANOVA, alpha = 0.05) among reaches are shown by P-values in bold text. Mean percents in the same row with the same letter are not significantly different (Tukey-Kramer HSD).

	Dam	Spec. Reg.	Bassett	Koehler	P-value
Boulder and Bedrock (>256 mm)	41% a	22% b	12% c	4% cd	0.00
Cobble (64-256 mm)	36% a	21% b	20% bc	9% d	0.00
Pebble (16-64 mm)	15% a	28% b	25% bc	18% cd	0.00
Gravel (2-16 mm)	7% a	10% a	9% a	10% a	0.14
Sand and Silt (<2 mm)	0% a	4% a	17% b	32% c	0.00

Table 1.5. Average weight (g) of sediment <2 mm intruded into vibert boxes at 4.2, 6.2, 13.1, 14.9, and 22.8 km below Philpott dam over 0.5 to 3 months. Significant differences (ANOVA, alpha = 0.05) are shown by P-values in bold text. Mean weights in the same row with the same letter are not significantly different (Tukey-Kramer HSD).

	4.2 km	6.2 km	13.1 km	14.9 km	22.8 km	P-value
0.5 month	36 a	52 a	92 ab	130 b	161 bc	0.00
1 month	34 a	49 a	102 b	113 b	105 b	0.00
1.5 months	38 a	51 a	76 ab	143 b	110 ab	0.01
2 months	47 a	47 a	71 a	83 a	153 b	0.00
3 months	25 a	55 ab	74 ab	110 bc	154 c	0.00

Table 1.6. Redd survey data including the number of redds found (which had not been present during previous surveys) for each date and location (km) that surveys were conducted; in parenthesis is the water temperature (°C) recorded during the survey. In bold text is the overall distance (m) surveyed for each location, total number of redds observed per site, and redd density (# redds/100 m) (some survey distance and redd density data for 2000 was not available due to a change in personnel).

	Distance Downstream of Philpott dam (km)												
	0.5	3.4	4.2	6.2	8.9	11.9	13.1	14.9	16.5	19.5	20.6	22.8	23.7
11/07/00							0	1		1		4	0
11/13/00				0 (8)	0								
11/14/00		1 (10)	0										
11/16/00								3 (8)				6 (7)	
11/20/00								0 (8)	4	0		1 (7)	3
11/28/00		2 (10)	5			3 (8)	7 (7)						
11/29/00				30 (7)	17								
11/30/00			1 (9)	13 (7)									
12/08/00							0 (8)	2 (7)		0		2	0
12/11/00				20 (6)									
12/13/00		18 (7)	24	11 (6)									
12/18/00		3	7 (7)										
Dist. (m)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Tot. # Redds	24	37	74	17	3	7	6	4	1			13	3
Redds/100m	N/A	8.5	3.5	N/A	N/A	1.3	1.5	N/A	N/A			3.3	N/A
11/28/01			0 (11)	0 (11)			0 (12)	0 (12)				0 (12)	
12/04/01			1 (11)	0 (10)									
12/07/01			0 (11)	2 (11)			0 (11)						
12/11/01			0 (11)										
12/21/01			0 (10)	0 (9)			0 (10)	0 (10)					
Dist. (m)			665	873			770	424				277	
Tot. # Redds			1	2			0	0				0	
Redds/100m			0.2	0.2			0.0	0.0				0.0	

Table 1.6 continued. Redd survey data including the number of redds found (which had not been present during previous surveys) for each date and location (km) that surveys were conducted; in parenthesis is the water temperature (°C) recorded during the survey. In bold text is the overall distance (m) surveyed for each location, total number of redds observed per site, and redd density (# redds/100 m).

	Distance Downstream of Philpott dam (km)												
	0.5	3.4	4.2	6.2	8.9	11.9	13.1	14.9	16.5	19.5	20.6	22.8	23.7
10/30/02			0										
11/05/02			0 (8)	0 (9)			0 (9)				0 (9)	0 (9)	
11/06/02			0	0				4 (10)					
11/07/02							0 (9)			3 (10)			
11/14/02							0		2 (9)	1			4 (10)
11/15/02					1 (8)				4 (9)				1 (10)
11/18/02		1 (8)	3 (9)				2 (9)						
11/19/02				0	2 (8)		2 (8)					0	
11/20/02				3 (9)			9	9 (7)					
12/03/02	1	6 (9)	9										
12/06/02				12			2	2				10	
12/12/02	1												
12/19/02			7				5						
12/20/02	2	5											
Dist. (m)	600	290	210	2690	170		820	480	760	770	560	380	170
Tot. # Redds	4	12	19	16	2		20	15	6	4	0	10	5
Redds/100m	0.7	4.1	9.0	0.6	1.2		2.4	3.1	0.8	0.5	0.0	2.6	2.9
10/28/03			0 (15)	0 (15)			0 (13)	0 (13)	0 (13)	0 (13)	0 (14)	0 (13)	0 (14)
11/05/03			0 (16)	0 (17)			0 (16)	0 (16)	0 (16)	0 (17)	0 (17)		
11/12/03			0 (15)	0 (15)			0 (13)	0 (14)	0 (13)	0 (13)	0 (15)	0 (14)	
12/02/03			2 (7)	8 (7)			5 (7)	2 (7)	0	18 (7)		0 (7)	1 (8)
12/07/03			0 (9)	8 (8)	1 (5)		1 (5)	1		4	0 (5)		1 (5)
12/21/03			7 (5)	5 (6)	0 (5)			2 (4)					
Dist. (m)		220	300	250			228	592	235	235	200	100	380
Tot. # Redds		9	21	1			6	5	0	22	0	0	1
Redds/100m		4.1	7.0	0.4			2.6	0.8	0.0	9.4	0.0	0.0	0.3

Table 1.7. Mean total length (mm) of age-0 brown trout sampled in May of 2000, 2001, 2002, 2003, and 2004 at 4.2, 6.2, 13.1, 14.9, and 22.8 km downstream of Philpott dam. Significant differences (ANOVA, alpha = 0.05) among sites are shown by P-values in bold text. Mean lengths in the same row with same letter are not significantly different (Tukey-Kramer HSD).

	4.2 km	6.2 km	13.1 km	14.9 km	22.8 km	P-value
2000	32.1 a	35.6 b	41.7 c	46.9 d	47.4 cd	0.00
2001	28.5 a	35.7 b	38.3 c	42.4 d	41.0 cd	0.00
2002	36.7 a	40.3 b	39.2 b	46.5 c	46.3 c	0.00
2003	24.9 a	26.4 a	30.5 b	32.1 b	27.0 ab	0.00
2004	29.7 a	47.8 b	43.2 c	45.0 bc	43.5 bc	0.00

Table 1.8. Mean total length (mm) of age-0 brown trout sampled in May of 2000, 2001, 2002, 2003, and 2004 at 4.2, 6.2, 13.1, 14.9, and 22.8 km downstream of Philpott dam. Significant differences (ANOVA, alpha = 0.05) among years are shown by P-values in bold text. Mean lengths in the same row with same letter are not significantly different (Tukey-Kramer HSD).

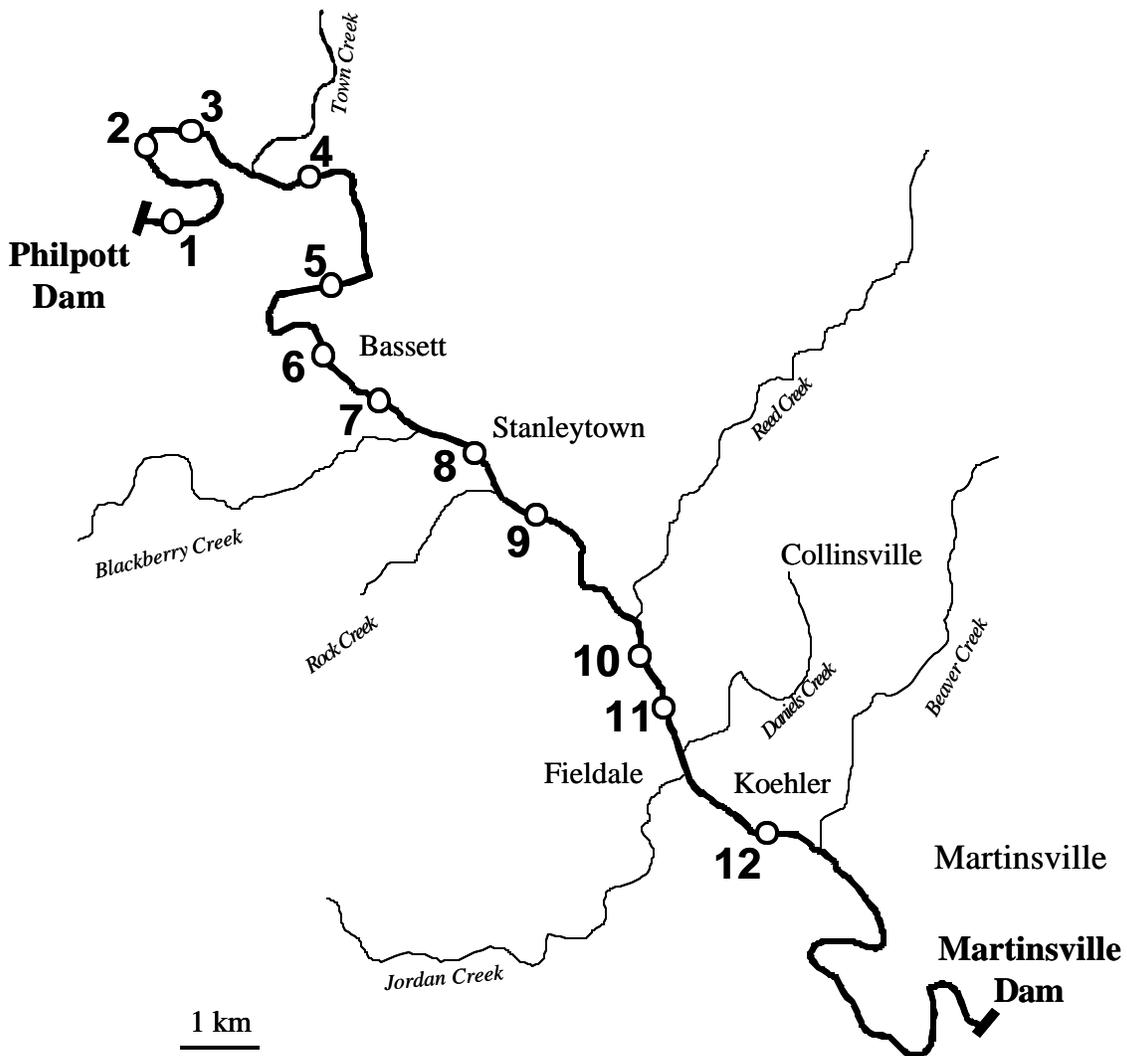
	2000	2001	2002	2003	2004	P-value
4.2 km	32.1 a	28.5 b	36.7 c	24.9 d	29.7 be	0.00
6.2 km	35.6 a	35.7 a	40.3 b	26.4 c	47.8 d	0.00
13.1 km	41.7 a	38.3 a	39.2 a	30.5 b	43.2 c	0.00
14.9 km	46.9 a	42.4 b	46.5 a	32.1 c	45.0 ab	0.00
22.8 km	47.4 ab	41.0 ad	46.3 bc	27.0 d	43.5 ac	0.00

Table 1.9. Habitat characteristics of the three 25 m reaches (0-25, 75-100, 150-175 m) within 3 m of the riverbank for the five monitored spawning sites (4.2, 6.0, 12.9, 14.1, and 22.8 km).

	Embeddedness (%)	Depth (cm)	Demersal Velocity (m/s)	Mean Column Velocity (m/s)	Bank angle (deg.)	Overstory Density (%)	D50 (mm)	D90 (mm)	Meso-Habitat	GPS
4.2 km										
0-25 m	50-75	15	0.09	0.14	117	65	11	21	Run	36 47.629; 80 01.011
75-100 m	0-25	17	0.03	0.07	149	17	50	288	Run	36 47.612; 80 00.969
150-175 m	25-50	14	0.03	0.05	144	60	30	65	Riffle	36 47.586; 80 00.941
Average	25-50	15	0.05	0.09	137	47	30	125		
6.0 km										
0-25 m	25-50	20	0.02	0.05	141	87	38	118	Run	36 47.215; 79 59.875
75-100 m	25-50	10	0.13	0.18	143	87	18	48	Run	36 47.223; 79 59.827
150-175 m	75-100	26	0.01	0.05	107	99	15	60	Run	36 47.238; 79 59.777
Average	50-75	19	0.05	0.10	130	91	24	75		
12.9 km										
0-25 m	50-75	29	0.04	0.06	152	89	8	22	Run	36 45.336; 79 59.224
75-100 m	25-50	5	0.03	0.04	164	64	15	32	Pool	36 45.317; 79 59.179
150-175 m	25-50	15	0.03	0.06	160	93	29	75	Riffle	36 45.275; 79 59.150
Average	25-50	16	0.03	0.05	159	82	17	43		
14.1 km										
0-25 m	75-100	31	0.03	0.07	121	53	0	6	Run	36 44.906; 79 58.223
75-100 m	25-50	12	0.02	0.05	156	68	35	162	Riffle & Run	36 44.881; 79 58.188
150-175 m	25-50	13	0.39	0.50	141	63	29	81	Riffle	36 44.840; 79 58.159
Average	50-75	18	0.15	0.21	140	61	21	83		
22.8 km										
0-25 m	75-100	19	0.02	0.04	106	79	1	4	Run	36 41.634; 79 55.521
75-100 m	50-75	29	0.16	0.33	104	98	9	19	Run	36 41.609; 79 55.479
150-175 m	75-100	32	0.06	0.20	152	79	0	0	Run	36 41.593; 79 55.436
Average	75-100	26	0.08	0.19	121	85	3	8		

Table 1.10. Macroinvertebrate wet weight (g/m²), density (#/m²), and Ephemeroptera-Plecoptera-Trichoptera (EPT) density (# EPT/m²) at twelve sites (0.5-23.7 km) during July 2000-2003. Site data is averaged among reaches for the dam reach (0.0-5.3 km), special regulations reach (spec. reg.; 5.3-10.0 km), Bassett reach (10.0-15.9 km), and Koehler reach (15.9-23.7 km). Significant differences among years and among reaches (ANOVA, alpha = 0.05) are shown by P-values in bold text.

	Wet weight (g/m ²)					Density (#/m ²)					EPT Density (# EPT/m ²)				
	2000	2001	2002	2003	<i>P</i>	2000	2001	2002	2003	<i>P</i>	2000	2001	2002	2003	<i>P</i>
0.5 km	1.20	0.29	1.02	1.96		404	927	1653	2003		2	5	15	7	
3.4 km	1.20	0.22	0.32	0.52		273	282	225	593		85	32	13	57	
4.2 km	0.70	0.49	0.26	1.29		158	407	600	965		64	67	13	85	
6.2 km	3.60	0.99	1.83	1.44		522	257	623	410		402	73	132	128	
8.9 km	1.60	0.56	1.67	4.11		408	458	377	217		338	273	168	147	
11.9 km	1.40	0.85	3.28	12.06		247	382	662	565		163	153	387	222	
13.1 km	0.50	1.45	14.39	3.53		132	353	873	125		65	152	572	35	
14.9 km	4.10	1.63	4.49	5.80		213	610	705	825		122	253	267	323	
16.5 km	1.00	4.99	10.57	7.49		398	278	342	358		280	150	128	220	
19.5 km	2.50	1.38	27.24	2.64		368	284	455	490		300	151	285	98	
20.6 km	0.80	1.59	9.72	1.64		172	190	405	627		117	105	178	92	
23.7 km	2.10	7.99	10.41	0.22		165	297	518	253		112	85	177	57	
Dam	1.03	0.33	0.53	1.26	<i>0.11</i>	278	538	826	1187	<i>0.29</i>	50	34	14	50	<i>0.53</i>
Spec. Reg.	2.60	0.77	1.75	2.77	<i>0.42</i>	465	358	500	313	<i>0.55</i>	370	173	150	138	<i>0.10</i>
Bassett	2.00	1.31	7.39	7.13	<i>0.18</i>	197	448	747	505	<i>0.06</i>	117	186	408	193	<i>0.06</i>
Koehler	1.60	3.99	14.48	2.99	<i>0.01</i>	276	262	430	432	<i>0.08</i>	202	123	192	117	<i>0.25</i>
<i>P</i>	<i>0.54</i>	<i>0.13</i>	<i>0.06</i>	<i>0.18</i>		<i>0.12</i>	<i>0.35</i>	<i>0.53</i>	<i>0.13</i>		<i>0.01</i>	<i>0.07</i>	<i>0.00</i>	<i>0.32</i>	
Tailwater	1.73	1.87	7.10	3.56	<i>0.02</i>	288	394	620	619	<i>0.04</i>	171	125	195	123	<i>0.39</i>



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
Reach	Dam			Special Regulations		Bassett			Koehler			

Figure 1.1. Sampling sites for brown trout are numbered upstream to downstream in the Smith River tailwater, southwestern Virginia. The five sites sampled for age-0 recruitment are sites 3, 4, 7, 8, and a site at 22.8 km near the Henry County Upper Smith River wastewater treatment facility between sampling sites 11 and 12. For analysis and discussion the tailwater is divided into four reaches based on management regulations and gross morphometric habitat variations.

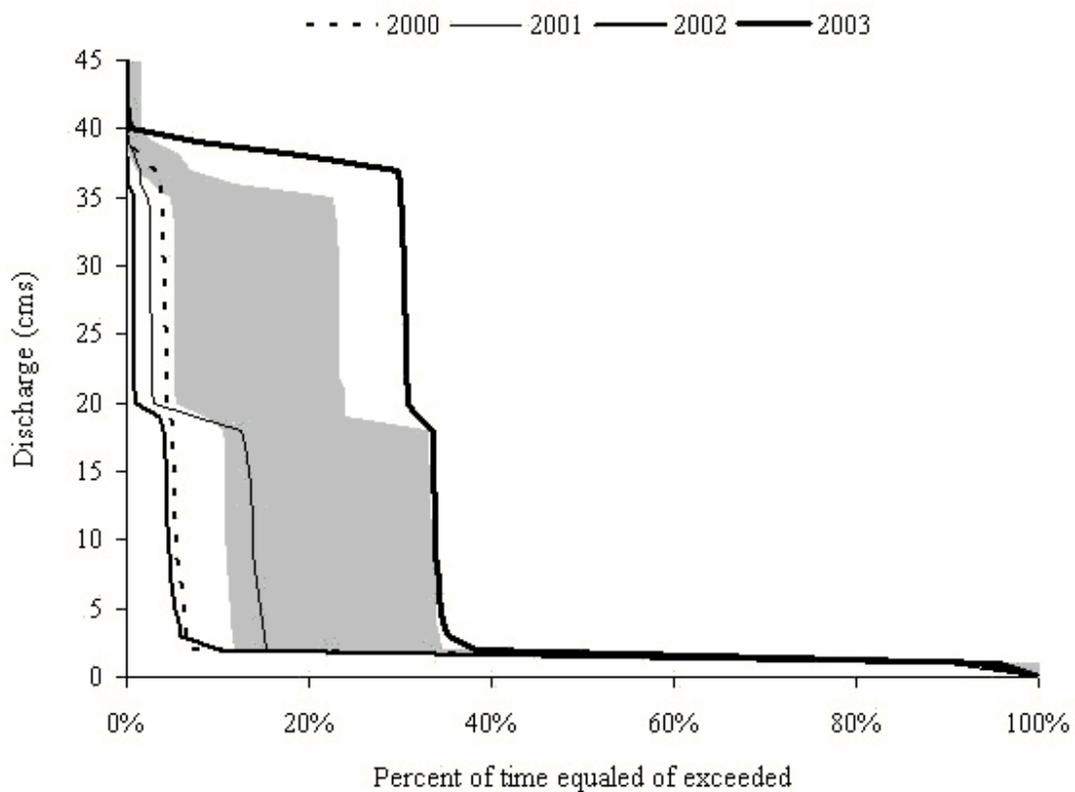


Figure 1.2. Annual flow-duration curves for Smith River study years (2000-2003) using 15 minute interval discharge data from the USGS Philpott gage located 0.4 km below Philpott dam, Smith River, Virginia. The shaded area indicates the range over which that flow-duration curves occurred during years 1991-1999.

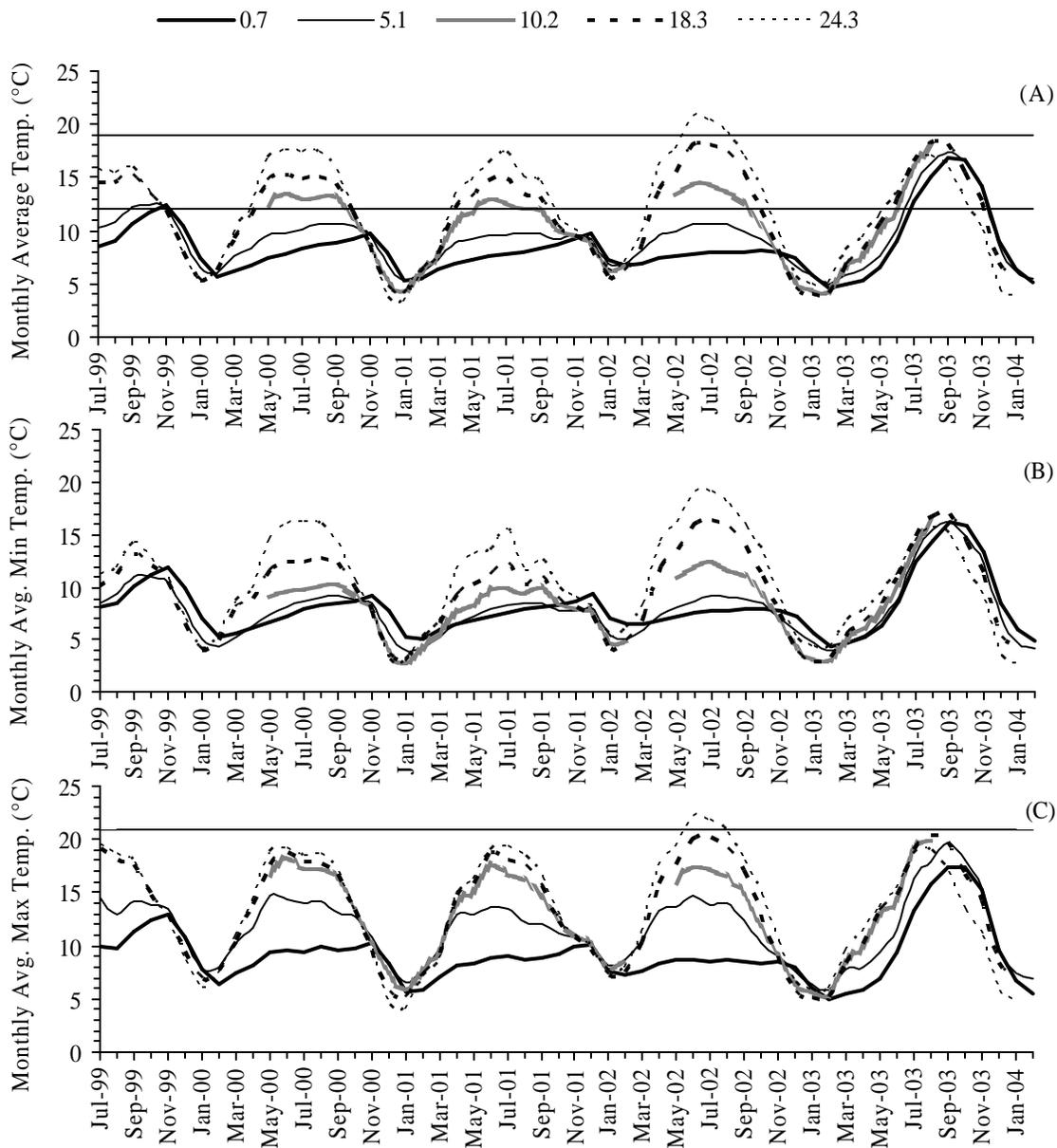


Figure 1.3. Summary of water temperature data measured throughout the tailwater (0.7-24.3 km) over time (Jul 99 - Feb 04). Water temperature data summarized as: (A) the monthly average temperature, (B) monthly average minimum temperature, and (C) monthly average maximum temperature. Horizontal lines represent: (A) the brown trout optimal growth temperature range (12-19°C) and (C) the DEQ 21°C maximum temperature standard for stockable trout waters.

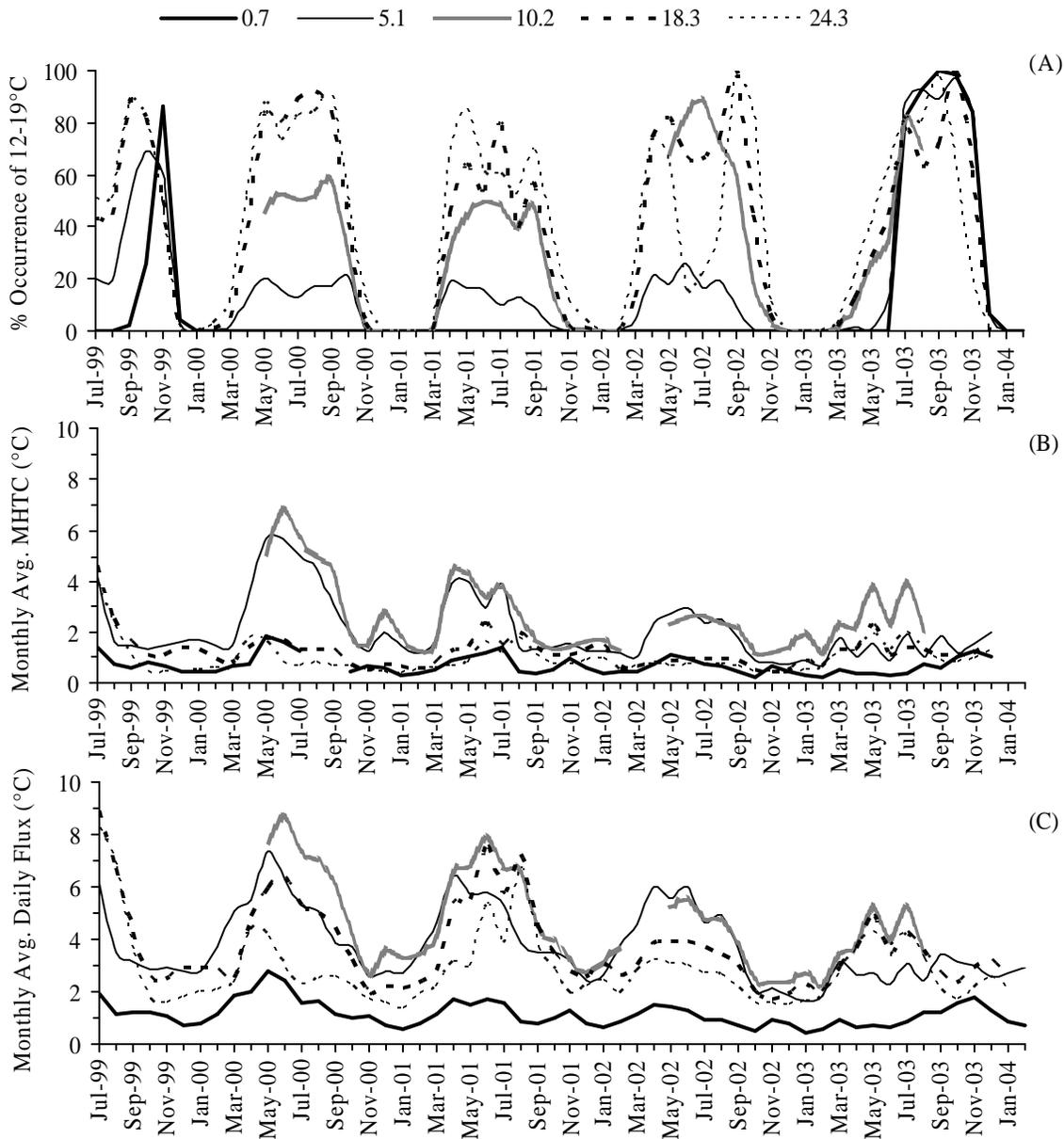


Figure 1.4. Summary of water temperature data measured throughout the tailwater (0.7-24.3 km) over time (Jul 99 - Feb 04). Water temperature data summarized as: (A) the percent monthly occurrence of 12-19°C (i.e. brown trout optimal growth temperature range), (B) monthly average maximum hourly temperature change (MHTC), and (C) monthly average daily temperature flux.

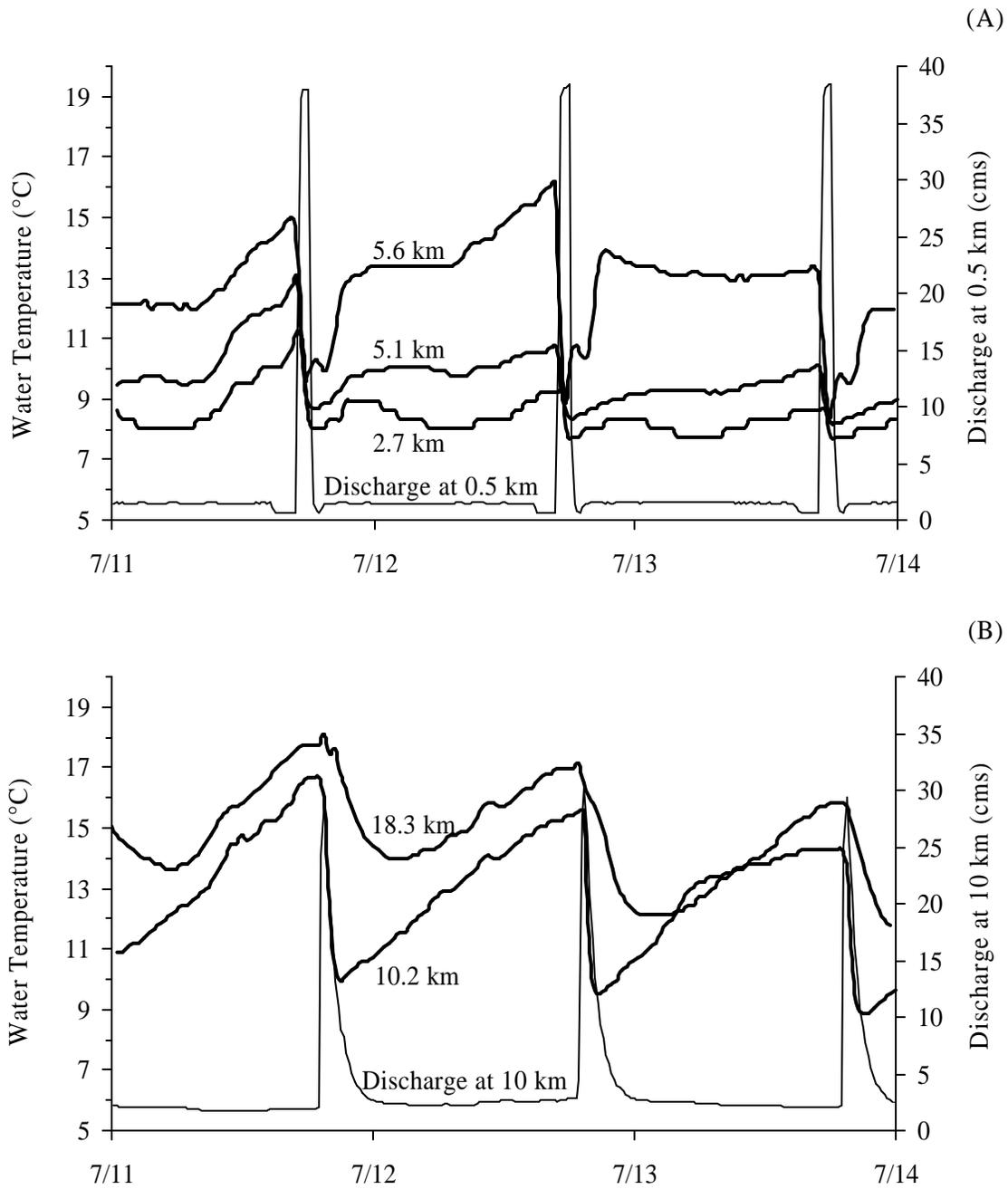


Figure 1.5. Declines in water temperature ($^{\circ}\text{C}$ recorded every 30 min) correspond to peaking discharge regime (cms recorded every 15 min). Data shown is from July 11-13, 2000 and panel (A) are upstream temperatures at 2.7, 5.1, and 5.6 km and Philpott gage discharge (0.5 km), and panel (B) are downstream temperatures at 10.2 and 18.3 km, and Bassett gage discharge (10 km). Measurement time of temperature data has been shifted to account for discharge travel time (5.1 and 5.6 km temperature data shifted 1 hr from 0 km discharge data, and 2 hrs for 18.3 km temperature from 10 km discharge data).

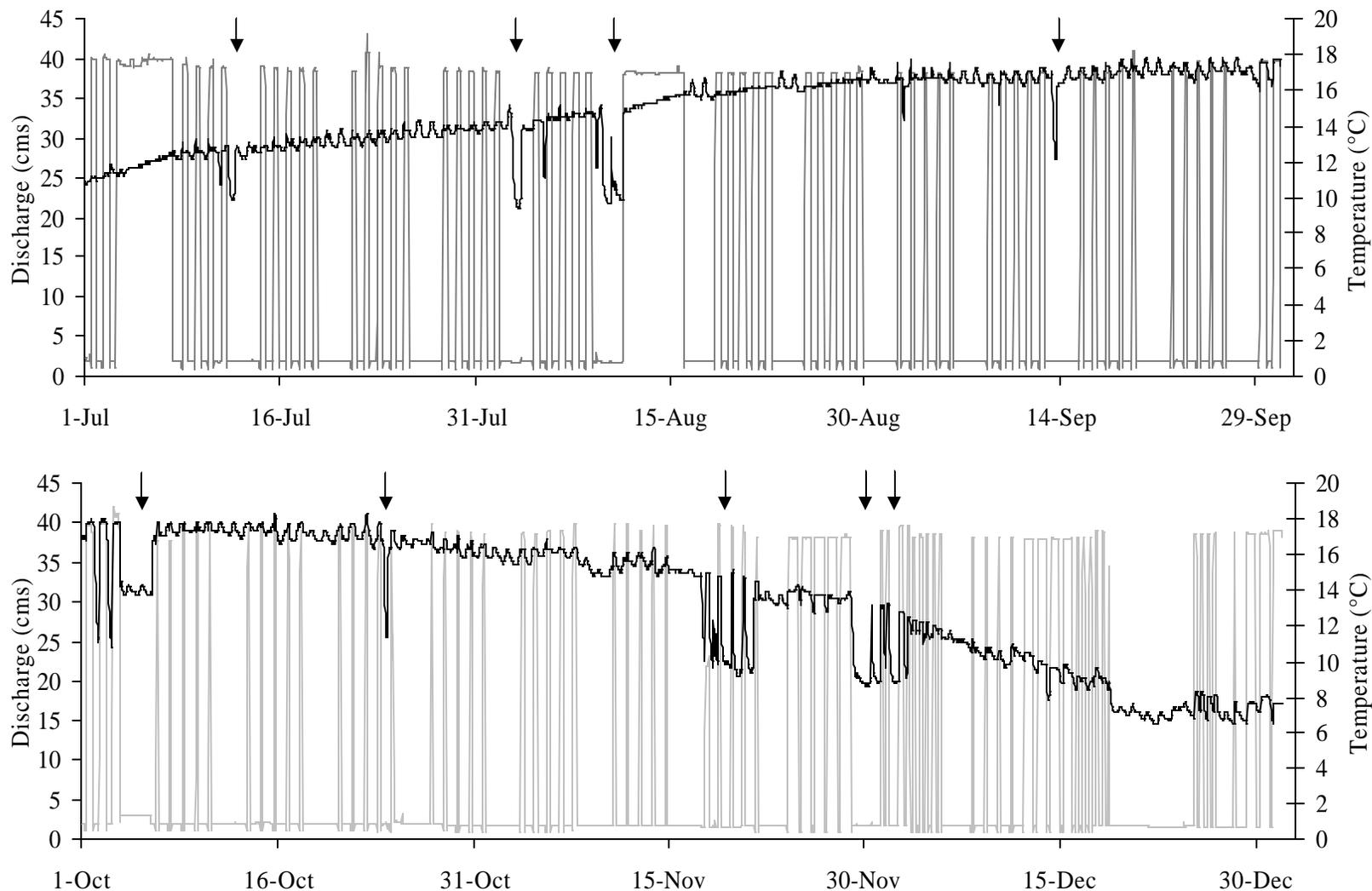


Figure 1.6. During the second half of 2003, large drops in water temperature occurred during baseflow conditions (denoted by arrows) due to use of Philpott dam low flow outlets which release water from near the bottom of the reservoir. Such drops in temperature were not observed in 1999-2002 temperature data. Water temperature recorded half-hourly at 0.7 km below Philpott dam shows temperature drops on Jul 12; Aug 3, 10-11; Sep 13; Oct 2-6, 24; Nov 17-21, 30; and Dec 1-3.

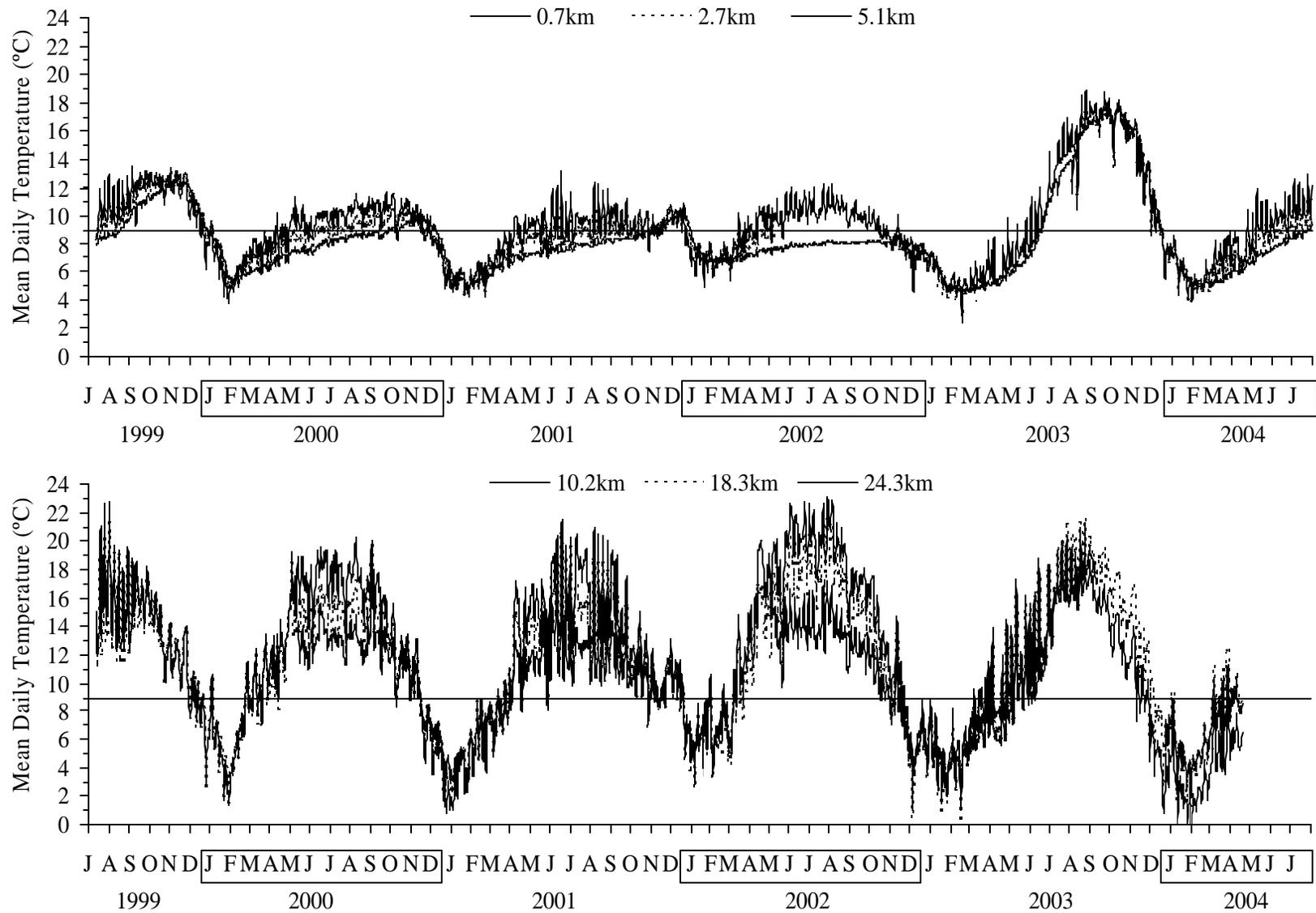


Figure 1.7. Mean daily water temperature (°C) at 0.7, 2.7, 5.1, 10.2, 18.3, and 24.3 km below Philpott dam in the Smith River, VA from July 1999 through July 2004. Horizontal line displays 9°C, which initiates spawning.

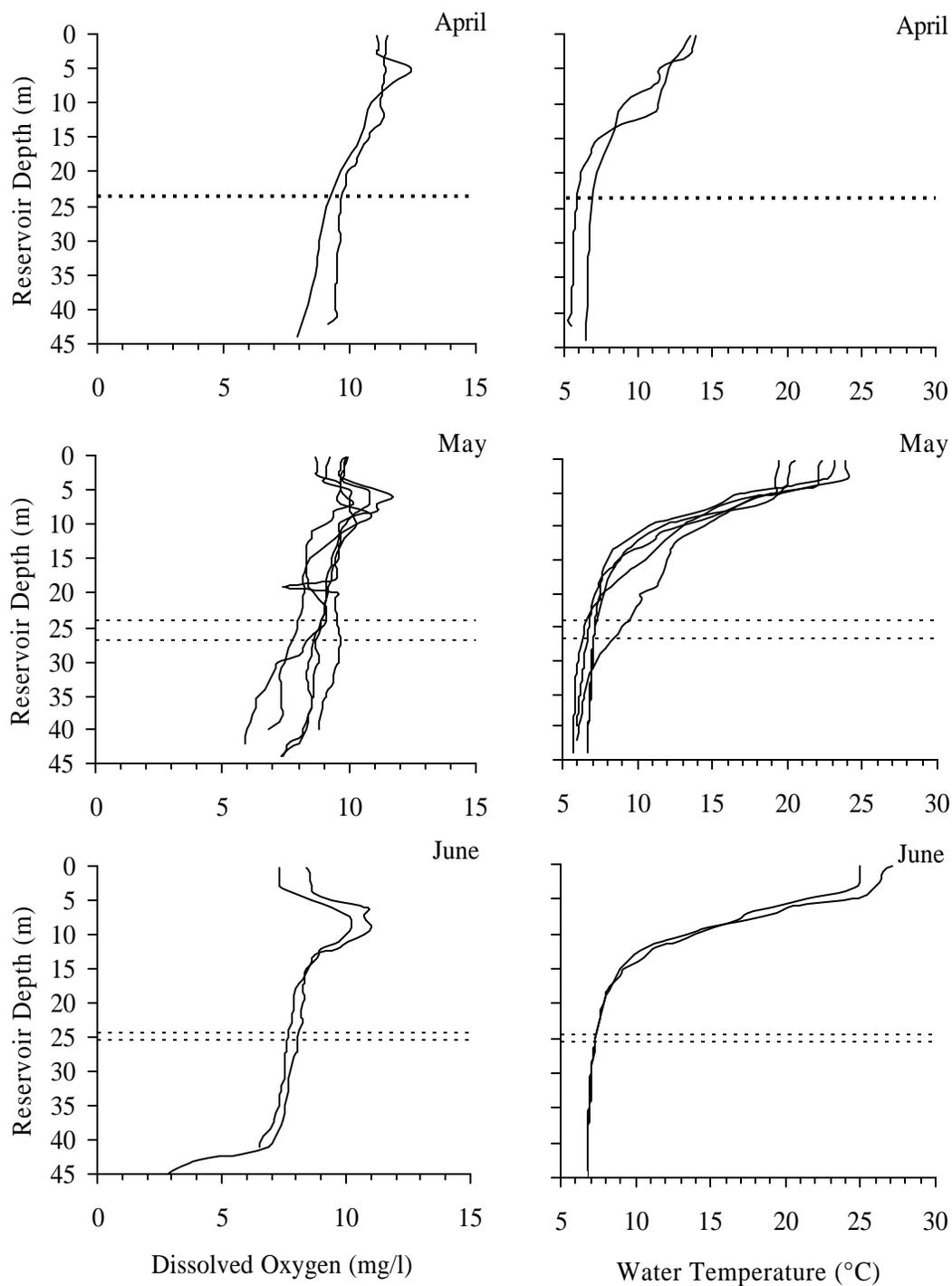


Figure 1.8a. Dissolved oxygen (mg/l) and water temperature ($^{\circ}\text{C}$) profiles measured in Philpott reservoir near the dam in April, May, and June in years ranging from 1995-2002. Dashed horizontal lines indicate the depth range of the hydropower intakes during the years of data collection. The minimum DEQ DO standard for trout waters is 6.0 mg/l.

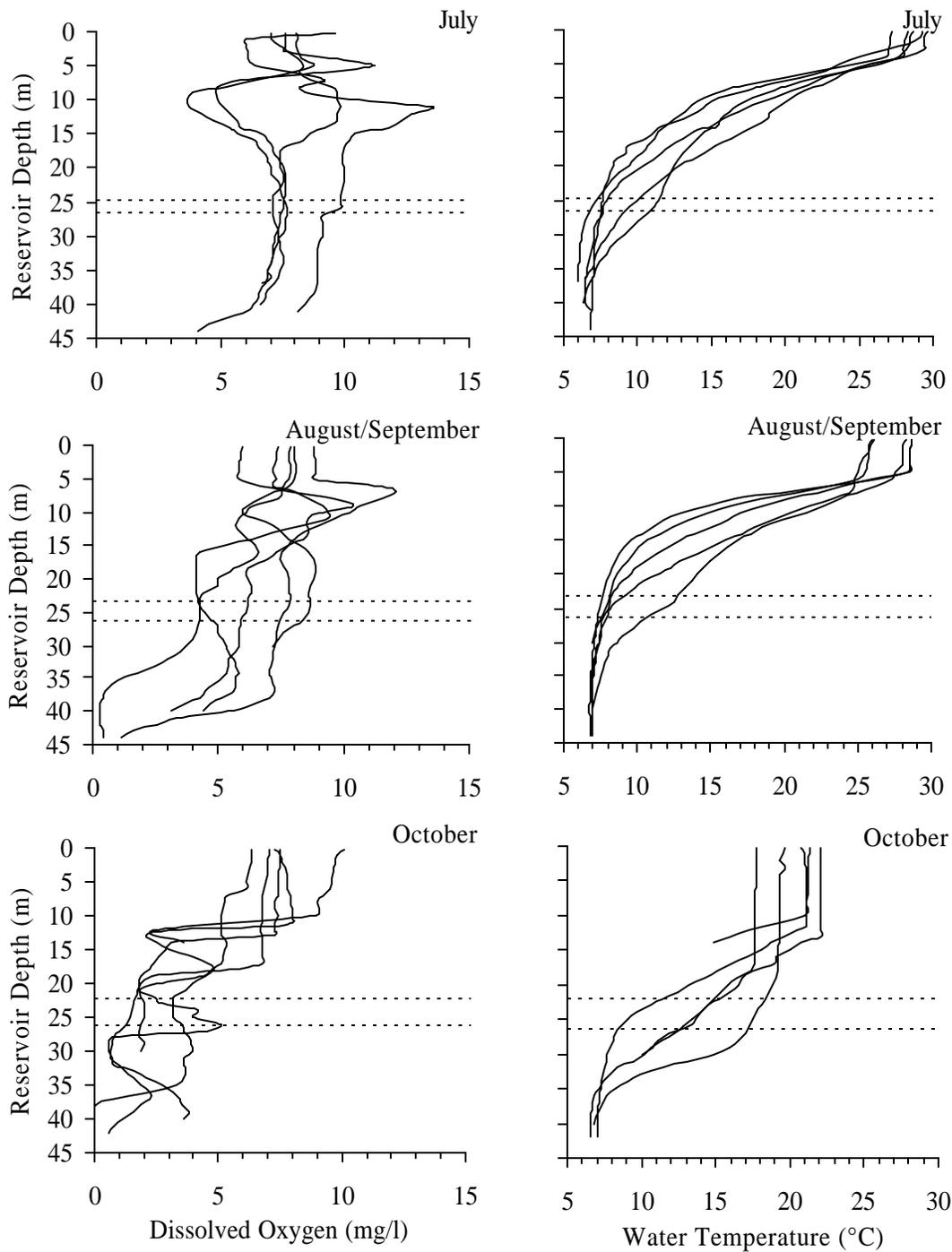


Figure 1.8b. Dissolved oxygen (mg/l) and water temperature ($^{\circ}\text{C}$) profiles measured in Philpott reservoir near the dam in July, August, September, and October in years ranging from 1995-2002. Dashed horizontal lines indicate the depth range of the hydropower intakes during the years of data collection. The minimum DEQ DO standard for trout waters is 6.0 mg/l.

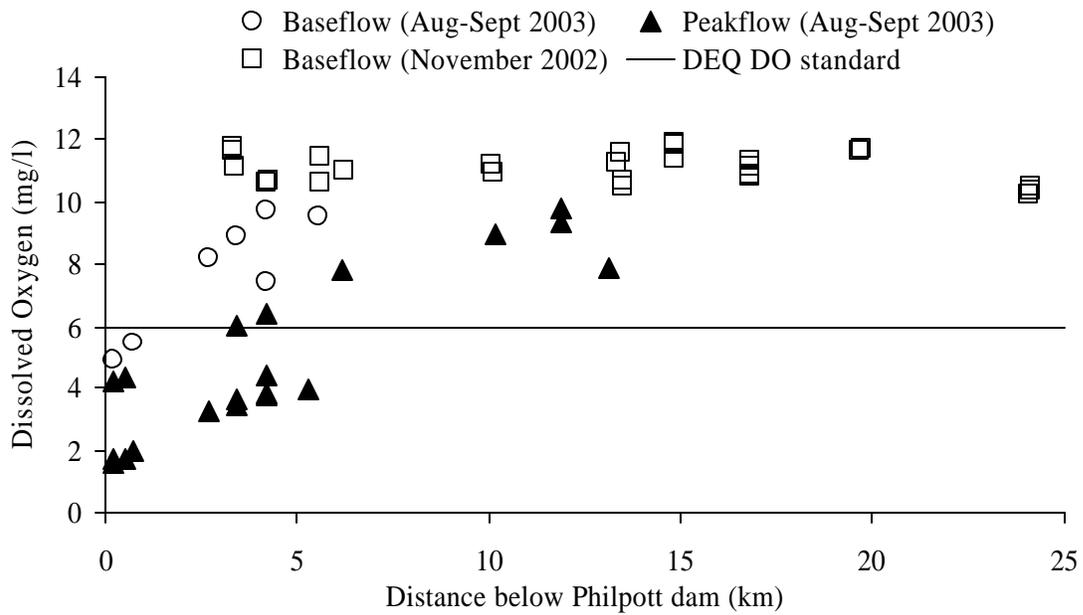


Figure 1.9. Dissolved oxygen (mg/l) measured during baseflow (Nov 2002 and Aug-Sept 2003) and peakflow (Aug-Sept 2003) at multiple locations downstream of Philpott dam. The horizontal line indicates the DEQ dissolved oxygen standard for trout waters.

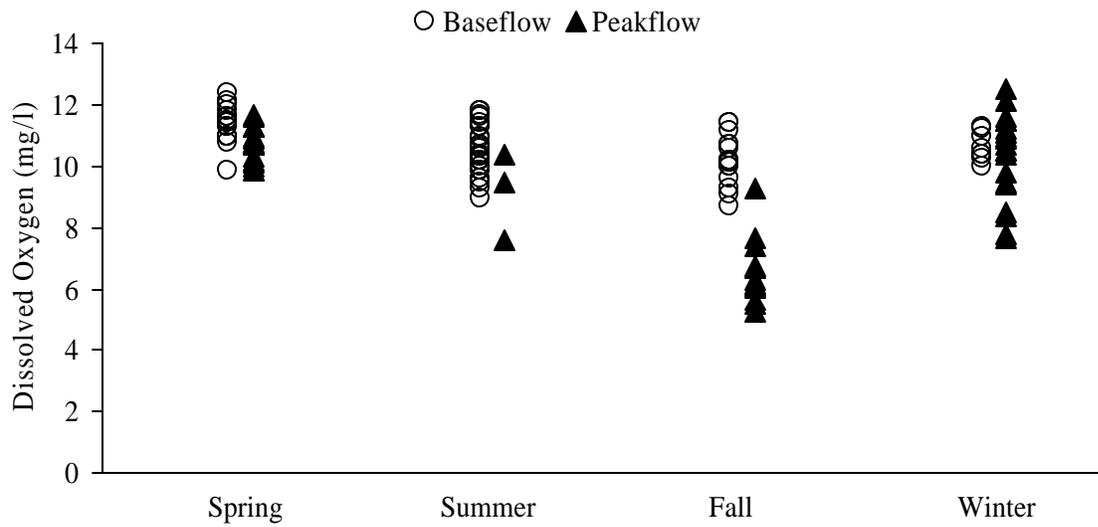


Figure 1.10. Dissolved oxygen (mg/l) measured by the VA DEQ from 1992 to 2003 at 5.1 rkm below Philpott dam in the Smith River is shown delineated based on season (Spring=Mar, Apr, May; Summer=Jun, Jul, Aug; Fall=Sep, Oct, Nov; Winter=Dec, Jan, Feb) and flow (base flow vs. peak flow).

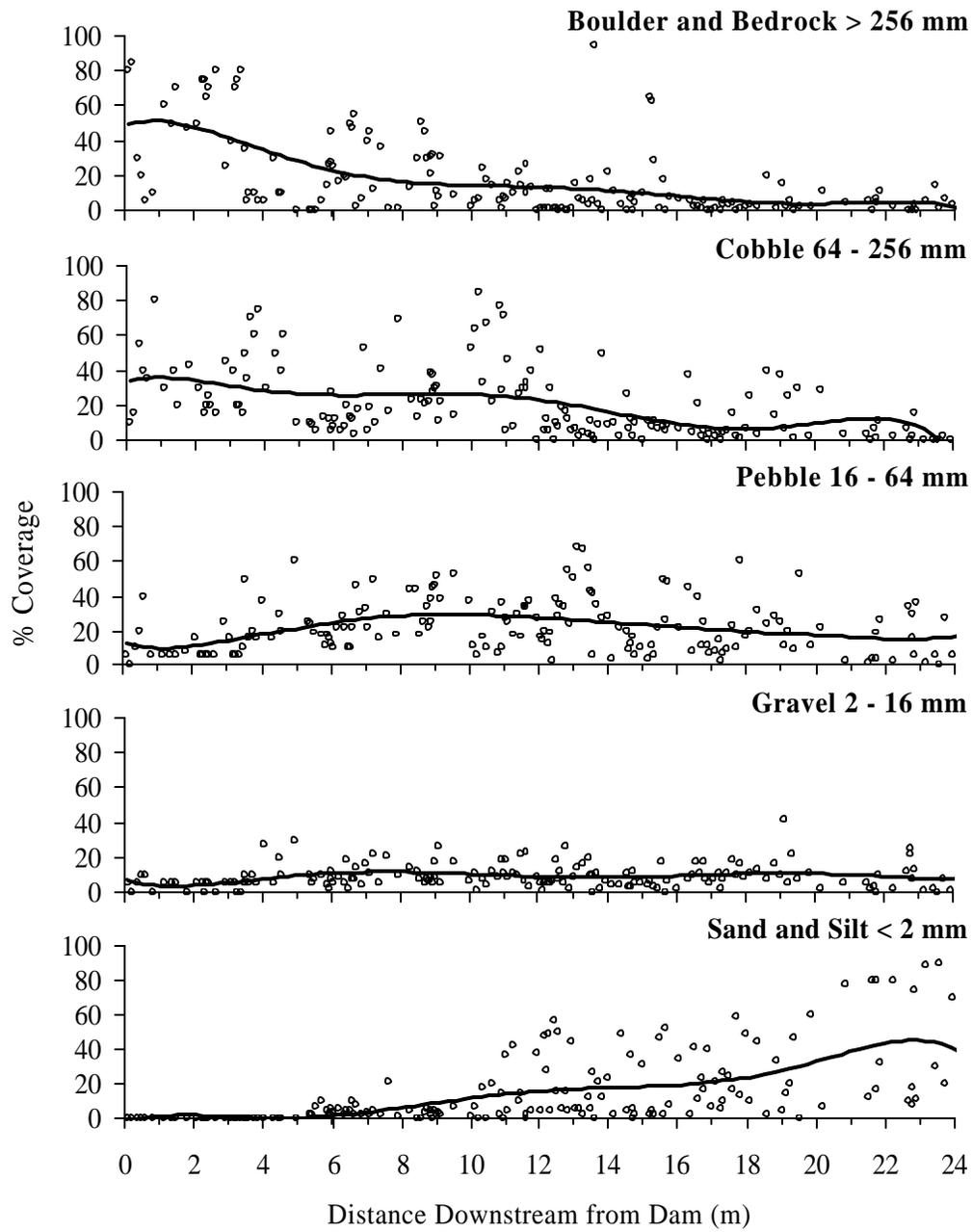


Figure 1.11. Trends in river bottom coverage of various-sized substrates throughout the Smith River tailwater.

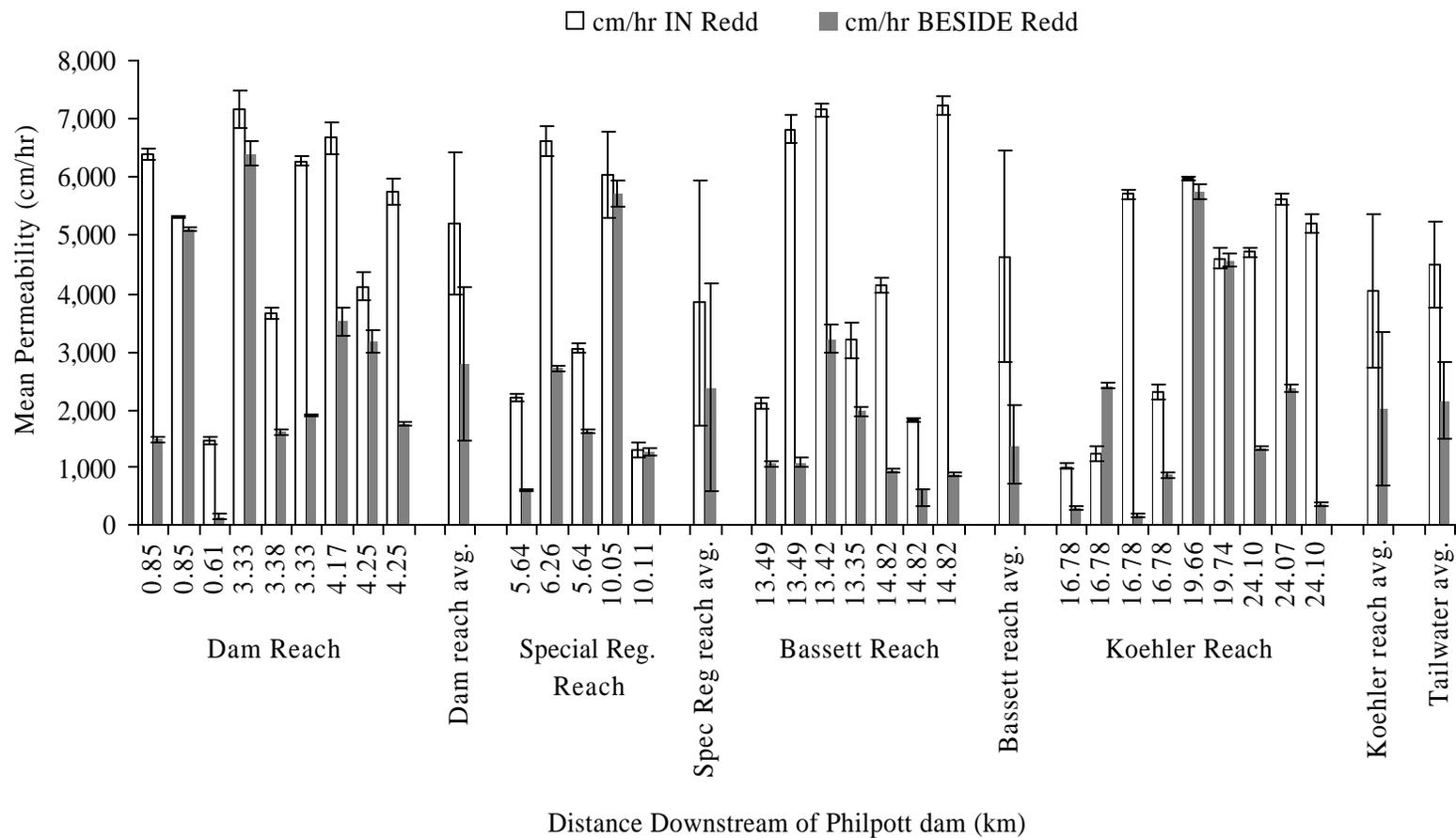


Figure 1.12. Substrate permeability (cm/hr) within the redd egg-pocket area (white bars) and in undisturbed substrate beside the redd (solid bars). Data is shown for individual redds delineated by reach as well as averages for reaches and the tailwater. Error bars are 95% confidence intervals and two standard errors for reach averages.

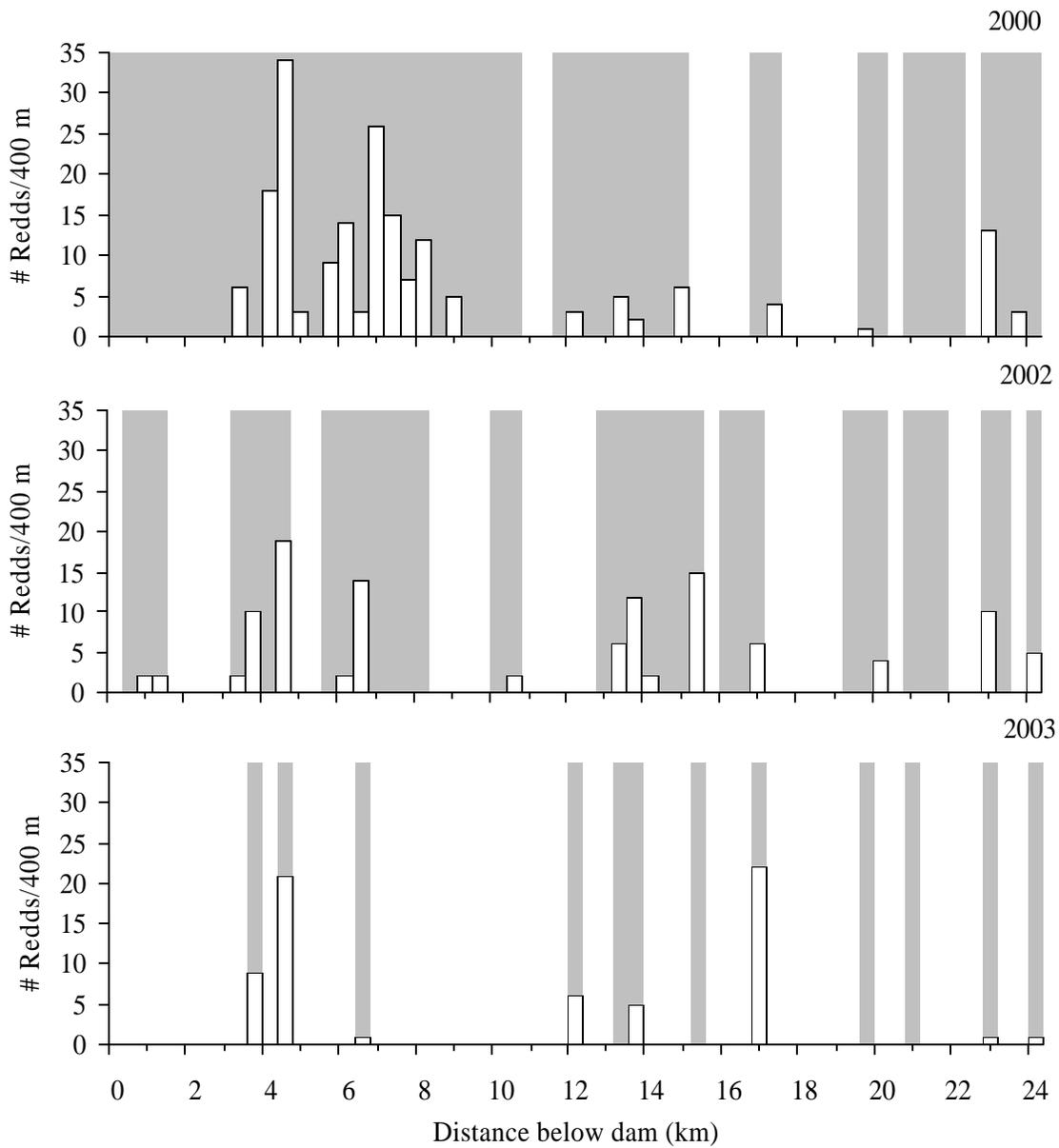


Figure 1.13. Distribution of redds downstream of Philpott dam, observed during November and December of 2000, 2002, and 2003 (2001 data not shown because only three redds were found most likely due to spawning occurring before surveying). White bars indicate the number of redds observed in 400 m reaches and gray bars represent the approximate areas surveyed.

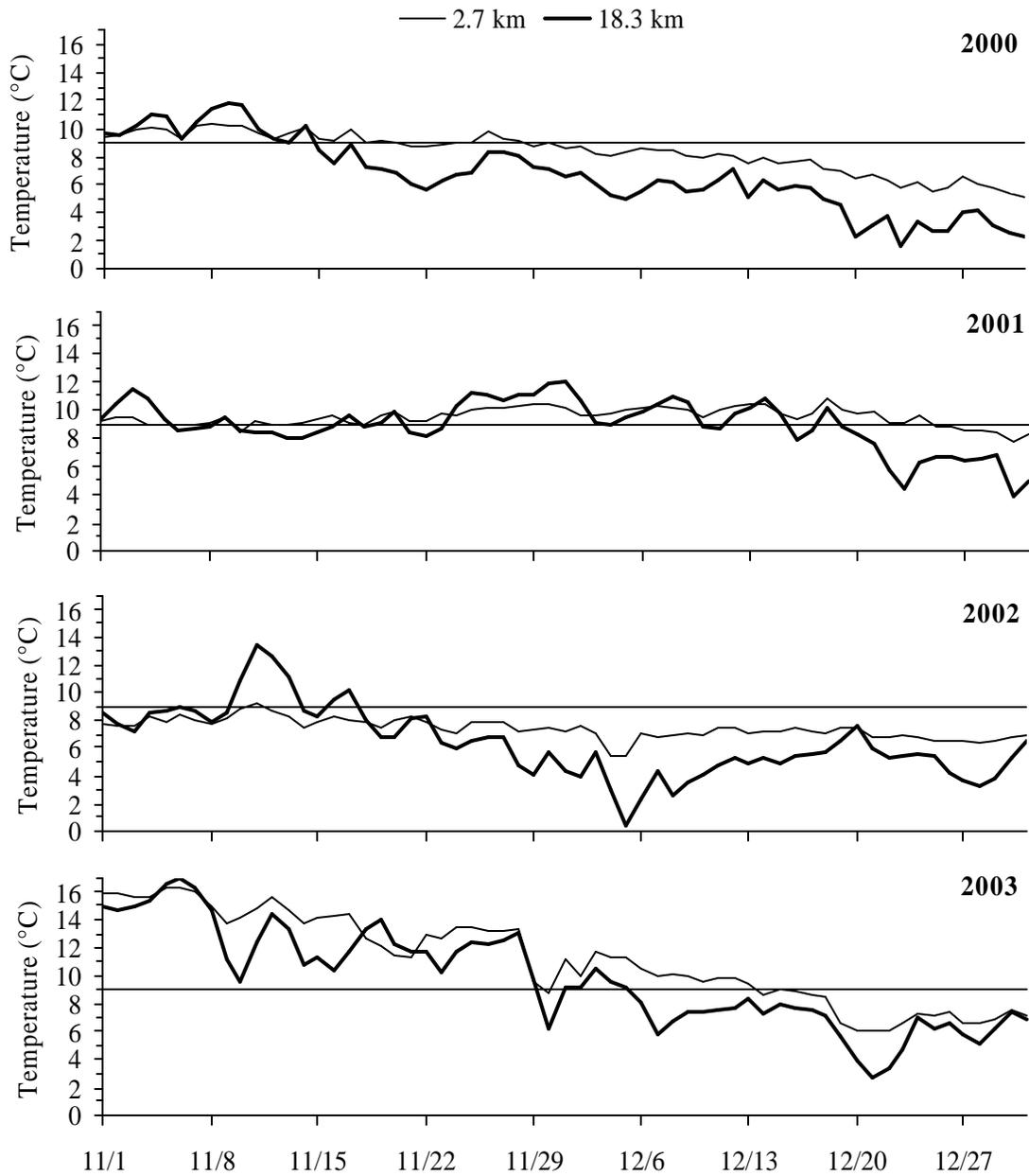


Figure 1.14. Daily water temperature recorded upstream at 2.7 km and downstream at 18.3 km below Philpott dam during the November and December spawning season of 2000-2003. Horizontal line at 9°C indicates the temperature at which spawning typically initiates.

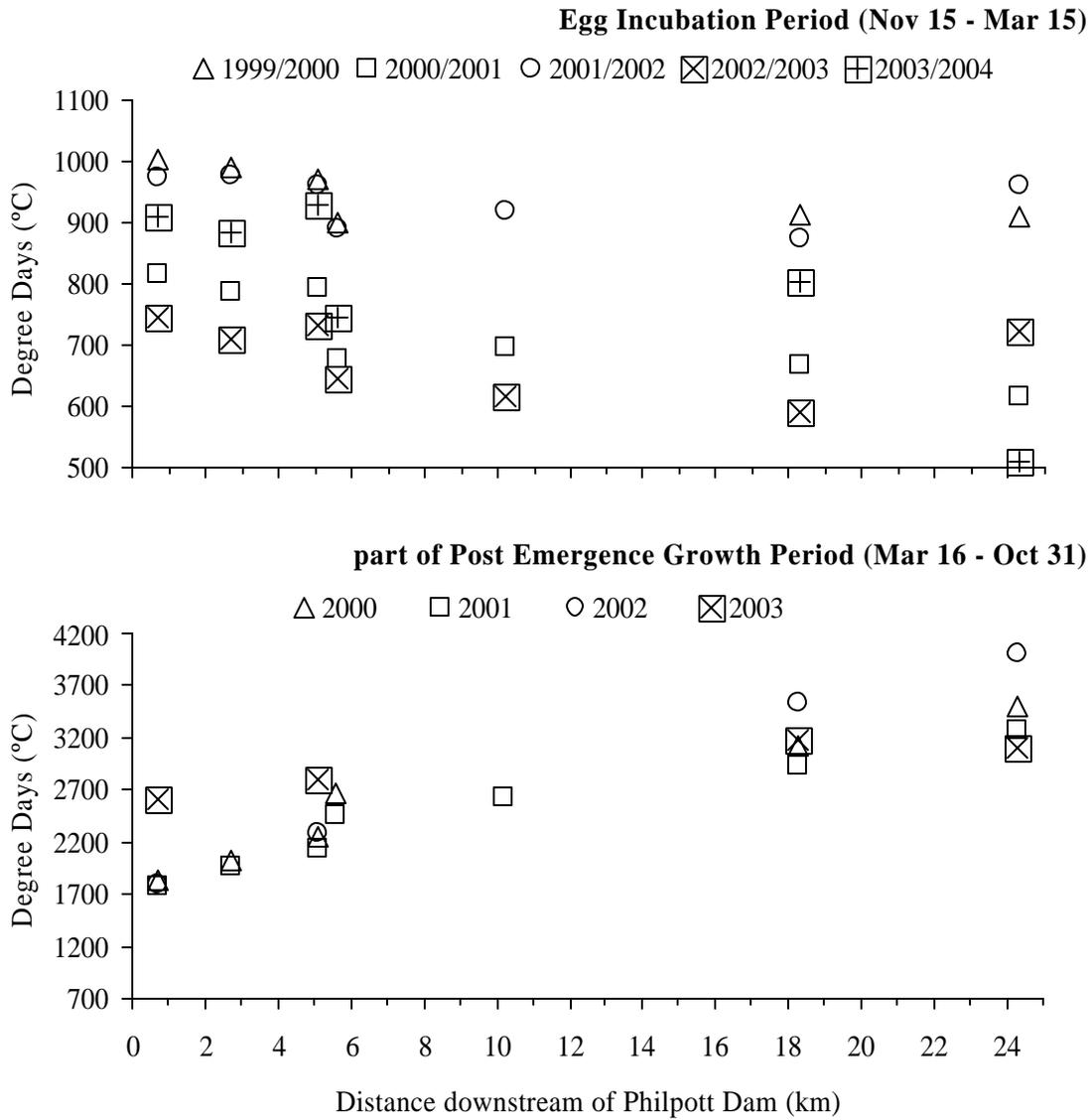


Figure 1.15. Degree days ($^{\circ}\text{C}$) (sum of daily mean water temperature) at 0.7, 2.7, 5.1, 5.6, 10.2, 18.3, and 24.3 km below Philpott dam in the Smith River during the typical egg incubation period (11/15 - 03/15; $n = 122$) and part of the growth period after emergence (03/16 - 10/31; $n = 230$).

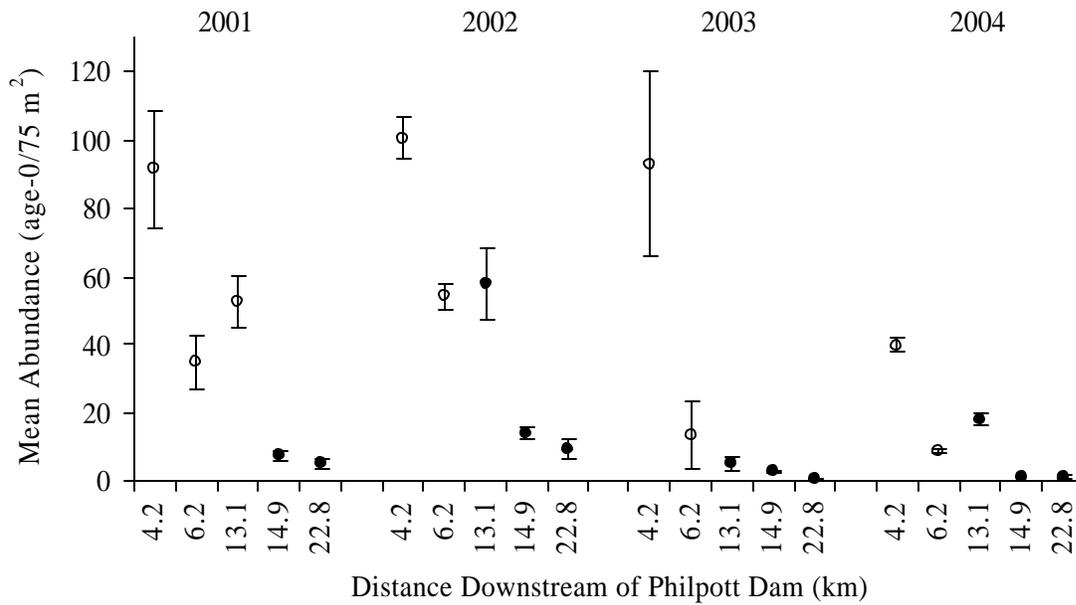


Figure 1.16. Mean abundance of age-0 brown trout per 75 m² in May of 2001-2004. Error bars are 95% confidence intervals. Abundance values are average population estimates (MicroFish software) of three 25 x 3 m reaches per site; 4.2, 6.2, 13.1, 14.9, and 22.8 km. Solid circles indicate non-descending catch depletions and/or cases where MicroFish did not compute a population estimate due to too few fish caught. In cases where MicroFish did not compute, the actual number of fish caught is shown.

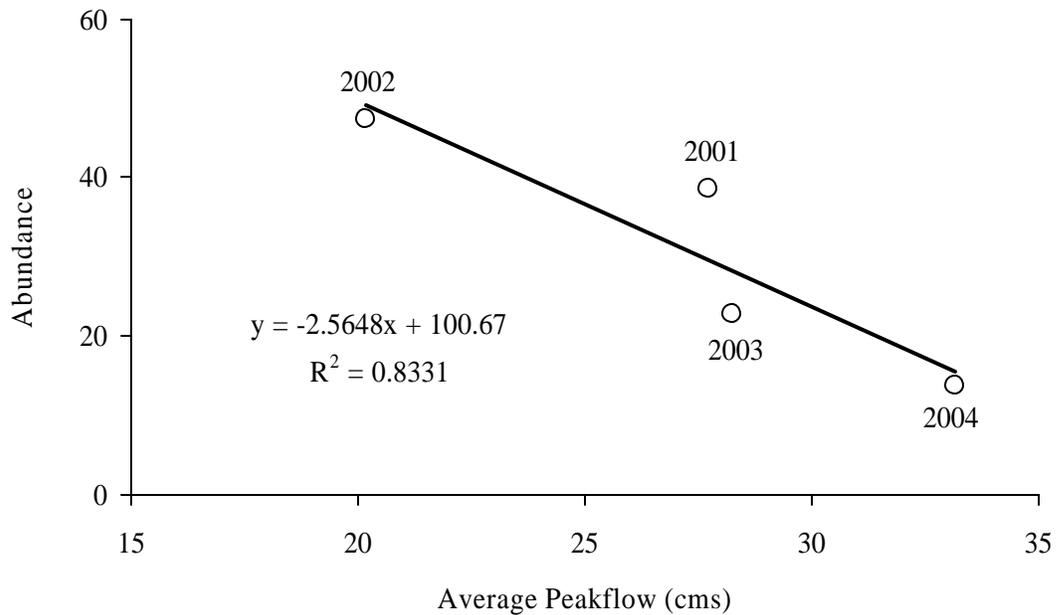
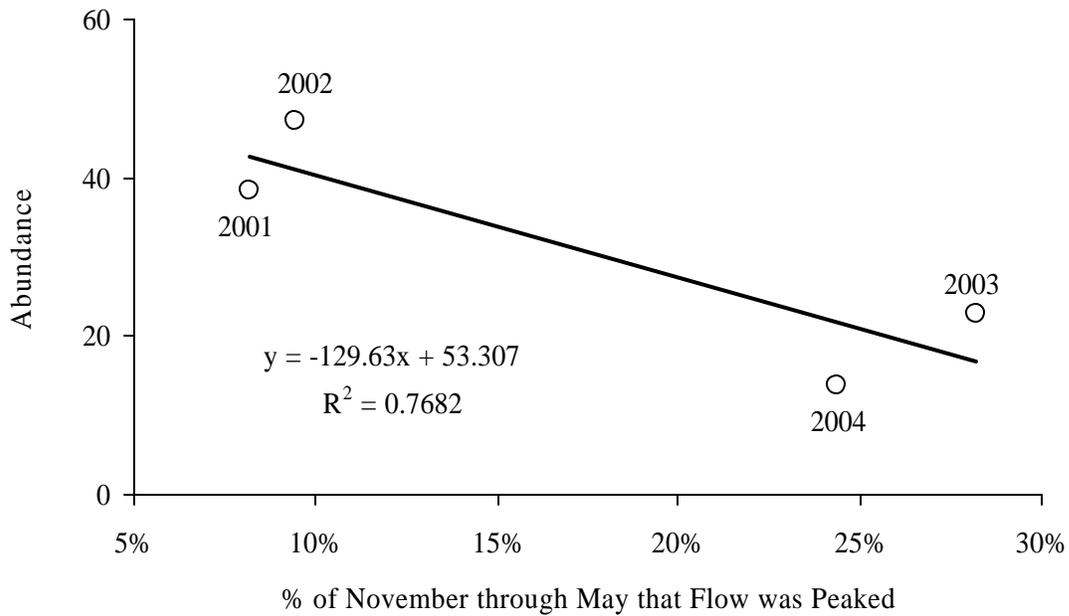


Figure 1.17. Average tailwater abundance of age-0 brown trout in May 2001-2003 versus the percentage that flow was peaked from November through May (upper graph) and the average peak flow (cms) during 2001-2003 (lower graph). Abundance values are the average population estimates (MicroFish software) from three 75 m² sections sampled per spawning area located 4.2, 6.2, 13.1, 14.9, and 22.8 km downstream of Philpott dam.

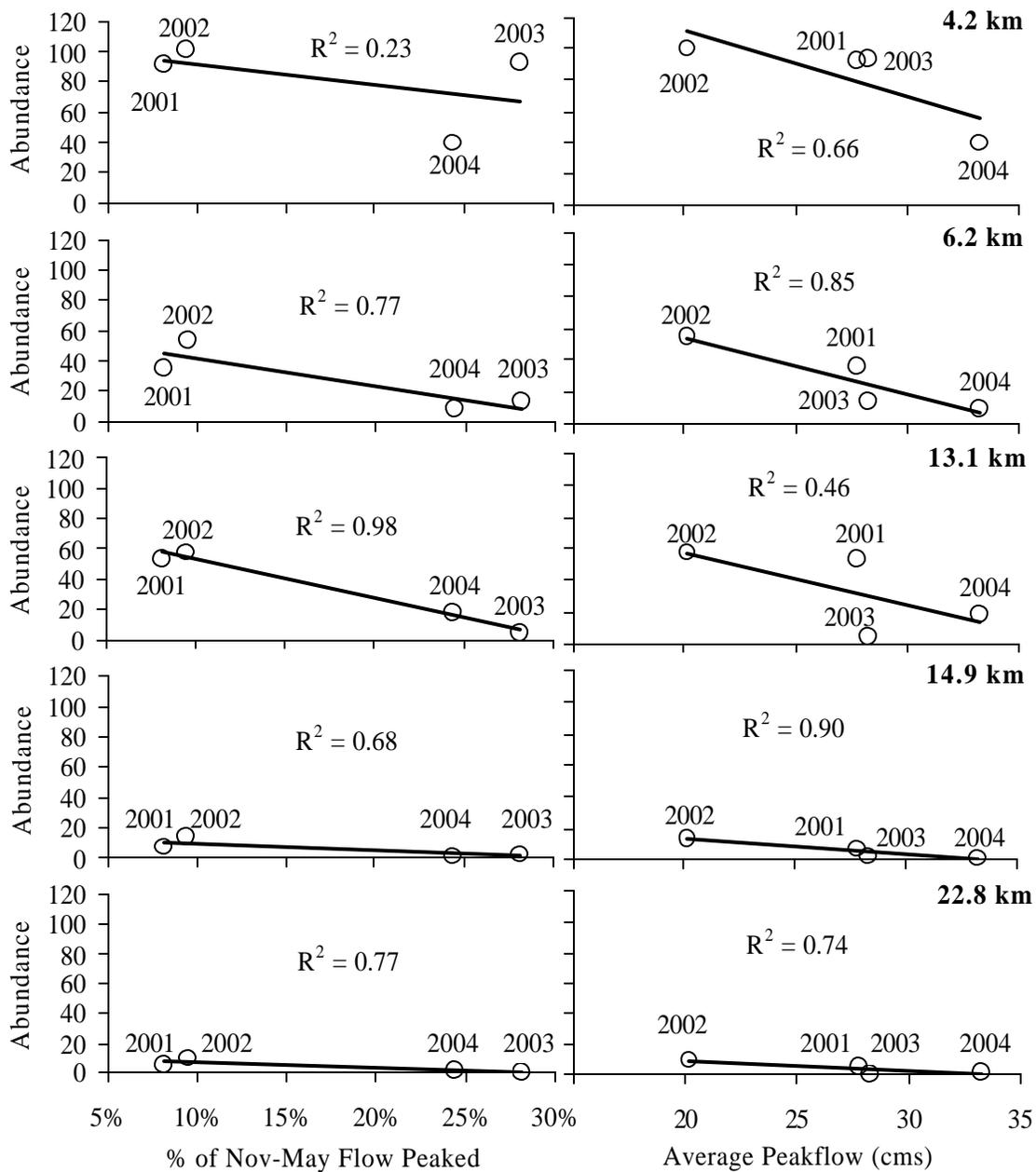


Figure 1.18. Abundance of age-0 brown trout per 75 m² (0.00375 hectare) in May 2001-2003 versus the percentage that flow was peaked from November through May (left panel), and the average peak flow (cms) during 2001-2003 (right panel). Abundance values are the average of three population estimates (MicroFish software) per site; 4.2, 6.2, 13.1, 14.9, and 22.8 km downstream of Philpott dam.

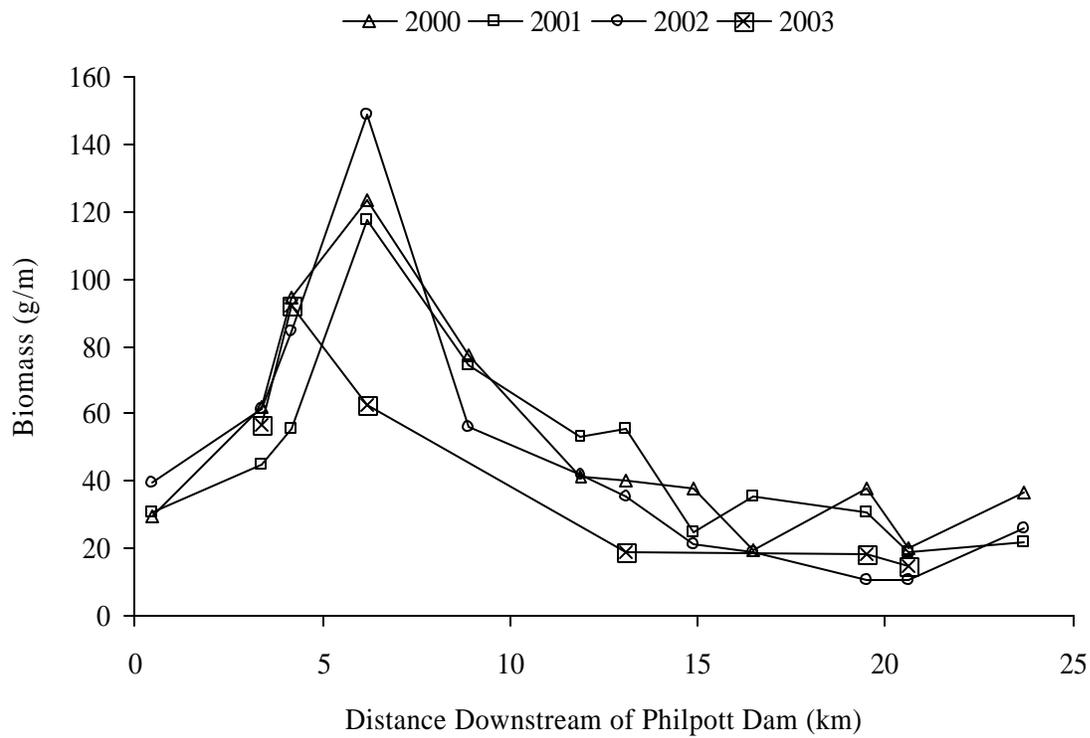


Figure 1.19. Biomass (g/m) of spawning size (= 203 mm) brown trout sampled in October 2000-2003, which is shortly before the November-December spawning season. In 2003, only six of the 12 sites were sampled.

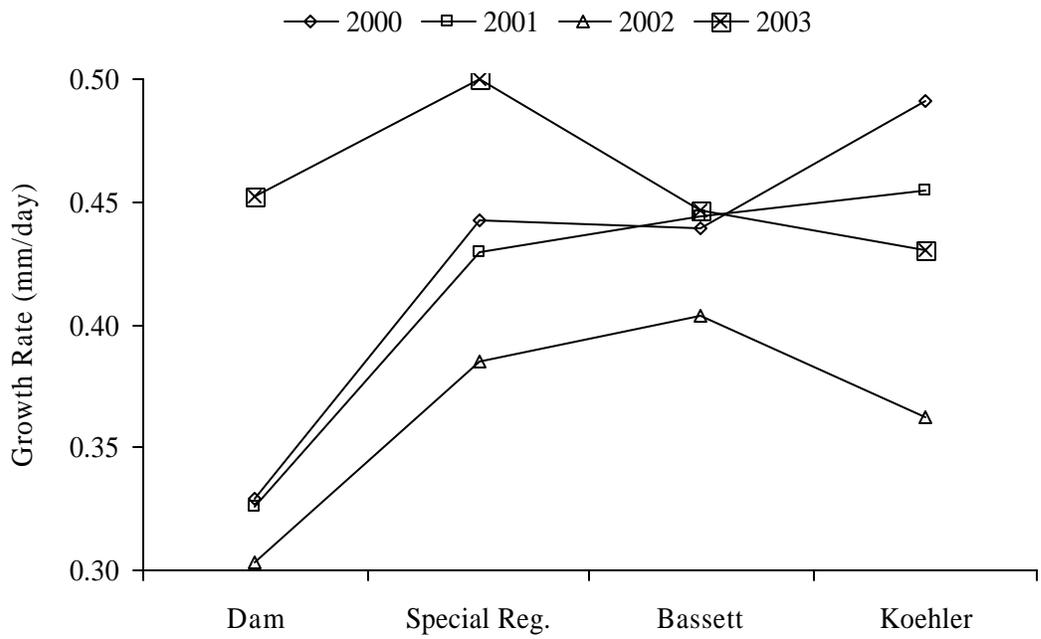


Figure 1.20. Growth rates (mm/day) of age-0 brown trout in the Dam (0.0-5.3 km downstream of Philpott dam), Special Regulations (5.3-10.0 km), Bassett (10.0-15.9 km), and Koehler (15.9-23.0 km) reaches.

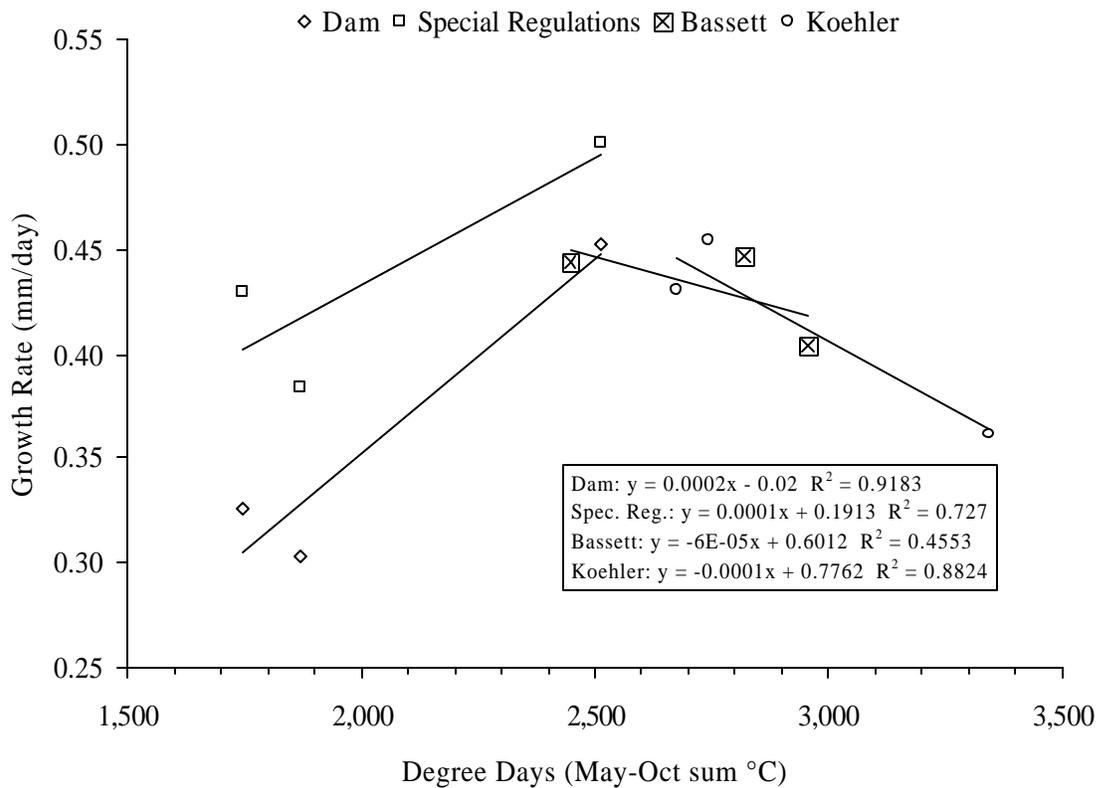


Figure 1.21. Age-0 growth rates (mm/day) calculated for the Dam, Special Regulations, Bassett, and Koehler reaches (based on May, Jun/Jul, and Oct length data; see Figure 21) are compared against degree days (°C) in 2001, 2002, and 2003 recorded at 5.1, 5.1, 18.3, and 24.3 km. Degree days are cumulative sums of daily temperature recorded by data loggers closest to the age-0 sampling sites. A positive trend, where growth rate increases with an increase in degree days, suggests warmer water temperatures improves growth.

◇ 2000 □ 2001 △ 2002 ⊠ 2003

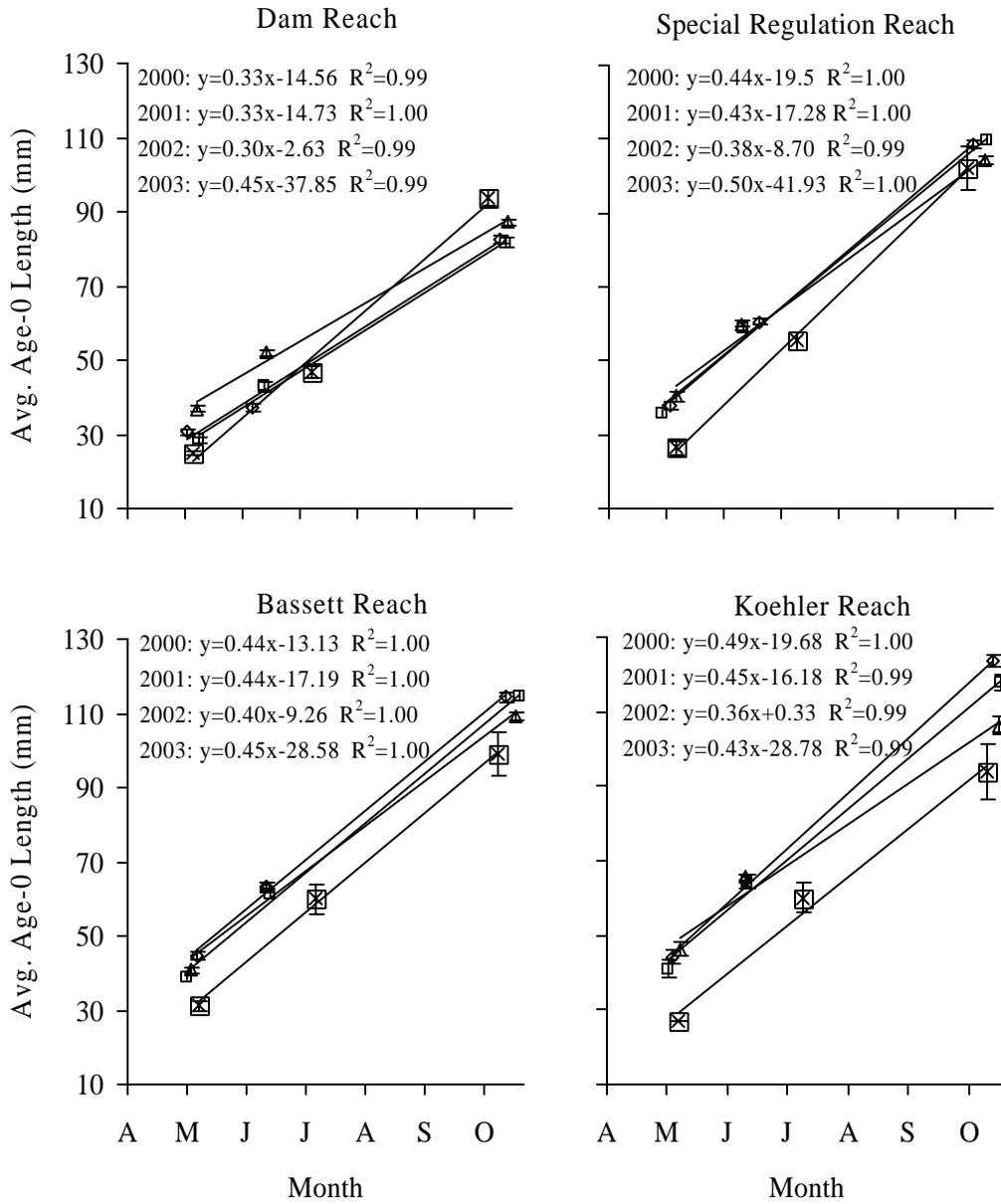


Figure 1.22. Age-0 brown trout average total length (mm) over time (May-Oct) during years 2000-2003 within the dam, special regulations, Bassett, and Koehler reaches. Error bars represent two standard errors. Slopes shown in the regression equations represent age-0 growth (mm/day). In 2003 age-0 lengths are initially smaller due to later emergence in April compared earlier emergence in February and March in 2002 and 2001 respectively.

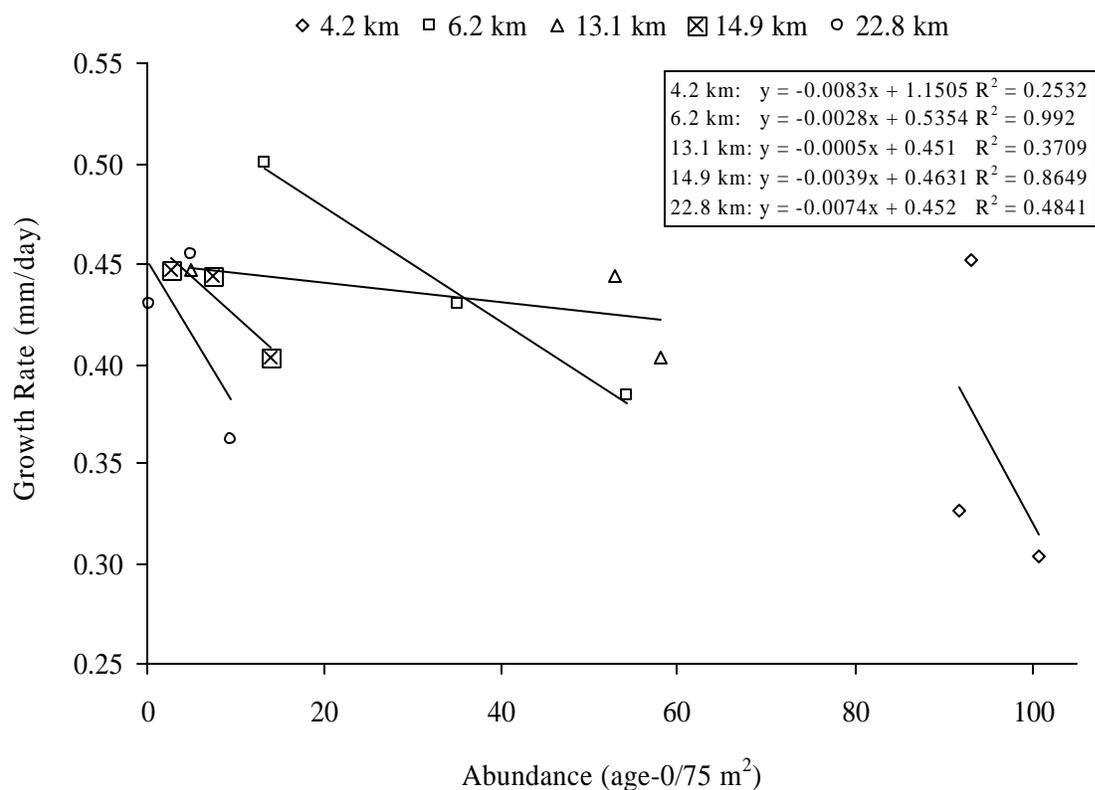


Figure 1.23. Age-0 growth rate (mm/day) (based on May, Jun/Jul, and Oct length data; see Figure 1.21) versus age-0 abundance (age-0/75 m²) in May (2001, 2002, and 2003) at the 5 sampling locations (4.2-22.8 km). A negative trend, where growth rate declines with an increase in abundance, suggests the presence of density dependence.

Job 2, Part A: 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 and proposed 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, and evaluation and modeling of forage and thermal constraints on the brown trout using bioenergetics modeling.

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 indirectly impact trout by influencing 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. 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 in the Smith River tailwater from four reaches from Philpott Dam (Rkm 0.5) to Martinsville (Rkm 23.0) beginning in June 2000 and continuing through October 2003. The tailwater was divided into four reaches to account for variation in the physical attributes of the channel that occur on a longitudinal pattern, and within each reach, 2-4 sites were selected for sampling (Table 2a.1). In 2000, fish were sampled in June and October, and in 2001 and 2002, trout were sampled in April, June, and October. In 2003, trout were sampled in July at 11 locations, with the site at Rkm 23.0 not being sampled due to increased water depths and decreased water clarity, and trout were sampled at a subsample of locations in October. Fish were collected with multiple mobile anode pulsed DC barge electrofishers. During the June/July sampling periods, three-pass depletion electrofishing was conducted on 100-m sections at each location that were enclosed with block nets, while single pass electrofishing was conducted in 200-400 m sections on other sampling dates.

Upon collection, brown trout were anesthetized, measured to the nearest mm total length (TL), and weighed to the nearest g. During June 2000, brown trout larger than 100 mm were implanted with PIT (Passive Integrated Transponder) tags (Biomark™, Inc.) and given an adipose fin clip to aid in identification of tagged trout. Tag recapture rates were low during the June 2001 sampling period (Table 2a.2), thus additional brown trout (>70 mm) were tagged in October 2001 (Table 2a.3). In subsequent sampling, trout were scanned for the presence of a PIT tag and length (mm) and weight (g) were recorded.

A subsample of trout were collected and returned to the lab for subsequent age analysis and cohort designation using otoliths during the duration of the project. Past studies suggested that otoliths are the preferred aging structure for trout and there have been known complications with scale structure of brown trout in the Smith River, hence otoliths, rather than scales, were used to age trout during this study (Smith 1996; Hining et al. 2000). After removal of the otoliths from the trout, otoliths were adhered to microscope slides using QuickTite super glue gel (Loctite, Avon OH) and sanded using 400 and 600 grit sandpaper to allow light to pass through the otolith. Prior to viewing, a drop of clove oil was placed on the otolith to help clarify the image. The otolith was viewed using an imaging system (Olympus SZ-ST scope with magnification range 1x-6.3x equipped with an Olympus SZ-CTV scope adapter; Samsung CCD SAC-410NA color camera; Image Pro-Plus® software), which allowed for digital enhancement of the image to aid in identification of the annuli and precise measures for distances from the focus to each annulus. All annuli were marked and measurements were taken along the dorsal axis from the focus to each annulus.

Brown trout were collected from each of the four reaches of the river during February, May, September, and December 2002 via backpack electrofishing to determine seasonal and longitudinal trends in diet composition and consumption rates. Five brown trout were collected every six hours 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 the trout that were 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 g. Fish were sacrificed and stomach contents were removed, preserved in a 10% formalin solution, and returned to the laboratory for further analysis.

Upon return to the laboratory, stomach contents were identified, enumerated, blotted dry, and weighed to the nearest 0.001 g. Fish in the stomach contents were identified to species when possible, using keys in Jenkins and Burkhead (1993), aquatic invertebrates were identified to family when possible using Merritt and Cummins (1996), and terrestrial insects and other aquatic organisms were identified to order (Daly et al. 1998). To aid in analyses, food items were grouped into 11 categories: Ephemeroptera, Plecoptera, Trichoptera, Diptera, Fish, Decapoda, Isopoda, Gastropoda, Terrestrial Insects, Other (comprised of insect families that occurred sporadically in the diets), and Vegetation. Partially digested unidentified insect matter was distributed among the insect categories based on percent occurrence in the diet. Food categories were expressed as a percentage of the overall total weight of the stomach contents (Hyslop 1980).

Analyses

Population Abundance - Population estimates for brown trout were made during June/July using three-pass multiple depletion samples at each site. Population estimates, standard errors, 95% confidence intervals, and capture probability were calculated using maximum-likelihood population estimates with Microfish 3.0 (Van Deventer and Platts 1983). From the population estimates, brown trout density and standing crop and 95% confidence intervals were calculated for each site. Brown trout density was calculated from the following formula:

Density = Population estimate / ha;
where density is the number ha⁻¹ and ha is the area shocked. Brown trout biomass was calculated by:

Biomass = (Density x mean fish weight) / 1000;
where biomass is kg ha⁻¹ and mean fish weight is in g. Differences in population estimates, density and biomass between sites and years were significantly different when 95% confidence intervals failed to overlap.

Relative trout abundance was calculated for all sampling periods for each site as the number of age-1 and older brown trout per 100 m by the equation:

$$\text{Relative Abundance} = (\# \text{ of trout} / \text{distance shocked}) \times 100;$$

where distance shocked is in m. The number of trout caught on the first pass was used for the June sampling periods. Jones and Stockwell (1995) determined that single pass estimates provided a reliable estimate of trout population when compared to three-pass removal estimates.

To test for significant differences in relative abundance 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.

Population Size Structure - 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:

$$\text{RSD} = (\# \text{ of fish} \geq \text{specified length} / \# \text{ of fish} \geq \text{minimum stock length}) \times 100;$$

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 standard errors were calculated for each reach. 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.

Brown Trout Condition - Relative weight (Wr; Wege and Anderson 1978) was used to assess condition of brown trout in the tailwater. Relative weight of trout greater than 140 mm was calculated by the formula:

$$W_r = (W / W_s) \times 100 ;$$

where W is the weight of the individual fish and W_s is the length-specific standard weight of the fish (Wege and Anderson 1978). The standard weight (W_s) equation was proposed by Milewski and Brown (1994) and is:

$$\log_{10} W_s \text{ (g)} = -4.867 + 2.960 \log_{10} \text{TL};$$

where W_s is the standard weight in grams and TL is the total length of the fish in mm. Analysis of variance was used to test for differences in brown trout condition between reaches within a sampling period and to test for differences between seasons within a reach. To determine if tagging influenced brown trout condition, ANOVA was used to test for differences in condition between tagged, untagged, and trout with shed tags. 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 or -4 were checked against data sheets to check for recording or data entry errors. If errors in data entry occurred, values were corrected. If no errors in data entry had occurred, values were excluded from subsequent analyses.

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 the 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 $\alpha=0.05$.

Brown Trout Growth - Brown trout annuli measurements from otoliths were used to back-calculate length-at-age using the formula:

(Focus to annulus distance \div Focus to edge distance) x Total fish 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. Length at age-at-time of capture obtained from otolith analyses was assessed to determine variation in growth between the four reaches. Total length and age-at-time of capture were log transformed, and regression analysis was used to determine growth patterns. Significant differences in the slope of the length-age equation were tested using PROC GLM in SAS to test for differences in growth patterns between reaches. All tests were considered significant at $\alpha=0.05$.

von Bertalanffy growth parameters were determined for each reach. Seasonal growth oscillations (C) and winter points (WP) were identified for each reach using data from recaptured tagged trout using Appledoorn's Method in the FAO-ICLARM Stock Assessment Tools program (FiSAT Version 1.1.0; Gayanilo et al. 2002). Seasonal growth oscillations and WP values were used to assess length-frequency data collected from all sampling periods to determine von Bertalanffy growth parameters, including length at infinity (L_{inf}) and the growth parameter (K), for each reach using ELEFAN I in FiSAT Version 1.1.0 (Gayanilo et al. 2002). Weight at infinity (W_{inf}) was calculated using L_{inf} values and length-weight regression values for each reach. Length and weight curves were developed for each reach based on the von Bertalanffy model.

Tagged brown trout that were collected during two consecutive sampling periods were used to calculate seasonal instantaneous growth rates. Instantaneous growth rates of brown trout were calculated by equations in Ricker (1975). Instantaneous growth in weight (G_W) was calculated by the formula:

$$G_W = ((\log_e w_2 - \log_e w_1) / \text{day}) \times 100;$$

where w_2 and w_1 are in g. Instantaneous growth rates in length (G_L) were calculated by the formula:

$$G_L = (b (\log_e l_2 - \log_e l_1) / \text{day}) \times 100;$$

where b is the constant from the reach-specific length-weight relationship at time 1 (Table 2a.4) and l_2 and l_1 are in mm.

Growth rates in weight and length were tested for differences between reaches within a time period, and growth rates in length were tested within a reach to determine seasonal growth patterns. Because growth rates decrease with increasing age of the fish, initial length was used as a covariate in subsequent analyses on growth rates. Significant differences in growth rates in length were tested using analysis of covariance where the length at the start of the time period was the covariate and reach and season or time period was the treatment.

Environmental characteristics were collected from the tailwater to use in regression analysis to determine the factors that influence the instantaneous growth rates of brown trout in the tailwater. Table 2a.5 lists the environmental variables and the values that were used in the regression analysis during the time periods. Water temperature was recorded at half-hour intervals at a downstream location in each sampling reach (Dam=5.1 Rkm; Special

Regulations=10.2 Rkm; Bassett=17.0 Rkm; Koehler=24.3 Rkm) using StowAway Tidbit Temp Loggers and Optic StowAway Temp Loggers (Onset Computer Corporation, Bourne, MA). Flow was determined at the 12 locations from RQUAL flow predictions. Nongame fish relative abundance (# 100 m⁻¹) was from Job 2 part B. Invertebrate data from 2000 and early 2001 was from Newcomb et al. 2001, and invertebrate data from mid-2001 and 2002 was from unpublished data collected during the duration of the project.

Using stepwise regression with the instantaneous growth rate in length as the dependent variable, environmental variables entered the model at level 0.05 and remained in the model at 0.05. Environmental variables that influenced growth rates were determined for each reach.

Mortality - Due to low numbers of older trout that were aged, use of catch curves to estimate mortality was not feasible due to the inflated mortality estimates due to the reduced number of older trout. Mortality (Z) and survival rates (S) of brown trout were calculated by log-transforming the number of recaptured tagged trout over time. Mortality rates were calculated for each reach based on June 2000 and October 2001 tagged trout. The log-transformed number of recaptured tagged trout was regressed against the log of the day of the recapture period, and the absolute value of the slope of the regression was equal to Z (Ricker 1975). Survival rates were calculated by the formula in Ricker (1975):

$$S=e^{-Z};$$

where S is survival, e=2.71828, and -Z is the negative value of the mortality rate. Differences in mortality rates between reaches were tested using PROC GLM in SAS. If overall tests between the four reaches were significant, pairwise tests between two reaches were conducted to determine which reaches were significantly different from the others. Reaches were significantly different at alpha=0.05.

Mortality rates and 95% confidence intervals were also calculated for each reach by assessing length frequency distributions of all trout collected during each sampling period using the length-converted catch curve method in the FAO-ICLARM Stock Assessment Tools program (FiSAT Version 1.1.0; Gayanilo et al. 2002). Values were corrected using C and P that were determined by assessing growth patterns of recaptured tagged trout. Failure of the confidence interval to overlap indicated that the reaches had significantly different mortality rates.

Brown Trout Diets - Seasonal and longitudinal trends in percent composition of items in the diet were assessed using the log-likelihood ratio test (G-test). Percent diet composition was divided into five categories: 0-0.1, 0.11-24.9, 25-49.9, 50-74.9, and 75-100%. Overall G-tests were performed to test for differences between all reaches and months. If the overall tests were significant, pair-wise G-tests were conducted to determine which reaches and months were significantly different from the other. All tests were considered significant at alpha=0.05.

Relative content wet weight (RWW; Kwak et al. 1992) was calculated for each trout collected using the formula:

$$RWW = (\text{weight of stomach contents/weight of fish}) \times 100;$$

where the weight of the stomach contents and weight of the fish are in g. Mean RWW' was determined for each six hour sampling period and were then used to calculate consumption rates using the following formula:

$$CD = 24 \times S \times R;$$

where CD is the daily ration in g g⁻¹, S is the mean stomach content weight in g, and R is the evacuation rate in % hour⁻¹ (Eggers 1977). S was determined from the formula:

$$S = ? S_i/4;$$

where S_i is the average stomach content weight at time I (RWW') and 4 is the number of samples collected over a 24-hour period. Evacuation rates (R) were calculated from the formula:

$$R=0.0362 e^{(0.114 * T)}$$

where T is the water temperature in °C (Elliott 1991).

Bioenergetics Modeling - The Wisconsin Bioenergetics Model 3.0 (Hanson et al. 1997) was used to test the influence of water temperature and consumption on the growth of brown trout. Because brown trout physiological parameters are unavailable in the bioenergetics software, the steelhead trout *Oncorhynchus mykiss* model parameters (developed by Rand et al. 1993) were used and configured for brown trout with parameters available in literature. The following parameters in the steelhead trout model were changed: 1) the slope coefficient of maximum consumption versus weight (CB) was changed from -0.30 to -0.241 (Stewart et al. 1983), 2) the temperature for maximum consumption (CTO and CTM) was changed from 20 °C to 17 °C (Raat 2003), the respiration temperature where the activity relationship changes (RTL) was changed from 25 °C to 20 °C (Raat 2003), and the excretion parameter that is the proportion excreted versus temperature and ration size (UA) was changed from 0.0314 to 0.0259 (Elliott 1976).

Age-0 and age-1 brown trout classes were identified by length-frequency analysis using Bhattacharya's Method and further defined using Normal Separation Procedures in FiSAT (FiSAT Version 1.1.0; Gayanilo et al. 2002). Mean weights of age-0 and age-1 brown trout in June and October were determined and used as starting and ending weights for the summer growth period. Water temperature values were identified for each reach using observed water temperature data collected at half-hour intervals at a downstream location in each sampling reach using StowAway Tidbit Temp Loggers and Optic StowAway Temp Loggers (Onset Computer Corporation, Bourne, MA). Average daily temperatures were calculated for each reach for each summer and fall-spring time period. Prey items for brown trout were identified during the diet assessment in the current project. Prey energy content values were taken from available literature (Cummins and Wuycheck 1971, Pope et al. 2001). Models were calibrated to fit the observed ending weights given the observed starting weight, temperature and prey proportions.

To determine the influence of water temperature and consumption on the growth of trout, years were modeled under two scenarios: 1) observed temperature values for each year without varying the consumption rate and 2) observed consumption values for each year with no variation in temperature (Railsback and Rose 1999). The influences of temperature and consumption rates were examined using the P parameter in the bioenergetics model. The P -value is a parameter that is calculated that determines the percent of the maximum daily consumption rate that is obtained by the fish. The P parameter is adjusted in the bioenergetics model to fit the observed initial and ending weights given the observed temperature values and prey items.

To assess the influence of changes in temperature on growth, a thermally-neutral P -value was developed. To develop the thermally-neutral P -value, the daily temperature from each reach for each year was averaged to obtain an overall averaged daily temperature for each reach. The overall averaged daily temperatures were used in the model along with the observed initial and ending weights from each year to obtain a year-specific thermally-neutral P -value. The yearly P -values were averaged to develop an overall thermally-neutral P -value. The overall thermally-neutral P -value was then assessed with the initial observed weight and observed temperature values to determine a predicted ending weight (G_T).

To assess the influence of consumption rates on growth, a year-specific P -value was determined for each reach and each year by fitting the model to observed initial and ending weights and observed daily temperature observations. The year-specific P -value was then assessed with the initial observed weight and overall averaged daily temperature values to determine a predicted ending weight (G_P).

The growth under yearly temperature variation (G_T) and the growth under yearly consumption variation (G_P) were regressed on the observed growth (G_O) of the trout to determine if temperature and/or consumption influence growth. The regression of G_T and G_P on G_O tests the hypothesis that variations in growth were driven by temperature or consumption. If the hypothesis is correct, the regression will have a high R^2 , a slope near one and a y -intercept near zero (Smith and Rose 1995).

Because yearly observed P -values are fit in the bioenergetics model under existing conditions, yearly fluctuations in P -values can be the result of changes in the environment, including temperature and flow variables, or changes in the diet composition, which can be the result of changes in prey availability or trout density. To assess the impacts of temperature, flow, food availability, and trout population dynamics on changes in P , stepwise linear regression was used (Railsback and Rose 1999). Variables used and their values are in Table 2a.6 for June-October for age-1 trout and Table 2a.7 for age-0 trout. Methods of collecting the variables were the same with those used in assessing growth values in the previous section.

To assess changes in temperature and consumption, observed yearly P -values were averaged and standard deviation values for each reach were obtained. To assess changes in consumption and temperature, normal consumption values (overall average P -value), high consumption values (averaged P -value plus one standard deviation) and low consumption values (averaged P -value minus one standard deviation) were assessed under varying thermal conditions. The thermal conditions assessed include the following alternative flow scenarios: 1) baseline (9 °C outflow with a baseflow of 1.5 cms, peakflow of 31.8 cms, with a 5 hour release duration), 2) 12 °C outflow (with a baseflow of 1.5 cms, peakflow of 31.8 cms, with a 5 hour release duration), 3) new turbines to reduce flow magnitude (9 °C outflow with a baseflow of 1.5 cms, peakflow of 15.9 cms, with a 10 hour release duration), and 4) steady baseflow (9 °C outflow with a baseflow of 5.5 cms) (Buhyoff et al. In press). Initial weight was averaged from the initial starting weights and modeled under each scenario of consumption (normal, high, or low) and temperature (baseline, 12 °C, reduced magnitude, or constant flow) to determine a predicted ending weight.

Results

Population Abundance

Population Estimates - Population estimates for age-1 and older brown trout in June 2000 were significantly different between the 12 sites. Sites in the Dam and Special Regulations reaches had higher population estimates than the Bassett and Koehler reaches (Table 2a.8). In June 2001, brown trout population estimates were higher in the first 13 km of the tailwater, with significantly lower population estimates being observed in the lower 10 km of the tailwater (Table 2a.9). In June 2002, population estimates were again significantly higher in the first 13 km of the tailwater (Table 2a.10). In 2003, the highest population estimates for age-1 and older trout was observed immediately below the dam, with the brown trout population estimates being significantly lower in downstream locations (Table 2a.11).

Population estimates for age-1 and older brown trout were significantly different between years within a site (Table 2a.12). Although there was no year that was consistently higher or lower among all the sites, population estimates were significantly lower in 2001 in 8 of 12 sites when compared to 2000, 2002, and 2003 (Table 2a.12). Population estimates increased from 2002 to 2003 at 7 of 12 in the tailwater; however, sites in the lower 7 km of the tailwater showed decreasing population estimates from 2002 to 2003 (Table 2a.12).

Population estimates for age-0 brown trout during the June samples were highly variable among the sites and between years. Age-0 brown trout in the Smith River had low capture probabilities, which resulted from non-descending catch rates and high numbers of trout being captured on consecutive passes. Low capture probabilities have also been shown in other studies on young brown trout (Habera et al. 2002). Low capture probabilities resulted in wide confidence intervals for the population estimates. In 2000 (Table 2a.13), 2001 (Table 2a.14), 2002 (Table 2a.15), and 2003 (Table 2a.16), population estimates for age-0 brown trout were highest from 3.4 to 8.9 km downstream of Philpott Dam, with lower population estimates being observed in the lower 8 km of the tailwater.

Population estimates for age-0 brown trout were higher in 2002 than other years sampled, while in 2003, population estimates were lower than in previous year (Table 2a.17). During 2002, flow magnitude was reduced, which caused average daily flows to be lower than in previous years, and in 2003, flow magnitude and duration were increased over previous years which resulted in increased averaged daily flow values. Age-0 brown trout population estimates in June in the Smith River were negatively related to daily flow rates from April to June in the tailwater, with lower average daily flows resulting in a higher population estimate (Figure 2a.1A).

Reduction of a year-class due to increased flow magnitude and duration during the first year can have an impact on the trout fishery in the years following increased flow events. In the Smith River, average daily flows during April to June when trout are age-0 were negatively correlated to population estimates of age-1 and older trout in the following year (Figure 2.A.1B). When daily flow values were high for age-0 trout from April to June, population estimates of age-1 and older in the following were low. Age-1 and older brown population estimates in 2003 were greater than in previous years, which resulted from low flows creating a strong year class in 2002. This also may indicate that year-class strength potential is formed during the first few months after swim-up based on flow rates.

Population density-Age-1 and older brown trout population densities were different between sites in 2000 (Table 2a.8), 2001 (Table 2a.9), 2002 (Table 2a.10), and 2003 (Table 2a.11). During all years, highest density estimates occurred in the first 11 km of the tailwater, with lower densities occurring the lower 13 km of the tailwater. There were variations in densities between the years, with the highest densities occurring in 2000, 2002, and 2003 and the lowest densities occurring in 2001 (Table 2a.18).

Population density of age-0 trout was highly variable among sites and years. From 2000-2003, densities were highest from 3.4 to 8.9 km downstream of Philpott Dam (Tables 2a.13-2a.16). Densities of age-0 brown trout were highest in 2002 and lowest in 2003 (Table 2a.19). *Brown trout standing crop*-The standing crop of age-1 and older brown trout was highest from 3.4 to 15.9 km in 2000 (Table 2a.8). In 2001, the standing crop was highest from 4.2 to 12.6 km (Table 2a.9). The standing crop was highest from 0.5 to 12.6 km in 2002 (Table 2a.10). In 2003, the standing crop of age-1 and older trout was highest from 0.5 to 14.3 km (Table 2a.11).

Between the years, standing crop was highest in 2000 and 2003, with a lower standing crop occurring in 2001 and 2002 (Table 2a.20).

The standing crop of age-0 brown trout in the Smith River was highest from 3.4 to 18.9 km in 2000 (Table 2a.13), 3.4 to 15.9 km in 2001 (Table 2a.14), 3.4 to 20.1 km in 2002 (Table 2a.15), and 3.4 to 6.2 km in 2002 (Table 2a.16). Standing crop of age-0 brown trout was highest in 2002 and lowest in 2001 (Table 2a.21).

Brown trout biomass and density estimates from the Smith River are high in comparison to other southeastern rivers (Table 2a.22). Biomass and density estimates in the Dam and Special Regulations reaches are consistently among the highest estimates reported for the region; however, biomass and density estimates in the Bassett and Koehler reaches are among the lowest reported (Table 2a.22).

Brown Trout Relative Abundance - Relative abundance of age-1 and older brown trout was significantly different between the four reaches during all sampling periods except the June 2002 sampling period ($P=0.0771$) and October 2003 ($P=0.0845$) (Table 2a.23). On the sampling date there were differences in relative abundance, abundances in the Koehler Reach were significantly different than the Dam or Special Regulations reaches (Table 2a.23). Relative abundance was greatest in the Dam and Special Regulations reaches during all sampling periods (Table 2a.23). The Dam and Special Regulations reaches were not significantly from each other during any of the sample dates.

Relative abundance of age-0 brown trout was not significantly different between reaches for all sampling periods except June 2000 ($P=0.0309$) (Table 2a.24). In June 2000, the relative abundance of age-0 brown trout was significantly higher in the special Regulations Reach than the Koehler Reach (Table 2a.24). Relative abundance of age-0 was highest in the Special Regulations Reach from 2000 through 2002; however, in 2003, relative abundance was highest in the Dam Reach (Table 2a.24).

Brown trout relative stock density- Significant differences were found between the reaches within 2000 ($P=0.0492$), 2002 ($P=0.0477$) and 2003 ($P=0.0128$) for brown trout greater than 230 mm (Table 2a.25). The Koehler Reach had significantly higher RSD values than the Dam Reach in 2000 and 2003. In 2001, there were no significant differences between the reaches ($P=0.5800$). In addition, no significant differences were found between years in the Dam ($P=0.2477$), Special Regulations ($P=.4277$), Bassett ($P=0.0878$), and Koehler ($P=0.8129$) reaches.

Relative stock density indices for brown trout greater than 300 mm were significantly different between reaches in 2000 ($P<0.0001$), with the Dam and Special Regulations reaches having significantly lower RSD values than trout from the Bassett and Koehler reaches (Table 2a.26). There was also significant differences between the RSD-300 values in 2001 ($P=0.0021$), with the Dam Reach having significantly lower RSD values than the Koehler and Bassett reaches (Table 2a.26). In 2002, there was a significant difference between reaches ($P<0.0001$), with the Koehler Reach having significantly higher RSD values than the Dam, Special Regulations, and Bassett reaches (Table 2a.26) No significant differences in RSD-300 were observed between reaches in 2003 ($P=0.6020$). No significant differences in RSD-300 were observed between years in the Dam ($P=0.4657$), Special Regulations (0.2775), and Koehler ($P=0.4693$) reaches; however, significant differences between years were observed in the Bassett Reach ($P=0.0024$), with the RSD-300 values in 2002 being significantly lower than 2000 (Table 2a.26).

The size distribution of brown trout in the Smith River is dominated by small sized individuals. The low RSD-300 values indicate that few trout in the Dam and Special Regulations

reaches are attaining sizes of 300 mm and larger. Higher RSD values in the Bassett and Koehler reaches indicate that there is a greater percentage of the overall trout population that attain larger sizes in the downstream reaches. In the 1980s, RSD-230 values for brown trout in the Smith River ranged from 60-70, but have steadily declined over the last two decades (Anderson et al. 2003). Although no significant differences in RSD-230 were observed between the years in any of the reaches, the RSD-230 values decreased in the Dam, Special Regulations, and Bassett Reaches from 2000 to 2003.

Brown Trout Condition

Relative Weight - Relative weight values for brown trout in the Smith River were significantly different between the reaches during the sample dates (Table 2a.27). No consistent trends between the reaches were identified over all sampling periods. During 2000 and 2001, W_r values in the Dam Reach were lower than the downstream reaches, with trout in the Koehler Reach having the highest W_r values; however, in 2002 and 2003, W_r values were higher in the Dam Reach than in the downstream reaches (Table 2a.27).

Relative weight values between the sample months were not significantly different in the Dam Reach ($P=0.0632$). Relative weight values in the Special Regulations Reach were significantly different between months ($P<0.0001$), with April having significantly higher W_r values than June and October (Table 2a.28). The W_r values in the Bassett Reach were also significantly different between months ($P<0.0001$), with April being significantly higher than June and October and October being significantly lower than June (Table 2a.28). The Koehler Reach also had significant differences in W_r between sample months ($P<0.0001$), with the W_r values being significantly lower in June (Table 2a.28).

Relative weight values for nontagged, tagged, and trout with shed tags were not significantly different in the Dam ($P=0.4332$) and Special Regulations ($P=0.8368$) reaches (Table 2a.29). There were significant differences between tagged and nontagged trout in the Bassett Reach ($P=0.0011$), with nontagged trout having a higher W_r (Table 2a.29). There was also a significant difference in the Koehler Reach ($P=0.0017$), with tagged trout having higher W_r values than nontagged or trout who had shed the tag (Table 2a.29). Although differences were detected that indicated that tagged trout in the Bassett Reach had lower W_r values, trout condition in the other reaches were not negatively impacted by tagging activities.

Values for W_r of 93 are considered average for fish species because of the techniques that are used in the development of the standard weight equation (Murphy et al. 1990, Hebdon and Hubert 2001). Average W_r values for brown trout in the Smith River ranged from 84 in the Bassett Reach in October 2003 to 93 in the Koehler Reach during October 2001. The W_r values in the Smith River are below the values that have been reported for other tailwater salmonid species, which ranged from 93-104 in the North Platte River, Wyoming, 92-110 in the Big Horn River, Wyoming, and from 85-94 in the Shoshone River, Wyoming, for rainbow and cutthroat trout *Oncorhynchus clarki* during the winter months (Hebdon and Hubert 2001). The W_r values for Smith River brown trout are low in relation to other populations, which may indicate that food resources may be limited in the Smith River. However, studies on rainbow trout have indicated that mortality due to starvation was unlikely unless the W_r values fell below 75 (Hebdon and Hubert 2001). While the trout in the Smith River have lower W_r values than average, it is unlikely that the low condition is leading to additional mortality due to starvation.

Length-weight regressions- Brown trout length-weight regressions were significantly different between the reaches within each sampling period during all periods except October 2002. The

Dam Reach was significantly different from the Koehler Reach during six of the 11 sampling periods, with the Koehler Reach having a steeper length-weight regression slope than the Dam Reach (Table 2a.4). Although the brown trout length-weight regression slope was significantly different during three sampling periods between the Bassett and Koehler reaches (June and August 2000 and July 2003), the Bassett and Koehler reaches were most similar in length-weight relationships during all time periods when compared to the relationships observed between the other reaches (Table 2a.4).

The Koehler Reach had the greatest regression slopes during eight of the 11 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 nine out of the 11 sampling periods, which indicates that brown trout in the upper 10 km have a lower body weight than trout in the lower 13 km of the tailwater for a given length.

Brown Trout Growth

Brown Trout Length-at-Age - Brown trout otoliths were collected from 1066 brown trout from June 2000 to October 2003, which included 319 otoliths from the brown trout diet study. 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. The checks can be misidentified as annuli, thus potentially over-aging the trout. Otoliths from 165 brown trout (15% of the total aged) were aged by two readers for validation of reader accuracy. There was a 78% reader agreement rate on the trout that were double aged. Only 2% of the trout that were aged were 4 years of age or older, and the oldest trout aged was age 5.

Mean back-calculated length-at-age was lowest in the Dam Reach for trout ages 1 through 3 (Table 2a.30). 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 (Table 2a.30). 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 the reaches ($P = 0.0009$), with trout from the Dam Reach being smaller than trout from all other reaches (Table 2a.30). 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 (Table 2a.30).

When compared to back-calculated length-at-age for other trout populations, age 1 trout in the Smith River are within the range of sizes that are reported for other populations (Table 2a.31). However, by age-2, brown trout in the Smith River are in the lower range of sizes when compared to other populations (Table 2a.31). By age 3, trout in the Smith River are 30-80 mm smaller in size than brown trout in other populations (Table 2a.31). In the Smith River, increases in length are minimal after the trout reach 200 mm in length.

Significant differences in length at time of capture between reaches were detected with regression analysis. Trout in the Dam Reach had a significantly different regression slope than the Special Regulations ($P < 0.0001$), Bassett ($P < 0.0001$), and Koehler ($P = 0.0002$) reaches (Figure 2a.2). No significant differences in slope occurred between the Special Regulations, Bassett, or Koehler reaches. Trout in the Dam Reach were smaller than brown trout in the three downstream reaches at the same age at capture.

von Bertalanffy Growth Parameters - Seasonal growth oscillations (C) and WP values were identified for each reach using data from recaptured tagged trout. Trout in the Dam and

Bassett reaches had higher values for C that trout in the Special Regulations and Koehler reaches (Table 2a.32). Seasonal growth oscillation values indicate that there is a higher seasonal fluctuation in growth in the Dam and Bassett reaches than in the Special Regulations and Koehler Reaches. Winter point values decreased with increasing distance from Philpott Dam (Table 2a.32). Winter point values represent the time of the year when growth is slowest. The high WP value in the Dam Reach indicates that the time when growth is slowest is later in the year (November), with the lower WP values in the Special Regulations, Bassett, and Koehler reaches indicating that growth is slowest early in the year (February and April) (Table 2a.32). Length at infinity and W_{inf} values were higher at downstream locations and were lowest in the Dam Reach (Table 2a.32). Increased L_{inf} and W_{inf} values downstream were expected because fish in the lower portions of the river attain a larger size at age than trout in the Dam Reach, so if a trout were to live to infinity, they would be of larger sizes in the downstream reaches of the tailwater. Growth parameter (K) values showed no longitudinal pattern in the tailwater; although, K was higher in the Dam Reach than the downstream reaches (Table 2a.32). Growth curves indicate that little growth in length occurs after age 4 (Figure 2a.3A), and growth in weight slows after age 6 (Figure 2a.3B).

Instantaneous Growth in Weight - Instantaneous growth rates in weight for brown trout in the Smith River tailwater were significantly different between the four reaches during all time periods sampled. During June 2000 to October 2000, growth rates were significantly higher in the Bassett and Koehler reaches than in the Dam and Special Regulations reaches (Table 2a.33). Significant differences in growth rates in weight were observed during the October 2000 to April 2001 time period with the trout in the Koehler Reach having significantly higher growth rates than trout in the other three reaches (Table 2a.33). The trout in the Dam Reach had significantly slower growth rates than trout in the three downstream reaches (Table 2a.33). Instantaneous growth rates in weight in the Smith River were lower than growth rates observed in other river systems. In Laurel Fork, Tennessee, growth rates were highest during the late summer, with lower growth rates observed during winter months (Strange et al. 2000) (Table 2a.34).

Instantaneous Growth in Length - Significant differences were detected between the reaches during all time periods except the June 2001 to October 2001 timeperiod (Table 2a.35). During the June 2000-October 2000 time period, instantaneous growth rates in length were significantly different in the Bassett and Koehler reaches than in the Dam and Special Regulations reaches, with the brown trout in the Bassett and Koehler reaches having higher growth rates than trout in the Dam and Special Regulations reaches (Table 2a.36). During the October 2000-April 2001 time period, all reaches were significantly different than the other reaches, with the Koehler Reach having the highest growth rate, followed by the Special Regulations and Bassett reaches, while the trout in the Dam Reach had the lowest growth rates (Table 2a.36).

From April 2001 to June 2001, the Bassett Reach was significantly different than the Dam Reach, with trout in the Dam Reach having the slowest growth rates (Table 2a.36). No other significant differences between reaches occurred in the growth rates during the April 2001-June 2001 time period. During the June 2001-October 2001 time period, the instantaneous growth rates in length were not significantly different between the reaches. From October 2001 to April 2002, trout in the Koehler Reach had significantly higher growth rates than trout in the other three reaches (Table 2a.36). While the Special Regulations and Bassett reaches were not significantly different from each other, trout in the Dam Reach had significantly lower growth

rates than the Special Regulations and Bassett reaches during the October 2001-April 2002 time period (Table 2a.36).

During April 2002-June 2002, the Bassett Reach had significantly lower growth rates than the Koehler Reach (Table 2a.36). No other reaches had significantly different growth rates during the April 2002-June 2002 time period. From June 2002-October 2002, trout in the Koehler Reach had significantly higher growth rates than trout in the Special Regulations and Bassett Reaches (Table 2a.36). Trout in the Dam Reach had significantly higher growth rates than trout in the Special Regulations Reach during the June 20002-October 2002 time period (Table 2a.36).

Seasonal Trends in Growth Rates in Length - Brown trout in the Smith River had seasonal patterns of growth in the Dam, Special Regulations, and Bassett reaches, but no seasonal trends in growth were observed in the Koehler Reach (Table 2a.37). In the Dam Reach, instantaneous growth rates in length were significantly different among all seasons with the highest growth rates during the April to June period and lowest during the October to April period (Table 2a.38). In the Special Regulations Reach, instantaneous growth rates in length were highest during the April to June period, but there was no significant difference in growth rates between June to October and October to April seasons (Table 2a.38). In the Bassett Reach, growth rates were highest during the June to October time period, and no significant differences were observed between the April to June and October to April time periods (Table 2a.38).

Overall seasonal trends indicate that trout in the Dam and Special Regulations reaches have higher growth rates during the April to June time period, while trout in the Bassett and Koehler reaches have higher periods of higher growth during June to October. In the Dam, Bassett, and Koehler reaches, growth was slowest during October to April, while the slowest growth period for trout in the Special Regulations Reach was from June to October. The low growth rates during the October to April time periods were consistent with seasonal growth patterns observed in the von Bertalanffy parameters.

Seasonal growth rates in weight for brown trout in the Smith River were lower in October to April than during other time periods. Optimal water temperatures for brown trout growth range from 10-15.5°C (Jobling 1981). Water temperatures during the winter months in the Smith River are reduced below the thermal regime that provides optimum growth for brown trout, thus reductions in growth rates of brown trout would be expected during winter months. Similar studies have shown reduced growth of brown trout during the winter months when water temperatures are below the thermal optimum for growth (Cada et al. 1987b; Elliott 1994; Strange et al. 2000).

Growth rates for brown trout in the Smith River during June to October were lower in relation to other streams (Table 2a.34). Water temperatures in the Smith River during the June to October time period are generally within the thermal optimal growth range in lower portions of the tailwater; however, in the portion of the tailwater immediately below the dam, water temperatures remain below the thermal optimal range throughout the summer months under normal generation release schedules. In addition to the low water temperatures in the first 5 km downstream of the dam, food resources are also reduced. Relative abundance of nongame fish in the first 5 km of the Smith River was low, averaging 3 fish 100 m⁻¹ (range 0-9), from 2000 to 2002 (See Job 2 part B). In addition to low nongame fish abundance, invertebrate biomass is also reduced in the first 5 km of the tailwater; however, invertebrate density in the area in the Dam Reach is high due to the substantial number of Chironomid larvae that are present (Newcomb et al. 2001).

Environmental Factors that Influence Growth in Length - In the four reaches, stepwise regression indicated that temperature was important in determining the growth in length of brown trout. In the Dam Reach, the regression model was:

$$\text{Growth} = 0.0339 + 0.115 (\% \text{ time } >15.5 \text{ } ^\circ\text{C}) + 0.000211 (\text{Brown trout density}) - 0.0344 (\text{Hourly temperature flux});$$

where brown trout density is the density of age-1 and older trout (#/ha) and hourly temperature flux is the maximum decrease in temperature ($^\circ\text{C}$) observed in a one hour time period. The model only explained 11% of the variation in growth.

In the Special Regulations Reach, the model that explained variations in growth was:

$$\text{Growth} = 0.391 + 0.000422 (\text{Mean invertebrate density}) + 0.0625 (\text{Hourly temperature flux}) - 0.000543 (\text{Mean EPT density}) - 0.00513 (\% \text{ time } 10\text{-}15.5 \text{ } ^\circ\text{C}) - 0.0356 (\text{Mean daily flow});$$

where invertebrate density is the $\#/m^2$, hourly temperature flux is in $^\circ\text{C}$, mean EPT is the mean number of Ephemeroptera, Plecoptera, and Trichoptera per m^2 , and mean daily flow is in cms. The model failed to explain a large percent of the variation in growth rates (11%).

The environmental factors model that explained changes in growth rates in the Bassett Reach was:

$$\text{Growth} = 0.136 + 0.00302 (\% \text{ time } 10\text{-}15.5 \text{ } ^\circ\text{C}) - 0.000166 (\text{Mean invertebrate density});$$

where the mean invertebrate density is the $\#/m^2$. The model explained 20% of the variation in growth.

In the Koehler Reach, the model that best described variations in growth was:

$$\text{Growth} = 0.224 - 0.0102 (\% \text{ time } 10\text{-}15.5 \text{ } ^\circ\text{C}) - 0.00401 (\% \text{ time } <10 \text{ } ^\circ\text{C}) + 0.0117 (\text{Maximum daily flow});$$

where maximum daily flow is in cms. The model explained 19% of the observed variation in growth.

While temperature variables were important in explaining the variations in growth in all four reaches, the percent time when the temperature was in the optimal range (10-15.5 $^\circ\text{C}$) was negatively related to growth in the Special Regulations and Koehler reaches. No temperature variable was consistently positively or negatively related to growth in all four reaches. Elliott (1994) stated that water temperature is the main factor that influences growth in brown trout, so it understandable that temperature variables should have appeared in all models to explain growth in Smith River brown trout.

Parameters that assessed available food items for brown trout, including invertebrates and nongame fish, were only in the regression models in the Special Regulations and Bassett reaches. While the relationship between mean invertebrate density was positive in the Special Regulations Reach, it was negatively related to growth in the Bassett Reach. Variables that addressed changes in flow were in the regression model in Special Regulations and Koehler Reaches. Invertebrates rates that were explained on the Smith River were obtained from benthic studies; however, Bachman (1984) argued that growth was a function of available drift-prey and the available drift was dependent upon water velocities. Drift rates were not assessed in the Smith River, so it is unclear if drift rates would have important in determining growth rates.

Trout density was only in the model in the Dam Reach. Several authors have indicated that growth in trout is density-independent, so it should be expected that trout population characteristics would not be important variables in the growth models. Elliott (1994) indicated that growth rates of brown trout were not dependent upon trout densities. In addition, Mortensen

(1982) observed density-independent growth in brown trout. Other studies on salmonid species have also indicated that growth was not density-dependent (Cooper et al. 1962, Chapman 1965).

Mortality

Regression of Recaptured Trout Over Time - Mortality rates for trout tagged in June 2000 ranged from 0.779 ($S=0.459$) in the Dam Reach to 1.145 ($S=0.318$) in the Koehler Reach (Figure 2a.4). Mortality rates increased with increasing distance downstream of Philpott Dam; however, homogeneity of slopes tests indicated that the mortality rates between reaches were not significantly different ($P=0.5357$).

Mortality rates for trout tagged in October 2001 were higher than the mortality rates for trout tagged in June 2000. Mortality rates for trout tagged in October 2001 ranged from 1.628 ($S=0.196$) in the Dam Reach to 2.329 ($S=0.097$) in the Koehler Reach (Figure 2a.5). As with the trout tagged in June 2000, mortality rates for trout tagged in October 2001 were higher in downstream reaches than in upstream reaches. Although mortality rates increased with increasing distance downstream of the dam, the mortality rates between the reaches were not significantly different ($P=0.8445$).

Mortality (length-frequency data) - Mortality rates were also calculated for each reach by assessing the length-frequency data for brown trout collected by electrofishing that was corrected for seasonal growth oscillations and winter point values (Table 2a.32). Mortality rates calculated using length frequency data for brown trout indicated a lower mortality rate in the Dam Reach ($Z=0.88$) when compared to the lower three reaches in the Smith River tailwater (Figure 2a.6). Unlike the mortality rates from the tagged trout, mortality rates, based on length-frequency distributions, were higher in the Special Regulations and Bassett reaches and showed no increasing trend in mortality rates with increasing distance from Philpott Dam. Mortality rates were significantly different between the reaches, with the Special Regulations Reach having a significantly higher mortality rate than trout in the Dam, Bassett, and Koehler reaches (Figure 2a.6). Brown trout mortality rates in the Bassett Reach were significantly higher than mortality rates in the Dam and Koehler reaches (Figure 2a.6). There were no significant differences in mortality rates for trout in the Dam and Koehler reaches.

High mortality rates, coupled with slow growth rates, limit the potential for brown trout in the Smith River tailwater to attain large sizes before removal from the population due to natural causes. Previous creel surveys in the tailwater indicate that anglers in the Smith River harvest a small percentage (5%) of brown trout from the river (Hartwig 1998). Low harvest rates coupled with high overall mortality rates, indicate that natural mortality is having a greater impact on the brown trout population than harvest mortality.

Diet Study

In 2002, 320 brown trout were collected over four months from four reaches in the Smith River for stomach content analysis. Total lengths of the brown trout had a wide distribution over all sampling periods (Table 2a.39). Fifty-two different aquatic invertebrate families were identified in the stomach contents of the brown trout. A total of 26 different orders of aquatic invertebrates, terrestrial invertebrates, and aquatic organisms were identified in the stomach contents. The number of trout with empty stomachs was low, with only 1 empty stomach being observed in February and 2 empty stomachs in May (Table 2a.39).

Longitudinal shifts in trout diet composition - There were longitudinal differences in percent composition in Ephemeroptera during May ($G=25.63$) and December ($G=30.97$; Table

2a.40). In May, pairwise tests between the reaches for Ephemeroptera indicated that there were significant differences between the Dam Reach and the Special Regulations and Koehler reaches (Table 2a.41). Trout in the Dam Reach had a higher percentage of Ephemeroptera in their diets than trout in the Special Regulations and Koehler reaches (Appendix A). In December, Ephemeroptera was significantly different between the Dam Reach and the Special Regulations and Bassett reaches, and the Special Regulations Reach was significantly different than the Koehler Reach (Table 2a.41). Higher percentages of Ephemeroptera were observed in the diets of trout from the Special Regulations and Bassett reaches (Appendix A).

Plecoptera had significant differences between the reaches during September ($G=29.50$; Table 2a.40). Significant differences were observed in Plecoptera between the Dam Reach and the Special Regulations and Bassett reaches (Table 2a.41). A higher percentage of Plecoptera was observed in the diets of trout in the Special Regulations and Bassett reaches than in trout from the Dam Reach (Appendix A).

There were longitudinal differences in percent composition of Trichoptera in May ($G=44.77$), September ($G=28.59$), and December ($G=32.25$; Table 2a.40). In May, pairwise tests between the reaches for Trichoptera indicated that the Dam Reach was significantly different than the Special Regulations, Bassett and Koehler reaches, while the three downstream reaches had no significant differences between them (Table 2a.41). Trichoptera were not observed in the stomach contents in the Dam Reach during May, but Trichoptera was a dominant item in the stomach contents of trout from the three downstream reaches (Appendix A). In September, Trichoptera composition was significantly different between trout in the Dam Reach and the Special Regulations and Bassett reaches, with trout in the Special Regulations and Bassett reaches consuming a greater percentage of Trichoptera than trout in the Dam Reach (Table 2a.41, Appendix A). Trichoptera was significantly different in December between the Dam Reach and the Special Regulations and Bassett reaches, and the Special Regulations Reach was significantly different than the Koehler Reach (Table 2a.41). Higher percentages of Trichoptera were observed in the diets of trout from the Special Regulations and Bassett reaches (Appendix A).

Diptera had longitudinal differences in September ($G=47.99$) and December ($G=26.06$; Table 2a.40). In September, Diptera was significantly different between the Koehler Reach and Dam, Special Regulations, and Bassett reaches, with trout in the Koehler Reach consuming a lower percentage of Diptera than trout in the other reaches (Table 2a.41, Appendix A). In December, trout in the Dam Reach had a significantly different percentage of Diptera in their diets than trout in the Bassett and Koehler reaches (Table 2a.41). Trout diets in the Dam Reach were composed of a greater percentage of Diptera than trout diets in the Special Regulations and Bassett reaches (Appendix A).

There were significant differences between the reaches in fish composition in December ($G=24.73$; Table 2a.40). There were significant differences in the percentage of fish in the diets between the Koehler Reach and the Special Regulations and Bassett reaches (Table 2a.41). Trout diets in the Koehler Reach were composed of a higher percentage of fish than trout in the Special Regulations and Bassett reaches (Appendix A).

Decapoda had significant longitudinal trends in December ($G=35.01$; Table 2a.40). Decapoda composition was significantly different between the Dam Reach and the Bassett Reach (Table 2a.41). Decapoda was not observed in the trout diets in the Dam Reach during December but were present in the trout diet in the Bassett Reach (Appendix A).

There were longitudinal differences in diet composition of Isopoda during February (G=57.24), May (G=43.41), September (G=72.30), and December (G=58.63; Table 2a.40). During all four sampling months, the Dam Reach was significantly different than the Special Regulations, Bassett, and Koehler reaches (Table 2a.41). Trout diets in the Dam Reach were composed of a higher percentage of Isopoda than trout diets in the three downstream reaches during all months of the study (Appendix A).

Gastropoda had longitudinal differences during May (G=30.53; Table 2a.40). The Koehler Reach was significantly different than the Dam and Special Regulations reaches for Gastropoda (Table 2a.41), with Gastropoda being absent in the diets of trout in the Koehler Reach (Appendix A).

Terrestrial insects were significantly different between the reaches during May (G=23.07; Table 2a.40). The Koehler Reach was significantly different than the Dam and Special Regulations reaches for terrestrial insects (Table 2a.41). Trout diets in the Koehler Reach were composed of a higher percentage of terrestrial insects than diets in the Dam and Special Regulations reaches (Appendix A). No other significant differences were observed for terrestrial insects in May.

Trout diets in the Dam Reach were different than the Special Regulations, Bassett, and Koehler reaches. Thermal regimes and flushing of the area during generation periods limits the types of fauna that are present in the Dam Reach. Invertebrate composition in the Dam Reach of the Smith River is dominated by Chironomidae and Isopoda (Newcomb et al. 2001), which were both important diet items for the trout in the Dam Reach. As distance downstream of Philpott Dam increases, Ephemeroptera, Plecoptera, and Trichoptera increase in abundance, while Chironomidae and Isopoda decrease in abundance (Newcomb et al. 2001). These trends in invertebrate composition in the Smith River help explain why the trout diets in the Dam Reach were different than the three downstream reaches.

The abundance of non-game fish in the Smith River increases with increasing distance downstream of Philpott Dam (See Job 2 part B). In the trout diets, fish occurred in the diet of 35% of the trout collected from the Koehler Reach during May, September, and December. In contrast, fewer than 10% of trout in the Special Regulations and Bassett reaches contained fish in their diets, even though fish were available in the reaches. Based upon scatter-plots of nongame fish abundance and diet composition, it appears that relative abundance of 100 nongame fish per 100 meters is a minimum threshold before fish appear in the brown trout diets.

Seasonal Shifts in Diet Composition - Seasonal shifts in diet composition were observed Ephemeroptera in all four reaches (Table 2a.42). In the Dam Reach, percent composition of Ephemeroptera in February and May were significantly different than September and December (Table 2a.43). Trout diets in the Dam Reach were composed of a higher percentage of Ephemeroptera in February and May than in the later sampling months (Appendix A). In the Special Regulations and Koehler reaches, February was significantly different than May, September, and December (Table 2a.43), with diets in February being composed of a higher percent of Ephemeroptera than in later months (Appendix A). In the Bassett Reach, February was significantly different than May, September, and December, and May was significantly different than September (Table 2a.43). Trout diets in February were composed of a higher percent of Ephemeroptera than in later months, and May trout diets had a higher percentage of Ephemeroptera than the diets in September (Appendix A).

Seasonal differences were observed in the percent of Plecoptera in the diet contents in all four reaches (Table 2a.42). In the Dam Reach, trout in December had a significantly higher

percentage of Plecoptera in the diet than trout during May and September (Table 2a.43, Appendix A). In the Special Regulations Reach, trout diets in May were comprised of a lower proportion of Plecoptera than in the other three months (Table 2a.43, Appendix A). Trout diets in May were comprised of a lower percentage of Plecoptera than trout diet in September and December in the Bassett Reach (Table 2a.43, Appendix A). In the Koehler Reach, a significantly higher percentage of Plecoptera was observed in the diets in February than in the later three months (Table 2a.43, Appendix A).

Trichoptera diet composition was seasonally different in the Special Regulations ($G=27.52$) and Bassett ($G=23.71$) reaches (Table 2a.42). In the Special Regulations Reach, trout in February had a lower proportion of Trichoptera in the diet than in September (Table 2a.43, Appendix A). Trout diets in the Special Regulations Reach during December were significantly different than May and September (Table 2a.43), with trout diets being comprised of a lower portion of Trichoptera in December than in May and September (Appendix A). In the Bassett Reach, diets in September had a significantly higher percentage of Trichoptera in the diets than in February (Table 2a.43, Appendix A).

In the Koehler Reach, there were seasonal differences in the percent composition of Diptera ($G=36.75$) and Fish ($G=25.54$; Table 2a.42). In September, trout diets were composed of a lower proportion of Diptera than in February and May (Table 2a.43, Appendix A). In December, trout diets in the Koehler Reach had a higher percentage of fish than in the other three months (Table 2a.43, Appendix A).

There was a seasonal shift in the diet composition of Decapoda in the Bassett Reach ($G=21.88$; Table 2a.42). Diet composition in December was significantly different than February and May (Table 2a.43), with trout diets being comprised of a higher percentage of Decapoda in December than in February and May (Appendix A).

Seasonal diets shifts occurred in the Dam Reach for Isopoda ($G=27.19$; Table 2a.42). Trout diets in May were comprised of a higher percentage of Isopoda than in February and September (Table 2a.43, Appendix A). In September, trout had a lower percentage of Isopoda in the diet than in December (Table 2a.43, Appendix A).

The percentage of Gastropoda in the trout diets were significantly different between the seasons in the Bassett ($G=35.79$) and Koehler ($G=26.70$) reaches (Table 2a.42). Trout in the Bassett Reach had a lower percentage of Gastropoda in the diet in February than in September and December (Table 2a.43, Appendix A). In May, trout in the Bassett Reach had a significantly lower percentage of Gastropoda in the diet than in December (Table 2a.43, Appendix A). In the Koehler Reach, Gastropoda comprised a greater portion of the diet in December than in February and May (Table 2a.43, Appendix A). In September, trout diets contained a greater percentage of Gastropoda than in May (Table 2a.43, Appendix A).

Seasonal shifts in the percentage of terrestrial insects in the diet occurred in all four reaches (Table 2a.42). In the Dam and Special Regulations reaches, terrestrial insects comprised a significantly higher portion of the diet in September than in the other three months (Table 2a.43, Appendix A). Brown trout in the Bassett and Koehler reaches had a significantly higher portion of terrestrial insects in the diets than in February and December (Table 2a.43, Appendix A).

Diet items that were consumed by brown trout in the Smith River were consistent with the findings of other diet studies on salmonids. Kreivi et al. (1999) noted Ephemeroptera, Diptera, and Trichoptera were common diet items for juvenile brown trout in Finland streams. Stream dwelling trout in Norway ate small Chironomidae when trout were young and switched to Plecoptera, Trichoptera, and Simuliidae at older ages (Jonsson and Gravem 1985). In a study on

trout in Wyoming tailwaters, trout consumed Diptera, Ephemeroptera, and Trichoptera during winter months (Hebdon and Hubert 2001). During the May and September samples, terrestrial insects were a significant part of the brown trout diets in the Smith River, which is consistent with work done by Cada et al. (1987a), who observed a high portion of terrestrial insects in brown trout diets during summer and autumn.

Brown trout in the Koehler Reach had a higher portion of fish in the diet compared to the three upstream reaches. Several papers have addressed the piscivorous nature of brown trout (Garman and Nielsen 1982, L'Abee-Lund et al. 2002). In a Virginia stream, large-sized stocked brown trout typically consumed fish, including fantail darters *Etheostoma flabellare* and central stoneroller *Campostoma anomalum* (Garman and Nielsen 1982). L'Abee-Lund et al. (2002) observed a shift from invertebrates to fish in the diets of brown trout after the introduction of two prey fish species; however, fish occurred less frequently in the trout diets several decades after the fish introduction. Although several authors have noted increased growth rates by piscivorous brown trout (Garman and Nielsen 1982, Aass et al. 1989, Jonsson et al. 1999), L'Abee-Lund (2002) failed to observe increased growth rates in piscivorous brown trout. If growth rates of brown trout in the Smith River are driven by diet items, then the observed higher growth rates in trout in the Koehler Reach may be due to the higher rate of piscivory by trout in the reach.

Consumption Rates - Consumption rates were highest in the Special Regulations and Koehler reaches in February, May, and September (Table 2a.44). In December, consumption rates were highest in the Bassett and Koehler reaches (Table 2a.44). Consumption rates were lowest in the Dam Reach in relation to all other reaches in all sampling months except December, in which the Special Regulations Reach had the lowest consumption rates (Table 2a.44). No significant differences in consumption rates were observed between the four reaches during the four months. There was a significant difference in consumption rates in the Special Regulations reach between February and December with consumption rates being higher in February than in December.

Because of the variability in RWW within a 6 hour sampling period was high, the confidence intervals around the consumption rates were high, and thus, no significant differences in consumption rates could be detected between the four reaches. However, consumption rates were lower in the Dam Reach than all other reaches over all seasons. Differences in consumption rates between the reaches are most likely based on water temperature through differences in evacuation rates. Evacuation rates are lower in colder water temperatures and increase with increasing water temperature. Trout in the Dam Reach had lower evacuation rates in May and September compared to the three downstream reaches, thus accounting for the lower consumption rate.

The low consumption rates could be contributing to the reduced growth in the Dam Reach; however, because of the colder water temperatures, maintenance energy requirements for trout in the Dam Reach are lower than energy requirements for trout in the downstream reaches that are experiencing warmer water temperatures. Cada et al. (1987b) observed increased growth rates in winter months when water temperatures were lower because the metabolic costs were lower in the cooler water than when compared to summer water temperatures. Seasonal water temperatures become increasingly important when food availability is limited. When food availability was limited throughout the year, high metabolic costs during warmer summer water temperatures dictated that limited energy intake went to fish maintenance rather than growth, thus accounting for decreased summer growth rates (Cada et al. 1987b).

Consumption rates were higher during May and September than in February and December for trout in the Smith River. Past research on Finland stream has observed increased consumption rates in brown trout during summer months for both age-0 and age-1 trout when water temperatures were high (Kreivi et al. 1999). Other research on salmonid species has also indicated that consumption rates are highest in the summer and then decline (Allan 1981, Walsh et al. 1988). Garman and Nielsen (1982) observed higher consumption rates during September and October and lower rates from May to August.

Bioenergetics Modeling

Assessing the Influence of Temperature and Consumption - Observed growth rates (G_O) for brown trout were highly variable between the reaches and years for age-1 brown trout. Observed growth rates in the tailwater ranged from 0.20 in 2002 in the Special Regulations Reach to 0.71 in the Dam Reach in 2003 (Table 2a.45). The Dam Reach had the highest G_O values in 2001 and 2003 and the Koehler Reach had the highest G_O values in 2000 and 2002 (Table 2a.45). Observed growth rates were generally lower in the Special Regulations Reach when compared to the other reaches (Table 2a.45). No years consistently had higher or lower G_O values among the reaches for age-1 trout in the summer. Observed yearly P -values for age-1 trout increased with increasing distance downstream of Philpott Dam (Table 2a.45).

Observed growth rates for age-0 trout in the summer were highly variable between reaches with G_O value decreasing with increasing distance down stream of Philpott Dam (Table 2a.46). Observed growth rates for age-0 trout ranged from 2.38 in the Dam Reach in 2003 to 1.11 in the Koehler Reach in 2002 (Table 2a.46). Highest G_O values for each reach were observed in 2003, while G_O values were lowest in the four reaches in 2002 (Table 2a.46). Observed P -values for age-0 brown trout were highest in the Dam Reach and decreased with increasing downstream distance.

Regression analysis of G_T values on G_O values for age-1 brown trout in the Smith River indicated through bioenergetics modeling that temperature had little influence in observed variation in growth in the Dam, Special Regulations, and Koehler reaches (Figure 2a.7, Table 2a.47). Variations in growth of age-1 trout in the Bassett Reach may be explained by yearly temperature fluctuations (Figure 2a.7, Table 2a.47). Regression values for G_T values on G_O values in the Bassett Reach yielded a high R^2 , a slope near one and an intercept near zero.

Regression analysis of G_P values on G_O values for age-1 brown trout in the Smith River indicated that observed variations in growth were explained by changes in consumption under a constant temperature in the Dam Reach (Figure 2a.7, Table 2a.47). In the Special Regulations, Bassett, and Koehler reaches, yearly variations in consumption failed to explain observed variations in growth (Figure 2a.7, Table 2a.47).

Observed variation in growth of age-0 brown trout in the Smith River was not explained by yearly observed water temperature in any of the reaches (Figure 2a.8, Table 2a.48). Regression analysis of G_T values on G_O values for age-0 trout had slopes that were greater than one and no reaches had intercept values near zero (Table 2a.48). Based on bioenergetics modeling, observed yearly variation in growth was explained by changes in consumption (Figure 2a.8, Table 2a.48). High R^2 values, slopes near one, and intercept values near zero were observed in the Dam, Special Regulations, and Koehler reaches when G_P values were regressed on G_O values (Table 2a.48).

In the Smith River, water temperature failed to explain observed variations in growth in age-0 and age-1 brown trout. Observed variations in growth in the Smith River were better

explained by changes in the consumption values. Railsback and Rose (1999) also noted that food consumption, not water temperature, explained summer growth of rainbow trout in California.

Environmental Factors Affecting Food Consumption (P) - Stepwise regression on yearly observed *P*-values to determine the influence of water temperature, flow, density dependence, and food availability for age-1 brown trout indicated that temperature, flow, and macroinvertebrate density explain 97% of the variation in *P*-values across years and reaches (Table 2a.49). The relationship between *P* and the explanatory variables were different between the reaches (Figure 2a.9). Consumption values were higher when a greater percentage of time had water temperatures >15.5°C in all reaches except the Special Regulations Reach (Figure 2a.9A). For mean daily flow, *P*-values increased in the Dam and Special Regulations reaches when flows increased, but *P*-values in the Bassett and Koehler reaches decreased under increased flow conditions (Figure 2a.9B). The relationship between *P*-values and the percentage of time <10°C indicated that when a greater amount of time was spent under colder water temperatures, *P*-values would be lower (Figure 2a.9C). In the Special Regulations Reach, *P*-values decreased with increased macroinvertebrate density, but in the remaining reaches, increased macroinvertebrate densities resulted in increased *P*-values (Figure 2a.9D).

For summer growth of age-0 brown trout in the Smith River, relative abundance of nongame fish was the only variable that explained observed variations in *P*-values ($R^2=0.76$, $P=0.0002$). Relative abundance of nongame fish was negatively related to the observed *P*-values with higher *P*-values being observed in the Dam Reach were the relative abundance of nongame fish is lowest (Figure 2a.10).

In a study on rainbow trout, Railsback and Rose (1999) noted that *P*-values were negatively related to water temperature and rainbow trout. Although brown trout density and biomass were not in the stepwise regression model, there is a weak negative trend between age-1 brown trout biomass and observed *P*-values for age-1 trout in summer ($R^2=0.0627$). No trend in *P*-values over changes in trout biomass or density was observed for age-0 brown trout in the Smith River. Although no trend was observed for age-0 brown trout with temperature, *P*-values for age-1 brown trout in the Smith River were positively related to mean daily water temperature ($R^2=0.4564$).

Predicted Growth under Alternative Flow Scenarios - Under the three alternative flow scenarios, no alternative scenario resulted in increased growth in all reaches for age-1 brown trout (Tables 2a.50-2a.51). Under the 12 °C outflow alternative, brown trout in the Dam Reach had higher predicted ending weights under the 12 °C outflow for both age-0 and age-1 brown trout than under baseline thermal conditions (Tables 2a.50-2a.51). Age-0 brown trout in the Dam Reach had a 7-13% increase in predicted ending weight under the 12 °C outflow scenario, which was the highest percentage increase in predicted ending weight from all scenarios. Although the 12 °C outflow scenario resulted in increased growth potential for trout in the Dam Reach, predicted ending weights were lower than predicted baseline ending weights for trout in the Special Regulations, Bassett, and Koehler reaches for both age-0 and age-1 brown trout (Tables 2a.50-2a.51). Decreased growth potential under the 12 °C outflow scenario for trout in the Special Regulations, Bassett, and Koehler reaches ranged from 1-7% and had the highest observed reductions predicted under the three alternative scenarios.

The installation of new turbines, to reduce the peakflow magnitude but with increased release duration, resulted in the smallest change in predicted ending weight. Age-1 trout in the Special Regulations Reach and age-0 and age-1 trout in the Bassett Reach had higher predicted

ending weights under the new turbine scenario than baseline conditions; however, these increases were small and were less than a 1.5% increase in ending weights (Tables 2a.50-2a.51). Although the trout in the Dam and Koehler reaches had decreased ending weights under the new turbine scenario when compared to the baseline conditions, the changes were minimal and were less than 1% of the baseline conditions (Tables 2a.50-2a.51).

The steady baseflow scenario resulted in the highest increases in predicted ending weights for trout in the Special Regulations and Bassett reaches compared to the other two alternative scenarios (Tables 2a.50-2a.51). The increases in ending weights over baseline conditions ranged from 1-5% for trout in the Special Regulations Reach and from 2-4% for trout in the Bassett Reach. Brown trout in the Dam and Koehler reaches had reduced predicted ending weights under the steady baseflow compared to baseline conditions; although the reductions in predictions in ending weights for trout in the Koehler Reach were minimal (less than 0.5%), reductions in ending weights for trout in the Dam Reach were greater for age-0 trout and ranged from 3-5% (Tables 2a.50-2a.51).

Changes in the thermal regime under the various flow scenarios were minimal in relationship to the predicted changes due to changes in food consumption. As was noted earlier, food consumption, rather than temperature, had a greater impact on observed variation in growth of brown trout in the Smith River. If consumption rates were to increase under baseline thermal conditions, increases in predicted ending weights for age-1 trout would range from 2% in the Bassett Reach to 32% for trout in the Dam Reach (Table 2a.50), and increases in predicted ending weights for age-0 trout would range from 14% in the Bassett Reach to 45% for trout in the Dam Reach (Table 2a.51). Railsback and Rose (1999) also noted that changes in *P*-values, rather than changes in temperature, had a greater effect on changes in growth.

Although bioenergetics modeling has been used in previous studies to examine the influence of temperature and consumption effects on growth in fish (Railsback and Rose 1999, Hayes et al. 2000), there are currently no bioenergetics models that can incorporate daily temperature fluctuations into the model. Bioenergetics models are based on a single averaged daily temperature value, so although the bioenergetics model failed to observe the influence of temperature on the yearly variation in growth, it is unknown what influence the water fluctuation during generation releases has on observed growth.

Although diet composition can influence the growth rates of brown trout, potential changes in diet composition were not assessed with bioenergetics models. Historic accounts on the Smith River indicated that brown trout near the dam preyed on alewives that were dead or dying after coming through the turbines (Cochran 1975); however, alewives were not observed in the tailwater during this study from 2000-2003. With the lack of alewives coming from the reservoir, trout in the Dam Reach now consume Chironomidae and Isopoda. The amount of energy in Diptera larvae is approximately one-fifth the energy of an alewife (Cummins and Wuycheck 1971, Stewart and Binkowski 1986). The reduction in energy content in prey items can lead to a reduction in the growth of the trout. Although potential changes in diet composition were not assessed, if more items with higher energy content were consumed, growth rates would potentially increase.

Discussion

Brown trout population estimates for age-0 brown trout and age-1 and older trout in the following year were negatively related to average daily flow values during the April to June time period. Based on these relationships, population abundance appears to be under the control of

the flow regime that occurs during late spring. Age-1 and older brown trout in the Smith River exhibit a high mortality rate in all reaches of the tailwater, so reduced year-class strength due to high flows in consecutive years may greatly reduce the brown trout population in the Smith River.

Mortality rates for age-1 and older brown trout in the Smith River are high, and although this study was not designed to differentiate between natural mortality and angling mortality, past creel surveys on the Smith River tailwater indicate that anglers harvest approximately 5% of the brown trout caught in the Smith River (Hartwig 1998). Due to the low percentage of brown trout harvested by anglers, natural mortality may be impacting the brown trout population in the Smith River. Under current harvest regulations, there is a 406 mm (16 in) minimum length limit in the Special Regulations Reach for brown trout. During the study from 2000-2003, no brown trout collected in the Special Regulations Reach were over 406 mm. While the 406 mm minimum length limit was designed to promote a trophy brown trout fishery, it currently does not provide anglers the opportunity to legally harvest any trout.

With high mortality rates and low harvest rates by anglers, changes in the harvest regulations may need to be considered. Currently, brown trout in the Special Regulations Reach fail to reach the minimum size limit before the trout are removed from the population due to natural causes. If the minimum length limit in the Special Regulations Reach was lowered or a slot length limit was initiated, it would allow for the occasional harvest of brown trout by anglers.

In the Dam, Bassett, and Koehler reaches, stocked rainbow trout provide additional angling opportunities. In 1995, anglers in the Smith River caught an estimated 23,000 rainbow trout, of which 90% were harvested (Hartwig 1998). In the entire tailwater, approximately 6,700 brown trout were caught in 1995, with anglers in the lower section (Bassett and Koehler reaches) harvesting 43% of the brown trout caught while less than 2% of the brown trout caught in the Dam and Special Regulations reaches were harvested (Hartwig 1998). Increased harvest rates in the Bassett and Koehler reaches may be due to increased sizes of brown trout in the downstream reaches. During a 1995 creel survey on the Smith River, 47% of the brown trout captured in the lower section of the tailwater were 254-406 mm (10-16 in), where as only 4 and 12% of the brown trout caught in the upper (Dam Reach) and middle (Special Regulations Reach) sections, respectively, were 254-406 mm (Hartwig 1998).

While the stocked rainbow trout provide an alternative angling opportunity, stocking rates may need to be adjusted. Although studies on the persistence of rainbow trout in tailwaters indicate that there is low potential for stocked rainbow trout to persist longer than 20 days (Bettinger and Bettoli 2002), it is unknown how long stocked rainbow trout persist in the Smith River. Currently, in the Dam Reach, the biomass and density estimates for brown trout are high, but studies on the invertebrate community indicate that food resources may be lower in the Dam Reach than in other reaches in the Smith River (Newcomb et al. 2001). If the rainbow trout stocking level is increased in the Dam Reach, there may be additional competition for food resources. With the low brown trout biomass and density estimates and large size of trout in the Bassett and Koehler reaches, it does not appear that the stocking of rainbow trout in these reaches are impacting the brown trout population. Further research into potential competition between rainbow and brown trout would be needed to determine the impact of continued stocking of rainbow trout at the current level (31,000 rainbow trout of harvestable size per year).

Food availability in the Smith River tailwater may be limiting the brown trout. Past reports indicated that brown trout in the Smith River tailwater would consume alewives that

came through the turbines during generation periods, which may have lead to the increased growth of brown trout (Cochran 1975). Previous research on tailwater systems has shown that trout consuming threadfin shad *Dorosoma petenense* and other forage fish that originated in the upstream reservoir had high growth rates (Pfitzer 1967). From 2000-2003, no alewives were collected during this study in the tailwater; however, it is unclear what changes have occurred that would prevent the alewives from passing through the dam. Changes in reservoir trophic status may have occurred since the completion of Philpott Dam in 1953. With the connectivity of tailwaters and the upstream reservoir, trophic changes that occur in the reservoir will often result in changes in the downstream tailwater productivity. After completion of reservoir projects, the reservoir is highly productive during the initial years; however, productivity will decrease as the reservoir ages until a productivity equilibrium is reached (Kimmel and Groeger 1986). Oligotrophication may occur in the reservoir as it ages, reducing not only the productivity levels downstream of the reservoir but also the amount of forage fish in the reservoir (Ney 1996), which may explain a potential reduction of alewives in Philpott Reservoir that pass through the dam. Data on reservoir productivity and forage fish abundance in Philpott Reservoir is lacking, so it difficult to determine what changes, if any, have occurred in the reservoir since completion of Philpott Dam.

In addition to changes in the reservoir, the cold water temperature in the Smith River limits the distribution of non-salmonid fish in the tailwater. In areas where prey fish are lacking, alternative food items, including Chironomidae, Isopoda and drift, are utilized; however, macroinvertebrates are impacted by changing flows. Flow regimes in the tailwater may be limiting the abundance of macroinvertebrates below the dam by daily flushing of the substrate during generation periods. Although the tailwater area may be productive in the initial years after completion of a dam, species richness, total density and biomass of macroinvertebrates decreases as the system ages (Trotzky and Gregory 1974, Garcia de Jalon et al. 1994). Macroinvertebrate abundances in the Smith River are below levels that would be observed in unregulated streams in Virginia (Newcomb et al. 2001), thus potentially limiting the food availability for trout.

Based on bioenergetics modeling, it appears that food consumption rates have a greater impact than temperature in determining yearly variations in growth rates. Increasing food consumption resulted in the greatest increase in predicted ending weight. Under the three alternative flow scenarios, there was no scenario that benefited the entire tailwater based on changes in the thermal regime. Although the impacts, both in increases and decreases in predicted growth, were lowest with the addition of new turbines that reduce the magnitude of the flow, the potential for increased growth was minimal (1%) compared to the other alternatives. Trade-offs will need to be evaluated to determine which areas of the tailwater should be enhanced through changes in the flow regime. If the overall goal is to improve growth of trout in the Dam Reach, then having a 12 °C outflow would provide the greatest benefit, but it would potentially reduce growth of trout in the Special Regulations Reach. With the Special Regulations Reach being managed as a trophy trout area, it may be beneficial to focus on improving growth in this section by having a steady baseflow; however, growth of trout in the Dam Reach would potentially be reduced. Because no scenario benefits the entire tailwater, selecting alternatives that maximize the potential to increase growth in one section while minimizing the potential for reduction in growth in other sections of the river should be considered.

Table 2a.1. Location in the tailwater, Virginia Department of Game and Inland Fisheries (VDGIF) management regulation (creel limit and minimum length limit), and general habitat description of the four reaches in the Smith River, Virginia. Non-game relative abundances (# 100 m⁻¹) and standard errors (SE) are mean values from June 2000 to October 2002 (see Job 2 part B).

Reach	Location from Philpott Dam (km)	Number of Sampling Sites	VDGIF trout regulation	General habitat	Non-game fish abundance # 100 m ⁻¹ (SE)
Dam	0-5.3	3	6 trout; 178 mm	Short riffle:run segments; substrate dominated by bedrock, boulders, and cobble	2.71 (0.50)
Special Regulations	5.3-10	2	2 trout; 406 mm	Short riffle:run segments; substrate dominated by bedrock, boulders, cobble, and pebbles	45.63 (6.05)
Bassett	10-15.9	3	6 trout; 178 mm	Long run segments; substrate dominated by pebbles and sand	106.08 (20.35)
Koehler	15.9-24	4	6 trout; 178 mm	Long, deep pools and short riffles; dominant substrate is silt and sand; few boulders	221.78 (62.38)

Table 2a.2. Number of brown trout tagged in June 2000 and the number of trout recaptured from August 2000 to October 2003 at 12 sampling locations in the Smith River, Virginia, tailwater. NA indicates that the site was not sampled during the timeperiod. Reach D=Dam; SR=Special Regulations; B=Bassett; and K=Koehler.

Distance from Philpott Dam (km)	Reach	<u>Date</u>										
		June 2000	August 2000	October 2000	April 2001	June 2001	October 2001	April 2002	June 2002	October 2002	July 2003	October 2003
0.5	D	77	15	7	7	6	4	2	1	3	0	NA
3.4	D	251	76	70	30	45	37	20	16	11	0	0
4.2	D	422	214	128	56	73	44	25	19	19	3	3
6.2	SR	235	131	79	51	43	31	17	3	18	0	3
8.9	SR	270	124	124	60	65	37	22	11	14	1	NA
11.3	B	170	67	54	29	15	18	6	5	3	1	NA
12.6	B	17	5	4	3	3	0	0	0	0	0	0
14.3	B	101	47	36	17	6	6	9	4	1	0	NA
15.9	K	90	25	16	4	1	3	2	1	1	0	NA
18.9	K	71	12	34	8	4	3	1	1	0	0	1
20.1	K	83	22	22	23	18	16	7	0	3	0	1
23	K	86	42	36	8	5	7	3	2	0	NA	NA
Total		1873	780	610	296	284	206	114	63	73	5	8

Table 2a.3. Number of brown trout tagged in October 2001 and the number of trout recaptured from April 2002 to October 2003 at 12 sampling locations in the Smith River, Virginia. NA indicates that the site was not sampled during the timeperiod. Reach D=Dam; SR=Special Regulations; B=Basset; and K=Koehler.

Distance from Philpott Dam (km)	Reach	<u>Date</u>					
		October 2001	April 2002	June 2002	October 2002	July 2003	October 2003
0.5	D	105	35	26	12	6	1
3.4	D	236	110	114	70	3	17
4.2	D	239	92	75	64	10	19
6.2	SR	248	98	49	53	10	11
8.9	SR	318	108	56	57	4	1
11.3	B	275	67	55	32	11	NA
12.6	B	148	38	26	20	7	3
14.3	B	112	24	20	12	5	NA
15.9	K	95	11	6	2	1	NA
18.9	K	94	25	6	5	0	4
20.1	K	56	8	12	4	0	1
23	K	100	10	21	10	NA	NA
Total		2026	626	466	341	57	57

Table 2a.4. Length-weight regression slopes (*b*) and intercepts (number of observations in parentheses) brown trout collected from four reaches and 11 sampling periods between 2000 and 2003 in the Smith River, Virginia, tailwater. Slopes in the same date with the same letter are not significantly different between the reaches (alpha=0.05).

Date	<u>Reach</u>							
	<u>Dam</u>	<u>Special Regulations</u>	<u>Bassett</u>	<u>Koehler</u>				
	Slope (<i>b</i>) (N)	Intercept	Slope (<i>b</i>) (N)	Intercept	Slope (<i>b</i>) (N)	Intercept	Slope (<i>b</i>) (N)	Intercept
June 2000	2.912 a (761)	-4.822	2.942 c (534)	-4.893	2.986 b (311)	-4.968	3.005 ac (380)	-5.027
August 2000	2.992 b (1184)	-5.003	2.947 b (663)	-4.903	3.038 b (461)	-5.102	3.21 a (217)	-5.535
October 2000	2.982 bc (1143)	-4.981	3.013 ab (852)	-5.054	2.986 ac (720)	-4.986	3.026 a (681)	-5.077
April 2001	2.851 b (700)	-4.679	3.285 a (538)	-5.675	3.035 a (554)	-5.079	2.997 a (276)	-4.991
June 2001	2.961 b (998)	-4.928	2.957 c (889)	-4.917	2.959 a (371)	-4.925	2.989 a (241)	-4.992
October 2001	2.979 b (1003)	-4.964	3.006 b (851)	-5.038	3.025 a (856)	-5.071	3.105 a (470)	-5.247
April 2002	2.956 a (1371)	-4.912	2.956 b (911)	-4.916	2.976 b (615)	-4.959	2.979 ab (291)	-4.977
June 2002	2.960 a (989)	-4.922	2.923 b (602)	-4.854	2.926 b (607)	-4.857	2.939 ab (364)	-4.891
October 2002	2.938 a (1068)	-4.875	2.940 a (761)	-4.888	2.937 a (574)	-4.886	2.944 a (294)	-4.899
July 2003	3.029 b (663)	-5.079	2.986 c (201)	-4.991	3.088 a (224)	-5.217	2.969 c (39)	-4.933
October 2003	2.965 a (1297)	-4.938	2.890 b (266)	-4.773	2.975 a (116)	-4.980	2.986 a (79)	-5.001

Table 2a.5. Environmental characteristics collected from the Smith River, Virginia, from June 2000- October 2002. Reach D=Dam; SR=Special Regulations; B=Bassett; and K=Koehler.

Reach	Distance downstream of Philpott Dam (km)											
	<u>0.5</u>	<u>3.4</u>	<u>4.2</u>	<u>6.2</u>	<u>8.9</u>	<u>11.3</u>	<u>12.6</u>	<u>14.3</u>	<u>15.9</u>	<u>18.9</u>	<u>20.1</u>	<u>23.0</u>
	D	D	D	SR	SR	B	B	B	K	K	K	K
June2000-October 2000												
Mean Daily Flow (cms) ^a	3.3	3.3	3.3	3.9	4.0	4.3	4.5	4.9	5.0	5.7	5.9	6.3
Maximum Daily Flow (cms) ^a	39.4	36.5	33.8	38.1	33.3	30.2	30.6	28.8	29.0	26.1	26.1	25.7
Nongame Relative Abundance (# 100 m ⁻¹) ^c	6	0	0	128	12	64	149	197	64	193	153	146
Brown Trout Biomass (kg ha ⁻¹)	10.2	54.1	68.7	98.8	35.2	31.7	11.8	24.9	30.3	11.5	1.5	12.4
Brown Trout Density (# ha ⁻¹)	248	807	742	799	384	301	73	184	217	128	17	114
Mean Invertebrate Wet Weight (g m ² - ¹) ^d	1.2	1.2	0.7	3.6	1.6	1.4	0.5	4.1	1.0	2.5	0.8	2.1
Mean Invertebrate Density (# m ² - ¹) ^d	404	273	158	522	408	247	132	213	398	368	172	165
Mean Density EPT (# m ² - ¹) ^d	2	85	64	402	338	163	65	122	280	300	117	112
Mean Daily Temperature (°C) ^e	10.4	10.4	10.4	13.0	13.0	14.5	14.5	14.5	16.5	16.5	16.5	16.5
Hourly temperature Fluctuation (°C) ^e	4.1	4.1	4.1	5.0	5.0	1.3	1.3	1.3	0.6	0.6	0.6	0.6
Percent Time < 10 °C ^e	59.9	59.9	59.9	9.9	9.9	3.2	3.2	3.2	3.4	3.4	3.4	3.4
Percent Time 10-15.5 °C ^e	38.5	38.5	38.5	69.9	69.9	62.8	62.8	62.8	25.8	25.8	25.8	25.8
Percent Time > 15.5 °C ^e	1.6	1.6	1.6	20.3	20.3	34.0	34.0	34.0	70.8	70.8	70.8	70.8

Table 2a.5. Continued.

Reach	Distance downstream of Philpott Dam (km)											
	<u>0.5</u> D	<u>3.4</u> D	<u>4.2</u> D	<u>6.2</u> SR	<u>8.9</u> SR	<u>11.3</u> B	<u>12.6</u> B	<u>14.3</u> B	<u>15.9</u> K	<u>18.9</u> K	<u>20.1</u> K	<u>23.0</u> K
October 2000-April 2001												
Mean Daily Flow (cms) ^a	3.2	3.2	3.3	4.2	4.2	4.5	4.8	5.2	5.3	6.0	6.2	6.7
Maximum Daily Flow (cms) ^a	40.9	37.4	34.6	45.0	42.5	40.8	39.8	40.3	39.4	37.3	37.3	36.4
Nongame Relative Abundance (# 100 m ⁻¹) ^c	2	7	9	57	25	32	59	113	92	111	113	277
Brown Trout Biomass (kg ha ⁻¹)	15.2	30.1	41.1	45.1	35.0	17.0	15.9	14.5	7.0	13.5	7.1	13.2
Brown Trout Density (# ha ⁻¹)	155	320	350	291	337	115	96	89	44	67	32	80
Mean Invertebrate Wet Weight (g m ² ⁻¹) ^d	1.3	0.4	0.9	4.7	4.7	4.1	4.2	9.7	3.1	5.3	6.0	18.1
Mean Invertebrate Density (# m ² ⁻¹) ^d	557	80	142	863	427	332	343	843	232	395	528	895
Mean Density EPT (# m ² ⁻¹) ^d	3	58	110	757	395	283	290	805	192	317	425	752
Mean Daily Temperature (°C) ^e	7.4	7.4	7.4	7.0	7.0	7.0	7.0	7.0	6.8	6.8	6.8	6.8
Hourly temperature Fluctuation (°C) ^e	0.7	0.7	0.7	0.8	0.8	0.4	0.4	0.4	0.4	0.4	0.4	0.4
Percent Time < 10 °C ^e	83.8	83.8	83.8	85.3	85.3	83.4	83.4	83.4	81.6	81.6	81.6	81.6
Percent Time 10-15.5 °C ^e	16.3	16.3	16.3	14.5	14.5	16.1	16.1	16.1	17.8	17.8	17.8	17.8
Percent Time > 15.5 °C ^e	0.0	0.0	0.0	0.2	0.2	0.5	0.5	0.5	0.6	0.6	0.6	0.6

Table 2a.5. Continued.

Reach	Distance downstream of Philpott Dam (km)											
	<u>0.5</u> D	<u>3.4</u> D	<u>4.2</u> D	<u>6.2</u> SR	<u>8.9</u> SR	<u>11.3</u> B	<u>12.6</u> B	<u>14.3</u> B	<u>15.9</u> K	<u>18.9</u> K	<u>20.1</u> K	<u>23.0</u> K
April 2001-June 2001												
Mean Daily Flow (cms) ^a	4.5	4.5	4.5	5.7	5.8	6.4	6.7	7.4	7.6	8.8	9.2	10.0
Maximum Daily Flow (cms) ^a	39.1	36.2	33.6	36.8	31.4	30.2	31.3	33.4	33.7	36.4	37.5	39.6
Nongame Relative Abundance (# 100 m ⁻¹) ^c	6	0	2	117	17	44	95	74	51	126	91	192
Brown Trout Biomass (kg ha ⁻¹)	19.9	13.3	25.1	32.9	22.0	18.9	14.5	12.1	7.0	9.4	7.5	10.8
Brown Trout Density (# ha ⁻¹)	208	198	371	258	291	260	163	114	64	67	43	95
Mean Invertebrate Wet Weight (g m ² ⁻¹) ^d	1.3	0.4	0.9	4.7	4.7	4.1	4.2	9.7	3.1	5.3	6.0	18.1
Mean Invertebrate Density (# m ² ⁻¹) ^d	557	80	142	863	427	332	343	843	232	395	528	895
Mean Density EPT (# m ² ⁻¹) ^d	3	58	110	757	395	283	290	805	192	317	425	752
Mean Daily Temperature (°C) ^e	9.3	9.3	9.3	11.9	11.9	13.5	13.5	13.5	15.0	15.0	15.0	15.0
Hourly temperature Fluctuation (°C) ^e	3.9	3.9	3.9	4.2	4.2	1.4	1.4	1.4	0.8	0.8	0.8	0.8
Percent Time < 10 °C ^e	70.3	70.3	70.3	38.1	38.1	15.0	15.0	15.0	4.6	4.6	4.6	4.6
Percent Time 10-15.5 °C ^e	27.8	27.8	27.8	45.3	45.3	55.9	55.9	55.9	45.9	45.9	45.9	45.9
Percent Time > 15.5 °C ^e	1.9	1.9	1.9	16.6	16.6	29.1	29.1	29.1	49.5	49.5	49.5	49.5

Table 2a.5. Continued.

Reach	Distance downstream of Philpott Dam (km)											
	<u>0.5</u> D	<u>3.4</u> D	<u>4.2</u> D	<u>6.2</u> SR	<u>8.9</u> SR	<u>11.3</u> B	<u>12.6</u> B	<u>14.3</u> B	<u>15.9</u> K	<u>18.9</u> K	<u>20.1</u> K	<u>23.0</u> K
June 2001-October 2001												
Mean Daily Flow (cms) ^a	5.0	5.0	5.0	5.4	5.4	5.7	5.9	6.3	6.4	7.1	7.3	7.8
Maximum Daily Flow (cms) ^a	24.1	22.0	21.4	21.7	21.6	22.3	22.8	23.6	23.8	25.2	25.6	26.4
Nongame Relative Abundance (# 100 m ⁻¹) ^c	1	1	3	28	24	40	129	243	69	126	171	387
Brown Trout Biomass (kg ha ⁻¹)	7.2	12.4	33.2	83.3	72.0	27.7	38.7	6.9	11.9	9.9	1.2	8.8
Brown Trout Density (# ha ⁻¹)	105	369	484	632	736	301	296	63	91	82	10	57
Mean Invertebrate Wet Weight (g m ² ⁻¹) ^d	0.3	0.2	0.5	1.0	0.6	0.8	1.5	1.6	5.0	1.4	1.6	8.0
Mean Invertebrate Density (# m ² ⁻¹) ^d	927	282	407	257	458	382	353	610	278	284	190	297
Mean Density EPT (# m ² ⁻¹) ^d	5	32	67	73	273	153	152	253	150	151	105	85
Mean Daily Temperature (°C) ^e	9.5	9.5	9.5	11.9	11.9	13.4	13.4	13.4	15.1	15.1	15.1	15.1
Hourly temperature Fluctuation (°C) ^e	2.0	2.0	2.0	2.5	2.5	1.5	1.5	1.5	1.2	1.2	1.2	1.2
Percent Time < 10 °C ^e	73.7	73.7	73.7	31.6	31.6	20.0	20.0	20.0	2.5	2.5	2.5	2.5
Percent Time 10-15.5 °C ^e	25.8	25.8	25.8	56.3	56.3	50.5	50.5	50.5	50.5	50.5	50.5	50.5
Percent Time > 15.5 °C ^e	0.5	0.5	0.5	12.1	12.1	29.6	29.6	29.6	47.0	47.0	47.0	47.0

Table 2a.5. Continued.

Reach	Distance downstream of Philpott Dam (km)											
	<u>0.5</u> D	<u>3.4</u> D	<u>4.2</u> D	<u>6.2</u> SR	<u>8.9</u> SR	<u>11.3</u> B	<u>12.6</u> B	<u>14.3</u> B	<u>15.9</u> K	<u>18.9</u> K	<u>20.1</u> K	<u>23.0</u> K
October 2001-April 2002												
Mean Daily Flow (cms) ^a	4.2	4.2	4.3	4.9	4.9	5.1	5.2	5.4	5.5	5.9	6.0	6.2
Maximum Daily Flow (cms) ^a	42.7	42.7	42.7	43.4	43.4	43.2	43.3	43.1	42.2	40.9	40.3	39.6
Nongame Relative Abundance (# 100 m ⁻¹) ^c	1	3	1	58	21	31	67	72	187	107	227	115
Brown Trout Biomass (kg ha ⁻¹)	12.9	28.1	27.0	42.3	31.0	23.2	22.4	9.5	12.6	11.2	6.8	8.0
Brown Trout Density (# ha ⁻¹)	115	377	246	264	280	190	151	53	68	70	33	55
Mean Invertebrate Wet Weight (g m ² ⁻¹) ^d
Mean Invertebrate Density (# m ² ⁻¹) ^d
Mean Density EPT (# m ² ⁻¹) ^d
Mean Daily Temperature (°C) ^e	8.2	8.2	8.2	8.0	8.0	7.8	7.8	7.8	8.6	8.6	8.6	8.6
Hourly temperature Fluctuation (°C) ^e	0.5	0.5	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.4	0.4	0.4
Percent Time < 10 °C ^e	81.1	81.1	81.1	78.1	78.1	78.1	78.1	78.1	67.0	67.0	67.0	67.0
Percent Time 10-15.5 °C ^e	18.9	18.9	18.9	21.9	21.9	21.9	21.9	21.9	32.1	32.1	32.1	32.1
Percent Time > 15.5 °C ^e	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.1	0.8	0.8	0.8	0.8

Table 2a.5. Continued. April 2002 to June 2002

Reach	Distance downstream of Philpott Dam (km)											
	<u>0.5</u> D	<u>3.4</u> D	<u>4.2</u> D	<u>6.2</u> SR	<u>8.9</u> SR	<u>11.3</u> B	<u>12.6</u> B	<u>14.3</u> B	<u>15.9</u> K	<u>18.9</u> K	<u>20.1</u> K	<u>23.0</u> K
April 2002-June 2002												
Mean Daily Flow (cms) ^a	2.3	2.3	2.3	2.8	2.9	3.2	3.4	3.8	3.9	4.6	4.8	5.2
Maximum Daily Flow (cms) ^a	20.4	19.3	18.9	20.2	18.9	18.0	18.2	17.8	16.9	15.6	15.4	15.2
Nongame Relative Abundance (# 100 m ⁻¹) ^c	6	1	1	46	27	33	60	32	60	103	99	182
Brown Trout Biomass (kg ha ⁻¹)	31.2	35.3	28.1	57.3	42.0	18.9	14.0	17.0	12.4	7.6	6.5	5.9
Brown Trout Density (# ha ⁻¹)	445	625	368	564	601	273	174	142	140	61	34	48
Mean Invertebrate Wet Weight (g m ² ⁻¹) ^d	1.0	0.3	0.3	1.8	1.7	3.3	14.4	4.5	10.6	27.2	9.7	10.4
Mean Invertebrate Density (# m ² ⁻¹) ^d	1653	225	600	623	377	662	873	705	342	455	405	518
Mean Density EPT (# m ² ⁻¹) ^d	15	13	13	132	168	387	572	267	128	285	178	177
Mean Daily Temperature (°C) ^e	9.9	9.9	9.9	12.5	12.5	15.4	15.4	15.4	17.7	17.7	17.7	17.7
Hourly temperature Fluctuation (°C) ^e	2.4	2.4	2.4	2.1	2.1	0.6	0.6	0.6	0.5	0.5	0.5	0.5
Percent Time < 10 °C ^e	63.7	63.7	63.7	21.2	21.2	3.5	3.5	3.5	0.3	0.3	0.3	0.3
Percent Time 10-15.5 °C ^e	34.8	34.8	34.8	61.4	61.4	44.6	44.6	44.6	24.0	24.0	24.0	24.0
Percent Time > 15.5 °C ^e	1.5	1.5	1.5	17.2	17.2	51.9	51.9	51.9	75.7	75.7	75.7	75.7

Table 2a.5. Continued. June 2002 to October 2002

Reach	Distance downstream of Philpott Dam (km)											
	<u>0.5</u> D	<u>3.4</u> D	<u>4.2</u> D	<u>6.2</u> SR	<u>8.9</u> SR	<u>11.3</u> B	<u>12.6</u> B	<u>14.3</u> B	<u>15.9</u> K	<u>18.9</u> K	<u>20.1</u> K	<u>23.0</u> K
June 2002-October 2002												
Mean Daily Flow (cms) ^a	2.2	2.2	2.2	2.5	2.5	2.6	2.7	2.9	2.9	3.2	3.3	3.5
Maximum Daily Flow (cms) ^a	24.6	18.9	18.7	17.8	16.6	13.7	12.7	12.3	11.9	10.2	10.2	10.0
Nongame Relative Abundance (# 100 m ⁻¹) ^c	4	1	4	90	25	36	124	209	181	391	264	263
Brown Trout Biomass (kg ha ⁻¹)	27.4	15.3	27.0	28.4	13.3	38.7	23.0	13.4	19.0	7.4	3.8	19.3
Brown Trout Density (# ha ⁻¹)	452	306	453	250	150	488	265	136	179	95	52	147
Mean Invertebrate Wet Weight (g m ² ⁻¹) ^d	1.0	0.3	0.3	1.8	1.7	3.3	14.4	4.5	10.6	27.2	9.7	10.4
Mean Invertebrate Density (# m ² ⁻¹) ^d	1653	225	600	623	377	662	873	705	342	455	405	518
Mean Density EPT (# m ² ⁻¹) ^d	15	13	13	132	168	387	572	267	128	285	178	177
Mean Daily Temperature (°C) ^e	10.3	10.3	10.3	13.2	13.2	16.4	16.4	16.4	18.4	18.4	18.4	18.4
Hourly temperature Fluctuation (°C) ^e	1.9	1.9	1.9	1.2	1.2	0.7	0.7	0.7	0.5	0.5	0.5	0.5
Percent Time < 10 °C ^e	62.5	62.5	62.5	5.0	5.0	0.9	0.9	0.9	0.0	0.0	0.0	0.0
Percent Time 10-15.5 °C ^e	36.3	36.3	36.3	76.0	76.0	34.8	34.8	34.8	14.8	14.8	14.8	14.8
Percent Time > 15.5 °C ^e	1.3	1.3	1.3	19.2	19.2	64.4	64.4	64.4	85.2	85.2	85.2	85.2

Table 2a.6. Environmental variables used in the regression model to determine the influence of environmental characteristics on the percent of maximum consumption (*P*) for the June-October time period from 4 reaches in the Smith River, Virginia, from 2000 to 2003 for age-1 brown trout. Reaches: D=Dam, SR=Special Regulations, B=Bassett, K=Koehler. (Bio.=Biomass, Den.=Density, R. A.= Relative Abundance; EPT=Ephemeroptera, Plecoptera, and Trichoptera)

Reach	Year	Mean Daily Flow (cms)	Max Daily Flow (cms)	Age-1 Trout Bio. (kg/hr)	Age-1 Trout Den. (#/ha)	R.A. Prey Fish (#/100)	Invert Wet Weight (g/m ²)	Invert Mean Den. (#/m ²)	Mean Den. EPT (#/m ²)	Mean Daily Temp (oC)	Hourly Flux (oC)	Time <10°C (%)	Time 10-15°C (%)	Time >15°C (%)
D	2000	3.3	36.6	44.3	599	2	1.0	278	50	10.4	4.1	60	38	2
SR	2000	3.9	35.7	67.0	591	70	2.6	465	370	13.0	5.0	10	70	20
B	2000	4.6	29.9	22.8	186	137	2.0	197	117	14.5	1.3	3	63	34
K	2000	5.7	26.7	13.9	119	139	1.6	276	202	16.5	0.6	3	26	71
D	2001	5.0	22.5	17.6	320	2	0.3	538	34	9.5	2.0	74	26	1
SR	2001	5.4	21.6	77.6	684	26	0.8	358	173	11.9	2.5	32	56	12
B	2001	6.0	22.9	24.4	220	137	1.3	448	186	13.4	1.5	20	50	30
K	2001	7.2	25.3	8.0	60	188	4.0	262	123	15.1	1.2	3	50	47
D	2002	2.2	20.7	23.3	404	3	0.5	826	14	10.3	1.9	62	36	1
SR	2002	2.5	17.2	20.9	200	58	1.8	500	150	13.2	1.2	5	76	19
B	2002	2.8	12.9	25.0	296	123	7.4	747	408	16.4	0.7	1	35	64
K	2002	3.2	10.6	12.4	118	275	14.5	430	192	18.4	0.5	0	15	85
D	2003	12.0	29.0	56.6	814	.	1.3	1187	49	16.5	1.2	0	24	76
SR	2003	13.9	31.5	24.1	285	.	2.8	313	138	17.7	2.0	0	89	11
B	2003	16.0	34.7	28.4	294	.	7.1	505	193	17.8	1.0	0	95	5
K	2003	19.9	39.3	11.1	76	.	3.0	432	117	16.3	1.1	0	76	24

Table 2a.7. Environmental variables used in the regression model to determine the influence of environmental characteristics on the percent of maximum consumption (*P*) for the June-October time period from 4 reaches in the Smith River, Virginia, from 2000 to 2003 for age-0 brown trout. Reaches: D=Dam, SR=Special Regulations, B=Bassett, K=Koehler. (Bio.=Biomass, Den=Density, R. A.= Relative Abundance; EPT=Ephemeroptera, Plecoptera, and Trichoptera)

Reach	Year	Mean Daily Flow (cms)	Max Daily Flow (cms)	Age-0 Trout Bio. (kg/hr)	Age-0 Trout Den. (#/ha)	R.A. Prey Fish (#/100)	Invert Wet Weight (g/m ²)	Invert Mean Den. (#/m ²)	Mean Den. EPT (#/m ²)	Mean Daily Temp (oC)	Hourly Flux (oC)	Time <10°C (%)	Time 10-15°C (%)	Time >15°C (%)
D	2000	3.3	36.6	0.5	597	2	1.0	278	50	10.4	4.1	60	38	2
SR	2000	3.9	35.7	1.8	772	70	2.6	465	370	13.0	5.0	10	70	20
B	2000	4.6	29.9	0.9	321	137	2.0	197	117	14.5	1.3	3	63	34
K	2000	5.7	26.7	0.3	124	139	1.6	276	202	16.5	0.6	3	26	71
D	2001	5.0	22.5	0.7	728	2	0.3	538	34	9.5	2.0	74	26	1
SR	2001	5.4	21.6	3.2	1467	26	0.8	358	173	11.9	2.5	32	56	12
B	2001	6.0	22.9	0.4	176	137	1.3	448	186	13.4	1.5	20	50	30
K	2001	7.2	25.3	0.4	137	188	4.0	262	123	15.1	1.2	3	50	47
D	2002	2.2	20.7	1.8	1135	3	0.5	826	14	10.3	1.9	62	36	1
SR	2002	2.5	17.2	3.3	1369	58	1.8	500	150	13.2	1.2	5	76	19
B	2002	2.8	12.9	1.5	564	123	7.4	747	408	16.4	0.7	1	35	64
K	2002	3.2	10.6	0.9	297	275	14.5	430	192	18.4	0.5	0	15	85
D	2003	12.0	29.0	0.4	326	.	1.3	1187	49	16.5	1.2	0	24	76
SR	2003	13.9	31.5	0.3	200	.	2.8	313	138	17.7	2.0	0	89	11
B	2003	16.0	34.7	0.0	9	.	7.1	505	193	17.8	1.0	0	95	5
K	2003	19.9	39.3	0.0	11	.	3.0	432	117	16.3	1.1	0	76	24

Table 2a.8. Brown trout population estimates (number m⁻¹), trout density (number ha⁻¹), and standing crop (kg ha⁻¹) (95% CI) for age-1 and older brown trout at 12 sites in the Smith River, Virginia, sampled using 3-pass depletion electrofishing in June 2000. Means in the same column with the same letter are not significantly different (alpha=0.05). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Bassett; K=Koehler.

Date	Site	Reach	Population Estimate (# m ⁻¹) (95% CI)	Trout Density (# ha ⁻¹) (95% CI)	Mean Trout Weight (g)	Standing Crop (kg ha ⁻¹) (95% CI)
June 2000	0.5	D	85 (81-91) e	248 (236-267) e	41.0	10.2 (9.7-10.9) i
June 2000	3.4	D	177 (175-181) c	807 (798-823) a	67.0	54.1 (53.5-55.2) c
June 2000	4.2	D	190 (188-193) b	742 (734-756) b	92.6	68.7 (68.0-70.0) b
June 2000	6.2	SR	230 (217-243) a	799 (754-843) ab	123.7	98.8 (93.3-104.3) a
June 2000	8.9	SR	100 (99-103) d	384 (380-396) c	91.8	35.2 (34.9-36.4) d
June 2000	11.3	B	90 (90-92) e	301 (301-307) d	105.4	31.7 (31.7-32.4) e
June 2000	12.6	B	21 (19-27) i	73 (66-96) i	161.9	11.8 (10.7-15.5) gh
June 2000	14.3	B	58 (57-61) f	184 (181-194) f	135.6	24.9 (24.5-26.2) f
June 2000	15.9	K	57 (39-92) e	217 (148-349) def	140.0	30.3 (20.8-48.9) def
June 2000	18.9	K	39 (39-39) h	128 (128-129) g	89.9	11.5 (11.5-11.6) h
June 2000	20.1	K	5 *	17 *	84.4	1.5 *
June 2000	23.0	K	42 (41-45) g	114 (112-124) h	108.5	12.4 (12.1-13.4) g

Table 2a.9. Brown trout population estimates (number m⁻¹), trout density (number ha⁻¹), and standing crop (kg ha⁻¹) (95% CI) for age-1 and older brown trout at 12 sites in the Smith River, Virginia, sampled using 3-pass depletion electrofishing in June 2001. Means in the same column with the same letter are not significantly different (alpha=0.05). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Bassett; K=Koehler.

Date	Site	Reach	Population Estimate (# m ⁻¹) (95% CI)	Trout Density (# ha ⁻¹) (95% CI)	Mean Trout Weight (g)	Standing Crop (kg ha ⁻¹) (95% CI)
June 2001	0.5	D	36 (35-39) d	105 (102-114) f	68.5	7.2 (7.0-7.8) i
June 2001	3.4	D	81 (71-94) c	369 (324-431) d	33.5	12.4 (10.8-14.4) f
June 2001	4.2	D	124 (121-129) b	484 (473-504) c	68.5	33.2 (32.4-34.6) d
June 2001	6.2	SR	182 (181-185) a	632 (628-642) b	131.9	83.3 (82.9-84.7) a
June 2001	8.9	SR	192 (185-200) a	736 (709-765) a	97.7	72 (69.3-74.8) b
June 2001	11.3	B	90 (90-91) c	301 (301-304) e	92.1	27.7 (27.7-28.0) e
June 2001	12.6	B	85 (81-91) c	296 (282-318) e	130.9	38.7 (36.9-41.6) c
June 2001	14.3	B	20 (20-22) f	63 (63-70) i	109.2	6.9 (6.9-7.7) j
June 2001	15.9	K	24 (24-26) e	91 (91-99) g	130.7	11.9 (11.9-13.0) f
June 2001	18.9	K	25 (25-27) e	82 (82-88) h	121.1	9.9 (9.9-10.6) g
June 2001	20.1	K	3 (3-4) g	10 (10-14) k	113.3	1.2 (1.2-1.6) k
June 2001	23.0	K	21 (21-22) f	57 (57-61) j	154.6	8.8 (8.8-9.5) h

Table 2a.10. Brown trout population estimates (number m^{-1}), trout density (number ha^{-1}), and standing crop (kg ha^{-1}) (95% CI) for age-1 and older brown trout at 12 sites in the Smith River, Virginia, sampled using 3-pass depletion electrofishing in June 2002. Means in the same column with the same letter are not significantly different ($\alpha=0.05$). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Basnett; K=Koehler.

Date	Site	Reach	Population Estimate (# m^{-1}) (95% CI)	Trout Density (# ha^{-1}) (95% CI)	Mean Trout Weight (g)	Standing Crop (kg ha^{-1}) (95% CI)
June 2002	0.5	D	155 (146-165) a	452 (426-480) b	60.7	27.4 (25.8-29.1) b
June 2002	3.4	D	67 (67-69) d	306 (306-315) c	50.1	15.3 (15.3-15.8) e
June 2002	4.2	D	116 (109-125) b	453 (426-487) b	59.7	27.1 (25.4-29.1) b
June 2002	6.2	SR	72 (72-72) c	250 (250-251) d	113.5	28.4 (28.4-28.5) b
June 2002	8.9	SR	39 (39-40) h	150 (150-152) f	89.2	13.3 (13.3-13.6) f
June 2002	11.3	B	146 (145-149) a	488 (485-497) a	79.4	38.8 (38.5-39.4) a
June 2002	12.6	B	76 (72-83) c	265 (251-288) d	86.7	23.0 (21.7-25.0) c
June 2002	14.3	B	43 (43-45) g	136 (136-141) g	98.7	13.5 (13.4-13.9) f
June 2002	15.9	K	47 (46-50) f	179 (175-191) e	106.4	19.0 (18.6-20.3) d
June 2002	18.9	K	29 (29-29) i	95 (95-96) h	78.4	7.5 (7.4-7.5) g
June 2002	20.1	K	15 (15-15) j	52 (52-53) i	72.7	3.8 (3.8-3.9) h
June 2002	23.0	K	54 (54-56) e	147 (147-152) f	131.7	19.4 (19.3-20.0) d

Table 2a.11. Brown trout population estimates (number m⁻¹), trout density (number ha⁻¹), and standing crop (kg ha⁻¹) (95% CI) for age-1 and older brown trout at 12 sites in the Smith River, Virginia, sampled using 3-pass depletion electrofishing in July 2003. Means in the same column with the same letter are not significantly different (alpha=0.05). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Bassett; K=Koehler.

Date	Site	Reach	Population Estimate (# m ⁻¹) (95% CI)	Trout Density (# ha ⁻¹) (95% CI)	Mean Trout Weight (g)	Standing Crop (kg ha ⁻¹) (95% CI)
July 2003	0.5	D	242 (157-327) abc	706 (458-954) bc	65.1	46 (29.8-62.1) bc
July 2003	3.4	D	231 (217-245) a	1054 (989-1118) a	67.5	71.2 (66.8-75.5) a
July 2003	4.2	D	175 (175-176) b	684 (684-689) b	77.0	52.6 (52.6-53.1) b
July 2003	6.2	SR	101 (101-103) d	351 (351-358) d	83.0	29.1 (29.1-29.7) d
July 2003	8.9	SR	57 (56-60) e	219 (215-231) e	87.0	19.0 (18.7-20.1) e
July 2003	11.3	B	157 (146-168) c	525 (488-562) c	70.9	37.2 (34.6-39.8) c
July 2003	12.6	B	56 (40-86) e	195 (139-299) e	129.8	25.3 (18.1-38.8) cde
July 2003	14.3	B	51 (36-81) e	162 (114-256) e	141.0	22.8 (16.1-36.1) cde
July 2003	15.9	K	16 *	61 *	144.8	8.8 *
July 2003	18.9	K	6 (6-6) g	20 (20-21) g	176.8	3.5 (3.5-3.7)g
July 2003	20.1	K	10 (10-11) f	35 (35-38) f	126.0	4.4 (4.4-4.8) f
July 2003	23.0	K	NA	NA	NA	NA

Table 2a.12. Population estimates for age1 plus brown trout (# 100 m⁻¹) (95% CI). Means from the same site with the same letter are not significantly different between years (alpha=0.05). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Bassett; K=Koehler.

Site	Reach	Year			
		2000	2001	2002	2003
0.5	D	85 (81-91) b	36 (35-39) c	155 (146-165) a	242 (157-327) a
3.4	D	177 (175-181) b	81 (71-94) c	67 (67-69) d	231 (217-245) a
4.2	D	190 (188-193) a	124 (121-129) c	116 (109-125) c	175 (175-176) b
6.2	SR	230 (217-243) a	182 (181-185) b	72 (72-72) d	101 (101-103) c
8.9	SR	100 (99-103) b	192 (185-200) a	39 (39-40) d	57 (56-60) c
11.3	B	90 (90-92) b	90 (90-91) b	146 (145-149) a	157 (146-168) a
12.6	B	21 (19-27) b	85 (81-91) a	76 (72-83) a	56 (40-86) a
14.3	B	58 (57-61) a	20 (20-22) c	43 (43-45) b	51 (36-81) ab
15.9	K	57 (39-92) a	24 (24-26) b	47 (46-50) a	16 *
18.9	K	39 (39-39) a	25 (25-27) c	29 (29-29) b	6 (6-6) d
20.1	K	5 *	3 (3-4) c	15 (15-15) a	10 (10-11) b
23.0	K	42 (41-45) b	21 (21-22) c	54 (54-56) a	NA

Table 2a.13. Brown trout population estimates (number m^{-1}), trout density (number ha^{-1}), and standing crop (kg ha^{-1}) (95% CI) for age-0 brown trout at 12 sites in the Smith River, Virginia, tailwater sampled using 3-pass depletion electrofishing in June 2000. Means in the same column with the same letter are not significantly different ($\alpha=0.05$). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Basnett; K=Koehler.

Date	Site	Reach	Population Estimate (# m^{-1}) (95% CI)	Trout Density (# ha^{-1}) (95% CI)	Mean Trout Weight (g)	Standing Crop (kg ha^{-1}) (95% CI)
June 2000	0.5	D	160 (128-192) a	467 (374-560) b	0.3	0.15 (0.12-0.18) e
June 2000	3.4	D	166 (101-248) a	757 (461-1132) ab	0.8	0.61 (0.37-0.92) d
June 2000	4.2	D	29 *	113 *	1.1	0.12 *
June 2000	6.2	SR	143 (121-165) b	497 (420-573) b	2.4	1.2 (1.02-1.39) bc
June 2000	8.9	SR	273 (170-376) a	1047 (651-1443) a	2.3	2.43 (1.51-3.36) a
June 2000	11.3	B	42 (33-60) c	140 (110-201) c	2.4	0.34 (0.27-0.49) d
June 2000	12.6	B	121 (105-138) b	421 (366-482) b	2.1	0.89 (0.77-1.02) c
June 2000	14.3	B	127 (105-150) b	403 (333-477) b	3.4	1.35 (1.12-1.60) ab
June 2000	15.9	K	75 (42-146) bc	285 (160-554) bc	2.1	0.59 (0.33-1.15) bcd
June 2000	18.9	K	55 (47-68) c	180 (154-224) c	3.2	0.57 (0.49-0.71) d
June 2000	20.1	K	5 (5-8) d	17 (17-29) d	3.9	0.07 (0.07-0.11) f
June 2000	23.0	K	5 *	14 *	4.6	0.06 *

Table 2a.14. Brown trout population estimates (number m⁻¹), trout density (number ha⁻¹), and standing crop (kg ha⁻¹) (95% CI) for age-0 brown trout at 12 sites in the Smith River, Virginia, tailwater sampled using 3-pass depletion electrofishing in June 2001. Means in the same column with the same letter are not significantly different (alpha=0.05). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Basnett; K=Koehler.

Date	Site	Reach	Population Estimate (# m ⁻¹) (95% CI)	Trout Density (# ha ⁻¹) (95% CI)	Mean Trout Weight (g)	Standing Crop (kg ha ⁻¹) (95% CI)
June 2001	0.5	D	4 (4-7) e	12 (12-21) e	0.3	0.01 (0.00-0.01) e
June 2001	3.4	D	168 (115-226) a	766 (525-1029) a	0.7	0.51 (0.35-0.68) ab
June 2001	4.2	D	102 *	398 *	1.2	0.48 *
June 2001	6.2	SR	63 (61-68) c	219 (212-235) c	2.1	0.45 (0.44-0.49) b
June 2001	8.9	SR	262 *	1004 *	2.2	2.21 *
June 2001	11.3	B	25 (23-31) d	84 (77-104) d	2.3	0.19 (0.18-0.24) c
June 2001	12.6	B	77 (70-87) b	268 (244-303) b	2.1	0.57 (0.52-0.64) a
June 2001	14.3	B	56 (49-68) c	177 (155-214) c	2.9	0.51 (0.44-0.61) ab
June 2001	15.9	K	81 (59-114) bc	308 (224-432) bc	2.6	0.8 (0.58-1.12) a
June 2001	18.9	K	14 *	46 *	3.4	0.16 *
June 2001	20.1	K	1 *	3 *	2.1	0.01 *
June 2001	23.0	K	2 (2-7) c	5 (5-19) e	4.5	0.02 (0.02-0.08) d

Table 2a.15. Brown trout population estimates (number m⁻¹), trout density (number ha⁻¹), and standing crop (kg ha⁻¹) (95% CI) for age-0 brown trout at 12 sites in the Smith River, Virginia, tailwater sampled using 3-pass depletion electrofishing in June 2002. Means in the same column with the same letter are not significantly different (alpha=0.05). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Basnett; K=Koehler.

Date	Site	Reach	Population Estimate (# m ⁻¹) (95% CI)	Trout Density (# ha ⁻¹) (95% CI)	Mean Trout Weight (g)	Standing Crop (kg ha ⁻¹) (95% CI)
June 2002	0.5	D	0	0	0	0
June 2002	3.4	D	95 *	434 *	1.6	0.69 *
June 2002	4.2	D	317 (256-378) b	1238 (1000-1476) b	1.4	1.78 (1.44-2.13) cd
June 2002	6.2	SR	476 (455-497) a	1653 (1579-1727) a	2.2	3.61 (3.45-3.77) a
June 2002	8.9	SR	283 (237-329) bc	1085 (909-1261) b	2.8	2.99 (2.50-3.47) ab
June 2002	11.3	B	71 (68-77) e	237 (227-256) e	3.0	0.71 (0.68-0.77) f
June 2002	12.6	B	215 (185-245) c	749 (645-852) c	2.0	1.53 (1.31-1.74) de
June 2002	14.3	B	223 (199-247) c	707 (632-781) c	3.4	2.37 (2.12-2.62) bc
June 2002	15.9	K	143 (107-182) d	544 (407-691) c	2.4	1.3 (0.97-1.66) de
June 2002	18.9	K	108 (101-117) d	354 (331-383) d	3.4	1.21 (1.13-1.31) e
June 2002	20.1	K	58 (42-87) e	201 (145-300) e	3.0	0.61 (0.44-0.91) f
June 2002	23.0	K	19 *	52 *	3.9	0.20 *

Table 2a.16. Brown trout population estimates (number m⁻¹), trout density (number ha⁻¹), and standing crop (kg ha⁻¹) (95% CI) for age-0 brown trout at 12 sites in the Smith River, Virginia, tailwater sampled using 3-pass depletion electrofishing in July 2003. Means in the same column with the same letter are not significantly different (alpha=0.05). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Basnett; K=Koehler.

Date	Site	Reach	Population Estimate (# m ⁻¹) (95% CI)	Trout Density (# ha ⁻¹) (95% CI)	Mean Trout Weight (g)	Standing Crop (kg ha ⁻¹) (95% CI)
July 2003	0.5	D	0	0	0	0
July 2003	3.4	D	116 (99-135) a	529 (452-614) a	1.0	0.55 (0.47-0.64) a
July 2003	4.2	D	23 *	90 *	1.4	0.13 *
July 2003	6.2	SR	31 (25-45) b	108 (87-157) b	1.8	0.19 (0.16-0.28) b
July 2003	8.9	SR	21 *	80 *	1.7	0.14 *
July 2003	11.3	B	0	0	0	0
July 2003	12.6	B	3 (3-4) e	10 (10-14) e	1.5	0.02 (0.02-0.02) e
July 2003	14.3	B	5 (5-5) d	16 (16-17) d	2.0	0.03 (0.03-0.03) d
July 2003	15.9	K	7 (7-7) c	27 (27-28) c	2.2	0.06 (0.06-0.06) c
July 2003	18.9	K	1 *	3 *	3.4	0.01 *
July 2003	20.1	K	1 *	3 *	1.7	0.01 *
July 2003	23.0	K	NA	NA	NA	NA

Table 2a.17. Age-0 population estimates between years brown trout (# 100 m⁻¹) (95% CI). Means from the same site with the same letter are not significantly different between years (alpha=0.05). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Bassett; K=Koehler.

Site	Reach	<u>Year</u>			
		2000	2001	2002	2003
0.5	D	160 (128-192) a	4 (4-7) b	0	0
3.4	D	166 (101-248) a	168 (115-226) a	95 *	116 (99-135) a
4.2	D	29 *	102 *	317 (256-378)	23 *
6.2	SR	143 (121-165) b	63 (61-68) c	476 (455-497) a	31 (25-45) d
8.9	SR	273 (170-376) a	262 *	283 (237-329) a	21 *
11.3	B	42 (33-60) b	25 (23-31) c	71 (68-77) a	0
12.6	B	121 (105-138) b	77 (70-87) c	215 (185-245) a	3 (3-4) d
14.3	B	127 (105-150) b	56 (49-68) c	223 (199-247) a	5 (5-5) d
15.9	K	75 (42-146) a	81 (59-114) a	143 (107-182) a	7 (7-7) b
18.9	K	55 (47-68) b	14 *	108 (101-117) a	1 *
20.1	K	5 (5-8) b	1 *	58 (42-87) a	1 *
23.0	K	5 *	2 (2-7)	19 *	NA

Table 2a.18. Density (# ha⁻¹;95% CI) of age-1 and older brown trout in the Smith River, Virginia, captured from 2000-2003 using barge electrofishers. Means from the same site with the same letter are not significantly different between years (alpha=0.05). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Bassett; K=Koehler.

Site	Reach	<u>Year</u>			
		2000	2001	2002	2003
0.5	D	248 (236-267) b	105 (102-114) c	452 (426-480) a	706 (458-954) a
3.4	D	807 (798-823) b	369 (324-431) c	306 (306-315) d	1054 (989-1118) a
4.2	D	742 (734-756) a	484 (473-504) c	453 (426-487) c	684 (684-689) b
6.2	SR	799 (754-843) a	632 (628-642) b	250 (250-251) d	351 (351-358) c
8.9	SR	384 (380-396) b	736 (709-765) a	150 (150-152) d	219 (215-231) c
11.3	B	301 (301-307) b	301 (301-304) b	488 (485-497) a	525 (488-562) a
12.6	B	73 (66-96) c	296 (282-318) a	265 (251-288) ab	195 (139-299) b
14.3	B	184 (181-194) a	63 (63-70) c	136 (136-141) b	162 (114-256) ab
15.9	K	217 (148-349) a	91 (91-99) b	179 (175-191) a	61 *
18.9	K	128 (128-129) a	82 (82-88) c	95 (95-96) b	20 (20-21) d
20.1	K	17 *	10 (10-14) c	52 (52-53) a	35 (35-38) b
23.0	K	114 (112-124) b	57 (57-61) c	147 (147-152) a	NA

Table 2a.19. Density (# ha⁻¹; 95% C) of age-0 brown trout in the Smith River, Virginia, collected from 2000-2003 using barge electrofishers. Means from the same site with the same letter are not significantly different between years (alpha=0.05). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Bassett; K=Koehler.

Site	Reach	<u>Year</u>			
		2000	2001	2002	2003
0.5	D	467 (374-560) a	12 (12-21) b	0	0
3.4	D	757 (461-1132) a	766 (525-1029) a	434 *	529 (452-614) a
4.2	D	113 *	398 *	1238 (1000-1476)	90 *
6.2	SR	497 (420-573) b	219 (212-235) c	1653 (1579-1727) a	108 (87-157) d
8.9	SR	1047 (651-1443) a	1004 *	1085 (909-1261) a	80 *
11.3	B	140 (110-201) b	84 (77-104) c	237 (227-256) a	0
12.6	B	421 (366-482) b	268 (244-303) c	749 (645-852) a	10 (10-14) d
14.3	B	403 (333-477) b	177 (155-214) c	707 (632-781) a	16 (16-17) d
15.9	K	285 (160-554) a	308 (224-432) a	544 (407-691) a	27 (27-28) b
18.9	K	180 (154-224) b	46 *	354 (331-383) a	3 *
20.1	K	17 (17-29) b	3 *	201 (145-300) a	3 *
23.0	K	14 *	5 (5-19)	52 *	NA

Table 2a.20. Standing crop (kg ha⁻¹; 95% CI in parentheses) of age-1 and older brown trout in the Smith River, Virginia, collected from 2000-2003 using barge electrofishers. Means from the same site with the same letter are not significantly different between years (alpha=0.05). Asterisk values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Bassett; K=Koehler.

Site	Reach	<u>Year</u>			
		2000	2001	2002	2003
0.5	D	10.2 (9.7-10.9) c	7.2 (7.0-7.8) d	27.4 (25.8-29.1) b	46.0 (29.8-62.1) a
3.4	D	54.1 (53.5-55.2) b	12.4 (10.8-14.4) d	15.3 (15.3-15.8) c	71.2 (66.8-75.5) a
4.2	D	68.7 (68.0-70.0) a	33.2 (32.4-34.6) c	27.1 (25.4-29.1) d	52.6 (52.6-53.1) b
6.2	SR	98.8 (93.3-104.3) a	83.3 (82.9-84.7) b	28.4 (28.4-28.5) d	29.1 (29.1-29.7) c
8.9	SR	35.2 (34.9-36.4) b	72.0 (69.3-74.8) a	13.3 (13.3-13.6) d	19.0 (18.7-20.1) c
11.3	B	31.7 (31.7-32.4) b	27.7 (27.7-28.0) c	38.8 (38.5-39.4) a	37.2 (34.6-39.8) a
12.6	B	11.8 (10.7-15.5) c	38.7 (36.9-41.6) a	23 (21.7-25.0) b	25.3 (18.1-38.8) ab
14.3	B	24.9 (24.5-26.2) a	6.9 (6.9-7.7) c	13.5 (13.4-13.9) b	22.8 (16.1-36.1) a
15.9	K	30.3 (20.8-48.9) a	11.9 (11.9-13.0) c	19 (18.6-20.3) b	8.8 *
18.9	K	11.5 (11.5-11.6) a	9.9 (9.9-10.6) b	7.5 (7.4-7.5) c	3.5 (3.5-3.7) d
20.1	K	1.5 *	1.2 (1.2-1.6) c	3.8 (3.8-3.9) b	4.4 (4.4-4.8) a
23.0	K	12.4 (12.1-13.4) b	8.8 (8.8-9.5) c	19.4 (19.3-20.0) a	NA

Table 2a.21. Standing crop (kg ha⁻¹; 95% CI in parentheses) for age-0 brown trout in the Smith River, Virginia, collected from 2000-2003 using barge electrofishers. Means from the same site with the same letter are not significantly different between years (alpha=0.05). Asterisked values indicate nondescending removal patterns or fish only being caught during one pass. Reaches: D=Dam; SR=Special Regulations; B=Bassett; K=Koehler.

Site	Reach	<u>Year</u>			
		2000	2001	2002	2003
0.5	D	0.15 (0.12-0.18) a	0.01 (0.00-0.01) b	0	0
3.4	D	0.61 (0.37-0.92) a	0.51 (0.35-0.68) a	0.69 *	0.55 (0.47-0.64) a
4.2	D	0.12 *	0.48 *	1.78 (1.44-2.13)	0.13 *
6.2	SR	1.2 (1.02-1.39) b	0.45 (0.44-0.49) c	3.61 (3.45-3.77) a	0.19 (0.16-0.28) d
8.9	SR	2.43 (1.51-3.36) a	2.21 *	2.99 (2.50-3.47) a	0.14 *
11.3	B	0.34 (0.27-0.49) b	0.19 (0.18-0.24) c	0.71 (0.68-0.77) a	0
12.6	B	0.89 (0.77-1.02) b	0.57 (0.52-0.64) c	1.53 (1.31-1.74) a	0.02 (0.02-0.02) d
14.3	B	1.35 (1.12-1.60) b	0.51 (0.44-0.61) c	2.37 (2.12-2.62) a	0.03 (0.03-0.03) d
15.9	K	0.59 (0.33-1.15) a	0.8 (0.58-1.12) a	1.3 (0.97-1.66) a	0.06 (0.06-0.06) b
18.9	K	0.57 (0.49-0.71) b	0.16 *	1.21 (1.13-1.31) a	0.01 *
20.1	K	0.07 (0.07-0.11) b	0.01 *	0.61 (0.44-0.91) a	0.01 *
23.0	K	0.06 *	0.02 (0.02-0.08)	0.20 *	NA

Table 2a.22. Biomass and density estimates for age-1 brown trout in the Smith River and other rivers in the United States.

Location	Biomass (kg ha ⁻¹)	Density (# ha ⁻¹)
Smith River, Virginia ^a	14.5	381
Smith River, Virginia, Dam Reach	19.8	615
Smith River, Virginia, Special Regulations Reach	24.8	696
Smith River, Virginia, Bassett Reach	13.5	258
Smith River, Virginia, Koehler Reach	5.9	123
Tellico River, Tennessee ^b	13.47	339
North River, Tennessee ^b	8.21	364
Beaverdam Creek, Tennessee ^b	16.59	223
Laurel Creek, Tennessee ^b	25.58	274
Laurel Fork, Tennessee ^c	9.41	197
Minnesota streams ^d	162	2279

^a Current study, average from entire tailwater

^b Habera et al. 2002

^c Strange et al. 2000;

^d Kwak and Waters 1997; total salmonid biomass and density including brown, brook, and rainbow trout

Table 2a.23. Relative abundance (#/100 m) of age-1 and older brown trout from four reaches in the Smith River, Virginia, collected from 2000 to 2003. Reaches in the same sample date with the same letter are not significantly different. Probability levels are from analysis of variance on ranked relative abundance values.

Date	<u>Reaches</u>				Probability
	Dam	Special Regulations	Bassett	Koehler	
June 2000	114 (30.35) a	100 (26.50) ab	42 (17.90) ab	22 (7.45)b	0.0218
October 2000	75 (16.58) ab	92 (6.97) a	27 (2.10) b	16 (3.17) c	0.0004
April 2001	71 (14.88) a	80 (4.83) a	48 (11.53) ab	20 (3.11) b	0.0034
June 2001	50 (17.02) ab	137 (9.00) a	49 (18.72) ab	13 (3.59) b	0.0175
October 2001	67 (20.37) ab	79 (2.13) a	35 (10.88) ab	16 (2.42) b	0.0265
April 2002	131 (21.22) ab	169 (4.96) a	52 (10.42) bc	21 (6.95) c	0.0009
June 2002	75 (13.92) a	52 (17.00) a	67 (26.08) a	29 (5.91) a	0.0771
October 2002	90 (13.61) a	115 (1.97) a	36 (9.36) b	13 (2.78) c	<0.0001
July 2003	124 (27.53) a	60 (21.00) ab	41 (26.67) ab	6 (1.00) b	0.0133
October 2003	150 (15.78)	84 (-) a	29 (-) a	12 (1.93) a	0.0845

Table 2a.24. Relative abundance (#/100 m) of age-0 brown trout from four reaches in the Smith River, Virginia, collected from 2000 to 2003. Reaches in the same sample date with the same letter are not significantly different. Probability levels are from analysis of variance on ranked relative abundance values.

Date	Dam	<u>Reaches</u>			Probability
		Special Regulations	Bassett	Koehler	
June 2000	33 (15.53) ab	65 (1.50) a	44 (14.01) ab	11 (4.73) b	0.0309
October 2000	35 (16.94) a	64 (17.52) a	44 (6.42) a	30 (12.41) a	0.6519
April 2001	2 (2.15) a	6 (6.01) a	0 (0.28) a	0 (0.00) a	0.1402
June 2001	27 (15.01) a	74 (34.00) a	26 (7.80) a	7 (6.26) a	0.0989
October 2001	47 (21.50) a	68 (9.69) a	37 (11.50) a	26 (12.29) a	0.4704
April 2002	3 (2.69) a	2 (1.70) a	1 (0.69) a	1 (0.41) a	0.9186
June 2002	43 (32.52) a	193 (80.50) a	81 (18.28) a	34 (12.73) a	0.0998
October 2002	95 (44.39) a	146 (48.94) a	91 (21.41) a	23 (6.97) a	0.1105
July 2003	19 (17.52) a	8 (5.00) a	2 (1.15) a	3 (1.67) a	0.7576
October 2003	20 (2.61) a	10 (-) a	3 (-) a	1 (0.95) a	0.0845

Table 2a.25. Mean relative stock density indices (standard deviation; sd in parentheses) for brown trout greater than 230 mm from 4 reaches collected from 2000 to 2003 from the Smith River, Virginia. Mean values within a year followed by the same letter are not significantly different between reaches (alpha=0.05).

Reach	<u>Year</u>			
	2000 (sd)	2001 (sd)	2002 (sd)	2003 (sd)
Dam	37.1 (9.05) b	39.8 (12.28) a	33.2 (10.23) a	27.9 (8.53) b
Special Regulations	46.8 (12.58) ab	49.6 (16.77) a	37.1 (14.08) a	37.0 (0.60) ab
Bassett	47.1 (7.95) ab	45.9 (10.63) a	34.9 (10.56) a	42.6 (15.86) ab
Koehler	49.8 (10.59) a	48.0 (17.05) a	49.4 (14.98) a	56.8 (12.02) a

Table 2a.26. Mean relative stock density indices (standard deviation; sd in parentheses) for brown trout greater than 300 mm from 4 reaches collected during 2000, 2001, and 2002 from the Smith River, Virginia. Mean values within a year followed by the same letter are not significantly different between reaches ($\alpha=0.05$).

Reach	<u>Year</u>			
	2000 (sd)	2001 (sd)	2002 (sd)	2003 (sd)
Dam	1.6 (1.7) b	2.0 (1.39) b	1.0 (0.97) b	1.8 (1.85) a
Special Regulations	3.1 (2.89) b	5.6 (5.62) ab	3.1 (2.82) b	1.0 (1.71) a
Bassett	12.3 (5.27) a	7.5 (4.62) a	3.4 (2.91) b	6.4 (9.09) a
Koehler	17.1 (10.08) a	12.0 (8.54) a	12.1 (7.84) a	12.0 (11.31) a

Table 2a.27. Mean relative weight values (W_r) (standard deviation; sd in parentheses) and sample sizes for brown trout collected from four reaches in the Smith River, Virginia, from 2000 to 2003. Values in the same sample date with the same letter are not significantly different between the reaches ($\alpha=0.05$).

Sample Date	Reach			
	Dam W_r (sd)	Special Regulations W_r (sd)	Bassett W_r (sd)	Koehler W_r (sd)
June 2000	86.5 (7.3) c	85.8 (5.4) c	91.3 (7.7) a	88.3 (6.5) b
N	628	532	310	375
October 2000	86.3 (6.9) b	86.5 (6.1) b	88.1 (7.8) a	88.3 (6.9) a
N	765	509	283	322
April 2001	86.2 (6.9) b	90.8 (5.9) a	91.3 (7.7) a	91.9 (6.7) a
N	451	400	460	255
June 2001	87.6 (7.3) a	88.0 (9.2) a	87.1 (6.4) a	87.9 (6.8) a
N	576	567	231	172
October 2001	88.4 (6.7) b	86.8 (6.3) c	89.0 (7.4) b	93.2 (15.3) a
N	561	462	427	233
April 2002	87.1 (8.8) b	87.7 (6.4) a	88.6 (6.9) a	86.1 (7.3) a
N	840	749	538	246
June 2002	88.2 (6.1) a	84.8 (6.2) c	85.9 (5.7) b	84.4 (5.4) c
N	673	450	436	212
October 2002	87.0 (7.0) a	85.3 (6.6) b	85.2 (6.9) b	84.8 (7.2) b
N	795	601	389	134
July 2003	88.2 (6.4) a	86.3 (4.8) b	87.8 (6.9) ab	90.2 (7.3) a
N	422	150	200	30
October 2003	87.2 (7.0) a	85.7 (6.5) b	83.8 (7.3) b	84.6 (7.5) b
N	1131	238	105	72

Table 2a.28. Mean relative weight values (W_r) (standard deviation; sd in parentheses) and sample sizes for brown trout collected from four reaches in the Smith River, Virginia, from three sample months from 2000 to 2003. Values in the same reach with the same letter are not significantly different between the seasons ($\alpha=0.05$).

Reach	April	June/July	October
Dam	86.8 (8.21) a	87.3 (6.58) a	87.2 (6.96) a
N	1291	3441	3252
Special Regulations	88.8 (6.40) a	86.2 (6.61) b	86.1 (6.41) b
N	1149	2361	1810
Bassett	89.8 (7.43) a	88.2 (7.03) b	87.1 (7.55) c
N	998	1638	1204
Koehler	89.1 (7.60) a	86.9 (8.57) b	88.9 (10.77) a
N	501	1001	761

Table 2a.29. Mean relative weight values (W_r) (standard deviation; sd in parentheses) and sample sizes for tagged, untagged, and brown trout with shed tags collected from four reaches in the Smith River, Virginia, from 2000 to 2003. Values in the same reach with the same letter are not significantly different between the tag types ($\alpha=0.05$).

Reach	No Tag W_r (sd)	Tagged (sd) W_r (sd)	Shed Tag (sd) W_r (sd)
Dam	87.14 (6.92) a	87.39 (7.43) a	87.16 (7.07) a
N	6277	1573	134
Special Regulations	86.71 (6.65) a	86.61 (6.42) a	86.88 (6.35) a
N	3879	1266	175
Bassett	88.49 (7.46) a	87.46 (6.97) b	87.02 (6.67) ab
N	3089	651	100
Koehler	87.84 (8.91) b	89.20 (10.54) a	84.01 (7.33) b
N	1826	409	409

Table 2a.30. Mean back-calculated length-at-age (mm) of brown trout collected from four reaches in the Smith River, Virginia, tailwater from 2000-2003.

Reach	<u>Age</u>		
	1	2	3
Dam	116 c	172 b	215 c
Special Regulations	130 b	190 a	228 bc
Bassett	139 a	191 a	237 ab
Koehler	126 b	186 a	252 a

Table 2a.31. Length-at-age (mm) data for brown trout from the Smith River and other locations in the United States.

Location	<u>Age</u>				
	1	2	3	4	5
Smith River, Virginia ^a	128	186	237	269	
Tellico River, Tennessee ^b	138	229	305	315	
North River, Tennessee ^b	123	215	291	399	448
Beaverdam Creek, Tennessee ^b	98	202	294	358	455
Laurel Creek, Tennessee ^b	124	211	271	305	
Poplar Creek, North Carolina ^c	105	182	240	297	349
Laurel Fork, Tennessee ^d	112	184	268		
Northeastern United States ^e	173	235	287	363	401
North Central United States ^e	170	242	290	432	488
Western United States ^e	157	240	320	368	429

^a Current study, average from entire tailwater

^b Habera et al. 2002

^c Coulston and Maughan 1981

^d Strange et al. 2000; length in January

^e Carlander 1969

Table 2a.32. von Bertalanffy growth parameters for brown trout collected from four reaches in the Smith River, Virginia, from 2000-2003. Seasonal growth oscillations (C) and winter point values (WP) were determined from recaptured tagged trout. Maximum theoretical length at infinity (L_{inf}) and weight (W_{inf}) and the growth parameter (K) were determined based on length-frequency data adjusted for C and WP. Theoretical age at zero length (t_0) was determined from length-at-age data.

Reach	C	WP	von Bertalanffy Parameter		K	t_0
			L_{inf} (mm)	W_{inf} (g)		
Dam	0.45	0.40	340	370	0.42	0.16
Special Regulations	0.14	0.24	417	689	0.34	0.18
Bassett	0.30	0.15	400	610	0.40	0.10
Koehler	0.10	0.10	487	1102	0.27	0.08

Table 2a.33. Instantaneous growth rates in weight of recovered tagged brown trout in the Smith River from four reaches over seven timeperiods. Number in parentheses is the number of trout that were collected during the time period. Mean growth values within a timeperiod with the same letter are not significantly different between reaches (ANCOVA with length as covariate; alpha=0.05).

Time period	<u>Reach</u>			
	Dam	Special Regulations	Bassett	Koehler
June 2000-October 2000	0.1988 b (201)	0.2091 b (202)	0.2712 a (93)	0.2970 a (107)
October 2000-April 2001	0.0642 c (65)	0.1905 b (68)	0.1598 b (29)	0.2568 a (15)
April 2001-June 2001	0.1618 ab (48)	0.1195 b (62)	0.1965 ab (17)	0.2253 a (15)
June 2001-October 2001	0.1603 a (58)	0.0958 b (51)	0.1627 ab (11)	0.1476 ab (12)
October 2001-April 2002	0.1636 b (271)	0.2562 a (236)	0.2350 a (136)	0.2473 a (60)
April 2002-June 2002	0.2611 a (184)	0.2065 b (84)	0.1762 b (76)	0.1966 ab (18)
June 2002-October 2002	0.1471 abc (132)	0.1123 c (58)	0.1220 bc (38)	0.2248 a (7)

Table 2a.34. Instantaneous growth rates in weight of brown trout from the Smith River, Virginia, and Laurel Fork, Tennessee.

Location	Time period	Instantaneous growth rate
Smith River, Virginia ^a	June-October	0.21
	October-April	0.17
	April-June	0.23
Laurel Fork, Tennessee ^b	August-October	0.77
	November-January	0.13
	February- April	0.41
	May-July	0.18
	August-October	0.45

^a Current study; average from entire tailwater

^b Strange et al. 2000

Table 2a.35. Analysis of covariance values to test for differences in growth in length between reaches within a time period. Values in bold indicate a significant difference between reaches within a timeperiod ($\alpha=0.05$).

Time period	Pr >F
June 2000-October 2000	<0.0001
October 2000-April 2001	<0.0001
April 2001-June 2001	0.0130
June 2001-October 2001	0.6632
October 2001-April 2002	<0.0001
April 2002-June 2002	0.0142
June 2002-October 2002	0.0013

Table 2a.36. Instantaneous growth rates in length of brown trout collected from the Smith River, Virginia, tailwater over seven time periods and four reaches. Mean length-adjusted growth rates in the same time period with the same letter are not significantly different between reaches (alpha=0.05). Numbers in parentheses are the number of observations for each time period

Time period	<u>Reach</u>			
	Dam	Special Regulations	Bassett	Koehler
June 2000-October 2000	0.1972 b (203)	0.1952 b (202)	0.3094 a (93)	0.3256 a (107)
October 2000-April 2001	0.0798 d (65)	0.1699 b (68)	0.1252 c (29)	0.2249 a (15)
April 2001-June 2001	0.1345 b (48)	0.1950 ab (62)	0.2417 a (17)	0.2481 ab (15)
June 2001-October 2001	0.1058 a (58)	0.1066 a (51)	0.1196 a (11)	0.0801 a (12)
October 2001-April 2002	0.1808 c (274)	0.2527 b (236)	0.2325 b (137)	0.3034 ba (60)
April 2002-June 2002	0.2343 ab (186)	0.2324 ab (84)	0.1908 b (77)	0.2929 a (18)
June 2002-October 2002	0.1481 ab (132)	0.1131 c (58)	0.1240 bc (38)	0.2129 a (7)

Table 2a.37. Analysis of covariance values to test for differences in growth rates between seasons within a reach for brown trout in the Smith River, Virginia. Values in bold indicate a significant difference between season within a reach ($\alpha=0.05$).

Reach	Pr >F
Dam	<0.0001
Special Regulations	<0.0001
Bassett	<0.0001
Koehler	0.0565

Table 2a.38. Seasonal trends in brown trout growth rates in the Smith River, Virginia, tailwater from four reaches over three seasonal periods. Means within a reach with the same letter are not significantly different between seasons ($\alpha=0.05$). Number in parentheses is the number of observations

Reach	<u>Seasonal Period</u>		
	April-June	June-October	October-April
Dam	0.2670 a (234)	0.2061 b (393)	0.1514 c (339)
Special Regulations	0.2489 a (146)	0.1770 b (311)	0.1962 b (304)
Bassett	0.1987 b (94)	0.2487 a (142)	0.1794 b (166)
Koehler	0.2048 a (33)	0.2391 a (126)	0.2024 a (75)

Table 2a.39. 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 fish with food in their stomach.

Date	Reach	Time	N _T	N _F	Average total length (range) mm
2/5/02	Dam	5:55	5	5	167.6 (109-260)
2/5/02	Dam	11:30	5	5	165.5 (89-280)
2/5/02	Dam	17:30	5	5	174.2 (100-272)
2/5/02	Dam	23:45	5	5	163.6 (81-257)
2/8/02	Special Regulations	5:45	5	5	176.8 (127-232)
2/8/02	Special Regulations	11:45	5	5	180.6 (116-227)
2/8/02	Special Regulations	18:10	5	5	164.0 (110-252)
2/8/02	Special Regulations	23:40	5	5	190.4 (104-237)
2/5/02	Bassett	7:30	5	5	144.6 (128-157)
2/5/02	Bassett	13:30	5	5	154.8 (98-207)
2/5/02	Bassett	19:20	5	4	198.8 (120-303)
2/6/02	Bassett	1:15	5	5	157.0 (118-217)
2/8/02	Koehler	7:15	5	5	129.2 (116-154)
2/8/02	Koehler	14:25	5	5	164.6 (113-252)
2/8/02	Koehler	20:15	5	5	182.2 (121-294)
2/9/02	Koehler	2:00	5	5	172.2 (102-293)
5/21/02	Dam	5:45	5	5	176.6 (120-218)
5/21/02	Dam	11:45	5	5	184.6 (100-275)
5/21/02	Dam	17:50	5	5	173.6 (115-231)
5/21/02	Dam	23:45	5	5	200.0 (143-265)
5/21/02	Special Regulations	7:30	5	5	214.8 (151-297)
5/21/02	Special Regulations	13:30	5	5	209.2 (151-318)
5/21/02	Special Regulations	19:30	5	4	176.6 (140-250)
5/22/02	Special Regulations	1:20	5	5	233.6 (171-289)
5/23/02	Bassett	5:45	5	5	191.2 (166-246)
5/23/02	Bassett	11:45	5	5	203.4 (160-270)
5/23/02	Bassett	17:30	5	5	213.0 (128-295)
5/23/02	Bassett	23:40	5	4	201.8 (142-298)
5/23/02	Koehler	7:30	5	5	291.6 (165-400)
5/23/02	Koehler	13:30	5	5	222.6 (166-293)
5/23/02	Koehler	19:30	5	5	174.0 (167-190)
5/24/02	Koehler	1:30	5	5	183.4 (142-275)
9/17/02	Dam	6:30	5	5	210.6 (158-262)
9/17/02	Dam	12:30	5	5	173.2 (142-253)
9/17/02	Dam	18:40	5	5	150.4 (120-200)
9/18/02	Dam	0:30	5	5	191.8 (141-234)
9/17/02	Special Regulations	5:15	5	5	223.8 (192-271)
9/17/02	Special Regulations	11:10	5	5	229.2 (201-258)
9/17/02	Special Regulations	17:15	5	5	218.6 (179-270)

Table 2a.39. Continued.

Date	Reach	Time	N _T	N _F	Average total length (range) mm
9/17/02	Special Regulations	23:15	5	5	215.8 (180-301)
9/19/02	Bassett	5:15	5	5	209.2 (183-256)
9/19/02	Bassett	11:30	5	5	193.2 (98-250)
9/19/02	Bassett	17:20	5	5	204.6 (116-269)
9/19/02	Bassett	23:15	5	5	242.0 (204-279)
9/19/02	Koehler	6:40	5	5	198.0 (183-236)
9/19/02	Koehler	13:10	5	5	207.6 (128-274)
9/19/02	Koehler	19:10	5	5	262.8 (199-323)
9/20/02	Koehler	0:50	5	5	209.4 (173-269)
12/12/02	Dam	5:00	5	5	172.0 (139-223)
12/12/02	Dam	11:00	5	5	204.0 (145-266)
12/12/02	Dam	16:50	5	5	183.8 (150-240)
12/12/02	Dam	22:50	5	5	231.0 (202-254)
12/12/02	Special Regulations	6:00	5	5	216.6 (143-260)
12/12/02	Special Regulations	12:15	5	5	220.6 (134-260)
12/12/02	Special Regulations	18:00	5	5	212.2 (182-234)
12/12/02	Special Regulations	23:55	5	5	216.0 (185-272)
12/10/02	Bassett	5:30	5	5	201.6 (188-237)
12/10/02	Bassett	12:40	5	5	173.8 (133-239)
12/10/02	Bassett	18:40	5	5	264.0 (196-327)
12/11/02	Bassett	0:30	5	5	199.0 (121-223)
12/10/02	Koehler	7:15	5	5	179.6 (140-208)
12/10/02	Koehler	14:55	5	5	197.4 (143-241)
12/10/02	Koehler	20:30	5	5	201.8 (135-297)
12/11/02	Koehler	2:20	5	5	193.4 (142-282)

Table 2a.40. G-test results to assess significant differences in percentages of brown trout diet items between four reaches in the Smith River , Virginia, within a season. Values in bold are significantly different between the reaches (alpha=0.05).

Month	<u>Diet item</u>								
	Ephemeroptera	Plecoptera	Trichoptera	Diptera	Fish	Decapoda	Isopoda	Gastropoda	Terrestrial
February	14.95	16.92	13.64	7.85	2.79	2.79	57.24	6.34	7.35
May	25.63	4.72	44.77	10.64	20.06	13.99	43.41	30.53	23.07
September	18.94	29.50	28.59	47.99	16.30	18.64	72.30	13.29	19.39
December	30.97	20.42	32.25	26.06	24.73	35.01	58.63	9.06	2.81

Table 2a.41. G-test results to assess significant differences in percentages of brown trout diet items between four reaches in the Smith River, Virginia, within a season. Values in bold are significantly different between the reaches within a collection month (alpha=0.05). Reaches: D=Dam, SR=Special Regulations, B=Bassett, K=Koehler.

Month	Reaches Compared	Ephemeroptera	Plecoptera	Trichoptera	Diptera	Fish	Decapoda	Isopoda	Gastropoda	Terrestrial
February	D-SR	NA	NA	NA	NA	NA	NA	32.54	NA	NA
February	D-B	NA	NA	NA	NA	NA	NA	32.23	NA	NA
February	D-K	NA	NA	NA	NA	NA	NA	43.97	NA	NA
February	SR-B	NA	NA	NA	NA	NA	NA	1.76	NA	NA
February	SR-K	NA	NA	NA	NA	NA	NA	2.59	NA	NA
February	B-K	NA	NA	NA	NA	NA	NA	2.54	NA	NA
May	D-SR	14.69	NA	20.54	NA	NA	NA	18.70	5.01	0.12
May	D-B	8.92	NA	32.41	NA	NA	NA	38.14	4.12	7.91
May	D-K	13.95	NA	21.95	NA	NA	NA	27.36	14.27	11.21
May	SR-B	5.57	NA	8.77	NA	NA	NA	1.16	8.62	8.55
May	SR-K	1.26	NA	2.95	NA	NA	NA	4.58	20.98	11.08
May	B-K	7.67	NA	6.28	NA	NA	NA	1.47	9.05	4.10
September	D-SR	NA	13.59	17.12	1.42	NA	NA	42.05	NA	NA
September	D-B	NA	16.13	18.94	4.29	NA	NA	42.05	NA	NA
September	D-K	NA	2.69	6.00	33.82	NA	NA	42.05	NA	NA
September	SR-B	NA	2.88	2.28	5.56	NA	NA	0.00	NA	NA
September	SR-K	NA	8.74	7.23	33.82	NA	NA	0.00	NA	NA
September	B-K	NA	11.40	7.45	21.95	NA	NA	0.00	NA	NA
December	D-SR	20.20	NA	21.39	8.29	4.19	4.40	24.90	NA	NA
December	D-B	12.19	NA	17.00	17.34	3.14	15.13	30.65	NA	NA
December	D-K	3.95	NA	9.11	13.53	8.34	4.40	41.23	NA	NA
December	SR-B	3.81	NA	3.92	3.98	1.84	6.20	2.06	NA	NA
December	SR-K	12.33	NA	10.61	3.98	14.07	4.50	8.04	NA	NA
December	B-K	6.44	NA	4.15	2.77	18.21	7.89	3.38	NA	NA

Table 2a.42. G-test results to assess significant differences in percentages of brown trout diet items between seasons within four reaches in the Smith River, Virginia. Values in bold are significantly different between the seasons (alpha=0.05).

Reach	Diet item								
	Ephemeroptera	Plecoptera	Trichoptera	Diptera	Fish	Decapoda	Isopoda	Gastropoda	Terrestrial
Dam	51.34	23.31	11.21	10.77	5.70	0.00	27.19	18.23	52.84
Special									
Regulations	52.35	26.16	27.52	13.12	2.79	15.82	15.93	11.07	57.34
Bassett	60.23	21.65	23.71	18.01	10.10	21.88	12.93	35.79	37.47
Koehler	48.75	28.78	12.22	36.75	25.54	20.29	4.64	26.70	41.78

Table 2a.43. G-test results to assess significant differences in percentages of brown trout diet items between seasons within four reaches in the Smith River, Virginia. Values in bold are significantly different between the seasons within a reach (alpha=0.05).

Reach	Season Compared	Ephemeroptera	Plecoptera	Trichoptera	Diptera	Fish	Decapoda	Isopoda	Gastropoda	Terrestrial
Dam	Feb-May	8.50	9.38	NA	NA	NA	NA	12.82	NA	3.43
Dam	Feb-Sept	23.71	5.06	NA	NA	NA	NA	9.24	NA	29.00
Dam	Feb-Dec	29.85	3.83	NA	NA	NA	NA	3.56	NA	1.41
Dam	May-Sept	15.32	1.41	NA	NA	NA	NA	12.77	NA	26.51
Dam	May-Dec	16.63	15.13	NA	NA	NA	NA	6.90	NA	2.18
Dam	Sept-Dec	3.75	9.74	NA	NA	NA	NA	12.45	NA	32.52
Spec Regs	Feb-May	24.07	11.92	8.62	NA	NA	NA	NA	NA	0.00
Spec Regs	Feb-Sept	35.44	4.44	10.73	NA	NA	NA	NA	NA	29.17
Spec Regs	Feb-Dec	24.13	1.73	7.06	NA	NA	NA	NA	NA	4.40
Spec Regs	May-Sept	7.91	13.80	8.89	NA	NA	NA	NA	NA	27.99
Spec Regs	May-Dec	0.27	16.64	11.65	NA	NA	NA	NA	NA	4.58
Spec Regs	Sept-Dec	6.59	6.20	9.58	NA	NA	NA	NA	NA	42.05
Bassett	Feb-May	16.67	7.29	7.52	NA	NA	4.42	NA	3.10	12.57
Bassett	Feb-Sept	40.65	4.34	13.18	NA	NA	7.38	NA	17.51	18.36
Bassett	Feb-Dec	37.06	4.80	5.39	NA	NA	14.61	NA	18.65	1.47
Bassett	May-Sept	13.20	15.44	8.99	NA	NA	4.37	NA	7.43	0.70
Bassett	May-Dec	8.27	11.51	5.83	NA	NA	9.90	NA	12.15	18.12
Bassett	Sept-Dec	2.06	1.15	5.00	NA	NA	2.15	NA	7.24	24.55
Koehler	Feb-May	29.02	13.08	NA	5.04	7.22	NA	NA	4.19	17.26
Koehler	Feb-Sept	25.18	11.24	NA	21.95	5.97	NA	NA	5.54	19.53
Koehler	Feb-Dec	27.19	12.87	NA	5.56	14.07	NA	NA	12.08	0.00
Koehler	May-Sept	3.42	2.69	NA	24.55	5.73	NA	NA	11.30	5.03
Koehler	May-Dec	5.84	7.22	NA	8.83	9.61	NA	NA	15.79	17.26
Koehler	Sept-Dec	1.84	1.50	NA	7.65	9.52	NA	NA	4.33	19.52

Table 2a.44. Summary of brown trout diet consumption study conducted in four reaches of the Smith River, Virginia, in 2002. Data are sample size, total length (TL) (sd) and weight (sd), mean relative content wet weight (RWW) (sd), number of empty stomachs, gastric evacuation rates (R), and daily ration (C₂₄) (95% Confidence Intervals) presented as RWW.

Sampling Month Reach	Total N	Mean TL (mm) (sd)	Mean Weight (g) (sd)	Mean RWW (g*100 g ⁻¹) (sd)	# empty	Water Temp (°C)	R (h ⁻¹)	C ₂₄ (%) (95% CI)
Dam								
February	20	168 (65)	66 (66)	0.35 (0.26)	0	6.36	0.0843	0.706 (±0.333)
May	20	184 (52)	74 (55)	0.37 (0.39)	0	7.95	0.0947	0.830 (±0.583)
September	19	183 (44)	68 (47)	0.33 (0.16)	0	8.93	0.1017	0.823 (±0.281)
December	19	195 (41)	81 (52)	0.21 (0.24)	0	7.48	0.0915	0.467 (±0.409)
Special Regulations								
February	20	178 (49)	71 (50)	0.57 (0.35)	0	5.97	0.0819	1.131 (±0.514)
May	20	209 (60)	109 (88)	0.80 (0.92)	1	9.11	0.1031	1.983 (±1.626)
September	20	222 (35)	113 (55)	0.68 (0.73)	0	11.11	0.1193	1.951 (±1.222)
December	20	216 (38)	99 (44)	0.17 (0.17)	0	5.76	0.0807	0.339 (±0.233)
Bassett								
February	20	164 (47)	55 (54)	0.54 (0.40)	1	4.85	0.0755	0.977 (±0.452)
May	20	206 (50)	96 (68)	0.39 (0.47)	1	11.62	0.1238	1.160 (±1.020)
September	20	212 (46)	106 (60)	0.53 (0.40)	0	15.94	0.1294	1.639 (±0.841)
December	20	210 (52)	100 (70)	0.47 (0.53)	0	5.84	0.0808	0.910 (±0.692)
Koehler								
February	20	162 (60)	58 (69)	0.57 (0.66)	0	6.01	0.0822	1.131 (±0.906)
May	20	218 (71)	133 (148)	0.53 (0.58)	0	14.18	0.1492	1.915 (±1.588)
September	20	219 (47)	112 (75)	0.47 (0.57)	0	15.94	0.1697	1.192 (±1.558)
December	20	193 (49)	81 (65)	0.54 (0.61)	0	5.84	0.0812	1.044 (±0.859)

Table 2a.45. Temperature and consumption effects on growth of age-1 brown trout during the summer at four reaches in the Smith River, Virginia from 2000 to 2003; G_O =observed growth rate, G_T = growth rate with observed temperature and thermally-neutral P -value, G_P =growth rate with observed consumption and averaged thermal regime. Thermally-neutral P -values were as follow: Dam=0.272, Special Regulations=0.256, Bassett=0.291, and Koehler=0.355.

Year	Observed P -value	G_O (% change in weight/day)	G_T (% change in weight/day)	G_P (% change in weight/day)
Dam Reach				
2000	0.264	0.42	0.45	0.43
2001	0.293	0.54	0.46	0.60
2002	0.223	0.31	0.51	0.31
2003	0.370	0.71	0.25	0.90
Special Regulations Reach				
2000	0.266	0.34	0.30	0.32
2001	0.272	0.34	0.28	0.32
2002	0.234	0.20	0.29	0.20
2003	0.299	0.28	0.07	0.56
Bassett Reach				
2000	0.298	0.41	0.38	0.37
2001	0.300	0.44	0.41	0.39
2002	0.299	0.34	0.31	0.46
2003	0.290	0.33	0.33	0.57
Koehler Reach				
2000	0.374	0.44	0.37	0.46
2001	0.325	0.32	0.42	0.28
2002	0.372	0.49	0.43	0.76
2003	0.327	0.35	0.46	0.34

Table 2a.46. Temperature and consumption effects on growth of age-0 brown trout during the summer at four reaches in the Smith River, Virginia from 2000 to 2003; G_O =observed growth rate, G_T = growth rate with observed temperature and thermally-neutral P -value, G_P =growth rate with observed consumption and averaged thermal regime. Thermally-neutral P -values were as follow: Dam=0.331, Special Regulations=0.313, Bassett=0.281, and Koehler=0.267.

Year	Observed P -value	G_O (% change in weight/day)	G_T (% change in weight/day)	G_P (% change in weight/day)
Dam Reach				
2000	0.332	1.77	1.77	1.85
2001	0.314	1.41	1.50	1.57
2002	0.304	1.25	1.39	1.32
2003	0.432	2.38	1.67	2.44
Special Regulations Reach				
2000	0.297	1.34	1.43	1.33
2001	0.314	1.44	1.43	1.48
2002	0.284	1.25	1.42	1.25
2003	0.385	2.08	1.54	2.22
Bassett Reach				
2000	0.278	1.35	1.37	1.32
2001	0.284	1.43	1.41	1.42
2002	0.275	1.24	1.28	1.30
2003	0.310	1.75	1.50	1.91
Koehler Reach				
2000	0.302	1.54	1.29	1.56
2001	0.267	1.34	1.35	1.30
2002	0.256	1.11	1.19	1.18
2003	0.248	1.32	1.49	1.30

Table 2a.47. Regression coefficients for regression of observed growth (G_O) on consumption-independent predicted growth (G_T) and temperature-independent predicted growth (G_P) for age-1 brown trout from four reaches in the Smith River, Virginia.

Reach	Slope	Intercept	R^2	Probability	Figure
GT					
Dam	-1.332	1.053	0.8148	0.0973	2.1.6a
Special Regulations	0.068	0.274	0.0125	0.8880	2.1.6b
Bassett	1.134	-0.023	0.9016	0.0505	2.1.6c
Koehler	-0.694	0.691	0.1191	0.6549	2.1.6d
GP					
Dam	0.668	0.120	0.9890	0.0055	2.1.6e
Special Regulations	0.130	0.244	0.0864	0.7061	2.1.6f
Bassett	-0.508	0.606	0.7079	0.1586	2.1.6g
Koehler	0.347	0.241	0.8806	0.0616	2.1.6h

Table 2a.48. Regression coefficients for regression of observed growth (G_O) on consumption-independent predicted growth (G_T) and temperature-independent predicted growth (G_P) for age-0 brown trout from four reaches in the Smith River, Virginia.

Reach	Slope	Intercept	R^2	Prob	Figure
GT					
Dam	2.169	-1.726	0.5216	0.2778	2.1.7a
Special Regulations	6.479	-7.923	0.9754	0.0124	2.1.7b
Bassett	2.303	-1.758	0.9250	0.0382	2.1.7c
Koehler	0.446	0.735	0.1009	0.6824	2.1.7d
GP					
Dam	1.042	-0.166	0.9937	0.0032	2.1.7e
Special Regulations	0.852	0.188	0.9992	0.0004	2.1.7f
Bassett	0.752	0.325	0.9533	0.0236	2.1.7g
Koehler	1.070	-0.102	0.9241	0.0387	2.1.7h

Table 2a.49. Stepwise regression model parameters for June-October consumption (*P*) values for age-1 brown trout in the Smith River (N=16).

Variable	Coefficient	Model R^2	Probability
Intercept	0.2047	0.00	0.0000
% Time >15.5°C	0.0020	0.70	0.0007
Mean Daily flow (cms)	0.0076	0.85	0.0175
% Time < 10°C	0.00096	0.93	0.0131
Mean Invertebrate Density (#/m ²)	-0.00006	0.97	0.0172

Table 2a.50. Predicted ending weights (predicted percent change in body weight from baseline conditions) for age-1 brown trout during summer (June to October) under baseline conditions, 12 °C outflow, new turbines to reduce flow magnitude, and a steady baseflow in four reaches of the Smith River, Virginia, modeled under fluctuations in percent of maximum consumption (*P*-value).

Reach	<i>P</i> -value	Initial weight	Baseline conditions	Predicted ending weights (g)		
				12 °C outflow (% change)	New turbines (% change)	Steady baseflow (% change)
Dam	Normal (0.288)	27.2	53.5	53.8 (0.6)	53.3 (-0.3)	52.8 (-1.2)
Dam	High (0.350)	27.2	70.7	73 (3.3)	70.4 (-0.4)	69.3 (-2.0)
Dam	Low (0.226)	27.2	39.3	38.3 (-2.5)	39.3 (0.0)	39.2 (-0.3)
Special Regulations	Normal (0.268)	47.4	72.4	67.9 (-6.2)	72.9 (0.7)	75.4 (4.2)
Special Regulations	High (0.295)	47.4	82.2	77.6 (-5.6)	82.6 (0.6)	85.2 (3.7)
Special Regulations	Low (0.241)	47.4	63.4	59 (-6.9)	63.9 (0.9)	66.3 (4.7)
Bassett	Normal (0.297)	52.1	85.6	80 (-6.6)	86.6 (1.2)	89.2 (4.2)
Bassett	High (0.301)	52.1	87.5	81.8 (-6.5)	88.5 (1.2)	91.2 (4.1)
Bassett	Low (0.292)	52.1	83.8	78.2 (-6.6)	84.8 (1.2)	87.3 (4.2)
Koehler	Normal (0.350)	61.3	103.0	96.7 (-6.1)	102.7 (-0.3)	102.5 (-0.5)
Koehler	High (0.377)	61.3	115.6	108.8 (-5.9)	115.4 (-0.2)	115.2 (-0.4)
Koehler	Low (0.322)	61.3	91.3	85.5 (-6.4)	91.0 (-0.3)	90.8 (-0.5)

Table 2a.51. Predicted ending weights (predicted percent change in body weight from baseline conditions) for age-0 brown trout during summer (June to October) under baseline conditions, 12 °C outflow, new turbines to reduce flow magnitude, and a steady baseflow in four reaches of the Smith River, Virginia, modeled under fluctuations in percent of maximum consumption (*P*-value).

Reach	<i>P</i> -value	Initial weight	Baseline conditions	Predicted ending weights (g)		
				12 °C outflow (% change)	New turbines (% change)	Steady baseflow (% change)
Dam	Normal (0.346)	1.0	7.9	8.6 (10.1)	7.8 (-1.1)	7.5 (-3.9)
Dam	High (0.404)	1.0	11.4	12.8 (12.7)	11.2 (-1.2)	10.8 (-4.6)
Dam	Low (0.287)	1.0	5.2	5.5 (7.0)	5.1 (-0.8)	5.0 (-3.1)
Special Regulations	Normal (0.320)	2.1	13.6	13.3 (-2.1)	13.5 (-0.4)	13.7 (1.0)
Special Regulations	High (0.365)	2.1	18.5	18.3 (-1.0)	18.3 (-0.7)	18.5 (0.1)
Special Regulations	Low (0.275)	2.1	9.6	9.3 (-3.5)	9.6 (-0.1)	9.8 (1.9)
Bassett	Normal (0.286)	2.4	14.1	13.3 (-5.7)	14.3 (0.9)	14.7 (3.9)
Bassett	High (0.302)	2.4	16.1	15.3 (-5.3)	16.3 (0.9)	16.7 (3.8)
Bassett	Low (0.270)	2.4	12.3	11.6 (-5.9)	12.4 (1.0)	12.8 (4.1)
Koehler	Normal (0.268)	2.7	14.2	13.4 (-5.5)	14.1 (-0.2)	14.1 (-0.2)
Koehler	High (0.292)	2.7	17.4	16.7 (-5.2)	17.4 (-0.1)	17.4 (-0.1)
Koehler	Low (0.245)	2.7	11.4	10.7 (-6.0)	11.3 (-0.2)	11.3 (-0.3)

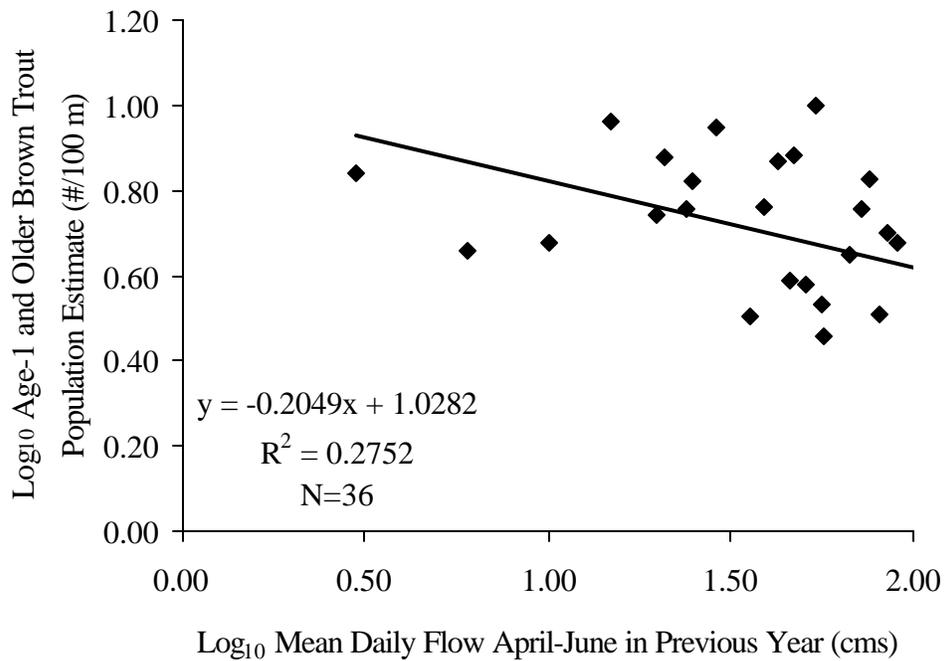
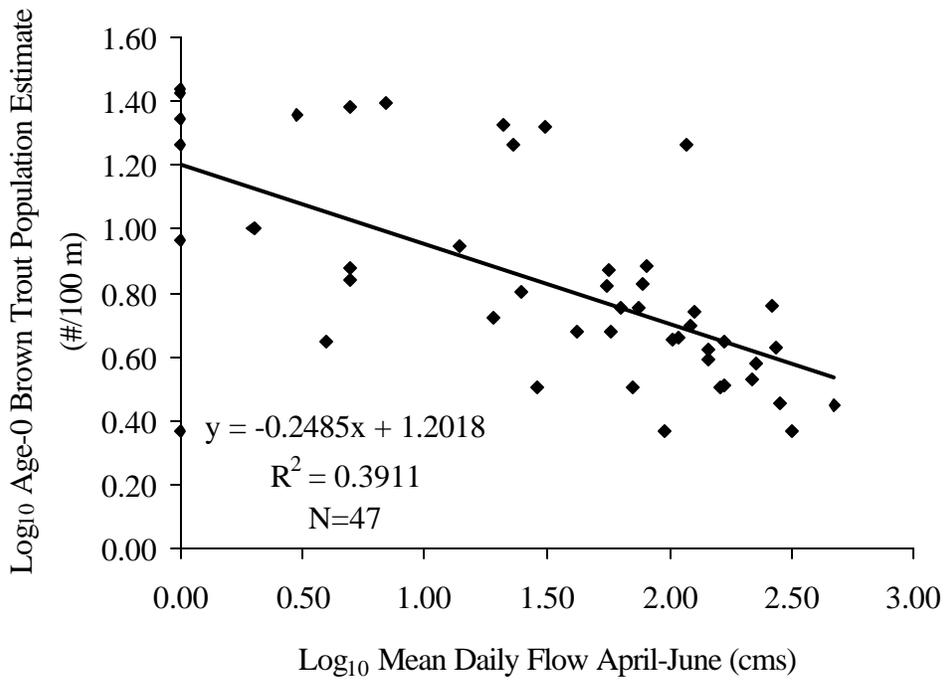


Figure 2a.1. Mean daily flows in April to June (cms) and population estimates for age-0 (Panel A) and age-1 and older (Panel B) brown trout in the Smith River, Virginia from 2000 to 2003

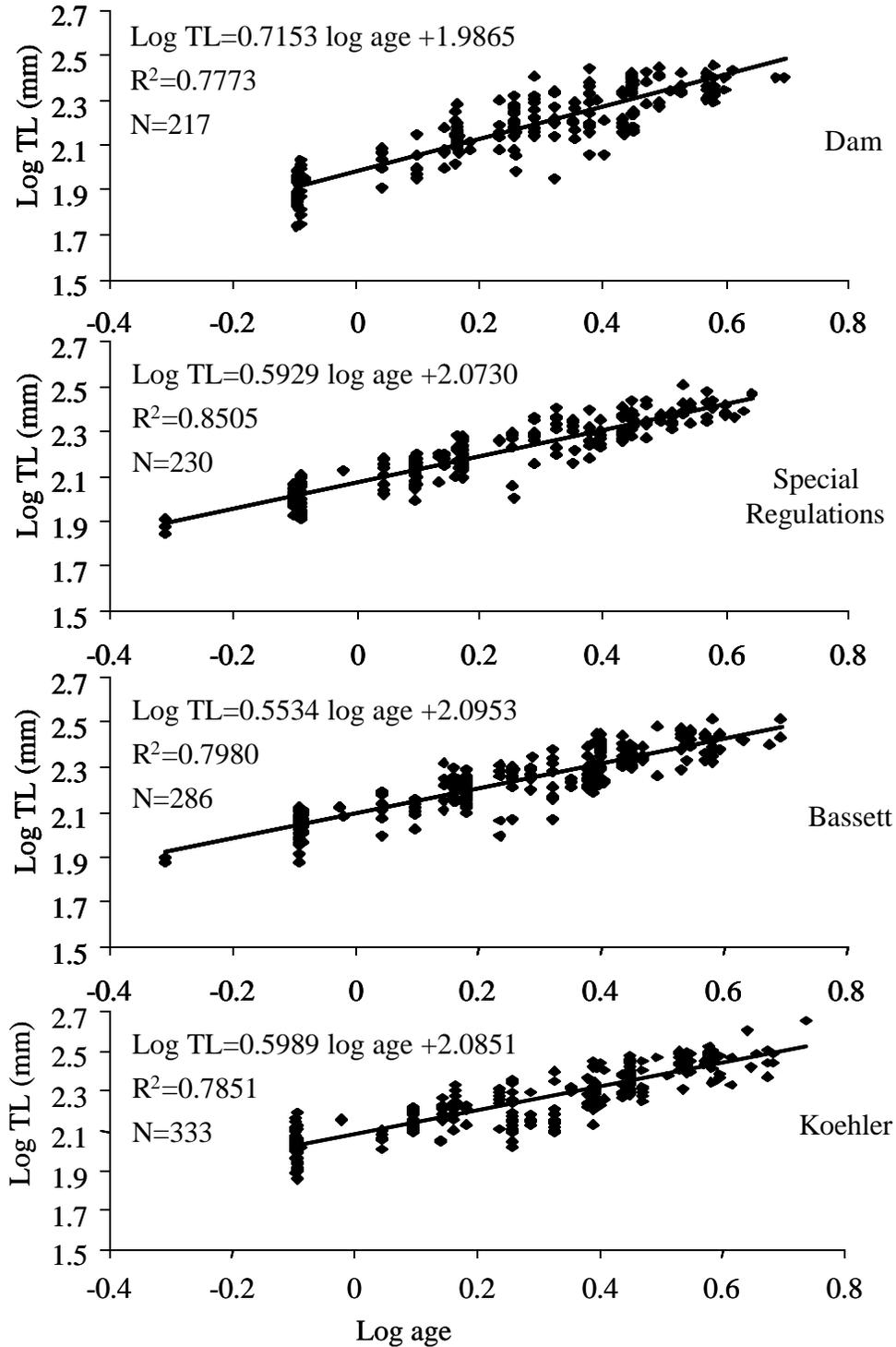


Figure 2a.2. Log₁₀ age at time of capture and log₁₀ total length at time of capture regressions for brown trout from four reaches in the Smith River, Virginia, tailwater collected from 2000-2003

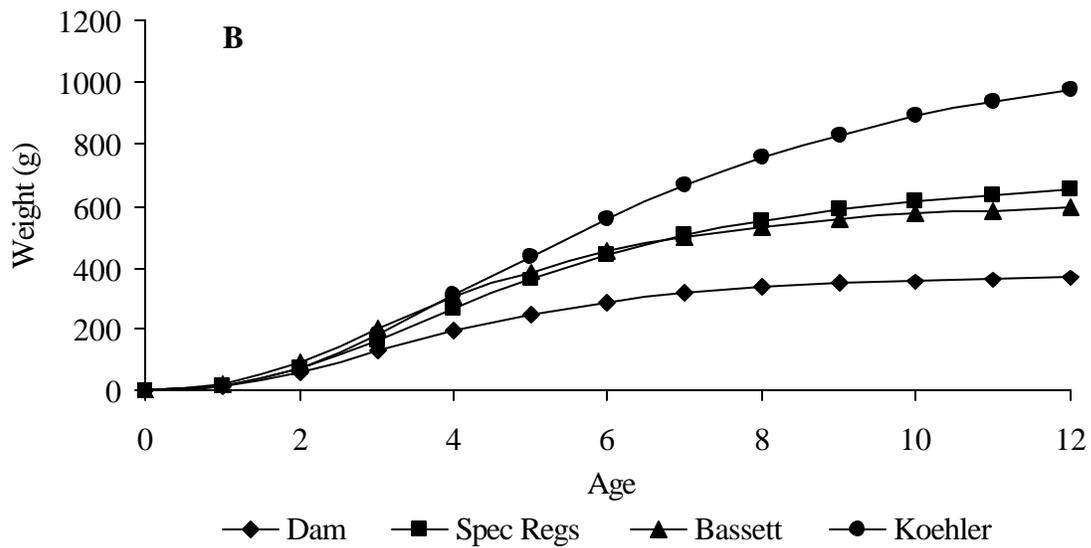
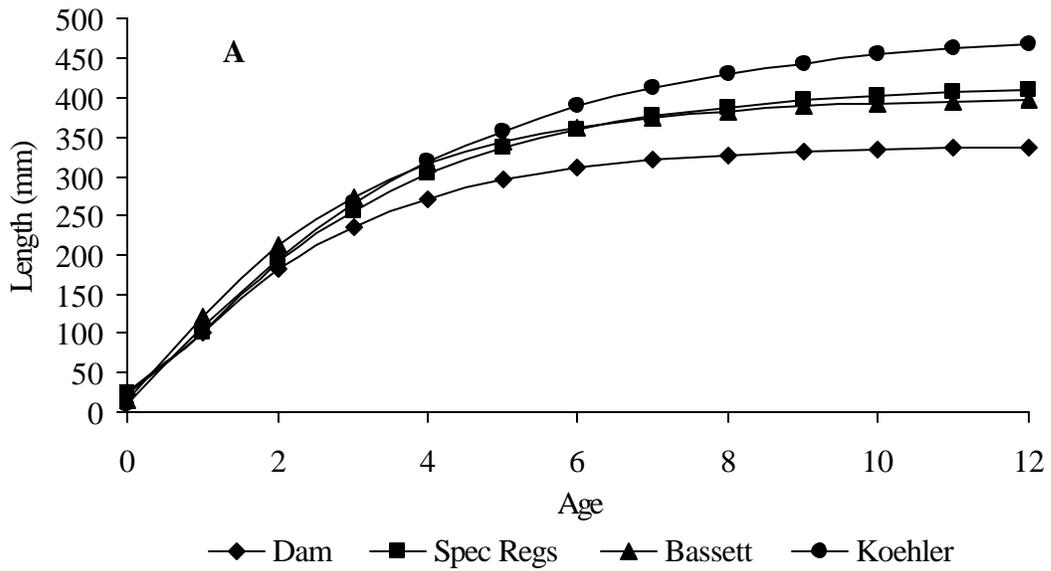


Figure 2a.3. Growth curves in length (A) and weight (B) for brown trout from four reaches in the Smith River, Virginia, based on von Bertalanffy models.

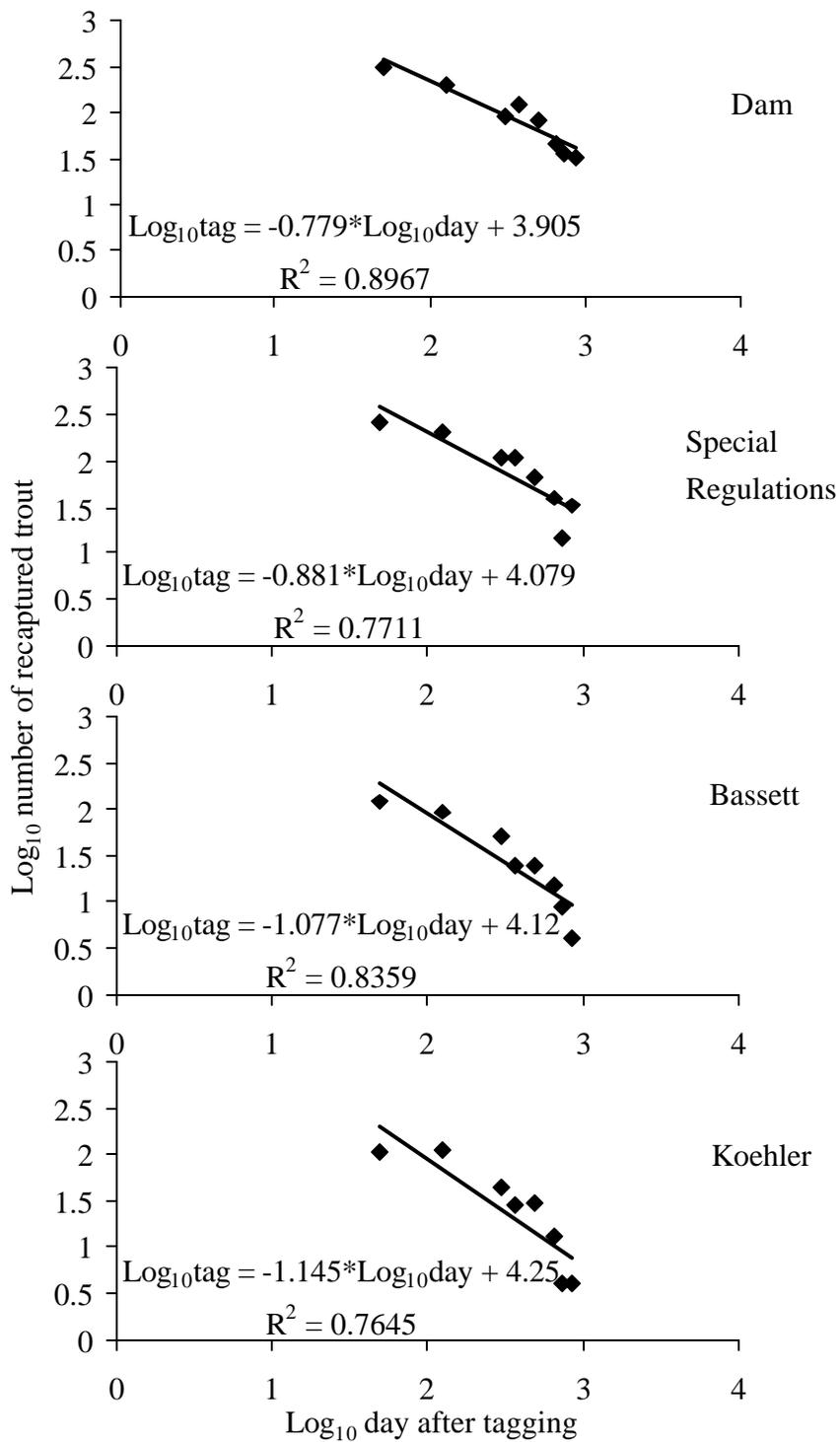


Figure 2a.4. Log₁₀ number of recaptured tagged trout that were tagged in June 2000 over time from four reaches in the Smith River, Virginia. The regression slope is equal to the mortality rate.

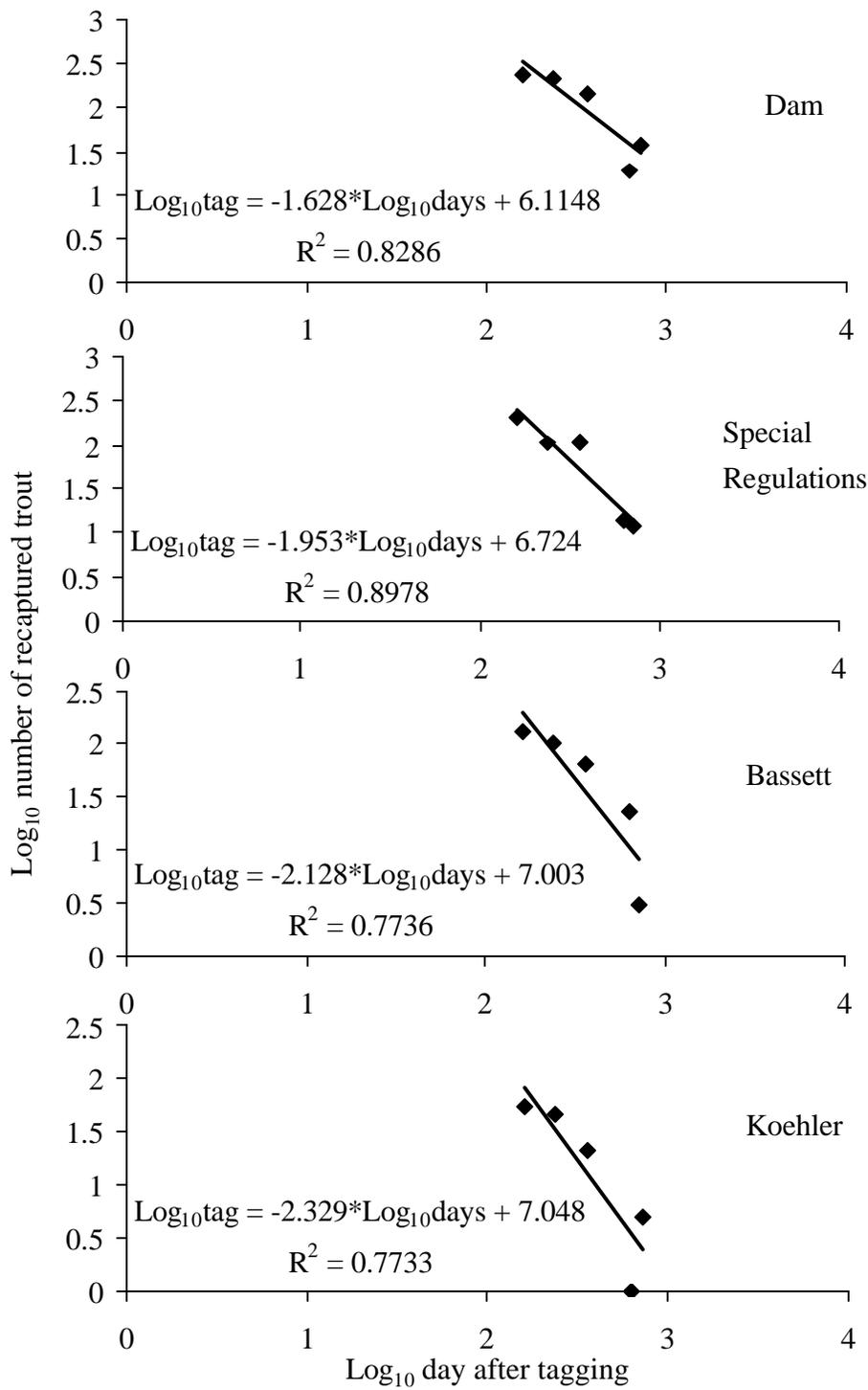


Figure 2a.5. Log₁₀ number of recaptured tagged trout that were tagged in October 2001 over time from four reaches in the Smith River, Virginia. The regression slope is equal to the mortality rate.

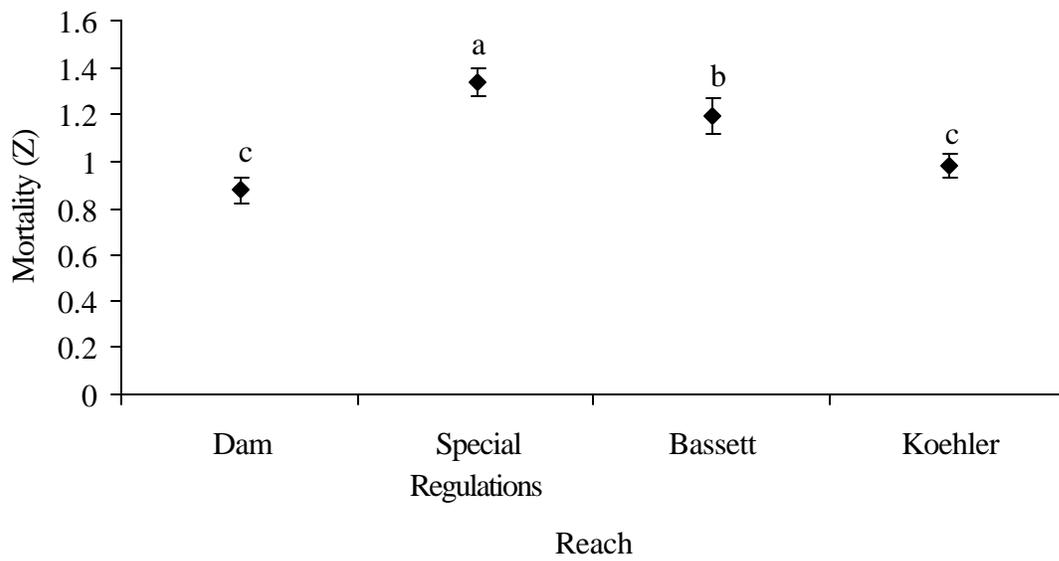


Figure 2a.6. Mortality rates of brown trout from four reaches in the Smith River, Virginia, calculated from length-frequency distributions from 2000 to 2003. Mortality rates with the same letter are not significantly different between reaches.

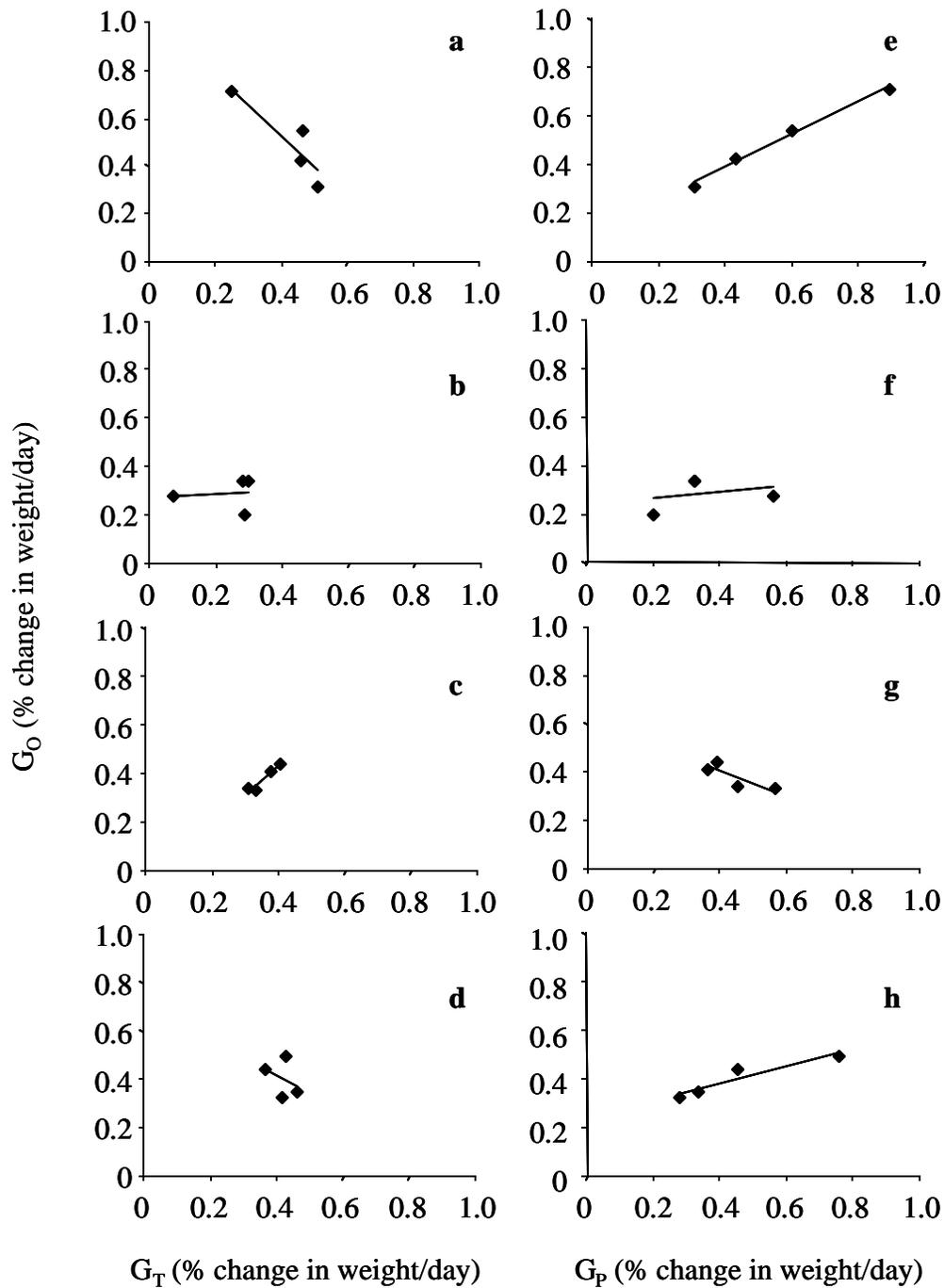


Figure 2a.7. The effects of observed temperatures with constant consumption (GT) on summer growth of age-1 brown trout in the Dam (a), Special Regulations (b), Bassett (c), and Koehler (d) reaches. The effects of observed consumption with averaged daily temperatures (GP) in the Dam (e), Special Regulations (f), Bassett (g), and Koehler (h) reaches in the Smith River, Virginia. The y-axis is the observed growth rate.

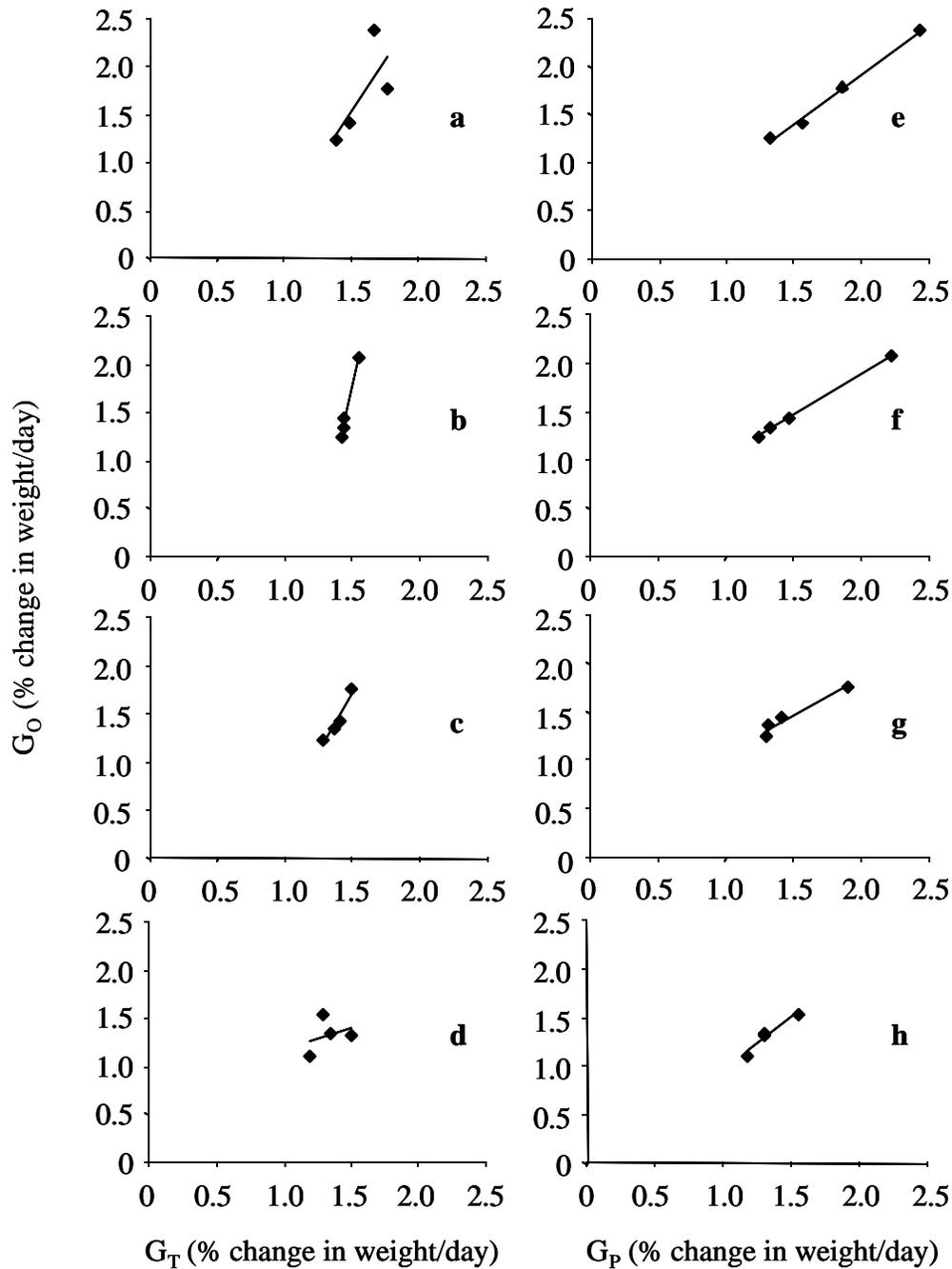


Figure 2a.8. The effects of observed temperatures with constant consumption (G_T) on summer growth of age-1 brown trout in the Dam (a), Special Regulations (b), Bassett (c), and Koehler (d) reaches. The effects of observed consumption with averaged daily temperatures (G_P) in the Dam (e), Special Regulations (f), Bassett (g), and Koehler (h) reaches in the Smith River, Virginia. The y-axis is the observed growth rate.

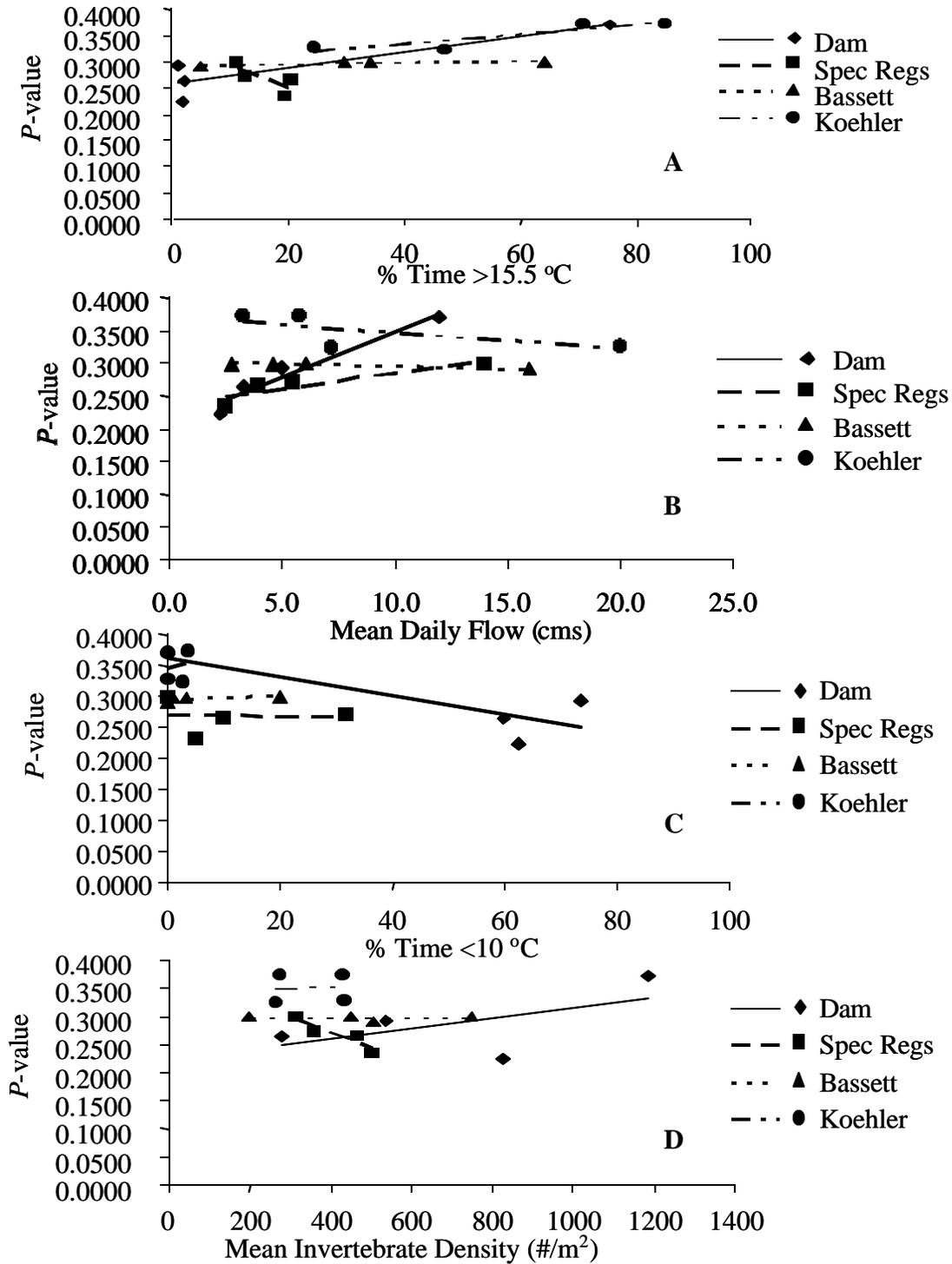


Figure 2a.9. The influence of the percent time >15.5°C (A), mean daily flow (B), percent time <10°C (C), and the mean invertebrate density (D) on consumption values (P) for age-1 brown trout from four reaches in the Smith River, Virginia from 2000-2003.

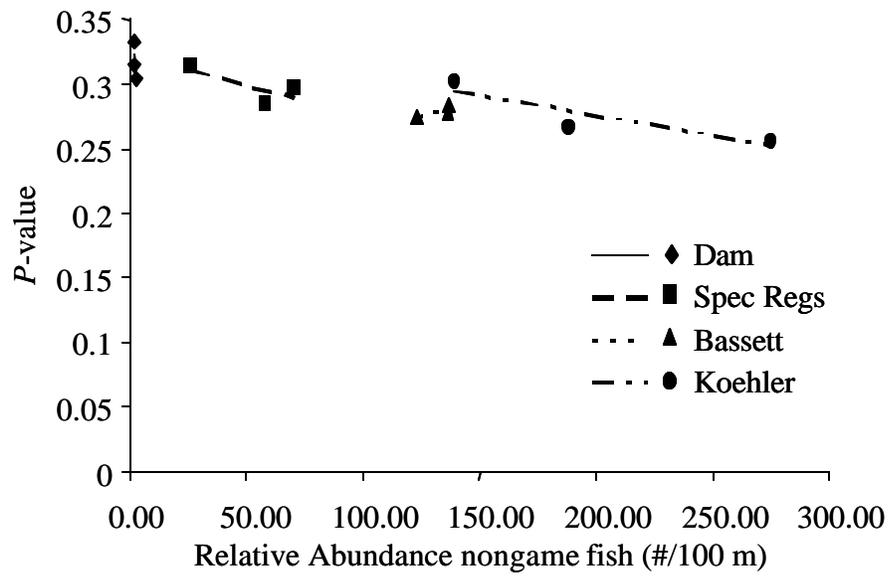


Figure 2a.10. The influence of the relative abundance of nongame fish on consumption values (P) for age-0 brown trout from four reaches in the Smith River, Virginia from 2000-2003.

Job 2, Part B. Longitudinal Patterns of Community Structure for Stream Fishes in a Virginia Tailwater

Section 1: Community structure of fishes in the Smith River in relation to longitudinal and environmental gradients below Philpott Dam.

Ecological Effects of Flow Regulation

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 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, water 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.

Fish Response to Flow Variability

Indeed, the quantity and timing of flow are crucial components of ecological function in river systems. It is increasingly recognized that the distribution and abundance of riverine species are limited by the effects of flow regulation (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 (Schlosser, 1985). Studies show that flow variability affects use of spatial resources or patterns of microhabitat use (Grossman et al., 1998). Therefore, evidence indicates a strong fish response to flow variability.

Fish Community Structure

The effects of flow regulation operate as a main structuring agent for fish abundance, diversity, and distribution. Thus, understanding how a fish community is structured in flow-regulated rivers has management implications for conservation of biodiversity. 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.

As natural resource management agencies face continued pressure to develop and defend recommendations to preserve aquatic resources, information on fish community structure is necessary, especially in relation to prominent issues such as stream flow regulation. Knowledge about the factors that most affect fish community characteristics can be incorporated into decisions to protect and enhance fisheries below dams. If analyses show adverse effects of varying flows on aquatic biota below a dam, then several mitigation strategies exist to minimize

the impact of the dam including operational changes, structural changes, or habitat modification (Cushman, 1985).

In this study, I evaluate the patterns of community structure among different populations of non-salmonid fish in a tailwater (Smith River, Henry Co., VA). The Smith River has a hydropeaking flow regime with a hypolimnetic release that dramatically depresses the temperature of the river, providing an excellent template for research on patterns in abundance, diversity, and distribution in relation to longitudinal and environmental gradients in a flow-regulated river. I specifically addressed the following questions:

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

Study Area, Species Present, and Current Management

The Smith River is a sixth order, regulated tributary of the Dan River, located in Virginia's Patrick and Henry Counties. The river is in the Roanoke drainage where forestry and agriculture are the dominant land uses in the upper sections, and urban and industrial development in the Bassett-Martinsville sections. Philpott Dam, constructed in 1952 by the U.S. Army Corps of Engineers and used for hydroelectric generation, flood control, and recreation, created the tailwater of the Smith River. The dam is a peaking hydropower production facility with an operational mode of power generation dictated by energy demands and water availability. The hydrology of the Smith River is dominated by regulated flows 12 months a year.

Discharge from the dam creates a flow regime that can fluctuate from 25 to 1400 cfs. The dam operated under different flow regimes during the sample years of 2000, 2001, and 2002 (Figure 2b.1). From January 2000 to May 2001, peaking flows were 1400 cfs, 7 days/wk, for 1-hour duration. From June 2001 to October 2001, peaking generation was at 700 cfs, 5 days/wk, for a duration of 2-10 hours. The peaking flows returned to 1400 cfs from November 2001 to February 2002, 5 days/wk, for a duration of 3-4 hours. Finally, generation releases were 700 cfs for a 1-hour duration from March 2002 to the end of the study in October 2002.

Because the discharge is hypolimnetic, the water releases influence the thermal regime of the Smith River on a regular basis. Temperatures directly below the dam are less than 10°C and increase with distance from the dam (Figure 2b.2). Maximum temperatures occur at the furthest downstream site in the study area. Daily maximum hourly temperature depression ranges from 0.16-8.39°C over all seasons with the greatest flux occurring in the summer. Variation in temperature depends on how greatly the ambient air temperature has warmed the water before the coldwater release.

A substrate profile below Philpott Dam indicates changes in types of substrate from upstream to downstream (Figure 2b.3). A high percent composition of bedrock, boulder, and cobble is present in the first 4 km below the dam (sites 1-3). Pebble and gravel, which are used by many fishes for spawning in the Smith River, are found in varying peaks throughout the longitudinal gradient. Sand and silt predominate in the lower reaches between 20 and 24 km below the dam (sites 11-12). Aquatic macrophytes increase as distance increases from the dam. Armoring of the channel exists near the dam due to loss of gravel recruitment, while further downstream the channel receives additions of fines from down-cutting of the channel and sediment input from tributaries. The average width of the channel is 30 m.

Despite such high environmental variability, five families of non-salmonid fish are present in the Smith River with fish boasting a wide range of life history strategies (Table 2b.1). The known non-salmonid fish fauna of the SR includes 34 species and is dominated by *Etheostoma flabellare*, *Nocomis leptocephalus*, *Notropis hudsonius*, and *Catostomus commersoni*. The most dominant fish group is the cyprinids, followed by the catostomids and centrarchids. Fish that are fairly common are *Clinostomus funduloides* and *Luxilus cerasinus*, while fish that are rare are *Exoglossum maxillingua* and *Percina rex*. *Percina rex* has an extant population in the Smith River and are a federally endangered species (Appendix B). A self-sustaining population of brown trout and stocked rainbow trout are present, creating a highly regarded trout fishery.

The Virginia Department of Game and Inland Fisheries designates a three-mile section of the Smith River to be a special trout regulation area with a 16-inch minimum, 2 fish-per-day limit. In 1995, anglers spent over 36,000 hours trout fishing in the Smith River, generating a total economic value of \$440,000/yr (Hartwig, 1998). During the 1970s, the Smith River became well-known for the catch of a state record brown trout. Today, there are smaller trout and fewer trophy catches. Biologists with the Virginia Department of Game and Inland Fisheries and the public would like to see more trophy catches due to the high economic and recreational value of the resource. Biologists also understand the ecological value of the entire fish community and seek to understand assemblage characteristics in relation to distance from the dam.

Procedures

Data Collection

Fish were sampled at 12 locations below Philpott Dam to 24 km downstream during the spring, summer, and fall (Figure 2b.4). The 12 locations were established as permanent sampling sites because of suitable access. Paired multiple anode, pulsed DC barge electrofishers were used to capture fish. Within the 12 sites, single-pass depletion sampling was performed in the spring and fall while three-pass depletion sampling was performed during the summer. All non-salmonid fish were identified to species, counted, and a subsample were measured to the nearest mm (total length), weighed to the nearest tenth of a gram, and released. Efforts were made to subsample each species across the range of sizes captured, measuring a representative sample of small to large individuals.

During June of 2000, 2001, and 2002, three-pass depletion electrofishing was conducted in 100-m sections enclosed by block nets at each location. From this data, population estimates and 95% confidence intervals were computed from maximum-likelihood estimates using the Microfish program (Van Deventer and Platts, 1989). 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 Analysis

I 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. A Kruskal-Wallis test was used to determine if a significant difference in relative abundance was present across the eight sampling periods (Zar, 1996). Likewise, I tested species richness across sampling periods using Kruskal-Wallis. A coefficient of variation (CV) was calculated for relative abundance and

species richness at each site across sampling periods and is reported as a percentage. To classify population variability for relative abundance and species richness, an arbitrary scheme proposed by Freeman et al. (1988) using CV values was followed: (1) $CV \leq 25\%$ = stable; (2) $26\% < CV \leq 50\%$ = moderately stable; (3) $51\% < CV \leq 75\%$ = moderately fluctuating; (4) $\geq 76\%$ = fluctuating.

Length data was used to show the age distribution over time and on a longitudinal basis. All species were classified as adult or juvenile based on information from Jenkins and Burkhead (1993). April, June, and October samples from 2001 and 2002 were used to show age distribution. Length data from 2000 was not used because it was not representative of more recent sampling periods. During June 2000, lengths were measured for only a few species. During October 2000, lengths were taken for fewer individuals of fish, because the distance sampled was set at 100 m, shorter than all other time periods.

I also used information from Jenkins and Burkhead (1993) to classify each of the 13 most common or numerically dominant 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, I averaged the number of fluvial specialists and habitat generalists by site within each season and plotted the longitudinal patterns with distance from the dam.

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 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).

Multiple linear regression was used to help discern the relationship between non-salmonid abundance and other biotic and abiotic variables in the Smith River. The primary utility of this analysis was inference about mechanisms that most influence relative abundance. Variables were carefully screened in order to avoid “data dredging” or “overfitting” commonly associated with multiple linear regression (Burnham and Anderson, 1998).

Relative abundance of all non-salmonid fish (number of fish per 100 m) 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 (Appendix C), temperature (Krause, 2002), and substrate composition were input into the model as the independent, predictor variables. Each independent variable had site specific values coinciding with the 12 fish sampling sites.

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. A second test for collinearity was performed by running a variance inflation factor (VIF) test in SAS for the full model (version 8 SAS). If the calculated VIF is 10, then the variable is most likely collinear with another variable. Variables which were determined to be collinear were removed from the model to prevent inclusion of redundant variables, or those sharing too much information. Finally, a stepwise regression procedure was used to obtain the final model (version 8 SAS).

To help summarize the relationship between the environmental variables selected by the regression model and the 13 most common species, canonical correspondence analysis (CCA) was performed using an Excel© macro developed by Lipkovich and Smith (2002). Relative abundance of the 13 species was used along with the corresponding environmental data. CCA has been used to relate species abundances to measured variation in the environment (Taylor, 1993; 2000). The technique selects the linear combination of environmental variables that maximizes the dispersion of the species scores (Jongman et al., 1987). By looking at the perpendicular intersection of a species with each environmental vector, I estimated the center of that species distribution along that environmental gradient (Taylor, 1993). Likewise, species and sites that were grouped together were noted.

Results

Overall Patterns

A total of 14,245 non-salmonid fish were caught in the Smith River tailwater, representing 5 families and 34 species. The longitudinal distribution of fishes exhibited much spatial variation and minimal temporal variation in terms of abundance and species richness. Longitudinal patterns of relative abundance (Figure 2b.5) and species richness occurred consistently over time across the 12 sites. Neither relative abundance nor number of species changed significantly between time periods (Kruskal Wallis, $p > 0.05$). There was a general trend of increasing abundance and numbers of species as distance increases from the dam. Marked increases or peaks in relative abundance and species richness consistently occurred in areas of the mainstem that are adjacent to four main tributaries, which I refer to as tributary junction sites (Figure 2b.6). The four main tributary junctions are found at 5.4, 13.4, 18.6, and 22.1 km from the dam.

June population estimates per 100 m ranged from a total of 2 individuals at site 1, 0.5 km downstream from the dam, to 1809 individuals at site 12, 24 km below the dam (Figure 2b.7). Numbers of individuals were generally higher across sites in the June 2002 sample compared to June 2000 and June 2001 with 8 out of 12 sites having more fish, on the order of 828 more individuals per 100 m at one site. The longitudinal pattern of population estimates was similar to relative abundance in that numbers of individuals increased with increasing distance from the dam. A comparison between the years showed that a distinctive peak in population estimates occurred at site 4, 6.2 km below the dam, for June 2002, and a distinctive drop in population estimates occurred at site 12, 24 km below the dam, for June 2000. Site 4 is adjacent to the largest tributary that flows into the mainstem, Town Creek.

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 2002. 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 (Table 2b.2).

Variability in species richness was moderately stable to stable for all sites across sampling periods with the exception of site 2, which highly fluctuated. Across sampling periods, I found relative abundance to be moderately fluctuating to fluctuating for sites 1-4 and 8-12 (Table 2b.3). However, sites 5,6, and 7 were moderately stable. CV values for relative abundance of the 13 most common species (Table 2b.4) were classified as moderately fluctuating to fluctuating for all species except *N. leptocephalus*, a moderately stable species.

Patterns in fish abundance were not driven by juvenile numbers during any sampling period. Variability in fish abundance was most attributable to spatial and temporal variation in adult fish numbers, primarily because adults comprised 75% to 90% of the population (Figure 2b.8). However, both age groups followed the same trend within each season such that no contrasting peaks occurred between adults and juveniles (Figure 2b.9). The lack of accentuated change in either age group indicates stability of age structure within each season. Though change in juvenile abundance in response to high flow regimes is a potential source of variation in community structure (Schlosser, 1985), this appears not to be a mechanism in the Smith River.

Of the 13 most common species, 6 were classified as fluvial specialists and 7 as habitat generalists (Table 2b.4). 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 with increasing distance from the dam (Figure 2b.10).

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 >2.00 (Table 2b.5). 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 of 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 >2.00 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. I 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 12. 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 2b.6). 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 non-salmonid fish during successive surveys and like seasons showed significant associations in site ranks across time (Table 2b.7). 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 2b.7). It is likely that this analyses was driven by the fact that 0 *N. hudsonius* were caught in June 2000, 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 2b.8). 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.

Regression Analyses

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/m², mean number of macroinvertebrates per 0.1 m, percent composition of sand and silt (< 2 mm), percent composition of aquatic macrophytes. I ran a correlation matrix for the eight variables (Table 2b.9) in the regression model to reveal the linear relationship between the variables as well as the strength of that relationship (Table 2b.10). The relationships marked by the strongest correlations with non-salmonid 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 non-salmonid 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:

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

Figure 2b.11 shows the changes in the response of the variables maximum hourly temperature flux and percent composition of sand and silt across the range of the observed values for those variables. By holding two of the variables constant in the above equation, the effect of one variable on non-salmonid abundance is shown, conditional on the other variables being fixed at their mean values. Using this method, predicted values of fish abundance are plotted with real numbers of fish abundance. Maximum hourly temperature flux influences fish abundance in that higher numbers of fish are found in areas with higher fluctuation in temperature. Percent composition of sand and silt positively influences fish abundance such that as sand and silt increases non-salmonid numbers also increase. The change in the response of fish abundance to tributary junction is not plotted with respect to the other two variables, due to the binary nature of tributary junction. Instead, refer to Figure 2b.6, which shows how tributary junctions are consistently marked with higher numbers of fish.

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 non-salmonid abundance, represented an inherent biological relationship with non-salmonid abundance, and could be easily measured in the field. Table 2b.11 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.

Canonical Correspondence Analysis

The CCA plot (Figure 2b.12) depicts dominant patterns of species in relation to tributary junction, maximum hourly temperature flux, and percent composition of sand and silt. The first axis (CC1) correlated most strongly with temperature flux, having a correlation coefficient of 0.82. The species with centers of distribution corresponding to high temperature flux were *Campostoma anomalum* (CES), *Luxilus cerasinus* (CRS), *Etheostoma flabellare* (FND), *Phoxinus oreas* (MRD), and *Clinostomus funduloides* (RSD). The species that had high correspondence with tributary junctions were *Nocomis leptocephalus* (BHC), *Campostoma anomalum* (CES), *Luxilus cerasinus* (CRS). High correlates of axis 2 (CC2) included sand and silt levels and tributary junction with coefficients of 0.68 and 0.64, respectively. The species corresponding to high levels of sand and silt were *Percina roanoka* (RND), *Catostomus commersoni* (WHS), and *Notropis hudsonius* (SPS). *Etheostoma podostemone* (RWD) was closely associated with sites 10, 11, and 12. *Catostomus commersoni* (WHS), *Moxostoma erythrurum* (GOR), and *Hypentelium nigricans* (NHS) grouped together near sites 10, 11, and 12. *Clinostomus funduloides* (RSD) and *Etheostoma flabellare* (FND) were found at sites 4, 5, and 6.

Discussion and Conclusions

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 a relevance of tributaries 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. Vannote et al. (1980) established the principle in the river continuum concept that tributaries have localized effects on the mainstem, which can be applied to the Smith River and its 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°C, comparable to the most downstream site at 23 km below the dam (Figure 2b.2). 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 the cooler temperatures found near the dam. The second most numerous species in the tailwater, *N. hudsonius*, showed a distinct distributional pattern over time suggesting thermal selectivity. Out of 5,526 *N. hudsonius*, only 12 individuals were caught in the first 12.6 km below the dam, and most of these individuals were caught at the Town Creek tributary junction. *Notropis* species are known to exhibit sharp range boundaries related to fixed thermal limits that regulate their distributional patterns (Matthews, 1987).

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 non-salmonid 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. The two temperature variables are directly related to dam operation. Consequently, flow management may be a viable tool to increase non-salmonid 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 changes in environmental variation. 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 2002. This variation was not observed between October 2000 and October 2002. 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; 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.

Section 2: Characterizing spawning microhabitat and testing transferability for *E. flabellare* and *N. leptocephalus* from an unregulated river to a regulated river.

Predictive Habitat Criteria as a Tool for Management

Identifying habitat characteristics selected by fish is an important component to fisheries management. Often, habitat characteristics are measured to develop criteria to predict habitat use, generally termed habitat suitability criteria. These criteria are accompanied by varying degrees of uncertainty, because fish do not select habitat solely on water depth, current velocity, and substrate type, which are the most commonly measured variables. Orth (1987) suggested that body size, risk of predation, presence and abundance of competitors, season, time of day, and thermal regime can alter microhabitat selection by fishes. For instance, shifts in microhabitat use might occur in the presence of competitors, making variation in microhabitat selection high between sites with different fish assemblages. Other investigators found that microhabitat use can be influenced by the energetics of foraging (Baker and Coon, 1997) or diel feeding habits (Kwak et al., 1992). If the chosen criteria do not reflect microhabitat selection, then predictions of how alteration of habitat will affect a species are likely to be inaccurate.

Quantification of all variables involved in selection of microhabitat would require costly, time-intensive research. Lacking all relevant data, researchers usually infer species requirements by observing habitat use (Freeman, 1999). Habitat suitability criteria such as developed for Instream Flow Incremental Methodology (IFIM) is one of several approaches to evaluate linkages between biota and instream habitat, especially under alternative flow regimes in regulated rivers (Freeman et al., 1997). These and other habitat models provide decision-makers with tools essential for fishery management. Predictive habitat criteria provide a good framework within which biologists can work until better models are available to evaluate and quantify relationships between aquatic fauna, flow, and habitat.

Why Transferability is Important: Regulated Rivers

Comprehensive strategies must be developed for improving biological function in flow-altered rivers. Since over half of the total streamflow in the world is regulated (Freeman, 1999), justification exists to understand habitat requirements for preservation of aquatic fauna below dams or diversions. Typically, site-specific habitat criteria are developed to use in instream flow analyses. However, developing criteria in streams with regulated flows or degraded habitat may not reflect true habitat requirements needed for survival and persistence of a target species (Freeman et al., 1997).

Transferability of habitat criteria would provide necessary information in streams where developing those criteria is not feasible and also would reduce the cost and time needed for research. A transferable fish-habitat model can be used to predict optimal habitat for the fish in a different system than the one in which the model was developed. Thomas and Bovee (1993) defined transferability as the ability of criteria to specify optimal habitat such that utilization of higher quality habitat occurs in greater proportion than lower quality habitat. Therefore, if criteria were correctly identified for a species, then predictions could be made about high quality habitat availability for that species in other stream environments, especially in regulated rivers or ones with degraded habitat. With confounding factors between stream environments such as varying competitive interactions and temperature regimes, there is reason to believe that a transferable model is not feasible. Thus, as Groshens and Orth (1993) stipulate, there is need for multiple transferability studies to define the degree of generality in different fish-habitat models.

Testing Transferability of Habitat Criteria with Functional Significance

Transferability of habitat criteria could be greatly beneficial for predicting functionally significant habitat requirements such as spawning microhabitat, especially in regulated rivers. Because animals preferentially occupy areas that best support survival, growth, and reproduction (Freeman et al., 1997), testing for transferability of criteria that have functional significance could more precisely reflect requirements for maintaining strong populations. Since the spawning requirements of most fish have evolved to be very specific, transference of criteria based on spawning microhabitat use might be more successful than criteria based on frequency-of-use data.

This study was designed to identify spawning microhabitat criteria for two fishes and to test the transferability of this criteria between an unregulated river (Roanoke River, VA) and a regulated river (Smith River, VA). *Etheostoma flabellare*, the fantail darter, and *Nocomis leptocephalus*, the bluehead chub, were chosen as target species because of their abundance in these rivers (Smith River abundance data, Appendix D). Transferability was defined as the ability of criteria to correctly specify optimal habitat or habitat utilized in greater proportion than nonoptimal habitat (Thomas and Bovee, 1993; Freeman et al., 1997).

Appropriate ranges of variables for the spawning microhabitat of *N. leptocephalus* and *E. flabellare* were developed by Smith (1999) from the North and South Forks of the Roanoke River system. Through multiple logistic regression models, Smith (1999) found the combination of variables which best-predicted spawning microhabitat use with at least three variables included in the model. Models were developed for reproductive guilds to be more generalized, and were validated. Models for egg-clusterers and mound-builders performed well with a high percentage of microhabitat use predicted. In this study, successful transference of these criteria was evaluated to provide evidence for the potential to develop broadly predictive criteria based on spawning microhabitat.

Procedures

Choice of Sample Reaches

Sections of river were identified as likely spawning areas from observations of target species made during electrofishing trips in June 2001, April 2001, and April 2002. Spawning adults of *E. flabellare* were noted if brightly-colored males with dorsal fin egg mimics and ripe females were present. The presence of individuals with fresh tuberculation and bright coloration signified likely areas for mound-building activity by *N. leptocephalus*.

Temperature data collected by Krause (2002) was reviewed to determine the upstream boundary of suitable spawning temperatures and the month spawning would most likely initiate. It was determined that mid-April would mark the beginning of the spawning season for *E. flabellare*, because of the occurrence of suitable spawning temperatures, 15°C and above. The spawning window for fantail darters was expected to be between April-June. *N. leptocephalus* spawning activity was predicted to begin in May and extend through July with the appropriate temperature range of 19 to 25°C (Jenkins and Burkhead, 1993).

From the temperature data and observations of spawning readiness, two sections of river were chosen to intensely survey for *E. flabellare*, coinciding with 12.3 and 12.6 km below the dam (Figure 2b.4). A site 6.2 km below the dam was chosen to survey, but not as intensely. Sites at 12.6, 14.3, 18.9-20.1, and 23 km below the dam were chosen to intensely survey for *N. leptocephalus*. Surveys were also completed in areas between these sites, the tributary junctions

of Reed and Jordan Creek (18.6 and 22.1 km below the dam), and sites 6.2-12.3 and 15.9 km below the dam, though not as intensely. In addition, approximately 2 km of the Town Creek tributary was surveyed upstream of its confluence with the mainstem of the Smith River. The surveys were completed in two main segments with the first being to find nests and measure occupied space and the second to relocate nests and measure unoccupied space, resulting in a paired-sampling design.

Survey Methods to Locate Spawning Microhabitat

Underwater observations were made to locate spawning microhabitat for *E. flabellare*. During periods of base flow (~ 50cfs), at least two individuals snorkeled 200 m from the downstream to the upstream end of a sample reach. Observers moved through the areas of greatest flow when in riffle and run habitat and midway between the middle of the river and the bank on both sides of pool habitat following the protocol of Leftwich et al. (1997). Each snorkeler selectively searched for large, disc-shaped rocks in the cobble size range, the preferred spawning habitat for *E. flabellare* (Smith, 1999). After a spawning site was located, the snorkeler dropped a flagged weight on the spawning rock and progressed upstream. A rock with attached eggs underneath was determined to be a spawning site if the male darter was present or if the eggs were identical to the eggs of nests where fish were present.

Wading surveys were performed to locate spawning microhabitat for *N. leptocephalus*. Two individuals moved midway between the middle of the river and the bank on both sides searching for mound structures. Each mound was flagged and labeled with a code. Mounds that appeared to have structural integrity (i.e. mound shape without rocks dispersed), and were clean of sediment and plant growth were considered to be active. All survey sites for *E. flabellare* and *N. leptocephalus* were repeatedly visited until no further sign of active spawning was apparent (i.e. no new nests found).

Measurement of Microhabitat Use

The microhabitat variables and measurement procedures used by Smith (1999) to predict spawning microhabitat for the target species were followed to minimize the likelihood that unsuccessful transference of criteria were based on discrepancies in methodology. To assess spawning microhabitat selection for the target species, a sampling frame or grid was used. A 60 by 60 cm grid was centered on the spawning site either on the nesting rock or on top of the mound. The grid was constructed of an aluminum bar frame and cord stretched horizontally and vertically, modified from the sampling frame design of Bunte and Abt (2001a). The cords were equally spaced every 10 cm around the frame creating 49 intersections at which measurements could be made.

At each *E. flabellare* nest, the grid was used to estimate percent areal coverage of silt. A visual estimate was made to the nearest 10% of how much of the grid area was covered with silt. At the spawning rock, I measured depth (cm), mean water column velocity (m/s), and demersal velocity (m/s). All velocity measurements were made using either a Marsh-McBirney flow meter or a Model 1220 Price Type "AA" current meter. The six-tenths method was used to estimate mean water column velocity (0.6 depth from the stream bottom), which produced reliable results within the depth range of the study. Demersal velocity was measured as close to the stream bottom as possible (usually within 2 cm). Embeddedness of the spawning rock was estimated to the nearest 10% as the percentage of the rock covered with sand or silt, and all 3 axes of the rock were measured using calipers. The number of healthy eggs and the number of

eggs infected with fungus were noted for each nest. Finally, survey coordinates were made for each nest using a tripod and leveler. From a benchmark, an angle was shot and the distance to the nest was determined with an optical range finder.

For *N. leptocephalus*, depth and mean water column velocity were measured at 9 locations on the grid (Figure 2b.13). Demersal velocity was measured at the center of the grid. The percent areal coverage of silt was estimated by counting the number of cells in the grid that were predominately covered with silt, out of 36 possible cells, and calculating a percent of coverage. At the 49 intersections of the grid, substrate particle diameter (intermediate axis) and embeddedness (percent of rock covered with sand or silt to the nearest 10%) were measured. Substrate particles were measured with a gravelometer or template. Bunte and Abt (2001b) suggests template use in determining particle size, because a template has higher accuracy than a ruler or caliper and templates reduce variability between operators. For each mound, the distance from the nearest shore, diameter of the mound (measured perpendicular to flow), and the type of cover (large woody debris, overhanging tree or bush, large rock) if present were noted.

Measurement of Microhabitat Availability

An attempt was made to sample microhabitat at 55 occupied and 200 unoccupied locations to reduce the possibility for Type I and Type II error when testing for transferability of habitat suitability criteria, as suggested by Thomas and Bovee (1993). To accomplish this, four unoccupied locations were measured to every spawning location with the goal of locating 55 spawning sites. The same microhabitat variables were measured for unoccupied locations as occupied locations to assess characteristics of spawning microhabitat availability.

To sample spawning microhabitat availability for *E. flabellare*, nests were relocated using the survey coordinates. From the nest location, I walked the line of a random angle shot with a compass. Angles were generated using a random number generator. I walked from 1 to 8 paces while searching for a disc-shaped, small or large cobble size rock. I placed the center of the grid on the chosen rock where a nest would seem likely to be present, but was not. If a new nest was located by using this method, I returned to the original nest and walked a different line using a new angle.

The same random angle protocol was used to measure spawning microhabitat availability for *N. leptocephalus*. However, at least one grid was placed in shoreline or midchannel habitat 2 to 15 paces from the mound. The number of paces was primarily determined by characteristics of depth and ability to place the grid. I was unable to effectively sample in depths greater than about 1 m and unable to place the grid on submerged woody debris, large boulders, or trash. Shoreline habitat was designated to be 5 m or less from the bank while midchannel habitat was greater than 5 m from the bank. At least 1 of the 4 unoccupied grid samples around each mound was placed in either shoreline or midchannel habitat in order to sample each type of habitat availability.

Data Analysis

I used the two-sample Kolmogorov-Smirnov test ($\alpha = 0.05$) to determine whether a microhabitat variable was uniformly distributed among available microhabitat (Sokal and Rohlf 1995). Those variables significantly different from the available distribution were considered to be the variables most selected by spawners. In addition to testing for spawning microhabitat selection, patterns of selection were evaluated with histograms showing the percent of

microhabitat used or available within specific classes of a variable. Each variable was divided into several classes following Smith (1999). Abbreviations and descriptions of variables are given in Table 2b.12.

Substrate was divided into classes using a modified Wentworth scale (Gordon et al., 1992) where 0.05 mm (corresponds to silt), 1 mm (sand), 2-16 mm (small gravel), 17-63 mm (large gravel), 64-128 mm (small cobble), 129-256 mm (large cobble), and greater than 256 mm (boulder). Class intervals were created for embeddedness and silt in 10% increments. The depth variable categories were set at 0-20, 21-31, 32-40, 41-51, and greater than 51 cm. The designated classes for mean water column velocity and demersal velocity were less than or equal to 0.1, 0.1-0.29, 0.3-0.49, 0.5-0.7, and greater than 0.7 m/s.

The measured variables were summarized in the following way: depth and water column velocity were averaged over the 9 measurements, the median particle diameter of the 49 grid measurements was estimated to be the median sediment particle diameter or D50, and substrate roughness or D90 was estimated by using the 90th percentile of the surface sediment particle size distribution. Hydraulic variables were calculated for both *E. flabellare* and *N. leptocephalus* data.

Froude number (Fr) was calculated by the formula: mean water column velocity / (g x depth)^{0.5} where g is the acceleration due to gravity (9.81 m/s²). Froude number can be used as an index to describe surface turbulence characteristics such that flow is subcritical if Fr is less than 1, critical if Fr is equal to 1, and supercritical if Fr is greater than 1 (Gordon et al., 1992). Relative depth (Rel) was calculated only for *N. leptocephalus* using the ratio of depth to substrate roughness. Relative depth can be used to characterize near bed flow conditions because the flow pattern is highly dependent on the height of the substrate relative to the depth of the water (Davis and Barmuta, 1989). The benthos experience turbulent flow when relative depth is high (Davis and Barmuta, 1989). Froude number was divided into classes of 0-0.1, 0.1-0.14, 0.15-0.22, 0.23-0.34, and greater than 0.34. The class ranges for relative depth were 0-1.4, 1.4-2.2, 2.3-4.1, 4.2-12.4, and greater than 12.4.

I used a one-sided chi-square goodness of fit test to evaluate the transferability of spawning microhabitat variables measured in the Roanoke River to those measured in the Smith River (Thomas and Bovee, 1993; Zar, 1996). For *N. leptocephalus*, separate chi-square tests were performed for depth, mean and demersal velocity, silt, embeddedness, substrate roughness (D90), median particle diameter (D50), Froude's number, and relative depth. For *E. flabellare*, tests were performed for depth, mean and demersal velocity, diameter of the spawning rock, silt, embeddedness, and Froude's number.

I classified the spawning microhabitat use and availability data as occupied, unoccupied, optimal or usable, after the study design of Thomas and Bovee (1993). Optimal conditions were designated as those that fell within the range of conditions occupied in the Roanoke River. Thus, occupied and unoccupied microhabitat was designated as usable if it fell outside the range of microhabitat occupied in the Roanoke River. The chi-square analysis tested the null hypothesis that optimal conditions were occupied in the same proportion as usable conditions (Thomas and Bovee, 1993). If the null hypothesis was rejected, it was determined that optimal conditions were occupied in greater proportion, suggesting successful transference of Roanoke River spawning microhabitat criteria to the Smith River.

The logistic regression models developed by Smith (1999) to predict spawning microhabitat for *E. flabellare* and *N. leptocephalus* in the Roanoke River were cross-validated using data collected in the Smith River. A predicted probability of presence was calculated for

each spawning site in the Smith River using the logistic regression models. The *E. flabellare* predictive model included the following parameters for the logistic regression function, where η is the linear predictor of the independent variables:

$$\eta = -3.457 + \text{diameter of spawning rock (0.044)} + \text{embeddedness (-0.465)} + \text{silt (0.011)}$$

The *N. leptcephalus* predictive model included the following parameters for the logistic regression function:

$$\eta = 14.87 + \text{diameter of the center particle (-0.042)} + \text{substrate roughness or D90 (0.018)} + \text{embeddedness (-2.404)} + \text{silt (-0.708)} + \text{demersal velocity (16.82)} + \text{Froude's number (-25.43)} + \text{Roughness Reynold's number (-0.118)} + \text{relative depth (-0.246)}$$

I was unable to calculate a Roughness Reynold's number, because I did not measure a velocity profile in the grid samples and therefore could not calculate the shear velocity of the grids. I substituted the mean for spawning habitat and available spawning habitat from Smith's data (13 and 88, respectively) into the predictive model for *N. leptcephalus*. The probabilities were estimated from the following equation adapted from Yu et al. (1995), Trexler and Travis (1993), and Knapp and Preisler (1999), where P_i is the probability of finding a spawning site at location i :

$$P_i = e^{\eta_i} / 1 + e^{\eta_i}$$

Model success for predicting spawning habitat in the Smith River was evaluated by a sensitivity and specificity analysis. I selected a probability cut-off of 0.50 such that model predictions greater than or equal to 0.5 indicated areas classified as used and predictions less than 0.5 indicated non-used areas. The sensitivity of the model was assessed by calculating the proportion of the use observations that were predicted correctly. The specificity of the model was assessed by calculating the proportion of the non-use areas predicted correctly. The models were considered to have high resolution for prediction if they both had high sensitivity and specificity.

Results

Microhabitat Use and Availability

A total of 86 *E. flabellare* nests and 292 habitat availability grid samples were measured in the mainstem of the Smith River, exceeding the sample size suggested by Thomas and Bovee (1993). Spawning activity was evident from mid-April through mid-May, after which few new nests were located. Substrate diameter or the measured b axis of the spawning rock was the only variable for which *E. flabellare* spawning microhabitat use was distinguishable from what was available ($p < 0.05$; Table 2b.13). Fish selected spawning rocks in the small and large cobble range (64-256 mm; Figure 2b.14). The microhabitat measurements for depth, silt, embeddedness, demersal velocity, and mean velocity were not distinguishable from what was available (Table 2b.13).

However, patterns of use were evident for each variable. The majority of *E. flabellare* spawners were found in depths ranging from 0 to 40 cm (Figure 2b.14). Areas with levels of silt greater than 20% were most used by spawners with the actual spawning rock most often 0 to 10% embedded (Figure 2b.15). However, these silt levels (>20%) were the dominant class, whereas the lowest silt class (0%) had the highest use-to-availability ratio indicating the value of silt-free sites for spawning. Microhabitat use matched the distribution of habitat availability well for not only depths and silt levels, but also water velocities. *E. flabellare* spawners most often

used demersal and mean water column velocities in the range of 0.1 to 0.29 m/s (Figure 2b.16). At base flow, all nest sites had a Froude number less than 1 indicating that spawning microhabitat was found in areas with subcritical flow or slow, tranquil flow (Gordon et al., 1992). From the sample size of 86 nests, egg counts varied from 10 to 300 with an average of 91 eggs per nest. Among the nests that were sampled, a small percentage exhibited high levels of fungal growth over the eggs suggesting minimal if any reproductive success at that nest. *E. flabellare* were observed frequently guarding their nests during the study.

A total of 44 *N. leptocephalus* mounds and 154 unoccupied sites were measured, under-representing the sample size suggested by Thomas and Bovee (1993). I observed evidence of active spawning in the mainstem from the beginning of June to the middle of July. The average diameter of each mound was 0.6 ± 0.1 m with the largest diameter measuring 0.95 m and the smallest 0.3 m. Mounds were found in shoreline habitat (5 m or less from shore) with an average distance from shore of 3.5 ± 4.1 m. Three mounds were located in the midchannel of the river measuring 7, 10, and 20 m from the closest shoreline. Mounds were found almost without exception adjacent to some type of cover including small boulders, submerged trees, overhanging shrubs, or trash (i.e. television, rusty barrel, metal wheel case). Few direct observations were made of spawning fish over a mound.

The variables for which spawning microhabitat use was distinguishable from available habitat for *N. leptocephalus* were demersal velocity and mean water column velocity ($p < 0.05$; Table 2b.13). Velocity was highly selected for in the 0.1 to 0.29 m/s range (Figure 2b.17). Although the microhabitat measurements for depth, embeddedness, silt, substrate roughness (D90), and median particle diameter (D50) were not distinguishable from available habitat (Table 2b.13), distinct patterns of habitat use for these variables were apparent. The use-to-available ratio was highest for 0% silt and 0% embeddedness (Figure 2b.18). The depth class most selected for fell within the range of 21 to 31 cm (Figure 2b.19). The median particle diameter for all mounds was most commonly in the small gravel category (Figure 2b.20). And, the 90th percentile of the particle size distribution was in the large gravel category (Figure 2b.20). While large gravels were used by bluehead chubs to construct the mound, the high use-to-availability ratio for boulders suggests that boulders may play a role in creating stable microhabitats during flow pulses (see Job 3 for more elaboration on this idea). The calculated relative depth for microhabitat use was not discernable from what was available, but was predominantly less than 1.4 (Figure 2b.21). At base flow, all mounds had a Froude number less than 1 indicating that spawning microhabitat was found in areas with subcritical flow or slow, tranquil flow (Gordon et al., 1992).

Tests of transferability

Criteria for spawning microhabitat successfully transferred for some variables from the unregulated Roanoke River to the regulated Smith River (Table 2b.14). Spawning microhabitat criteria transferred well for *N. leptocephalus* with the null hypothesis being rejected for several variables. For this species, optimal spawning microhabitat was occupied in greater proportion than usable spawning microhabitat for mean water column velocity, demersal velocity, silt, substrate roughness (D90), and median particle diameter (D50). Spawning microhabitat criteria did not transfer well for *E. flabellare* with only one transferable variable. Optimal spawning microhabitat was occupied in greater proportion than usable spawning microhabitat for embeddedness of spawning rock.

Unlike the single variable tests of transference, the logistic regression model for *E. flabellare* from the Roanoke River yielded high levels of sensitivity and specificity when cross-validated with data from the Smith River (Table 2b.15). The *E. flabellare* model predicted 59% of the spawning locations and predicted 64% of the unoccupied locations. The high predicted probabilities of use are most attributable to the egg-clustering reproductive strategy of *E. flabellare* having a strong affinity for small and large cobble spawning rocks as well as low embeddedness and silt. Unlike the single variable tests of transference, the *N. leptocephalus* logistic regression model predicted spawning habitat poorly, classifying 34% of the actual spawning locations correctly and 98% of the unoccupied locations correctly.

Discussion

I tested the hypothesis that a reproductive site was more likely to be found in optimal habitat as defined by spawning microhabitat criteria developed by Smith (1999). Although it is likely that multiple factors interact to determine an optimal state for spawning microhabitat, it is also likely that individual variables strongly influence microhabitat use (Freeman et al., 1997). Thus, single variables were tested for transferability to determine which were most critical for successful spawning of target species.

Freeman et al. (1997) suggested that microhabitat criteria for riffle fishes would be most likely to transfer in comparison to criteria for species occupying a variety of pool and riffle habitats. In this study, spawning microhabitat criteria for substrate and velocity transferred well for *N. leptocephalus* though this species is categorized as a habitat generalist. Spawning microhabitat conditions for the habitat specialist, *E. flabellare*, did not transfer well. These results are not similar to those of Freeman et al. (1997) who found nearly all criteria for two darter species to transfer between an unregulated stream to a regulated river. But, Freeman et al. (1997) did not test transferal of combined criteria based on spawning microhabitat.

By testing Smith's logistic regression models from the Roanoke River to predict spawning microhabitat in the Smith River, a combination of variables were tested for transference or successful differentiation of optimal spawning habitat between rivers. The *E. flabellare* model had a high level of predictive success with over half of the actual spawning sites and available sites correctly classified. The *N. leptocephalus* model had a much lower level of predictive success. Thus, the model for the habitat specialist proved to be the best predictive model using a combination of three variables to predict spawning location.

Other studies have had mixed success at predicting spawning location of fish based on a combination of spawning microhabitat criteria. Shirvell (1989) had poor prediction capability of chinook salmon spawning areas when using PHABSIM, a part of the Instream Flow Incremental Methodology, with both generic and river-specific habitat suitability criteria. In contrast, Knapp and Preisler (1999) used nonparametric logistic regression to identify spawning microhabitat criteria for California golden trout and then developed a parametric model to predict spawning sites. Their results indicated that spawning locations were predictable based on a combination of microhabitat criteria.

In this study, successful criteria transference indicated those habitat conditions most critical for spawning success, while poor transference of variables could indicate several possibilities. First, poor transference would have occurred if environmental factors that dictate optimal spawning microhabitat in the Smith River were different from the Roanoke River. If key variables that explain spawning microhabitat use in the Smith River were missing from the Roanoke River models (such as habitat conditions under peak flow), then they would have had

low resolution. Second, optimal conditions were not available in the Smith River. Third, the microhabitat scale at which the fish habitat models were tested was not the scale at which spawning individuals were most influenced. Finally, changes in critical variables under a peaking regime could have prevented target species from spawning in optimal habitat.

In the Smith River, this final consideration is the most probable cause for poor transference. Because of the daily flux in critical variables, it is possible that changes in velocity or temperature dictate the habitat fish are able to use for successful reproduction. Based on the results from this study, it is likely that this is the case for *N. leptocephalus*. If so, this species may not be able to use optimal habitat as defined by the Roanoke River models, but instead is forced to spawn in areas that fall into a usable range of critical variables.

Conclusions

I concluded that the best predictive model for spawning habitat was the logistic regression model for *E. flabellare*, because the parameters of the model provided high resolution in predicting the spawning microhabitat for this species. Because the model was developed with information from several egg-clusterer species, it is probable that this model represents a broadly transferable model. The ability of the model to predict spawning microhabitat suggests that diameter of the spawning rock, embeddedness, and silt are crucial microhabitat features for the spawning success of *E. flabellare* in regulated and unregulated rivers.

Summary

According to Petts (1984), a hierarchical framework can be used to assess the effect of flow regulation below a dam. By this framework, fish populations represent the most progressive impact in a regulated river. In this study, analysis of fish community structure provides evidence that the full effects of flow regulation are present in the Smith River. Because the non-salmonid fish community shows a persistent response to variable flows and temperature, I concluded that the biotic properties of the tailwater are in a highly impacted state. The community response to flow and temperature is most evident by the following patterns: low abundance and diversity in the first few kilometers below the dam, isolated peaks in abundance and diversity at tributary junctions, increasing abundance, diversity, and faunal similarity with increasing distance from the dam.

The fish community shows consistent longitudinal patterns of abundance, diversity, and distribution of species such that these community attributes do not markedly differ over time. The consistency in these results is surprising because the community patterns are well-developed, even under the highly variable flow conditions of 2000, 2001, and 2002. However, in contrast to a persistent longitudinal pattern of abundance, “stability” or constancy in numbers of individuals over time was not evident during this study (Ross et al., 1985).

Though relative abundance was not statistically different between time periods, the numbers of fish caught in the October 2002 sampling period were markedly higher than all other sampling periods. The high numbers of fish caught during this sampling period coincided with a discharge schedule that had lower magnitude and duration releases compared to time periods in 2000 and 2002. Variable flow years offered an opportunity to assess fish community response to different flow regimes. In effect, a natural experiment which tested recruitment under different flows took place with 2000 representing a harsh recruitment year, 2001 a medium recruitment year, and 2002 a mild recruitment year. Decreased discharge appeared to allow recruitment to be stronger during the mild flow year of 2002 compared to the previous years of the study, resulting

in strong year classes for 10 out of the 13 most common species. Results indicate that lower magnitude discharge and duration of release, provides more suitable conditions for the non-salmonid species in the Smith River.

Reduction of productivity in tailwaters has been observed (Cushman, 1985). I predict that the productivity in the Smith River is depressed, though no quantitative comparison was made between fish numbers in the Smith River and a similar unregulated river. My prediction is based on spawning surveys for *Nocomis leptocephalus* in the Smith River and Town Creek which likely suggest a depressed population of this species in the mainstem. I surveyed 55 active mounds in Town Creek, over a 3-day period, within approximately 2 km of stream from the tributary junction. In comparison, I surveyed 44 active mounds in the mainstem, over a 2-month period, within approximately 6 km of river. I would expect to find more mound-building activity in a higher order stream such as the mainstem, because of greater habitat availability. Yet, other regulatory factors besides habitat availability (i.e. variable flows, temperature) must be limiting mound-building activity in the mainstem.

An important component of productivity in the Smith River is reproductive success of those species that are present. In effect, the future of any fishery is linked with how successful species are able to propagate. In this study, I found that successful spawning for *Etheostoma flabellare* depended on presence of suitable small and large cobble size rocks. For *N. leptocephalus*, preference for slower velocity habitat determined successful spawning areas. Since Smith (1999) found that *E. flabellare* spawning microhabitat was very stable in areas of high bed movement, the preference of spawning rocks might be the main determining factor for spawning success of the most dominant fish in the Smith River. Likewise, the ability of *N. leptocephalus* to utilize both shoreline habitat for mound-building and cover, as found in this study, (i.e. submerged wood, small boulders) allows this species to occupy slower velocity habitat in which to spawn. These microhabitats must be able to persist during the regular flow pulses.

Table 2b.1. Species list and age distribution categories for juvenile and adult fish based on Jenkins and Burkhead (1993). Scientific and common name given with accompanying acronym.

Species		Acronym	Juvenile Length (mm)	Adult Length (mm)
<u>Catostomidae</u>				
<i>Catostomus commersoni</i>	White sucker	WHS	0-179	≥180
<i>Hypentelium nigricans</i>	Northern hogsucker	NHS	0-124	≥125
<i>H. roanokense</i>	Roanoke hogsucker	RHS	0-59	≥60
<i>Moxostoma erythrurum</i>	Golden redhorse	GOR	0-199	≥200
<i>M. pappilosum</i>	V-lip redhorse	VLR	0-229	≥230
<i>Scartomyzon cervinus</i>	Black jumprock	BLJ	0-84	≥85
<u>Cyprinidae</u>				
<i>Nocomis leptcephalus</i>	Bluehead chub	BHC	0-69	≥70
<i>Semotilus atromaculatus</i>	Creek chub	CRC	0-79	≥80
<i>Campostoma anomalum</i>	Central stoneroller	CES	0-59	≥60
<i>Exoglossum maxillingua</i>	Cutlips minnow	CUT	0-64	≥65
<i>Clinostomus funduloides</i>	Rosyside dace	RSD	0-49	≥50
<i>Luxilus cerasinus</i>	Crescent shiner	CRS	0-49	≥50
<i>Phoxinus oreas</i>	Mountain redbelly dace	MRD	0-39	≥40
<i>Notropis hudsonius</i>	Spottail shiner	SPS	0-59	≥60
<i>N. procne</i>	Swallowtail shiner	SWS	0-39	≥40
<i>N. chiliticus</i>	Redlips shiner	RES	0-39	≥40
<i>Notemigonus chrysoleucas</i>	Golden shiner	GOS	0-69	≥70
<i>Luxilus albeolus</i>	White shiner	WS	0-64	≥65
<i>Cyprinella galactura</i>	Whitetail shiner	WTS	0-49	≥50
<i>Lythrurus ardens</i>	Rosefin shiner	ROS	0-44	≥45
<u>Centrarchidae</u>				
<i>Micropterus salmoides</i>	Largemouth bass	LMB	0-229	≥230
<i>M. dolomieu</i>	Smallmouth bass	SMB	0-199	≥200
<i>Lepomis auritus</i>	Redbreast sunfish	RBS	0-89	≥90
<i>L. cyanellus</i>	Green sunfish	GSF	0-69	≥70
<i>L. macrochirus</i>	Bluegill	BLG	0-79	≥80
<i>Ambloplites cavifrons</i>	Roanoke bass	ROB	0-199	≥200
<i>Pomoxis nigromaculatus</i>	Black crappie	BLC	0-99	≥100
<u>Percidae</u>				
<i>Etheostoma flabellare</i>	Fantail darter	FND	0-34	≥35
<i>E. vitreum</i>	Glassy darter	GLD	0-34	≥35
<i>E. podostemone</i>	Riverweed darter	RWD	0-29	≥30
<i>Percina roanoka</i>	Roanoke darter	RND	0-34	≥35
<i>P. rex</i>	Roanoke logperch	ROL	0-79	≥80
<u>Ictaluridae</u>				
<i>Ameirus nebulosus</i>	Brown bullhead	BRB	0-139	≥140
<i>Noturus insignis</i>	Margined madtom	MAM	0-89	≥90

Table 2b.2. Species richness for non-salmonid fish in the Smith River, VA over eight sampling periods, where CV is the coefficient of variation for species richness across time periods within each site. There is no significant difference in species richness among time periods (Kruskal Wallis, $p > 0.05$).

Site	Distance from dam (km)	June 2000	October 2000	April 2001	June 2001	October 2001	April 2002	June 2002	October 2002	CV
1	0.5	5	1	3	1	3	4	4	3	0.47
2	3.4	1	5	0	1	3	2	2	2	0.76
3	4.2	2	5	4	6	4	2	2	5	0.42
4	6.2	17	14	16	12	17	11	15	7	0.25
5	8.9	4	4	7	6	14	8	9	9	0.43
6	11.3	10	6	9	6	11	9	5	11	0.28
7	12.6	15	11	16	22	18	13	14	18	0.22
8	14.3	11	13	16	14	17	13	13	19	0.18
9	15.9	8	11	14	12	13	12	10	15	0.19
10	18.9	17	14	16	17	21	18	19	20	0.13
11	20.1	12	11	20	15	26	19	11	21	0.33
12	23	8	20	19	20	17	14	17	23	0.27
Total species		26	24	26	29	28	26	27	29	

Table 2b.3. Abundance data for non-salmonid fish in the Smith River, VA over eight sampling periods where N = total abundance, 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. There is no significant difference in abundance among time periods (Kruskal Wallis, $p > 0.05$).

Site	Distance from dam (km)	June 2000		October 2000		April 2001		June 2001		October 2001		April 2002		June 2002		October 2002		CV
		N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	
1	0.5	18	6	2	2	24	6	1	1	3	1	17	6	11	4	6	2	0.66
2	3.4	1	0	7	7	0	0	2	1	7	3	3	1	8	1	3	1	1.32
3	4.2	5	0	9	9	5	2	9	3	4	1	2	1	10	4	9	3	0.99
4	6.2	228	128	57	57	351	117	68	28	141	58	122	46	250	90	47	18	0.59
5	8.9	53	12	25	25	56	17	67	24	68	21	73	27	58	25	100	37	0.31
6	11.3	145	64	32	32	151	44	82	40	128	31	115	33	62	36	222	62	0.31
7	12.6	311	149	59	59	333	95	267	129	260	67	256	60	301	124	552	168	0.40
8	14.3	311	197	113	113	265	74	383	243	263	72	138	32	425	209	1777	413	0.73
9	15.9	119	64	92	92	153	51	162	69	382	187	216	60	350	181	1779	868	1.41
10	18.9	393	193	111	111	468	126	267	126	372	107	359	103	650	391	949	399	0.65
11	20.1	254	153	113	113	336	91	391	171	800	227	371	99	504	264	1902	761	0.94
12	23	202	146	277	277	709	192	684	387	418	115	583	182	591	263	1716	528	0.53

Table 2b.4. Abundance data summed over all sites for the 13 most common or numerically dominant non-salmonid fish in the Smith River, VA over eight sampling periods where FS = fluvial specialist, G = habitat generalist, N = total abundance, 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.

13 Most Common Species	FS or G	June 2000		October 2000		April 2001		June 2001		October 2001		April 2002		June 2002		October 2002		CV
		N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	N	RA	
<i>N.leptocephalus</i>	FS	406	266	224	224	534	151	468	230	536	155	219	63	278	278	923	309	0.39
<i>C. anomalum</i>	FS	34	32	21	21	7	2	5	2	42	12	12	3	2	2	108	35	1.02
<i>L. cerasinus</i>	G	38	21	17	17	17	5	22	12	103	32	9	3	5	5	78	25	0.71
<i>P. oreas</i>	G	12	9	9	9	18	5	42	9	43	13	9	2	13	13	151	40	0.91
<i>C. funduloides</i>	G	48	37	49	49	84	24	123	85	173	51	71	18	62	62	398	92	0.51
<i>N. hudsonius</i>	G	1	0	48	48	641	174	431	296	525	167	733	216	114	108	2787	1132	1.35
<i>C. commersoni</i>	G	48	209	228	228	533	150	176	76	162	49	452	131	228	234	1226	386	0.58
<i>H. nigricans</i>	G	9	8	27	27	35	10	11	7	40	11	29	7	18	18	88	33	0.65
<i>M. erythrurum</i>	G	20	19	21	21	74	22	3	0	19	9	52	14	27	27	4	1	0.70
<i>E. flabellare</i>	FS	818	332	117	117	753	228	651	347	646	268	463	132	649	649	1576	466	0.56
<i>E. podostemone</i>	FS	26	17	59	59	29	9	136	48	236	81	77	21	47	47	805	280	1.25
<i>P. roanoka</i>	FS	118	53	19	19	32	9	58	42	50	15	22	6	43	74	350	121	0.94
<i>N. insignis</i>	FS	52	21	1	1	18	5	25	7	9	3	7	2	8	8	16	6	0.95

Table 2b.5. 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 2b.6. 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 2b.7. 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 2b.8. 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 2b.9. Environmental input variables for multiple linear regression models.

Sites	Trib junction	Temp flux (°C)	Mean temp (°C)	Brown trout per 100 m	Invertebrates per 0.1 m	Chl a mg/m ²	% Sand / silt	% Aquatic veg
<u>June 2000</u>								
1	0	-1.86	7.81	54	55.67	25.99	0	10
2	0	-2.56	8.80	136	8	4.39	0	0
3	0	-4.65	9.81	152	14.17	7.63	0	0
4	1	-5.31	13.16	126	86.33	3.33	4.25	38.75
5	0	-5.2	13.53	73	42.67	6.04	2	0
6	0	-6.51	15.31	73	33.17	3.81	26	4.25
7	0	-8.39	14.76	11	34.33	1.77	25.5	16.5
8	1	-5.52	15.35	42	84.33	5.18	3.25	27.75
9	0	-7.17	15.89	14	23.17	8.93	21	18.75
10	1	-1.75	15.31	35	39.5	15.7	19	6.5
11	0	-4.27	16.74	5	52.83	5.88	33.5	18.75
12	1	-0.8	17.62	34	89.5	1.07	43	14.25
<u>October 2000</u>								
1	0	-0.29	9.20	34	55.67	25.99	0	10
2	0	-0.93	9.89	125	8	4.39	0	0
3	0	-0.62	10.52	129	14.17	7.63	0	0
4	1	-0.93	10.56	103	86.33	3.33	4.25	38.75
5	0	-1.89	10.37	113	42.67	6.04	2	0
6	0	-1.39	11.69	86	33.17	3.81	26	4.25
7	0	-2.87	10.75	43	34.33	1.77	25.5	16.5
8	1	-2.43	10.93	52	84.33	5.18	3.25	27.75
9	0	-1.62	11.06	67	23.17	8.93	21	18.75
10	1	-0.31	11.69	27	39.5	15.7	19	6.5
11	0	-0.85	11.40	15	52.83	5.88	33.5	18.75
12	1	-0.31	11.85	63	89.5	1.07	43	14.25
<u>April 2001</u>								
1	0	-0.16	6.98	58	55.67	25.99	0	10
2	0	-3.24	8.03	54	8	4.39	0	0
3	0	-4.31	8.94	100	14.17	7.63	0	0
4	1	-2.18	10.11	75	86.33	3.33	4.25	38.75
5	0	-2.00	10.84	85	42.67	6.04	2	0
6	0	-4.50	10.68	86	33.17	3.81	26	4.25
7	0	-1.16	11.05	43	34.33	1.77	25.5	16.5
8	1	-0.79	11.19	31	84.33	5.18	3.25	27.75
9	0	-1.59	11.32	19	23.17	8.93	21	18.75
10	1	-0.95	12.06	20	39.5	15.7	19	6.5
11	0	-1.28	11.65	12	52.83	5.88	33.5	18.75
12	1	-0.3	13.32	28	89.5	1.07	43	14.25

Table 2b.10. 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
Non-salmonid 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 2b.11. 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 non-salmonid abundance	N	df	R ²	MSE	Regressor	Confidence interval for each regressor (±)
<u>Final Stepwise Model</u>						
(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
<u>2-Regressor Models</u>						
(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

Table 2b.12. Variable categories and summary variables measured in sampling grid.

Variable Category	Variable
Depth	Depth (D)
Substrate	Median particle diameter (D50)
	Substrate roughness (D90)
	Diameter of spawning rock (Dcen)
Sedimentation	Silt
	Embeddedness (Emb)
	Embeddedness of spawning rock (Emcen)
	Demersal velocity (Vbot)
Velocity	Mean water column velocity (V)
	Froude number (Fr)
Hydraulic	Relative depth (Rel)

Table 2b.13. Mean \pm standard deviation for spawning microhabitat variables for available microhabitat and used microhabitat by *E. flabellare* and *N. leptocephalus* spawners. Significant differences (Kolmogorov-Smirnov two-sample test $p < 0.05$) denoted by *.

N is sample size.

Variables	Available	N	Used	N
<i>E. flabellare</i>				
D (cm)	27 \pm 16	86	27 \pm 16	292
B axis (mm)	116 \pm 29	87	* 145 \pm 45	292
Silt (%)	45 \pm 36	87	29 \pm 29	287
Emcen (%)	12 \pm 14	86	6 \pm 7	292
Vbot (m/s)	0.15 \pm 0.13	86	0.14 \pm 0.11	291
V (m/s)	0.23 \pm 0.17	86	0.21 \pm 0.14	292
Fr	0.0015 \pm 0.001	86	0.0014 \pm 0.001	292
<i>N. leptocephalus</i>				
D (cm)	35 \pm 14	44	39 \pm 16	154
D50 (mm)	13 \pm 9	43	14 \pm 3	153
D90 (mm)	67 \pm 233	38	266 \pm 518	153
Silt (%)	57 \pm 36	44	26 \pm 18	154
Emb (%)	28 \pm 28	44	14 \pm 13	154
Vbot (m/s)	0.26 \pm 0.17	43	* 0.17 \pm 0.06	153
V (m/s)	0.49 \pm 0.27	43	* 0.18 \pm 0.07	153
Fr	0.28 \pm 0.18	43	0.09 \pm 0.03	153
Rel	47 \pm 294	36	1.40 \pm 1.15	157

Table 2b.14. Transferability of microhabitat variables between the unregulated Roanoke River and the regulated Smith River based on a one-sided chi-square goodness of fit test. Variables successfully transferred if optimal microhabitat was occupied in greater proportion than usable microhabitat ($p < 0.05$). Successful transference denoted with (+).

Variables	Test statistic	Conclusion
<u><i>E. flabellare</i></u>		
D (cm)	3.26	
D50 (mm)	0.01	
Silt (%)	0.87	
Emb (%)	6.74	+
Vbot (m/s)	0.06	
V (m/s)	0.00	
Fr	0.00	
<u><i>N. leptcephalus</i></u>		
D (cm)	2.09	
D50 (mm)	9.83	+
D90 (mm)	8.45	+
Silt (%)	11.55	+
Emb (%)	0.04	
Vbot (m/s)	10.44	+
V (m/s)	43.09	+
Fr	0.00	
Rel	1.58	

Table 2b.15. Classification table for logistic regression model cross-validation to estimate probability of presence. Classifications based on 0.5 probability cut-off.

Data set	Number of habitat units			
	Observed with fish	<u>Predicted with fish</u>		Correct
		Present	Absent	
<i>E. flabellare</i>				
Present	86	51	104	59%
Absent	287	35	183	64%
Total	373	86	287	
<i>N. leptcephalus</i>				
Present	38	13	3	34%
Absent	157	25	154	98%
Total	195	38	157	

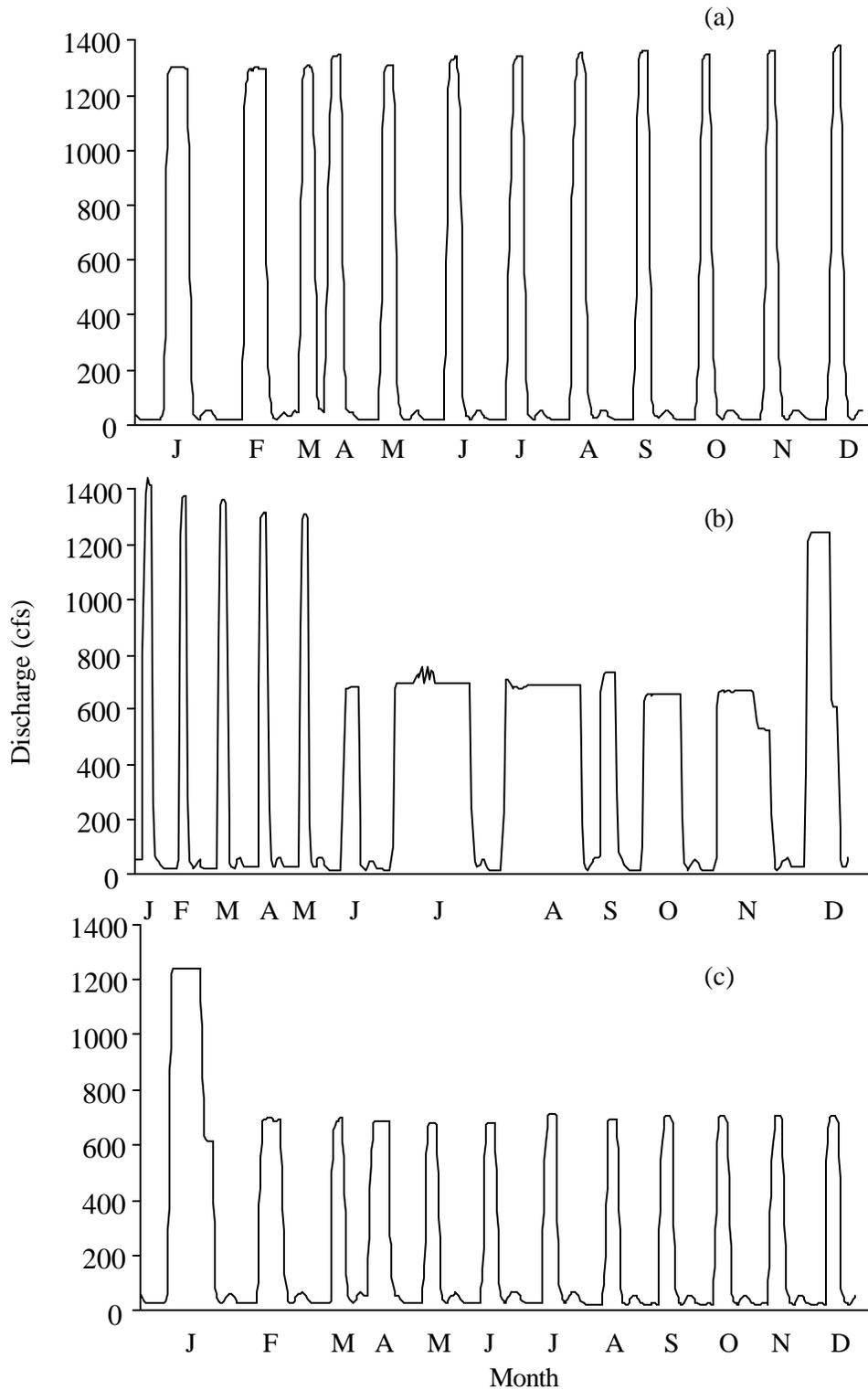


Figure 2b.1. Peaking discharge during one 24-hour time period by month for 2000 (a), 2001 (b), and 2002 (c) (USGS Philpott gage # 02072000). Wider peaks represent longer duration releases (i.e. 8 hour release in August, 2001).

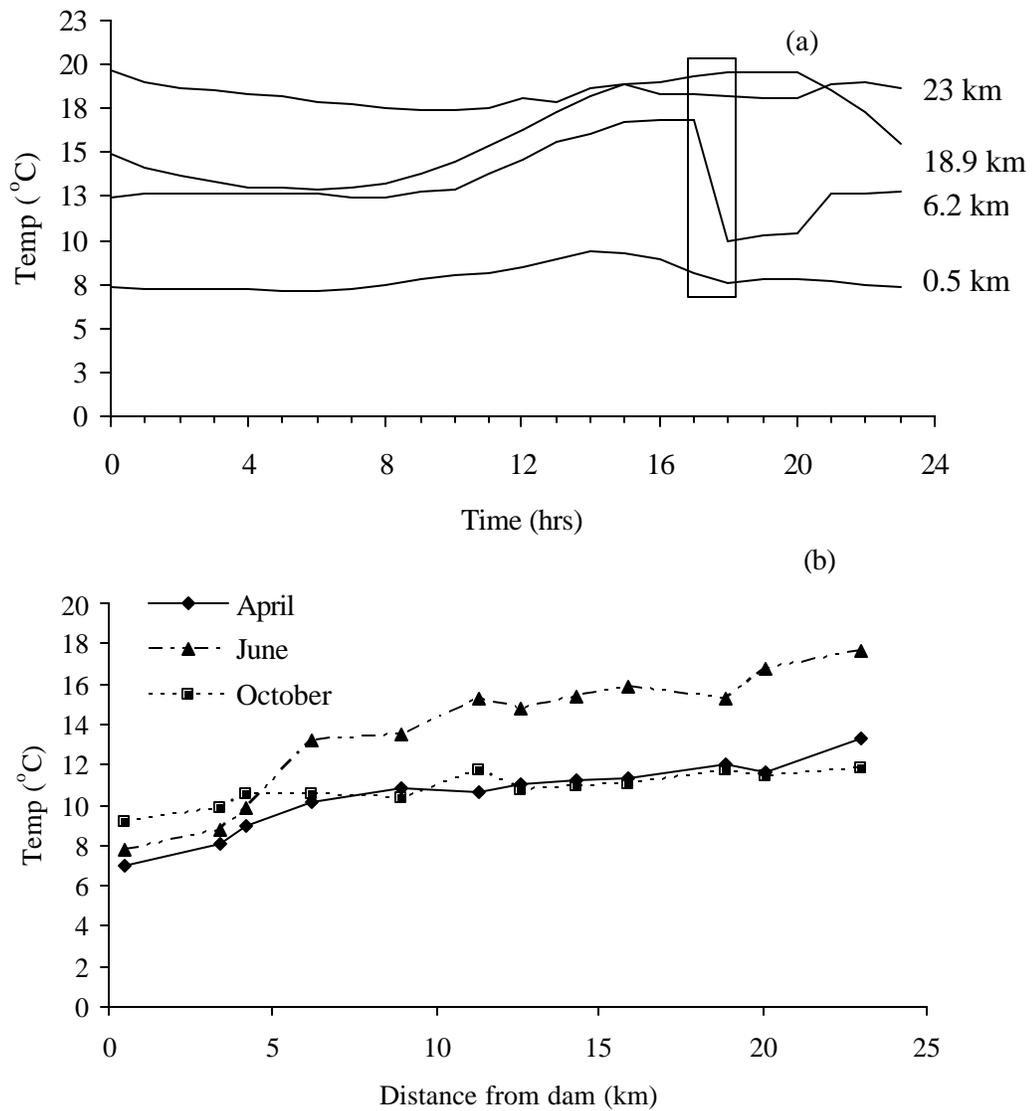


Figure 2b.2. Hourly temperature profile (a) for a typical day in June of 2000 at 4 sites, 0.5 km below the dam to 23 km below the dam. The box (a) represents the time of release from 1659 hrs to 1814 hrs (USGS Philpott gage # 02072000). Mean monthly temperature (b) at 12 sites below the dam for the months of April, June, and October in the year 2000 (Krause 2002).

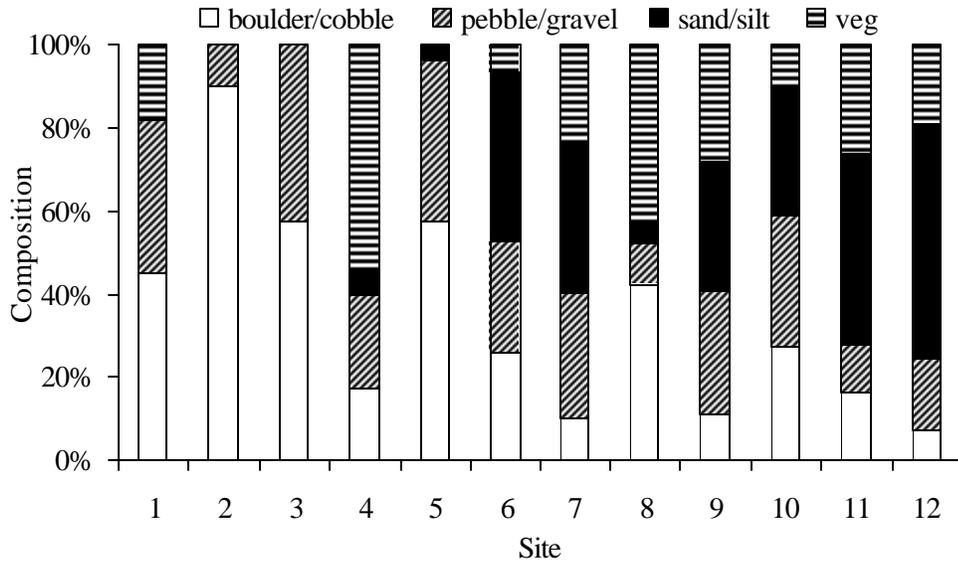


Figure 2b.3. Percent composition of substrate categories at each of the 12 sampling sites (Orth 2001).

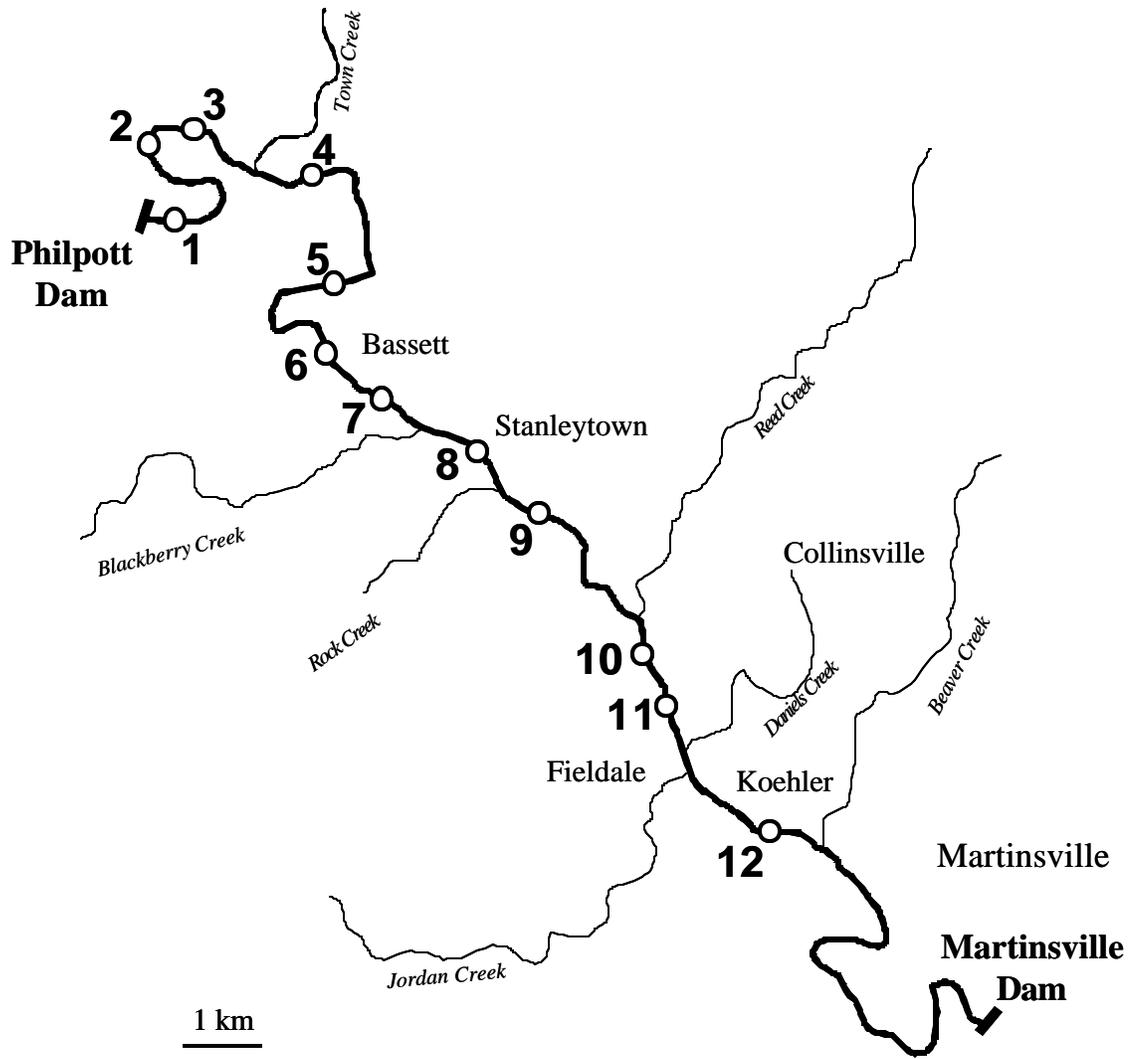


Figure 2b.4. Map of the Smith River tailwater between Philpott Dam and Martinsville Dam with sampling sites numbered upstream to downstream.

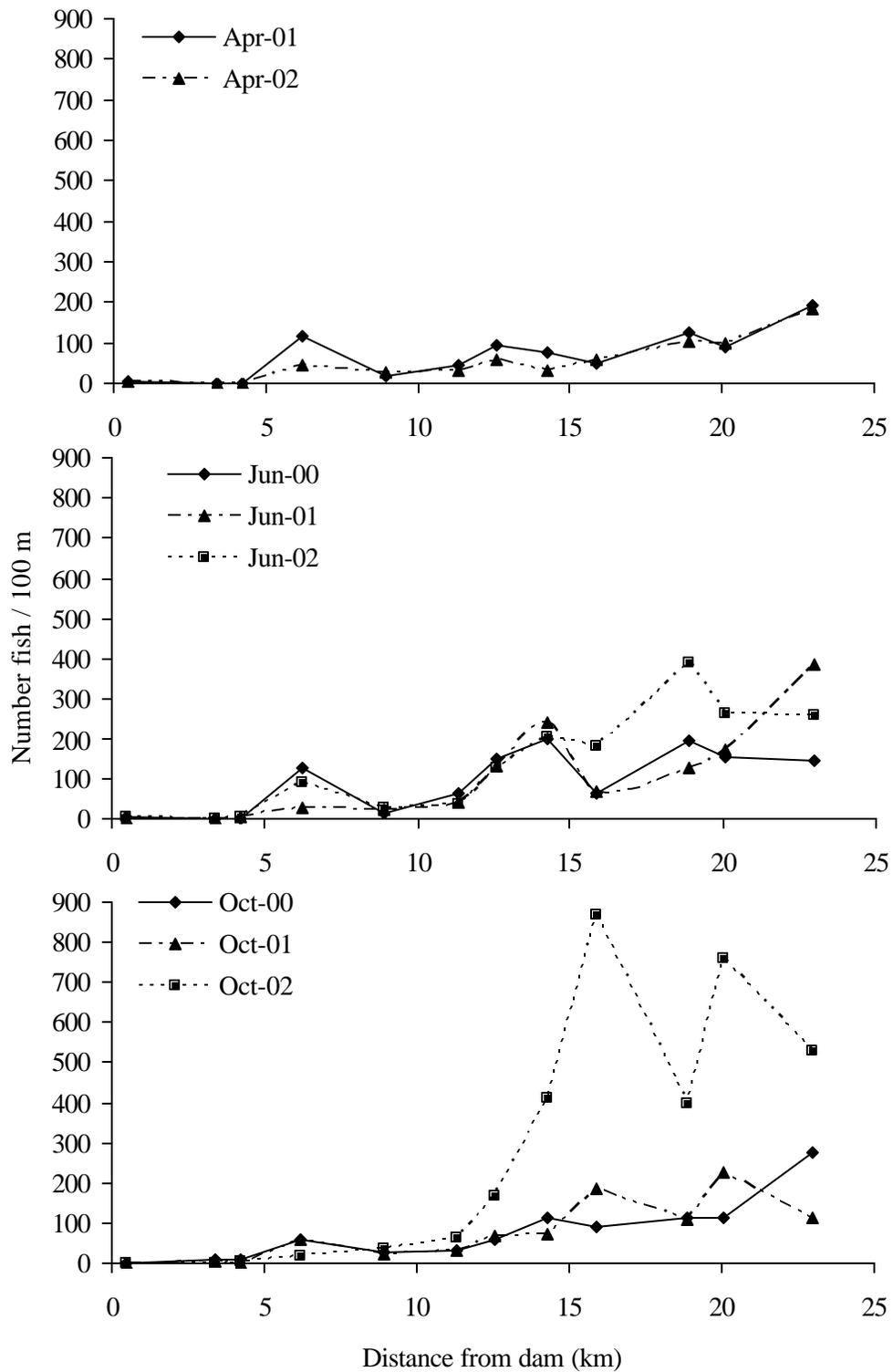


Figure 2b.5. Relative abundance for all non-salmonid fish as distance increases from the dam for all sampling periods.

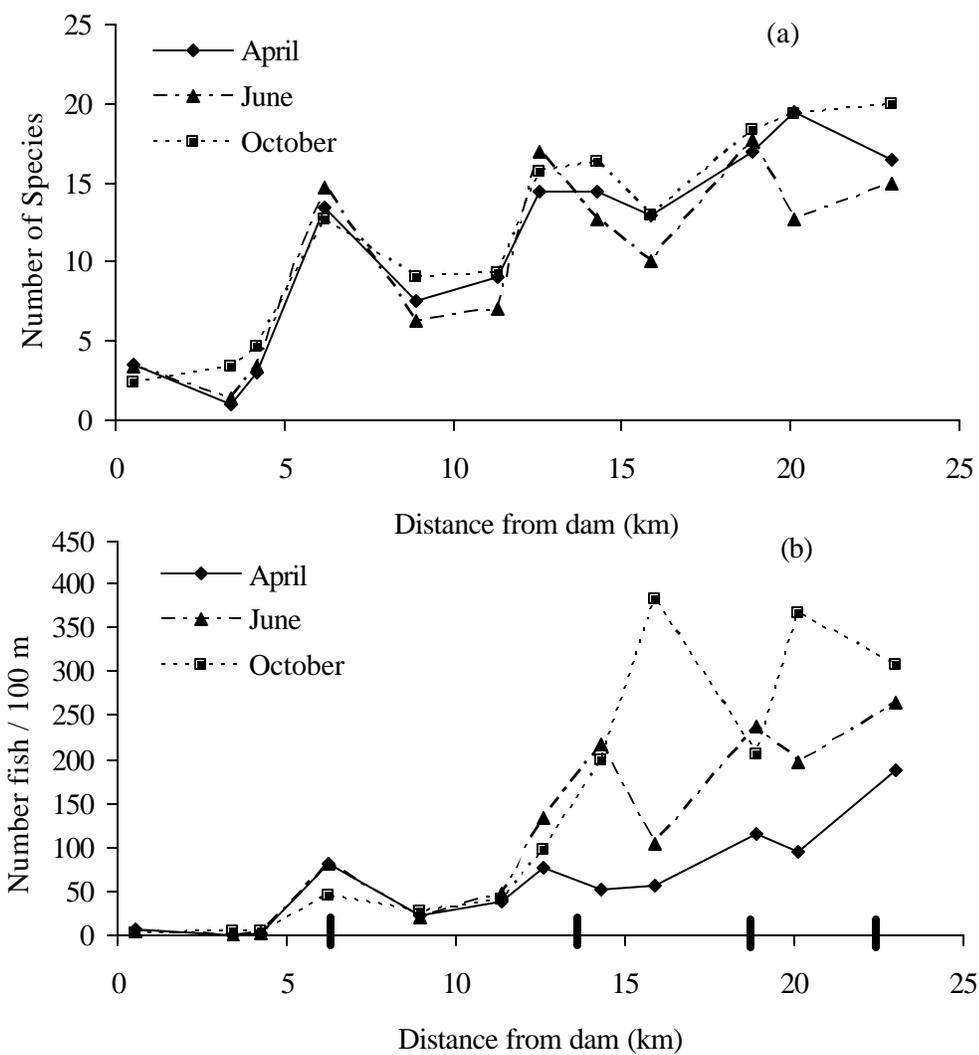


Figure 2b.6. Species richness (a) and relative abundance (b) for all species averaged within each season as distance increases from the dam for sampling periods in 2000, 2001, and 2002. Vertical bars (b) represent tributary junctions.

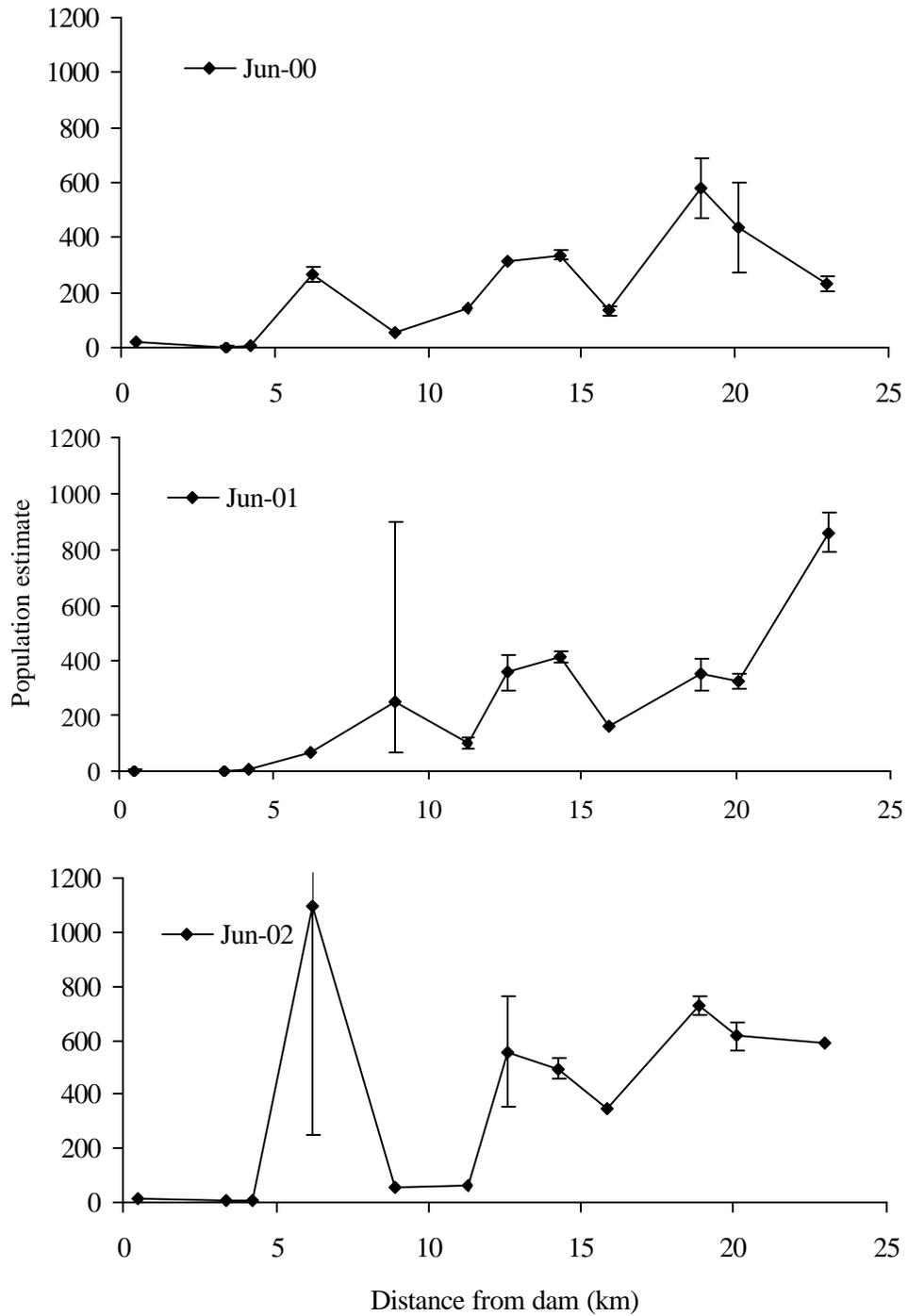


Figure 2b.7. Population estimates for fish per 100 m with 95% confidence intervals for June 2000, 2001, and 2002 using 3-pass depletion catch data (Microfish 3.0) for 12 sites. Total catch was used for sites with non-descending catch data (those without error bars).

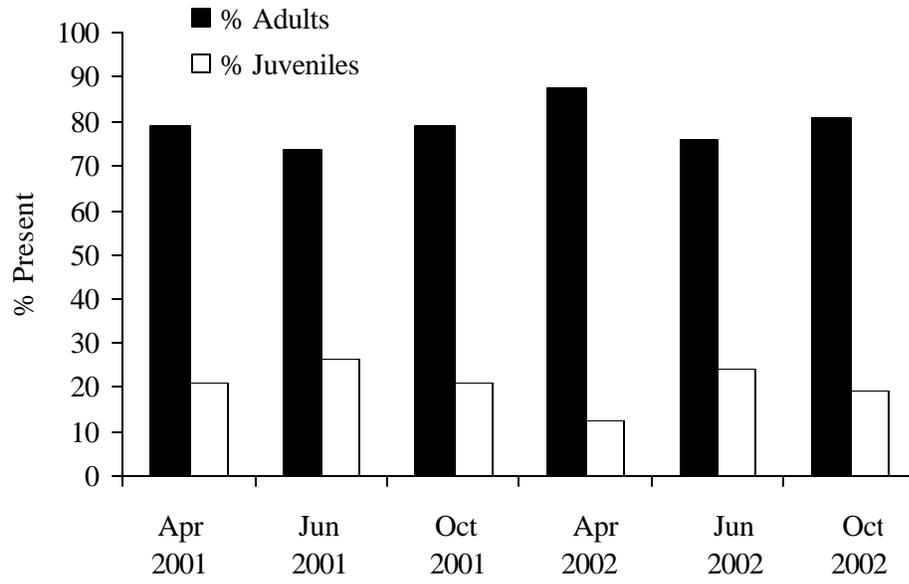


Figure 2b.8. Age distribution of the percentage of adults and juveniles in each sampling period for which length measurements were taken.

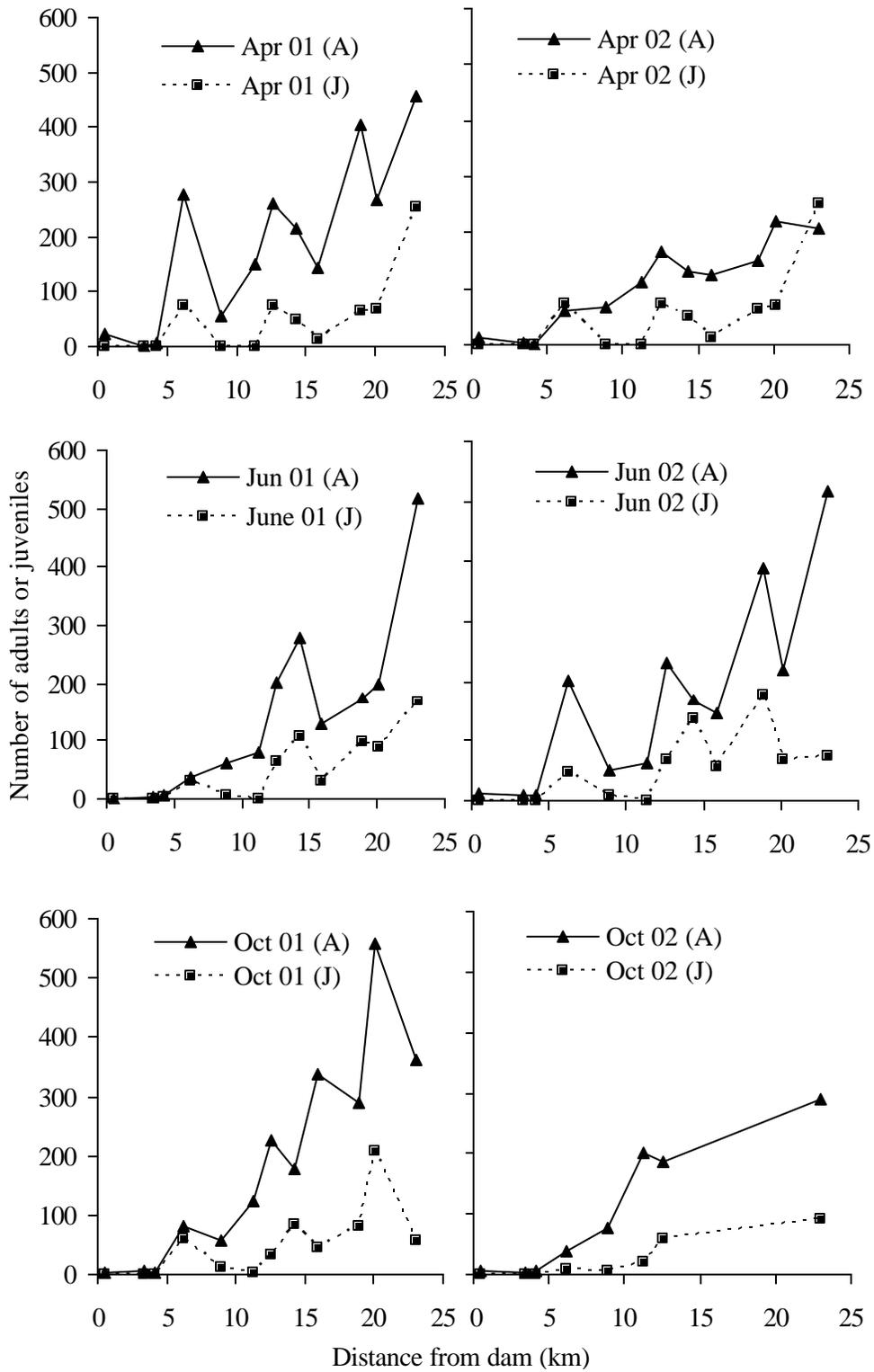


Figure 2b.9. Age distribution per 100 m as distance increases from the dam using a subsample of fish for which length measurements were taken in 3 seasons of 2001 and 2002. Length was not taken for sites 8-11 for October 2002. Note — A = Adults, J = Juveniles.

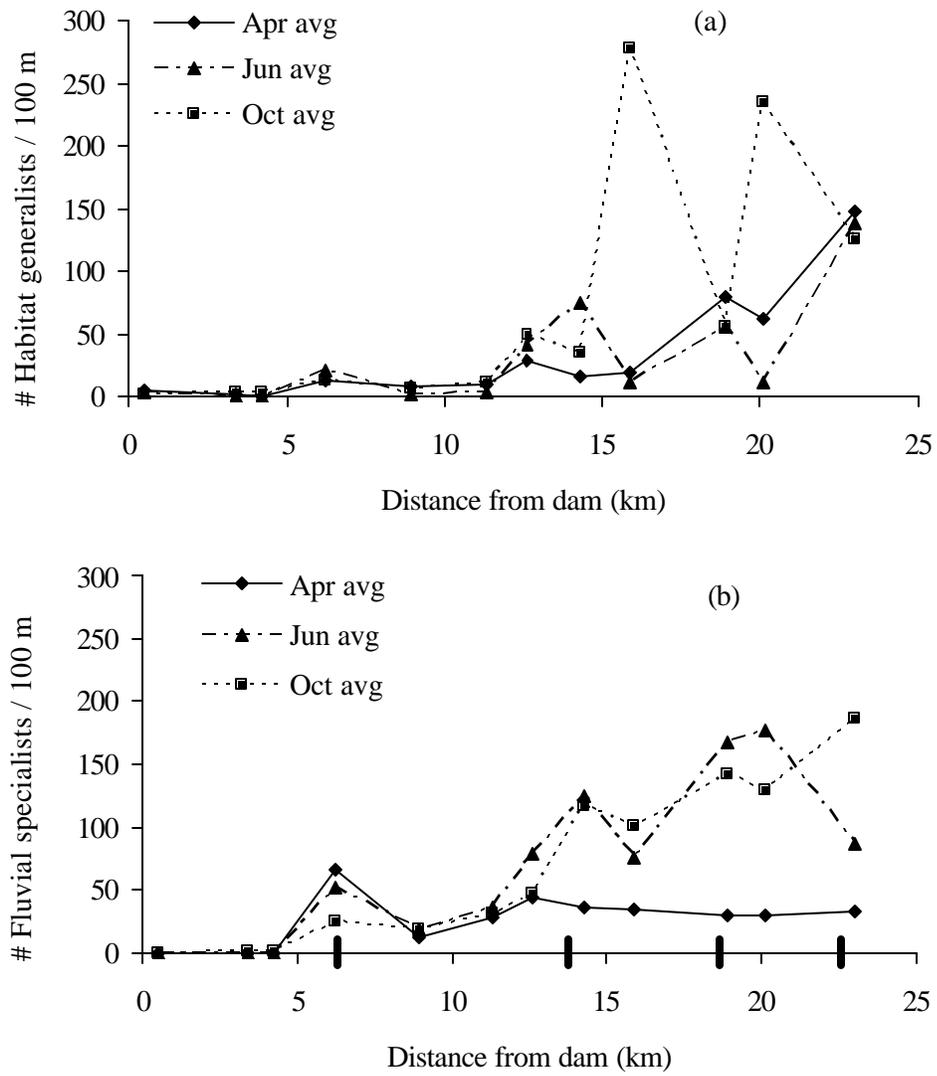


Figure 2b.10. Number of habitat generalists (a) and fluvial specialists (b) per 100 m for the 13 most common species averaged within each season across all sampling periods. Vertical bars (b) represent tributary junctions.

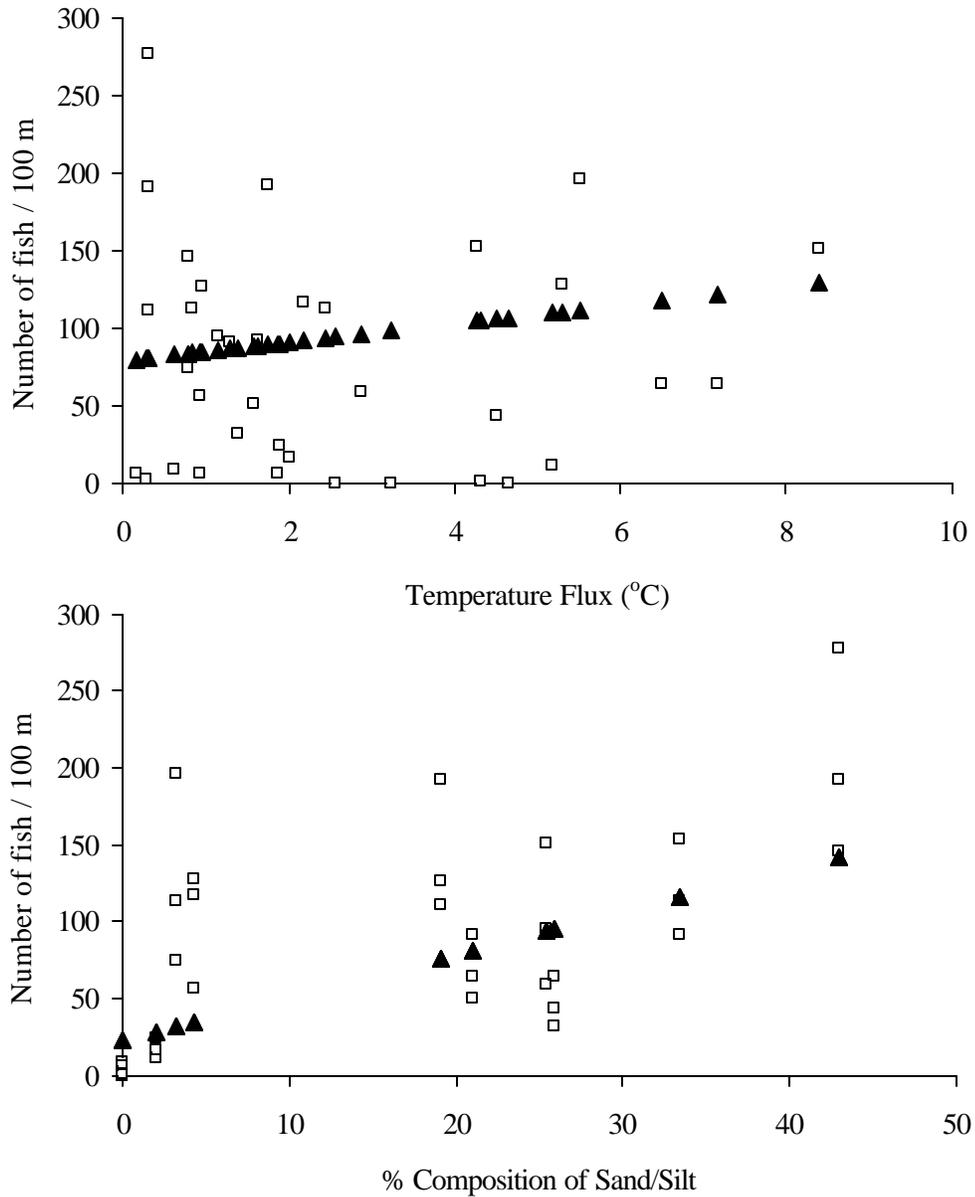


Figure 2b.11. Relationship of temperature flux (a) and percent composition of sand and silt (b) to non-salmonid abundance. Temperature flux is shown as the absolute value of a depression in degrees. Triangles represent the number of non-salmonid fish predicted by the multiple linear regression model [non-salmonid abundance = $-8.77 + \text{tributary junction } (94.30) + \text{temperature flux } (-5.97) + \% \text{ sand/silt } (2.75)$]. Open squares are actual numbers of fish per 100 m for June 2000, October 2000, and April 2001.

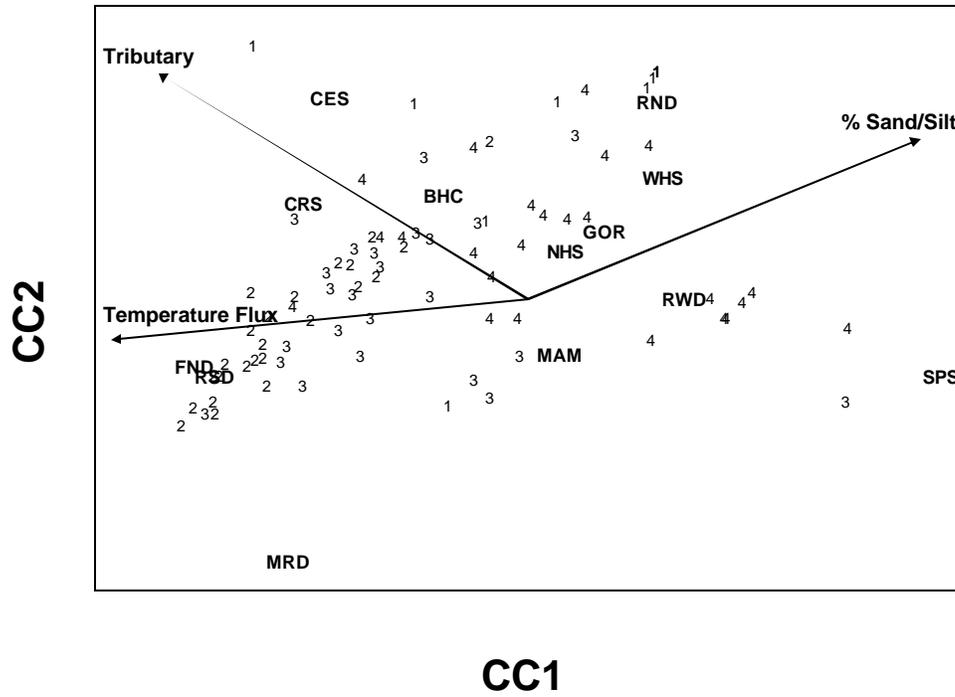


Figure 2b.12. Canonical correspondence analysis of species, reaches, and environmental variables. Numbers represent reaches such that 1 = sites 1-3, 2 = sites 4-6, 3 = sites 7-9, 4 = sites 10-12. Lines represent environmental variables.

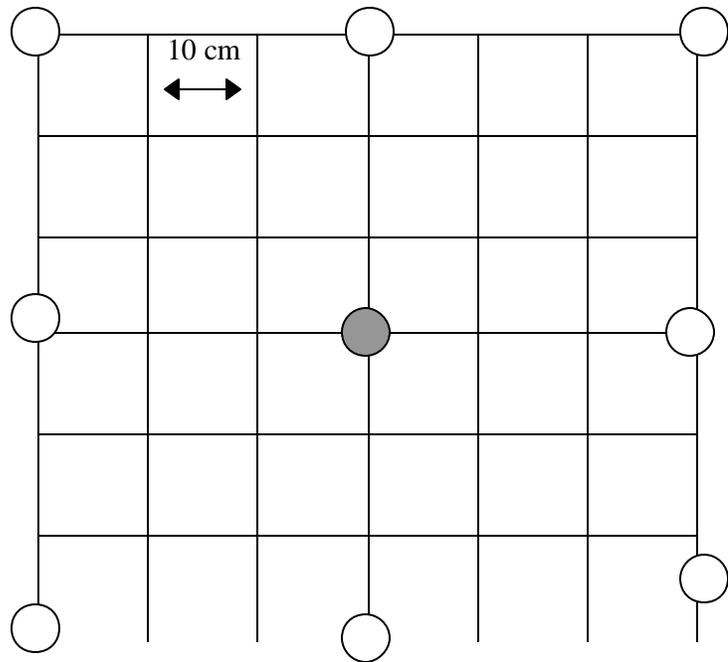


Figure 2b.13. Sampling grid used for microhabitat measurements. The grid is a 60x60 cm square with 10x10 cm cells. The circles represent areas where depth and water column velocities were measured. The shaded circle marks the center of the spawning site which served as the only location to measure demersal velocity. The 49 intersections of the grid marked the specific substrate particle that was measured.

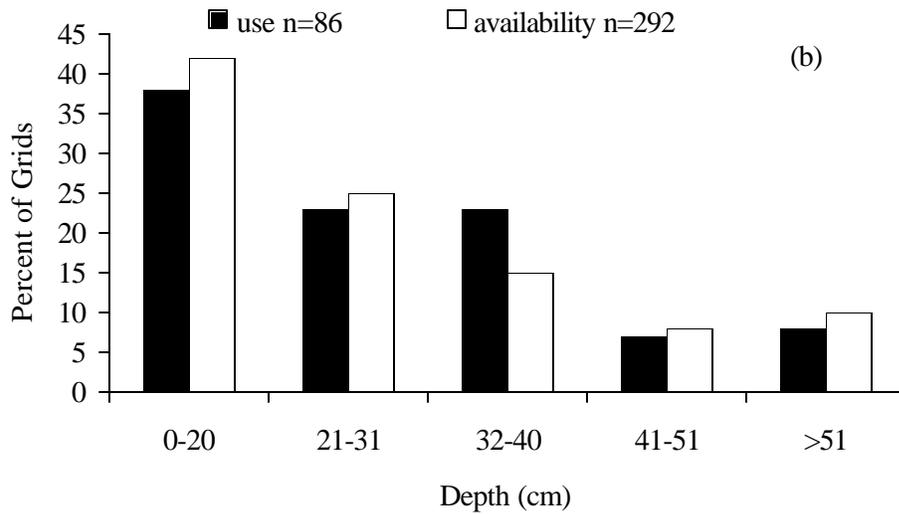
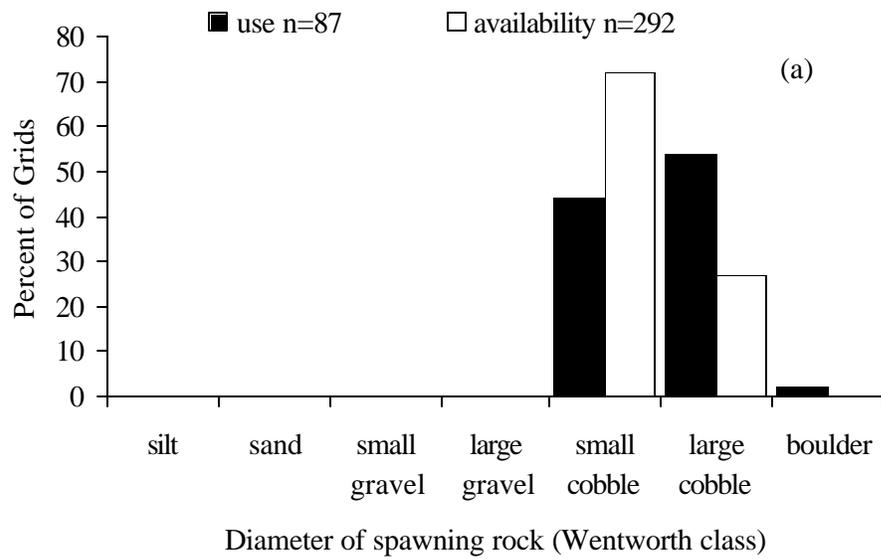


Figure 2b.14. Percent of substrate categories (a) used and available for *E. flabellare* spawning rocks and percent of depth classes (b) used and available for *E. flabellare* spawning microhabitat.

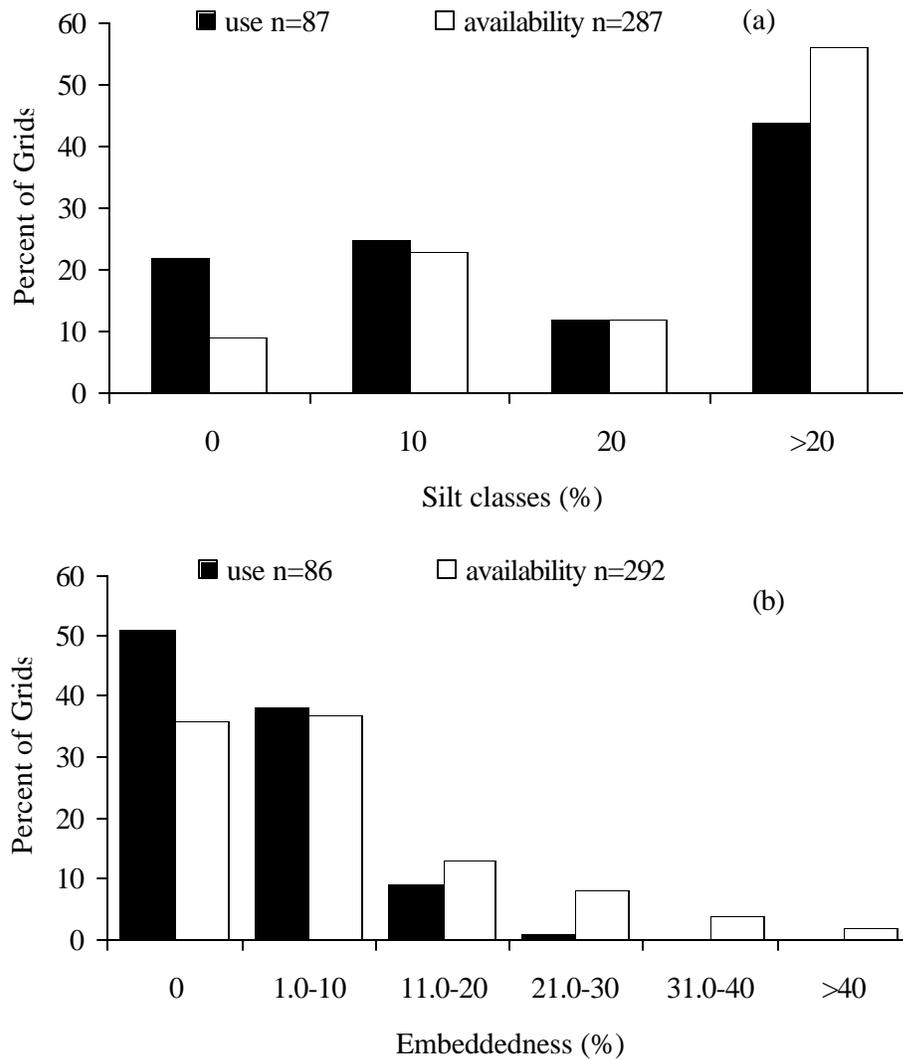


Figure 2b.15. Percent of silt classes (a) used and available for *E. flabellare* spawning microhabitat and percent of embeddedness classes (b) used and available for *E. flabellare* spawning rocks.

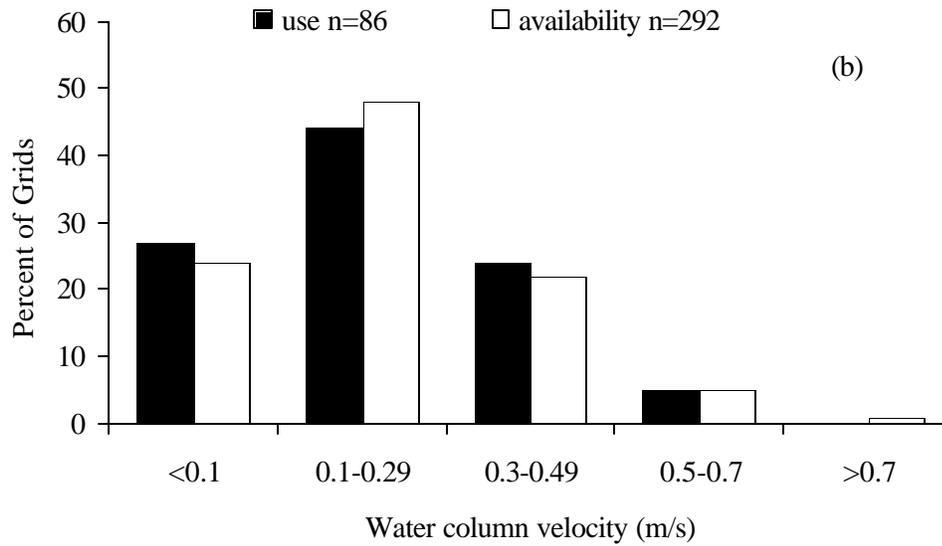
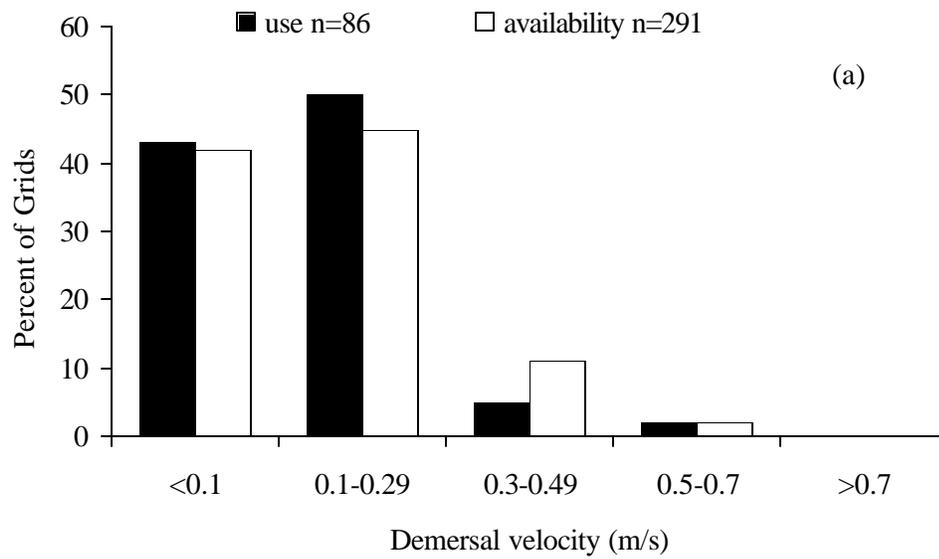


Figure 2b.16. Percent of demersal (a) and water column velocity (b) classes used and available for *E. flabellare* spawning microhabitat.

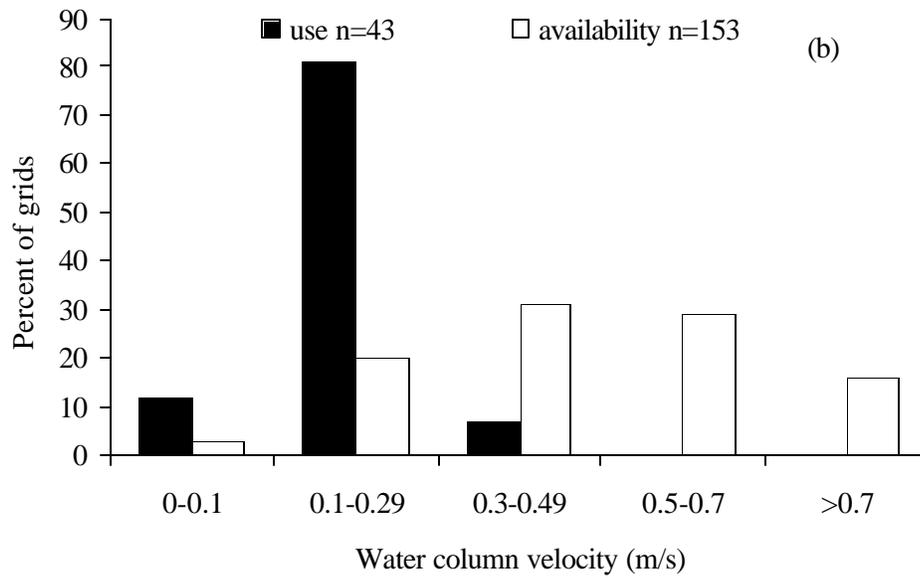
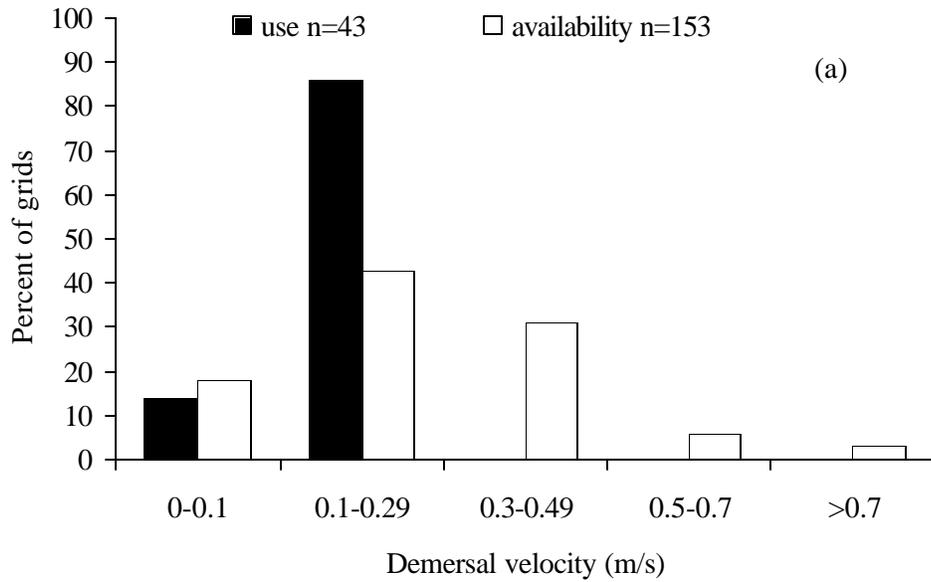


Figure 2b.17. Percent of demersal (a) and mean water column velocity (b) classes used and available for *N. leptcephalus* spawning microhabitat.

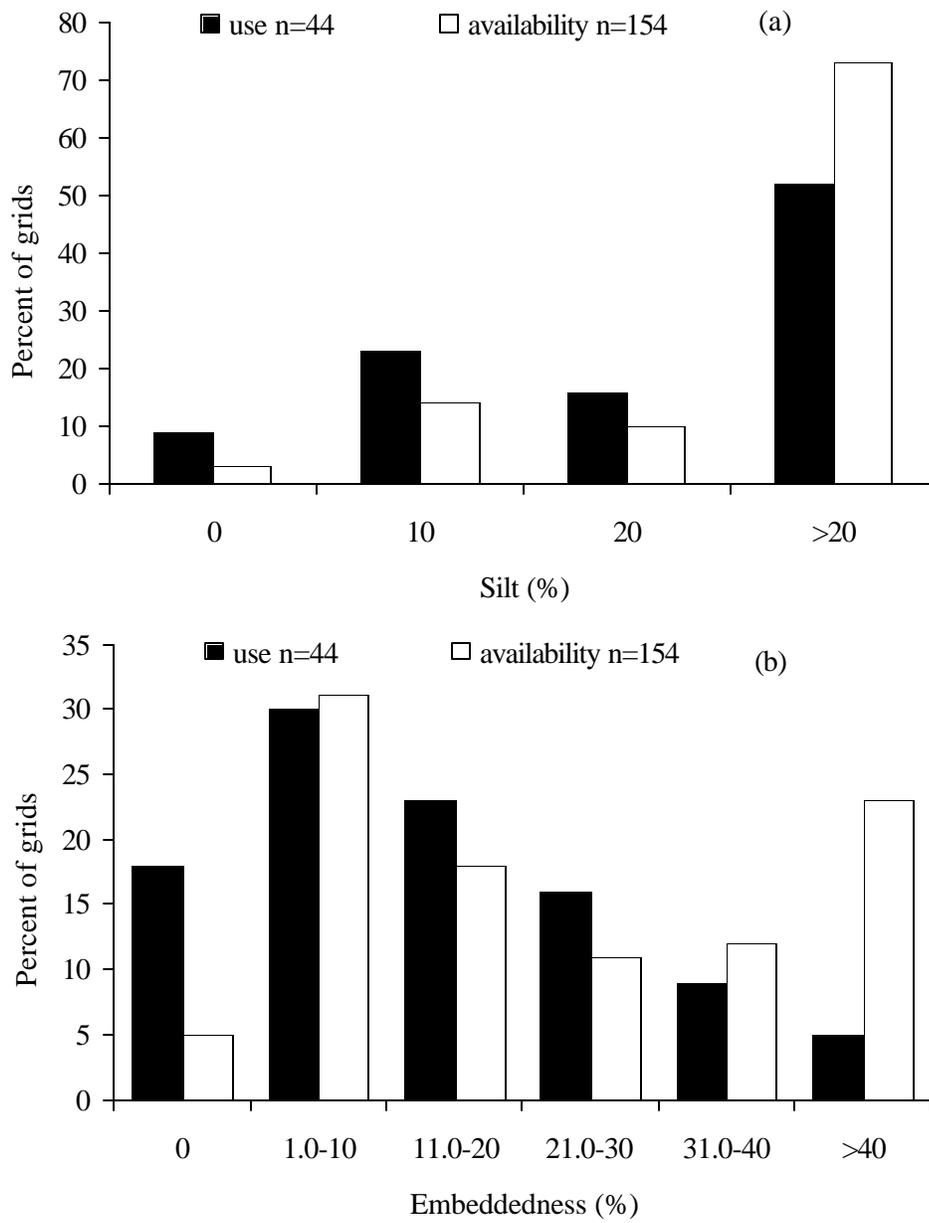


Figure 2b.18. Percent of silt (a) and embeddedness (b) classes used and available for *N. leptocephalus* spawning microhabitat.

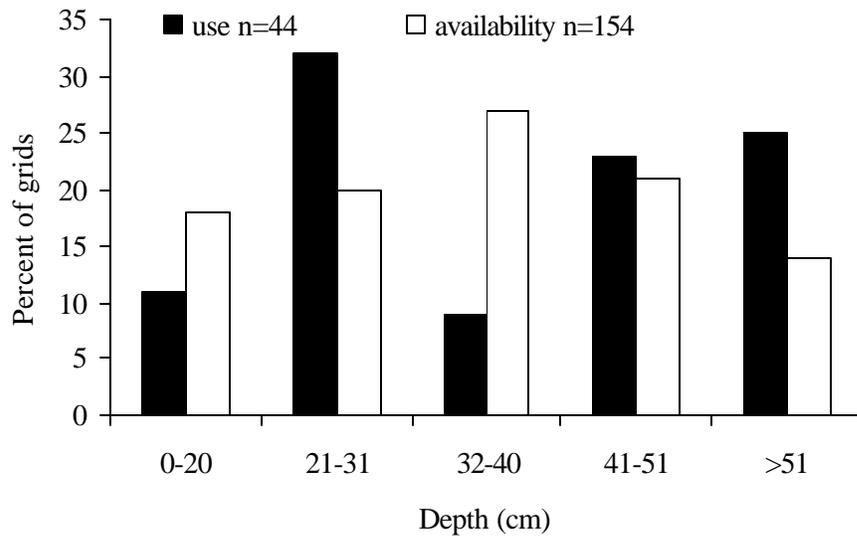


Figure 2b.19. Percent of depth classes used and available for *N. leptocephalus* spawning microhabitat.

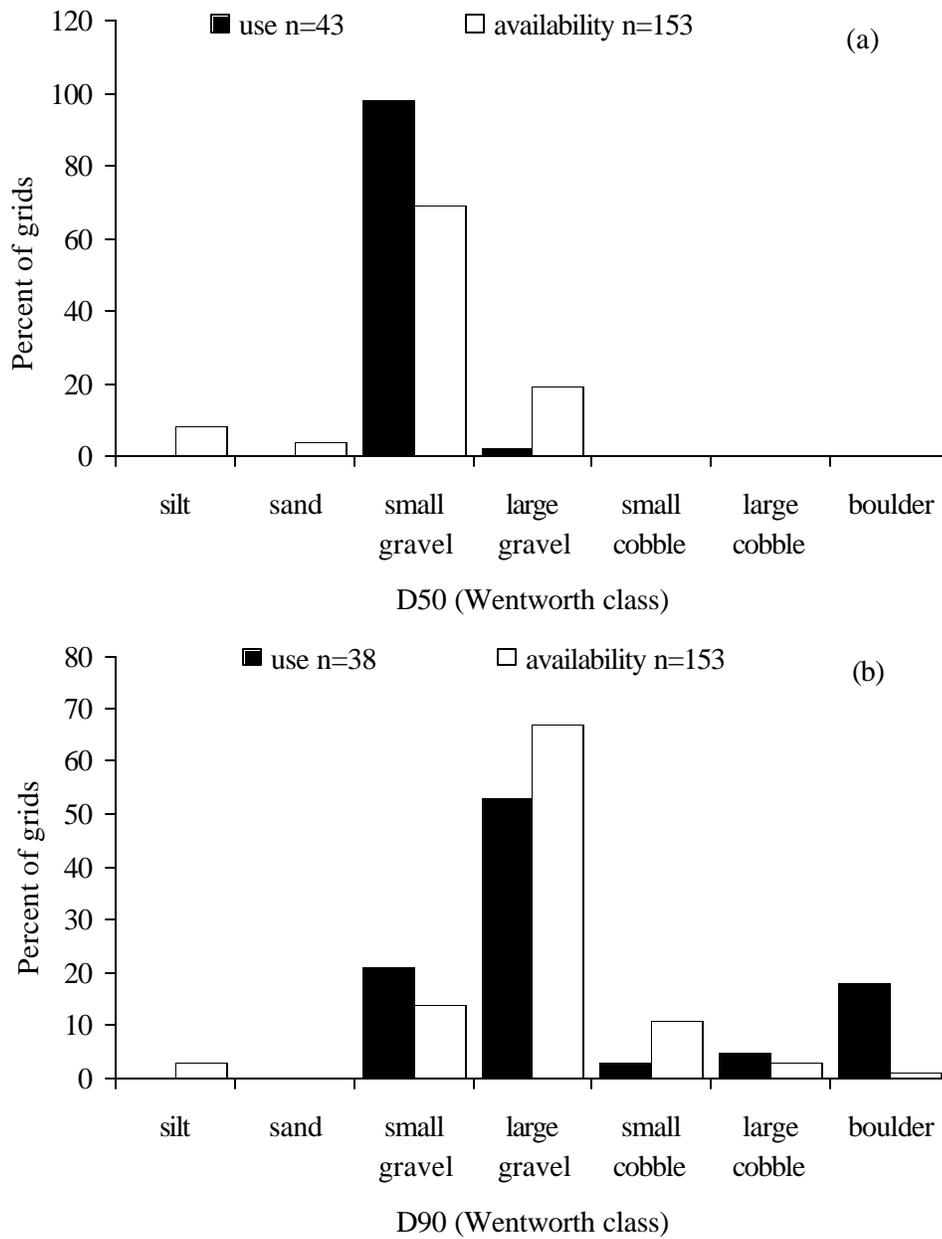


Figure 2b.20. Percent of median particle diameter (a) and substrate roughness (b) classes used and available for *N. leptocephalus* spawning microhabitat.

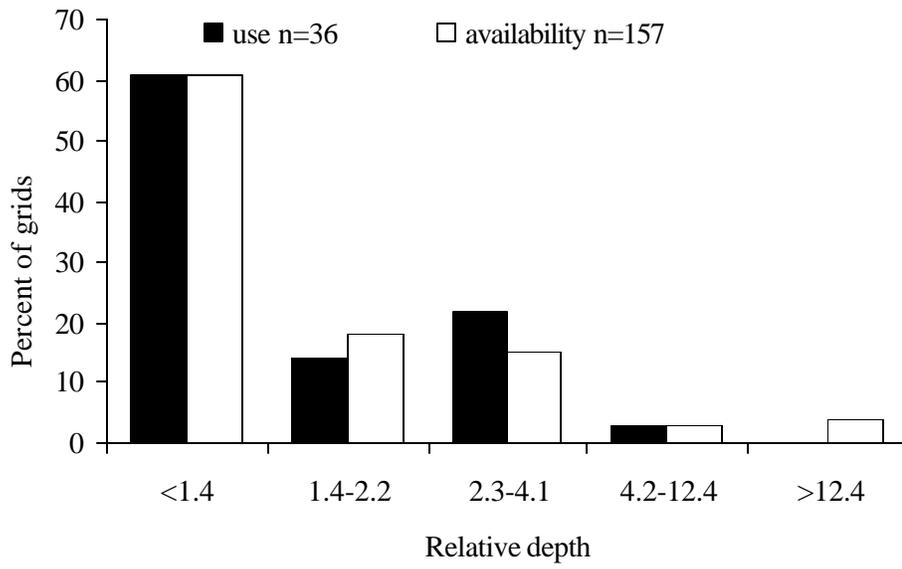


Figure 2b.21. Percent of relative depth classes used and available for *N. leptocephalus* spawning microhabitat.

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

Job Objective: 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 that would occur at sites in the tailwater.

The multiple flow regimes regulated in the Smith River often have detrimental effects on fish habitat, which includes areas of spawning and feeding. The initially suitable habitat can be changed to a harmful environment within minutes by releasing highly fluctuating flows from the upstream reservoir. Hence, measures need to be taken to minimize their impact on fish habitat, in terms of both fish diversity and biomass. One important remedial work may be fishrock installation. This habitat structure is often used in stream restoration projects in an effort to mimic natural flow features in creating hydraulic heterogeneity, velocity shelter, and turbulence wake for aquatic species.

To examine how the presence of the stream obstacles affects the local small scale flow patterns, the velocity patterns behind the boulders were explored using both 2-D (RMA2) and 3-D (CFX) hydraulic models. The biological significance of boulders on local fish habitat was also elaborated in light of habitat prediction from an existing bio energetic model for drift-feeding fish. Considering that fish and invertebrates often utilize heterogeneous hydraulics for feeding and other purposes, such information is crucial in studying the impact of flow complexity on fish habitat. In the reach scale, to investigate the effects of varying flows on brown trout, potential fish spawning habitat was predicted using both 1-D (PHABSIM) and 2-D (RMA2) hydraulic models. Finally, optimal reservoir release was recommended based on multiple flow simulations.

Procedures

The three hydraulic models mentioned above were applied in two selected study sites of the Smith River. Their channel topographies were first surveyed and discretized. The hydraulic boundary conditions were obtained from the nearest upstream USGS gage stations as well as the field measurements. For model calibration and validation purposes, water surface elevations and velocities at three different discharges were measured using a hand-held flow meter and an ADCP (Acoustic Doppler Current Profiler). Finally, assessment of suitable habitat was made by simulating various flows encountered in the Smith River. This section introduces the two study sites, presents the procedures of model simulations, and describes the methods for fish habitat assessment.

Study Sites Description

The study sites considered here are located in the tailwater below the Philpott Dam of the Smith River. The regulation generates a mean annual flow of 8.6 m³/s, with daily values fluctuating within a range from 1.78 m³/s to 42 m³/s. The average channel width is 30 m, with an average slope of 0.08%. Two study reaches (site 3 & 7 in Figure 3.1) have been selected. The first site (site 3 in Figure 3.1) is 160 meters long and is located 4.2 km below the Philpott Dam. This reach includes an island in the middle of the channel, several pool-riffle sequences, as well as numerous boulders at various locations, features that typically enhance fish habitat. Ten kilometers downstream of the first site is the second study site (site 7 in Figure 3.1) with a length of 150 meters. Unlike the upstream site, which is within the portion of the Smith River

having high abundance of spawning brown trout, the downstream site represents a poorer fish reproductive area.

Field Measurements

Physical (channel topography) and hydraulic data (flow depth and local velocity measurements) at the two study sites were collected along selected cross sections in both reaches. The transects were placed across observed optimal spawning areas as well as those areas having uniform and complex flow patterns. Eleven such transects were setup at the upstream site and ten at the downstream site. Each transect was oriented perpendicularly to the streamwise flow direction. Cell boundaries in the PHABSIM model were established halfway between each pair of adjacent measurement points so the center of each cell was exactly at a measurement point.

Measurements of Riverbed and Boulders - An electronic total station, Leica TC600, was used to georeference (in the form of XYZ site-specific coordinates) the geometry of both banks, the topography of the riverbed (approximately 2 sampling points per square meter), and the boulders in the two study sites. More than 2000 spot elevations were collected over each study site. In addition, the boulder near the fish habitat and redd locations were measured more intensively to capture their complex topographical features. In an effort to accurately represent the geometries for the boulders, in terms of their sizes and shapes, nine measurements were typically taken for each boulder. Among them, four points were taken at the base, another four points were obtained in the middle of the boulder, and one point was located on the top of the boulder. This survey procedure insures the dimensions of the obstruction are accurately incorporated into the whole reach geometry (Figure 3.2).

Measurement of Hydraulic Data - Depth and velocity data were collected at a peakflow of 42 m³/s, a moderate flow of 19 m³/s, and a baseflow of 1.79 m³/s. These three discharges represent the range of flow conditions commonly encountered in the Smith River and their magnitudes were determined from the readings obtained at a USGS gage station located immediately downstream of the dam. Measurements of water depths and velocities were taken along selected transects (Figure 3.1) as well as in the vicinity of some selected boulders. For example, to measure the velocities around a boulder, two points with different relative distance to the boulder (i.e. the ratio of actual distance from the boulder to the boulder width normal to the streamwise velocity), were defined in the field (Figure 3.3). Point 1 was located within the wake generated by the boulder, while point 2 was located further downstream (i.e. outside the wake region). At the baseflow, water depths at the two points were obtained using a wading rod, and velocities along various water depths were measured with a Marsh McBirney® Model 2000 flow meter. The flow meter was aligned with the direction of local maximum velocity and used to record the velocity angle using a compass. At higher flows, when the river became unswimmable, a SonTek® ADCP was deployed to record the corresponding water depths and velocity profiles. The ADCP was mounted to a cableway system set up across the channel and slid to the desired points for measurement (Figure 3.4). The positions of the ADCP were georeferenced using the total station before and after each measurement, and averaged values were used for data analysis. The ADCP was operated with a sampling frequency of 3 MHz, which is recommended by SonTek® for shallow water (< 3 m deep). Its transducers' sampling volume for the water velocity was divided into a maximum of 13 range cells along the local water depth, with each cell height of 0.15 m. The streamwise, lateral and vertical velocity components were combined within each range cell to obtain the 3-D velocity for that depth layer.

To reduce the turbulence fluctuation during the measuring period, the ADCP was kept stationary at each point for at least 10 minutes and the collected time-series velocities were time-averaged to get the mean velocity. Because the ADCP assumes the flow field is uniform within the sampling volume, it was not deployed in the immediate vicinity of any underwater obstructions (e.g. boulders) to avoid strong flow gradients. Aside from that, the substrate was inspected visually and the particles were classified into different size groups. This, in turn, was used to estimate the local channel bottom roughness.

Survey of Brown Trout Spawning Sites - Fish spawning locations were accurately georeferenced immediately after they were visually identified. Most brown trout redds were observed in slower water, but close to a fast current augmented by the boulders. By knowing the exact fish redd location, we were able to analyze the influence of boulders on local fish habitat.

Hydraulic Models

One-dimensional Hydraulic Model (PHABSIM) - PHABSIM is a 1-D hydraulic model based on the Instream Flow Incremental Methodology (IFIM) (Bovee et al., 1998). The model was set up using cross-sections to provide channel width and bed elevations. Each transect was divided into many small slices (i.e. cell or tile). The water depth and velocity within each cell/tile were assumed uniform. From the computations of water depth and velocity, and the measurements of channel substrate, PHABSIM was used to predict WUA (Weighted Usable Area) as an indicator of fish habitat quality based on the standard Habitat Suitability Criteria (HSC).

Two- and Three-dimensional Hydraulic Models (RMA2 and CFX) - 2-D and 3-D numerical approaches were employed to discretize the study sites, including the boulders, into meshes using either finite element method (in the 2-D RMA2) or finite volume method (in the 3-D CFX). The 2-D model (RMA2) was constructed with a mesh consisting of 46,191 quadratic triangular elements, whereas the 3-D model (CFX) was set up on a mesh composed of 2,114,126 tetrahedral elements, as illustrated in Figure 3.5. Mesh resolution varies spatially based on geometry complexity and the type of fluid solved. For both RMA2 and CFX models, finer meshes (i.e. elements with smaller sizes) were defined particularly near the boulder surfaces to resolve the small-scale turbulent flows surrounding the boulders.

Appropriate boundary conditions were specified for both models based on field measurements and observations. A non-slip condition was defined for all the solid boundaries in CFX, including the boulder surfaces, the river bottom, and the two banks; whereas a slip condition was set for the above boundaries in RMA2. In addition, all the boulder surfaces in CFX were taken as smooth walls, while the river bottom in CFX was treated as a rough wall, and assigned a roughness height 0.02 m. This value is equivalent to the mean size of sediment particles collected in the study sites. In RMA2, a Manning's n value 0.04 was specified in accounting for the river bottom friction. Finally, a recorded discharge was assigned to the upstream boundary for both models and a measured water surface elevation or equivalent water hydrostatic pressure was defined at the downstream boundary of the study sites.

Both RMA2 and CFX are considered as adequate tools in this study due to the following four reasons. First, the RMA2 model's working hypothesis, which is a 2-D horizontal flow field, satisfies the requirements of many free surface flow applications, provided a river has a large width-to-depth ratio (i.e. >20:1). In this study, the average width-to-depth ratios in the Smith River at base and peak flows are 30:1 and 25:1, respectively. On the other hand, the robust 3-D CFD solver CFX can adequately replicate the complex 3-D flow structure that is often found

near river obstructions (Salaheldin et al., 2004). Second, the finite element and finite volume methods adopted by the two models can adapt to complex streambed topography. Consequently, the mesh in each model can be refined intensively to include the complex river obstructions (i.e. the boulders). This function is important for the analysis of local complex flow patterns creating important fish habitat. Third, the wetting/drying method in RMA2 is capable of simulating the process of flooding and draining for a small area. This function is particularly useful when simulating low flows in the Smith River, because many boulders may just be partially submerged at the baseflow condition. Similarly, in the CFX model, VOF (Volume of Fluid) method is employed to define the interface between water and air throughout the study site at any flow conditions. Fourth, the turbulence wakes behind the boulders in the Smith River need to be adequately defined in both models. In RMA2, the turbulent stresses can be modeled using Boussinesq's eddy viscosity concept whereby the stresses are assumed proportional to velocity gradients. In contrast, the Reynolds Stress Method in the CFX model is utilized to estimate heterogeneous velocity distribution within the wake. Therefore, turbulence transport, flow separation, and vortex shedding from the boulders may be adequately quantified.

Influence of River Obstructions on Local Flow Patterns

To explore the extent that river obstructions impact local flow patterns at various flows, two discharges, which are the base and peak flows, were used to evaluate the impact of depth of submergence of boulders on local flow structures. Both the 2-D RMA2 and the 3-D CFX models were applied to this study.

The computed velocity distributions at the boulder-scale were compared to field data. In addition, vorticity and circulation, two spatial metrics for defining fish habitat proposed by Crowder and Diplas (2002), were also analyzed based on the results of the 2-D and 3-D models. Their suitability for use as metrics stems from their capability to quantify the complexity of small-scale flow structures, such as those found in the vicinity of boulders. The biological implications of vorticity and circulation are well documented. Researchers have reported that fish rocks, spur-dykes, and other flow obstructions create vortices and circulations that are utilized by fish and invertebrates for feeding and other purposes (Shields et al., 1995; Way et al., 1995).

In mathematics, vorticity is a measure of the rate of rotation of a fluid element about its three axes, which are in the x (streamwise), y (lateral) and z (vertical) directions (Munson et al., 1990). Although the 3-D CFX model is able to compute the vorticity in all the three directions, only the z (vertical) component can be calculated from the horizontal velocity distribution of the 2-D RMA2 model. Considering that the axes of the wake vortices behind flow obstacles are found almost vertical (Salaheldin et al., 2004), the vorticity in the vertical direction is used throughout this study, which can be defined as:

$$\mathbf{x} = \left(\frac{\partial v}{\partial x} - \frac{\partial u}{\partial y} \right) \hat{k} \quad (1)$$

where u and v are velocity components in the x and y directions, while \hat{k} is a unit vector in the z direction. To investigate the flow complexity within an arbitrary area, the absolute circulation, which integrates the absolute values of the above vorticity over a region of interest, can be estimated at a selected plane surface using the following expression:

$$\Gamma_{ABS} = \iint |\mathbf{x}| dA = \sum |\mathbf{x}| \Delta A \quad (2)$$

where GABS is the absolute value of circulation for a specific plane surface and ΔA represents a small element within that surface.

Biological Models

Bioenergetic Modeling - In a lotic system of a mountain river, some fish species (e.g. trout) often take up feeding stations where they remain relatively stationary and feed on food that drifts by them. Presumably, these fish occupy locations having “optimal” velocity conditions, where a fish’s energy expenditure (e.g. swimming cost) is minimized, while energy intake is maximized. In other words, drift-feeding fish will try to occupy a focal point velocity to maximize net energy intake (i.e. energy gain from the food minus energy cost for swimming). The relationship between the flow velocity (V) and fish’s net energy intake has led to the establishment of the following equation proposed by Grossman et al. (2002):

$$e^{(b+cV)} = 1/(cV - 1) \quad (3)$$

where b and c are fitting constants. These two empirical parameters can be obtained from the prey capture success curve (i.e. prey capture success P versus velocity V), which can be expressed as:

$$P = 1/(1 + e^{(b+cV)}) \quad (4)$$

In this study, a prey capture success curve for medium size trout (71 ~ 125 mm in length) (Hill and Grossman, 1993) was fit using Equation 4 (Figure 3.6). The obtained curve parameters b and c were then substituted into Equation 3 to calculate the optimal fish velocity, which is 23 cm/s in our study. Given that the RMS (root-mean-square) error of our hydraulic model simulation at baseflow is 4 cm/s, any location having velocity between 19 to 27 cm/s is deemed as a potential feeding station at that flow rate. Consequently, by providing a contour map of the velocities in the vicinity of boulders from the hydraulic models, potential drift-feeding fish habitat having optimal velocities is predicted. The results are then compared between the two study cases to evaluate the influence of boulders on local aquatic environments.

River-specific Habitat Suitability Criteria (HSC) - 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:

$$CSI = I_v^{0.49} \times I_s^{0.34} \times I_d^{0.17} \quad (5)$$

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.

Results and Discussion

Model Calibration and Validation

Among the three discharges simulated, the moderate flow (19 m³/s) was used for calibration of the 1- and 2-D hydraulic models, while the base (1.79 m³/s) and peak (42 m³/s) flows were used for validation purpose. During the calibration procedure, the roughness coefficient and the isotropic eddy viscosity were adjusted in the 2-D model until the predicted velocities and depths provided a good match to the corresponding measured values. In contrast, for the 3-D CFX model, we decided to validate directly the simulation results without any calibration efforts. We adopted such a modeling strategy for the following three reasons. First, in theory, because the 3-D code is more physics based than the 2-D code, it may model the nature of the flow more realistically than the 2-D code, and need less calibration effort (Lane et al., 1999). Second, the calibration procedure for the 3-D code is more complicated than the 2-D code due to complex mathematics manipulation, and will take much more computational time to complete the task. Third, over-calibration for one flow may sometimes make the adjusted model parameters unrealistic, so that they may not be appropriate for other flow simulations.

Water Surface Level - The relative errors between measured and simulated water surface elevations were computed at 186 random locations under each of the three modeled discharges. Overall, the agreement between the predicted and observed water surface elevations is good (Figure 3.7). All relative errors were within 10%. The mean absolute error of the predicted water depths throughout the sites is around 0.03 m for the 1- and 2-D models at baseflow with a mean water depth of 0.6 m, increasing to 0.04 m at peakflow with a mean water depth of 1.6 m.

Velocity Field - Individual values of water velocity predicted by the 1- and 2-D hydraulic models are well correlated with measured velocities at the three different flows. Figure 3.8 shows the comparison of velocity profiles at a selected transect between the calculated values, the corresponding measured values at base, and peak flows. At the baseflow, a close examination of the error distribution between the predicted and measured values illustrates that larger errors exist near the banks (Figure 3.8a). The error in the calculated velocity values near the channel sides may be attributed to the lack of information on riverbank friction (vegetative cover) and possibly insufficient mesh refinement in the near bank regions. Similar error trends were exhibited for the peakflow scenario (Figure 3.8b), though it was not possible to collect data very close to the banks because of equipment limitations (ADCP requires at least 0.5 m water depth for correct deployment). Comparison between the predicted total horizontal velocity from the 3-D model (i.e. $\sqrt{u^2 + v^2}$, where u, v are the streamwise and lateral velocities, respectively.) and the field data measured at the base and peak flows are shown in Figure 3.9. Each point in Figure 3.9 represents the location of a flow meter measurement within and out of the wake of the single boulder in the field. Considering that the simulation results from the 3-D model have not been calibrated to any field data, the correlation between the 3-D model prediction and field data is reasonably good.

Influence of River Obstructions on Local Flow Patterns

Both the 2- and 3-D hydraulic models were employed to examine the impact of boulders on local flow patterns of biological importance at various flows. The results of the depth-averaged velocity obtained from RMA2 are compared against the total horizontal velocity component obtained from CFX at the aforementioned two points behind the boulder (see Figure 3.3) as well as at a selected horizontal plane. This comparison is performed at two flow regimes (Table 3.1). The plane selected to plot the CFX results is located 15 cm above the river bottom,

which is found to be, on average, the favored location of adult brown trout (Bouckaert and Davis, 1998).

Figure 3.10 provides the vertical distribution of the streamwise velocity, V_x , at two locations behind the boulder at the peakflow. In the lower layer behind the boulder, the flow is separated to create a recirculation region (see Point 1). Inside the recirculation zone, the flow is backward towards the boulder. These backward flows exist from 0 to 20 cm above the river bottom, a range that overlaps with the optimal height sought by brown trout in the field. The depth-averaged velocity from the 3-D model is 1.01 m/s at point 1, while the corresponding velocity predicted by the 2-D model is 1.15 m/s, 0.14 m/s higher than that obtained from the 3-D model. The difference in the calculated values obtained by the two models is getting smaller as the point moves downstream from the flow obstruction (see Point 2), where the flow becomes more uniform.

At baseflow, at point 1, the streamwise velocity is facing backward over the entire flow depth (Figure 3.11). Interestingly, the maximum backward velocity occurs at about a height of 15 cm above the bed, which has been suggested as the optimal drift-feeding position for brown trout. Hence, it might be likely that fish will seek an optimal depth in the velocity shelter where maximum reversed flow occurs. The depth-averaged streamwise velocity predicted by the 3-D model is -0.06 m/s, whereas the 2-D model calculates a positive value. The depth-averaged V_x of the 2-D model is different from the value obtained from the 3-D model at point 1. Once again, as the point moves downstream, where there are no reversed flows, the depth-averaged velocity value predicted by the 2-D model agrees reasonably well with the corresponding depth-averaged value predicted by the 3-D model (see Point 2 in Figure 3.11).

Figure 3.12 illustrates the vorticity distribution around the single boulder computed by the 2- and 3-D models at the baseflow. The reversed flow in the wake impinges upon the obstruction piercing through the water surface, rotates, and generates vortices. Consequently, significant vorticity values ($> 0.4 \text{ s}^{-1}$) are found in the wake region from both hydraulic models. The result from the 3-D model shows that the wake extends approximately two obstruction diameters downstream from the boulder (Figure 3.12 left), which is consistent with the wake size behind other obstacles found by Shamloo et al. (2001) and Salaheldin et al. (2004). The high vorticity area calculated from the 2-D model is less than that of the 3-D model (Figure 3.12 right). According to the principles of fluid mechanics, flow passing by the boulder should separate from the sides of the boulder and generate vortex shedding. These vortices shed from the boulder are further transported downstream in the form of small-scale eddies. The distribution of the wake vortices shown in Figure 3.12 suggests that this feature is better characterized by the 3-D CFX than the 2-D RMA2.

At the peakflow, when the boulder becomes well submerged, the 2-D model only computed uniform velocities and did not detect any significant levels of vorticity (Table 3.2). In contrast, large high vorticity area of nearly 2 m^2 is found by the 3-D model at the defined plane (Table 3.2). Shamloo et al. (2001) studied the flow behavior behind a hemisphere at deep water and observed similar flow patterns.

To further estimate the flow complexity around the boulder, the circulation metric Γ_{ABS} was computed within the turbulence wake by both models (Table 3.2). Table 3.2 shows that at the baseflow, the defined plane has a circulation metric value of $0.44 \text{ m}^2\text{s}^{-1}$ from the 3-D model, which is greater than that calculated based on the depth-averaged velocity data from the 2-D model. This is consistent with the previous comparison result for the vorticity area between the two models, since the circulation is the integration of vorticity over that area. At the peakflow,

the 2-D model does not predict any circulation metric values based on the result of zero vorticity obtained previously. In contrast, a circulation metric value of $4.32 \text{ m}^2\text{s}^{-1}$ was computed by the 3-D model. Hence, the circulation metric values obtained by the 3-D model may adequately reflect the heterogeneous flow structures behind the boulder.

Influence of River Obstructions on Fish Drift-feeding Habitat

To further illustrate the biological importance of river boulders, hydraulic results obtained from the 2-D simulations are analyzed using the bioenergetic model in assessing potential drift-feeding habitat in the study reach. The results of optimal fish velocities are compared between the two cases (with and without boulders) at various flows. Suitable habitat is identified as the region having velocities between 19 and 27 cm/s, where fish can get a maximum energy intake while keeping a minimum swimming cost.

Two identical meshes, each having 46,191 elements, were created. In the first case (CASE 1), the data set used to generate the mesh is comprised of all the bathymetry information, including the boulders. For the second case (CASE 2), the data set consists of the same topography as the first case, except for selected boulders found near identified fish locations, which are purposely excluded. This modeling strategy has been adopted for the following three reasons. First, by keeping the same mesh resolution over the entire modeled site, any difference between the two cases can be attributed to the removal of the selected boulders. Second, since only a small number of boulders are excluded from the second mesh construction, the overall change in riverbed topography is small and localized in nature. Third, since all the selected boulders are close to observed fish habitat, the simulation results can help determine the biological influence of the inclusion or exclusion of river obstructions on local fish environments.

Figure 3.13 shows the comparison of predicted drift-feeding habitat over the entire site at the baseflow (relative depth < 1.0). Results show that, at such a low flow rate, the emergent boulders either increase or decrease the surrounding good drift-feeding habitat area, in comparison with the simulation without boulders. In Area A, with the incorporation of four boulders measured from the field, a total potential drift-feeding habitat of 20 m^2 is predicted around these obstructions. In contrast, when the four boulders are excluded from the river topography, flows in this region are not retarded by any river obstructions and become too fast for supporting drift-feeding fish. In the middle of the reach, after excluding the six boulders in Area B, the model results overestimate the potential drift-feeding habitat by 41%. In this portion, the channel width is reduced by the six boulders and a large part of flow is shifted to the left side (looking upstream). Consequently, the local velocities are accelerated, which provides a fast flow area that is difficult for fish to capture prey. However, with the absence of the six boulders, the simulation shows that flow becomes fairly uniform and local velocities are significantly reduced to levels suitable for drift-feeding activity. In contrast to the situation in Area B, the simulation without boulders underestimates the drift-feeding habitat in Area C by 58 m^2 . For peakflow, the present study shows that the inclusion or exclusion of small river obstructions does not affect the predicted drift-feeding habitat. As in such a condition with a high relative depth (which exceeds 4 at most boulder locations), theoretically a trout would be unable to feed in the water column because of high energetic cost and inability to withstand the harsh hydraulic conditions (Lagarrigue et al., 2002).

Brown Trout Habitat Suitability Analysis

Our results indicate that areas with more redds had higher habitat quality indices, which implied a positive relationship between redd densities observed in the river and habitat quality predicted by the models (Figure 3.14). Specifically, 78% of the redd locations were identified as areas having high suitability indices ($CSI > 0.9$) by the 2-D model, whereas 69% redd locations were predicted to have high CSI values by the 1-D model. The result of the 2-D model also shows that there were no redds found in the area having a CSI value less than 0.3. In contrast, the result from the 1-D model shows that there were three redds in the locations having CSI values below 0.1 (Table 3.3). 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 cell, although the remaining part of that cell 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 cells 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 3.14). For instance, at the baseflow 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 the baseflow, but differences of the wetted area calculated between the two models were reduced for higher flows (Table 3.4). Limited by the ability of the 1-D model in bathymetric approximation, the stream obstructions such as boulders between cross-sections were treated as rough elements and estimated as being submerged underwater even at baseflow. Under the same baseflow 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. Obviously, the 2-D model is more accurate than the 1-D model when dealing with local obstructions, since dry areas should be excluded from usable habitat due to unsuitable water depth. The comparison results of WUA between two models are listed on Table 3.4. At the baseflow, both 1- and 2-D models generated nearly identical WUA. This is because the 1-D model predicted more wetted area and a smaller 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 the moderate flow, total WUA of the study reach predicted by the 1-D model was 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 peakflow. 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 3.15). For example, at the moderate flow the reach-averaged velocity was 0.64 m/s corresponded to a velocity suitability index of 0.8. However, when the reach-averaged velocity increased to 1.02 m/s at peakflow, the velocity suitability index was reduced to 0.2.

Discrepancy in habitat prediction by the two 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. In contrast, the finite element mesh in the 2-D model was adjusted to adapt to the highly irregular channel

boundaries. Element 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.

Conclusions

Hydraulic Models Performance - Overall reach hydraulics and reach-scale fish habitat can be adequately evaluated by the 1- and 2-D hydraulic models with regard to the prediction of water depths and velocity distributions. However, the 1-D hydraulic model PHABSIM may not correctly represent hydraulic conditions between cross-sections and their associated potential fish habitat. Total WUA may be underestimated by PHABSIM at high discharges. Compared to the 1-D model, the 2-D hydraulic model RMA2 may allow a better representation of the physical environment of the fish habitat through the finite element method, if detailed topographic data are available to construct the mesh. The study also shows that both the 2- and 3-D models can be used to analyze small-scale flow patterns around the boulders, as long as the flow region is uniform and out of the turbulence wake created by the boulders. Within the wake region where flow complexity is greatly increased, it is recommended to use a 3-D model instead of a 2-D model, because the latter may overestimate local depth-averaged velocities in the wake.

Biological Importance of Boulders - The area occupied by the boulders of interest amount to a total area of 38 m², only 0.8% of the entire reach. However, the addition of such a small area of boulders augments the local habitat suitable for drift-feeding fish by 164 m² at baseflow conditions. These results suggest that under low flow conditions with a relative depth less than 1, the total amount of potential fish habitat may be dictated by the presence of properly placed flow obstructions, rather than the remaining large flat area of the river reach. Although these river obstructions are difficult to measure and are easily neglected during river restoration studies, it is crucial to correctly evaluate their effects on local flow behavior before proceeding further with fish habitat modeling.

Recommendation for Optimal Reservoir Release - 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 rather than a 1-D model. Based on our regression analysis on habitat quantity from multiple flows simulation, optimal reservoir release may occur at 12 m³/s for our study reach. Obviously, more research should be done on other reaches of the Smith River to test if this recommended flow would satisfy the local brown trout flow requirements.

Flow Management

Optimum flow range for brown trout spawning

Hydraulic models were used in predicting physical habitat for brown trout spawning under alternative flow scenarios. Results from the habitat simulation indicates significantly positive relationship between redd density and habitat quality predicated by the model. Based on our regression analysis, current base flow appears to be below the optimal reservoir release range (9-

15 m³/s), while the peak flow is too high to support suitable brown trout spawning environment. A 12 m³/s reservoir release scenario predicted the best suitable habitat availability in our study site. However, because the highly fluctuating flow causes temporal changes in the locations of suitable habitat, an adaptive monitoring program is needed to correctly evaluate the fish habitat under such a flow scenario.

Shear stress effect of unsteady flow increase

Our analysis from hydraulic models shows that gravels, which are important to redd construction, may be removed under current reservoir flow within its initial release period. This fluctuating flow also causes a higher stress to fish and may displace young brown trout as well. Our recommendation is to use a two-step flow release scenario. It is better to use one turbine to release flow for half an hour and then add another turbine to release additional flow if necessary, the ramp down should also be stepped. Study illustrates the shear stress acting on gravel and drag force exerted on fish may be greatly reduced under such a hypothetical reservoir release scenario, and hence a healthier stream can be maintained without affecting power generation requirement.

Table 3.1. The flow regimes modeled in this study.

<i>Flow Regime</i>	Discharge (m ³ /s)	Free Stream Mean Velocity (m/s)	Boulder Height h and Width w (m)	Relative Depth (d/h)
Peakflow	42	1.2	$h = 0.36$	3.7
Baseflow	1.78	0.25	$w = 0.5$	0.64

Table 3.2. Vorticity and circulation results from the 2-D and 3-D models for the single boulder.

	Baseflow			Peakflow		
	Vorticity area (m ²)	Circulation (m ² s ⁻¹)	Volume (m ³)	Vorticity area (m ²)	Circulation (m ² s ⁻¹)	Volume (m ³)
2-D RMA2	0.24	0.13	0.05	0	0	0
3-D CFX	0.46	0.44	0.087	1.8	4.3	0.55

Table 3.3. Relationship between redd density and CSI computed by the 1-D and the 2-D models (The redd density is calculated based on a unit area of 1000 m²).

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 3.4. 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
	10.0	4550.20	2375.62	52.21
	19.0	4629.11	2096.74	45.29
	30.0	4629.11	1702.41	36.78
	42.0	4629.11	841.74	18.18
2-D	1.79	4151.21	2039.55	49.10
	10.0	4633.61	2733.00	58.98
	19.0	4600.32	2428.31	52.79
	30.0	4633.61	2017.47	43.54
	42.0	4633.61	1011.71	21.80

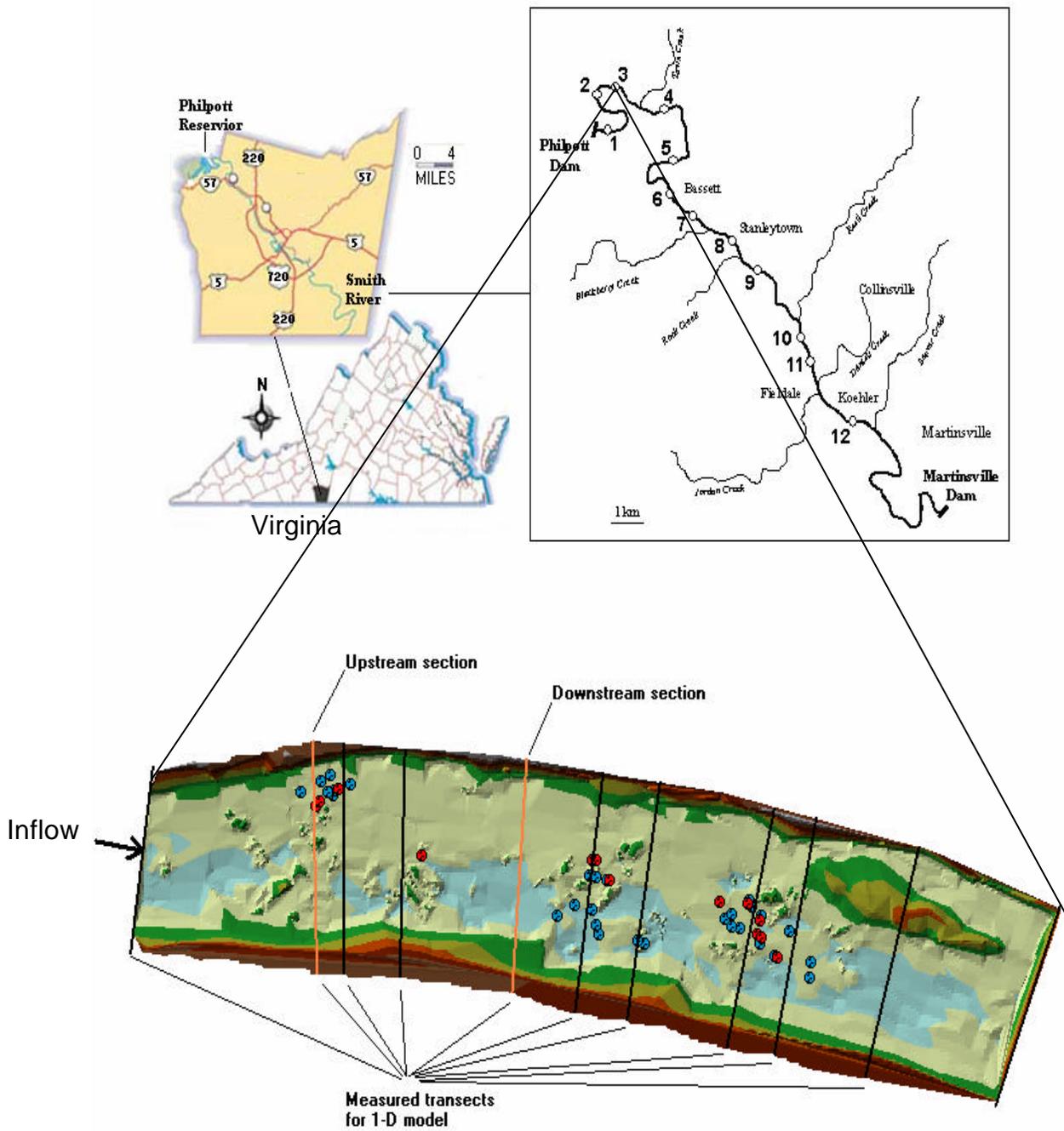


Figure 3.1. Map of the Smith River tailwater with sampling sites numbered upstream to downstream (Site 3 & 7 are the modeling sites). Color dots represent redd locations in 2000 (blue) and 2002 (red).

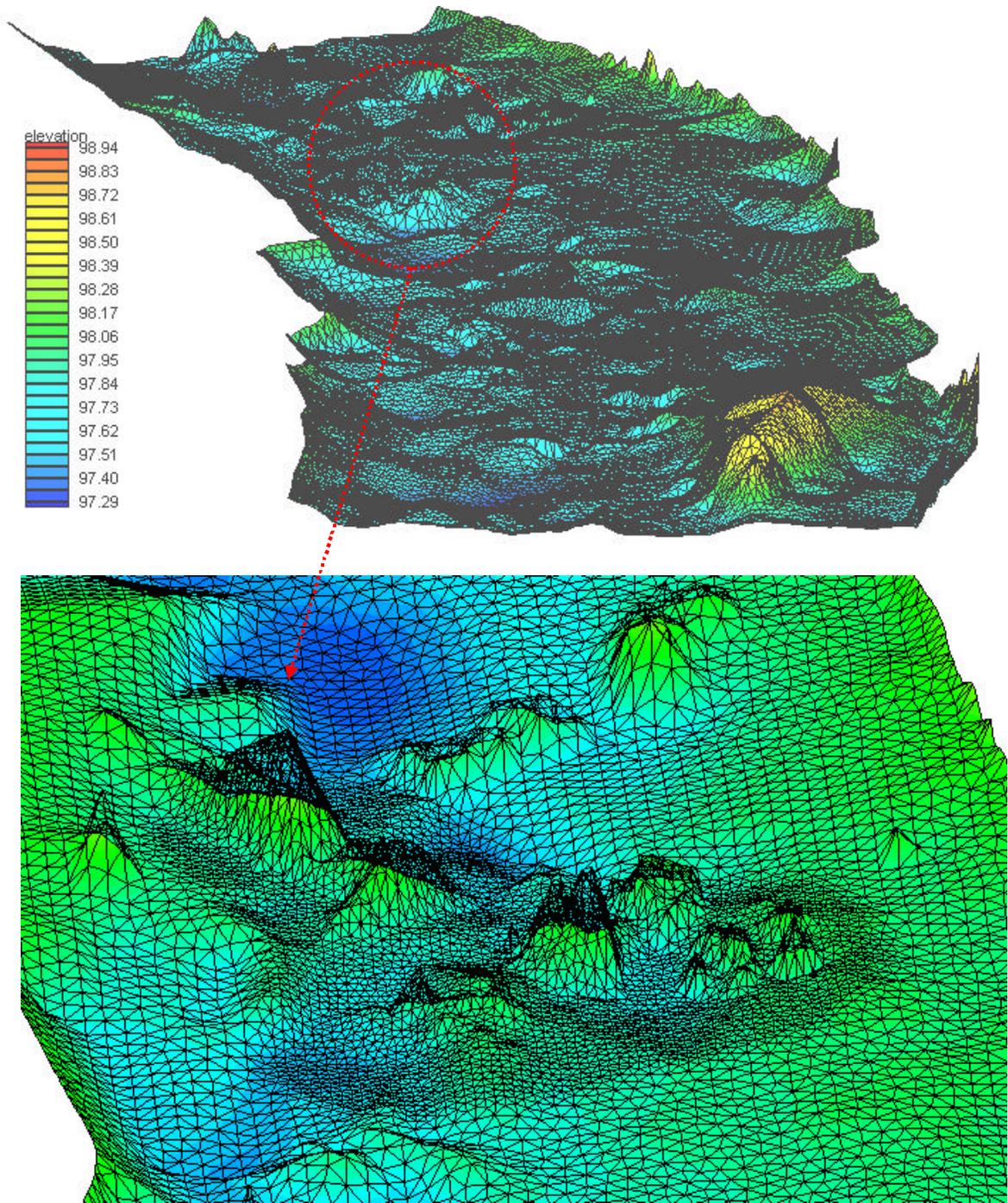


Figure 3.2. Channel topography discretized by finite element mesh in the two-dimensional model (RMA2). Mesh is especially refined near the boulders to capture their complex geometric features.

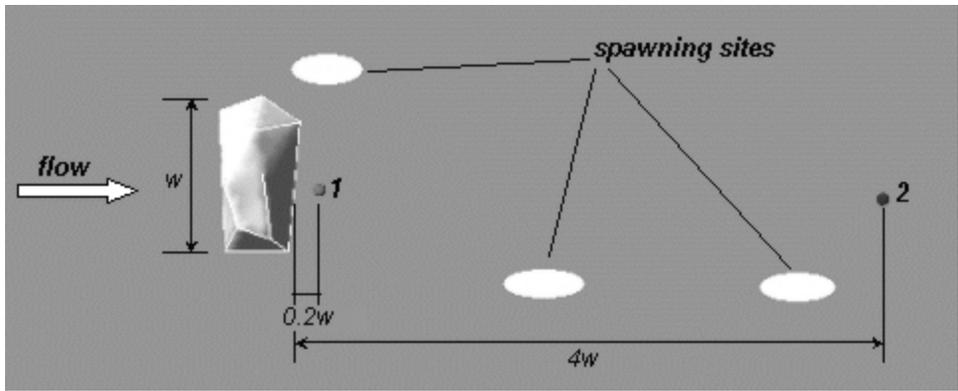
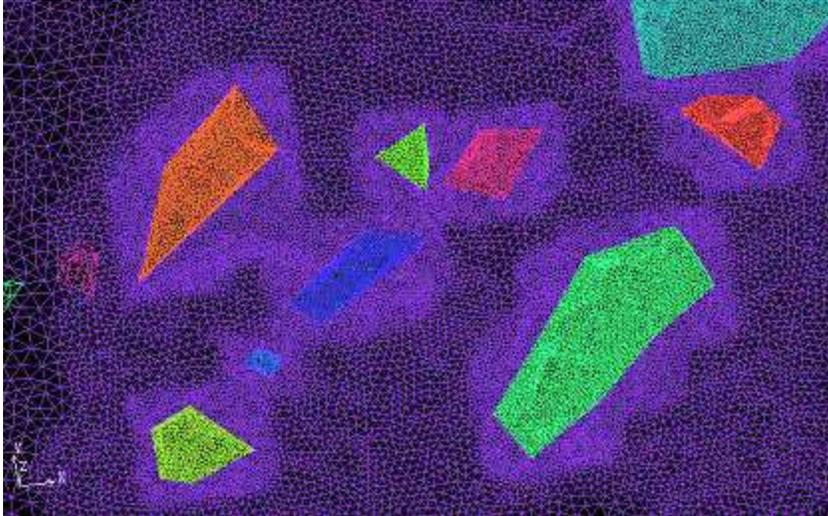


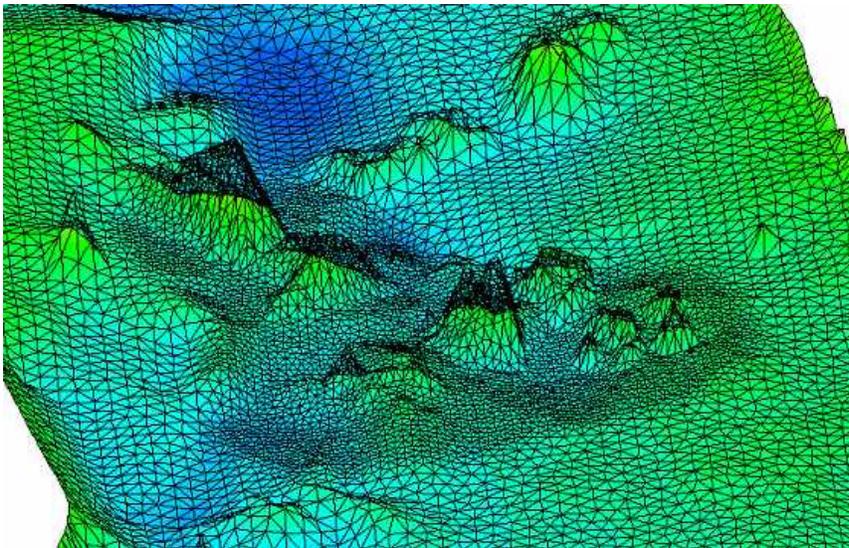
Figure 3.3. A plane view of the single boulder surrounded by three spawning sites (i.e. redds) and two defined points. The relative distances of the two points from the boulder are 0.2 and 4 for point 1 and 2, respectively.



Figure 3.4. Pulley system setup across the Smith River to which the ADCP attached.



(a) 2-D Mesh for the riverbed and boulder cluster, flow is from top to bottom.



(b) 3-D Mesh for the riverbed and boulders, flow is from left to right. Color polygons represent boulders.

Figure 3.5. 2-D Mesh and 3-D Mesh for the study site (including the boulders).

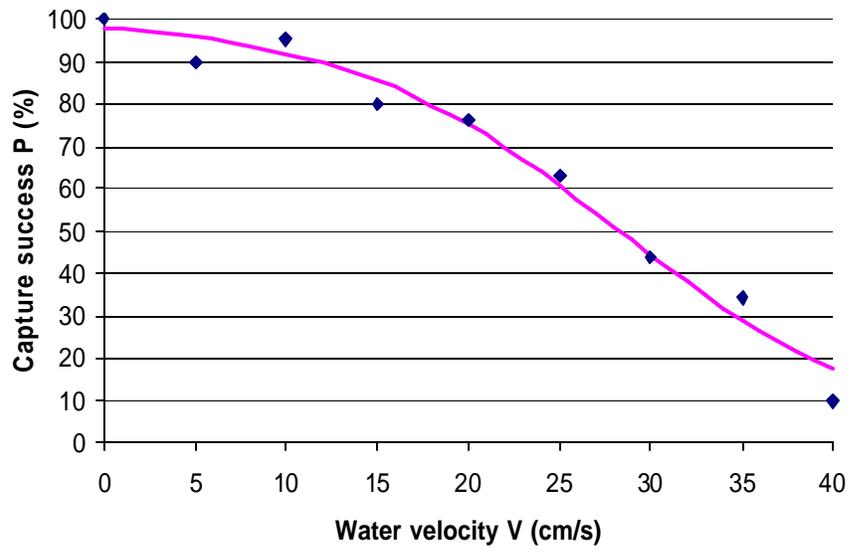


Figure 3.6. A prey capture success curve for medium trout, $P=1/(1+e^{(-3.76311+0.133102*V)})$, where $b = -3.76311$ and $c = 0.133102$.

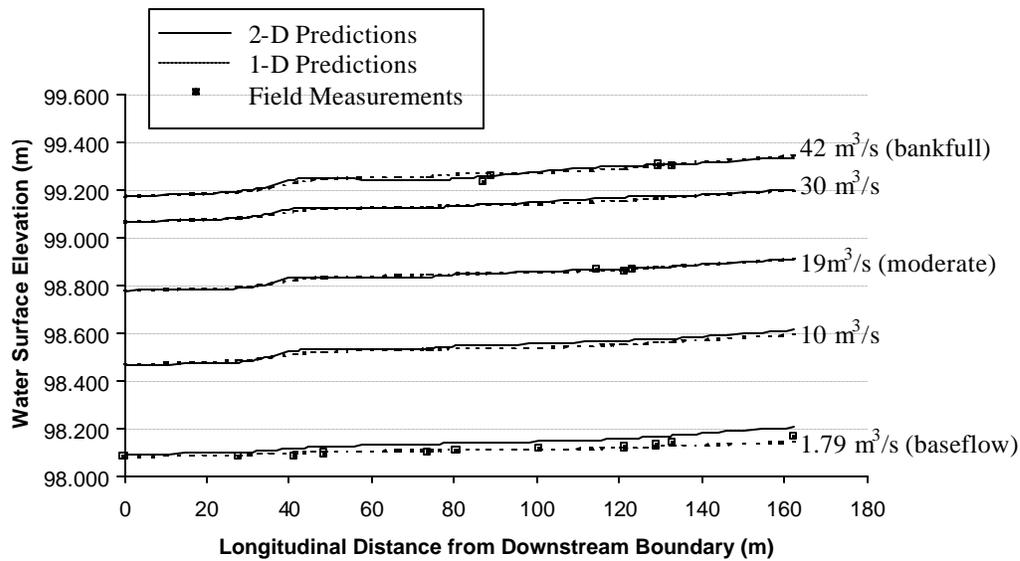
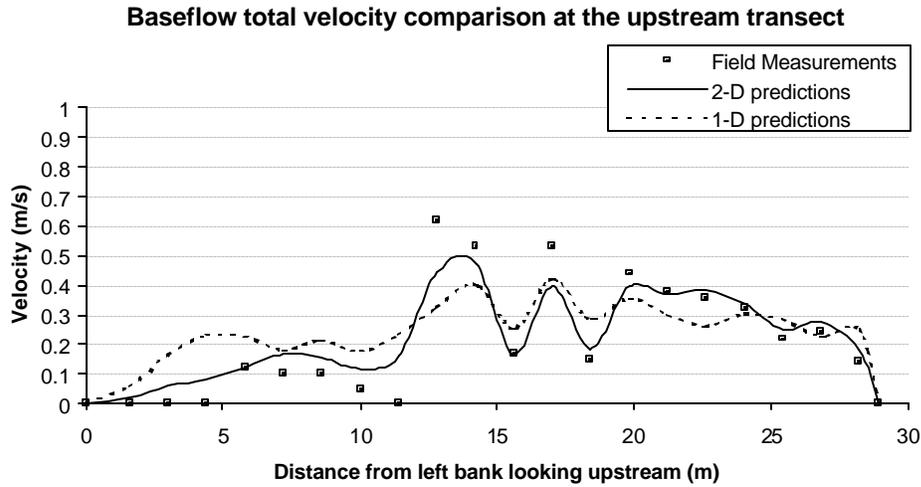
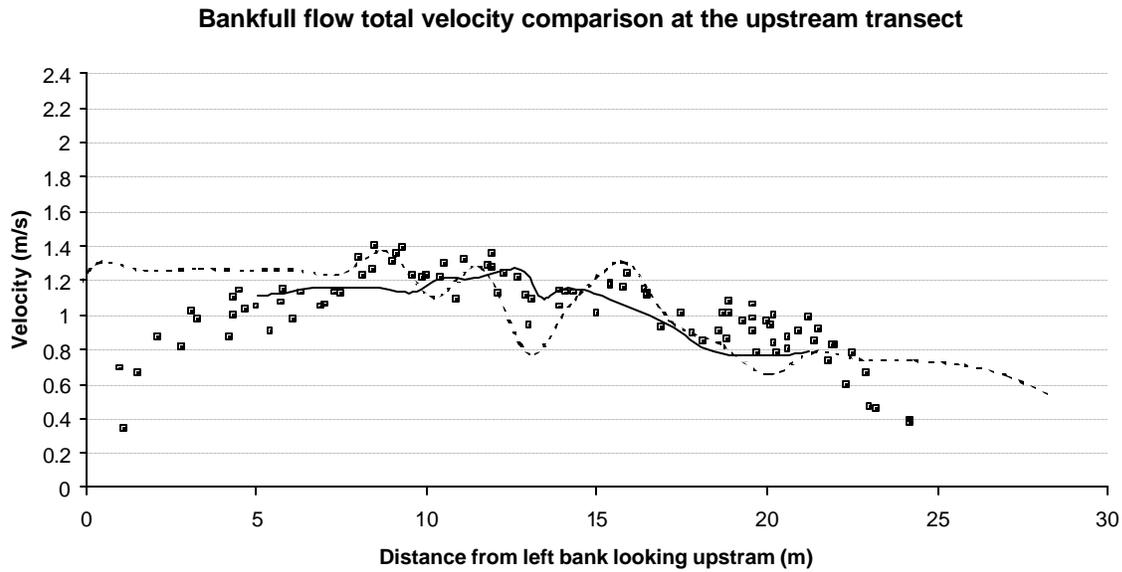


Figure 3.7. Comparison of longitudinal water surface elevation profiles along the study site, as predicted by 1-D (broken line) and 2-D (solid line) models at various flows. Black points depict the measured data using the wading rod and ADCP. For the convenience of comparison, the water surface elevations in the 2-D model were averaged in each cross section.



(a) The baseflow



(b) The peakflow

Figure 3.8. Comparison of the total velocity profiles between the hydraulic models' predictions and field data at the base and peak flows. The velocities are in the directions having maximum magnitudes. RMS (root-mean-square) value for base and peak flow simulations is 0.04 m/s and 0.09 m/s, respectively.

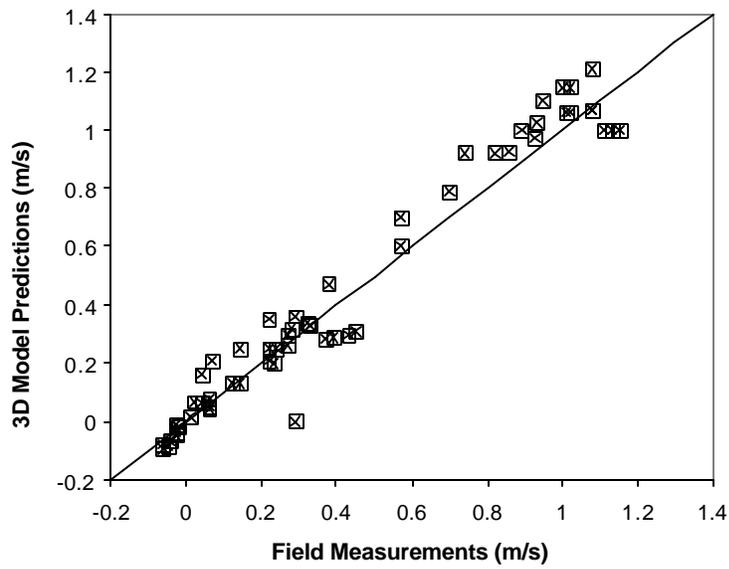
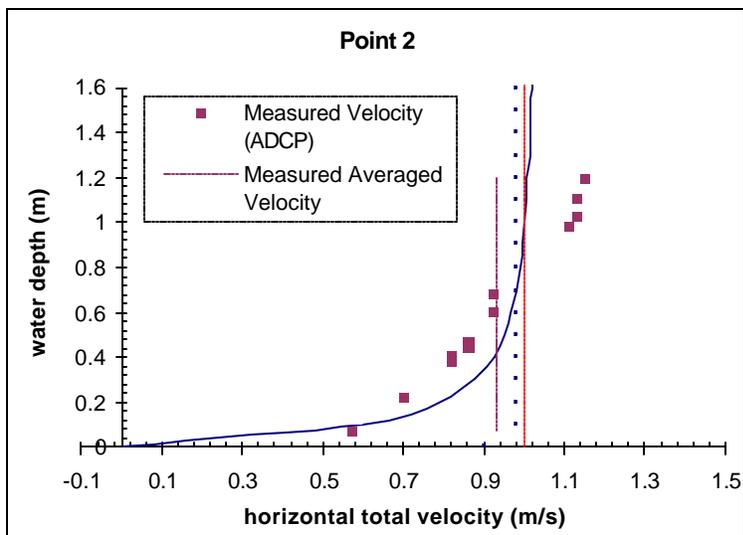
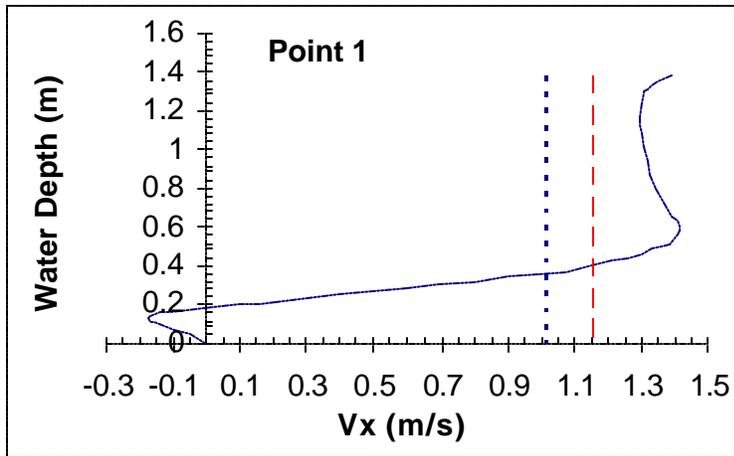


Figure 3.9. Comparison of the total horizontal velocities between the 3-D numerical results and field data at the base and peak flows using a random sample.



- velocity profile (3-D model)
- - - depth-averaged velocity (3-D model)
- - - depth-averaged velocity (2-D model)

Figure 3.10. Comparison between the 2- and 3-D model results for streamwise velocity V_x at the locations of two points at the peakflow.

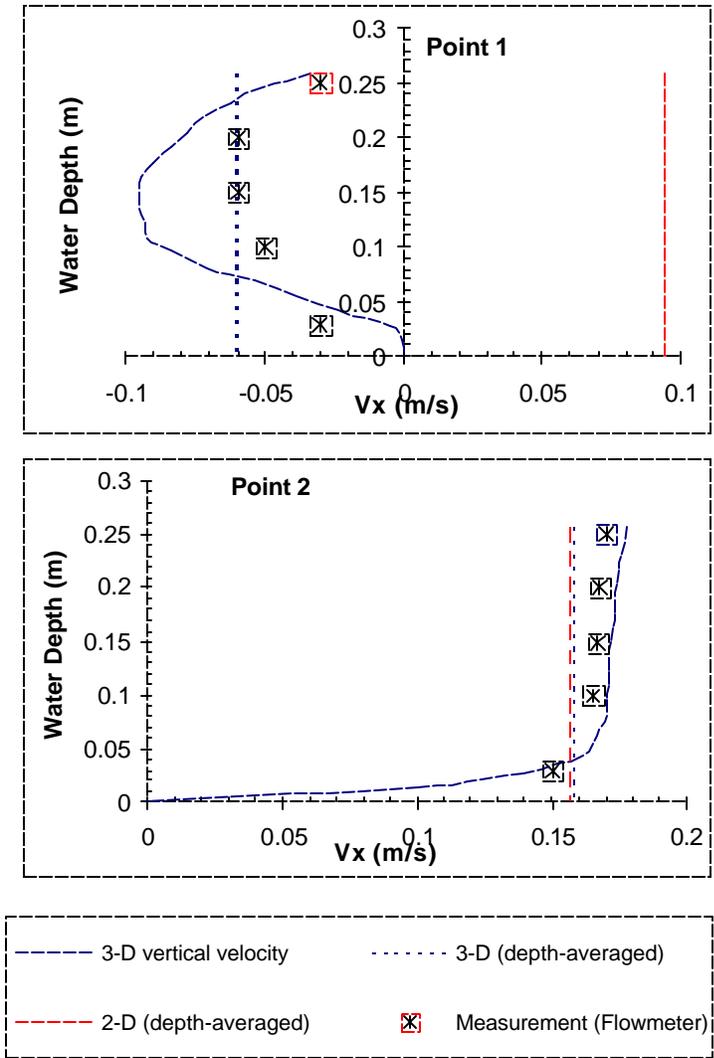


Figure 3.11. Comparison between the 2- and 3-D model results for streamwise velocity V_x at the two locations at the baseflow.

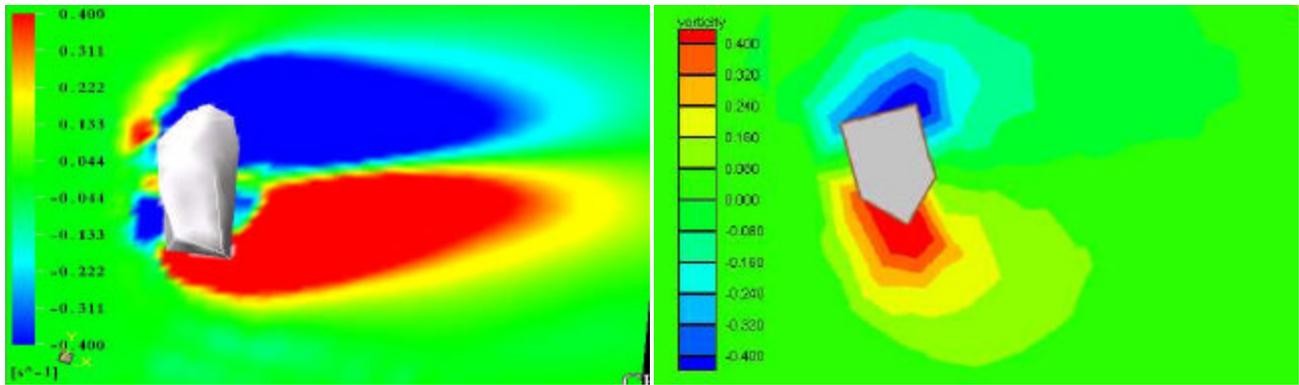


Figure 3.12. Comparison of vorticity values around the boulder at the baseflow between the 3-D (left) and the 2-D (right) models. Flow is from left to right and the vorticity unit is in 1/s.

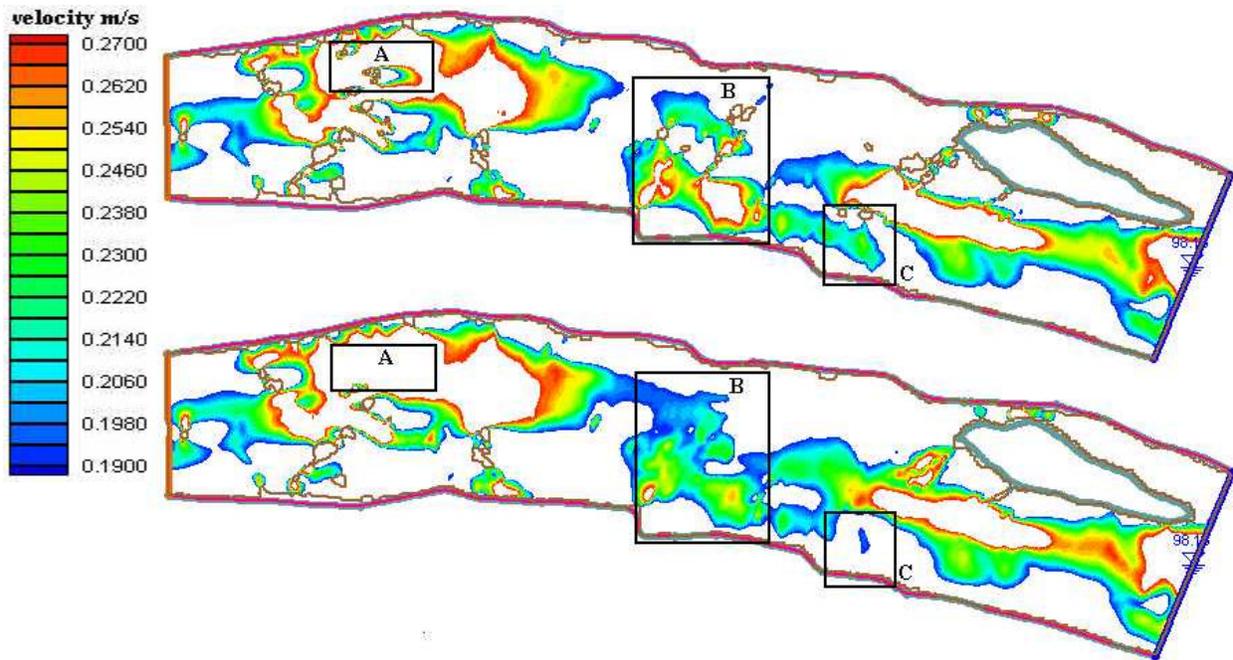


Figure 3.13. Comparison of the optimal velocity area for the drift-feeding fish at the baseflow. Upper graph has the selected boulders and lower graph is without the selected boulders. Areas A, B, and C represent locations where the boulder geometries are modified. White areas have velocities out of the suitable range for the drift-feeding fish. Flow is from left to right side.

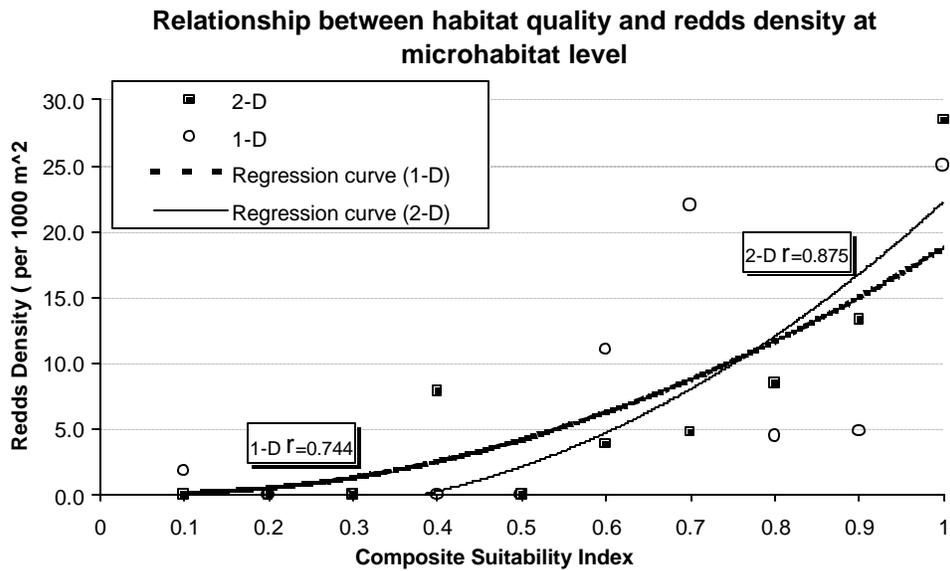


Figure 3.14. Comparison of habitat quality (CSI) at the baseflow between the 1-D (PHABSIM) and 2- D (RMA2) models at the mesohabitat level (i.e. cell or element). Redd density is calculated based on a unit area of 1000 m². Best fit curves (1-D r = 0.74; 2-D r = 0.88) are created through polynomial regression analysis by Microsoft Excel.

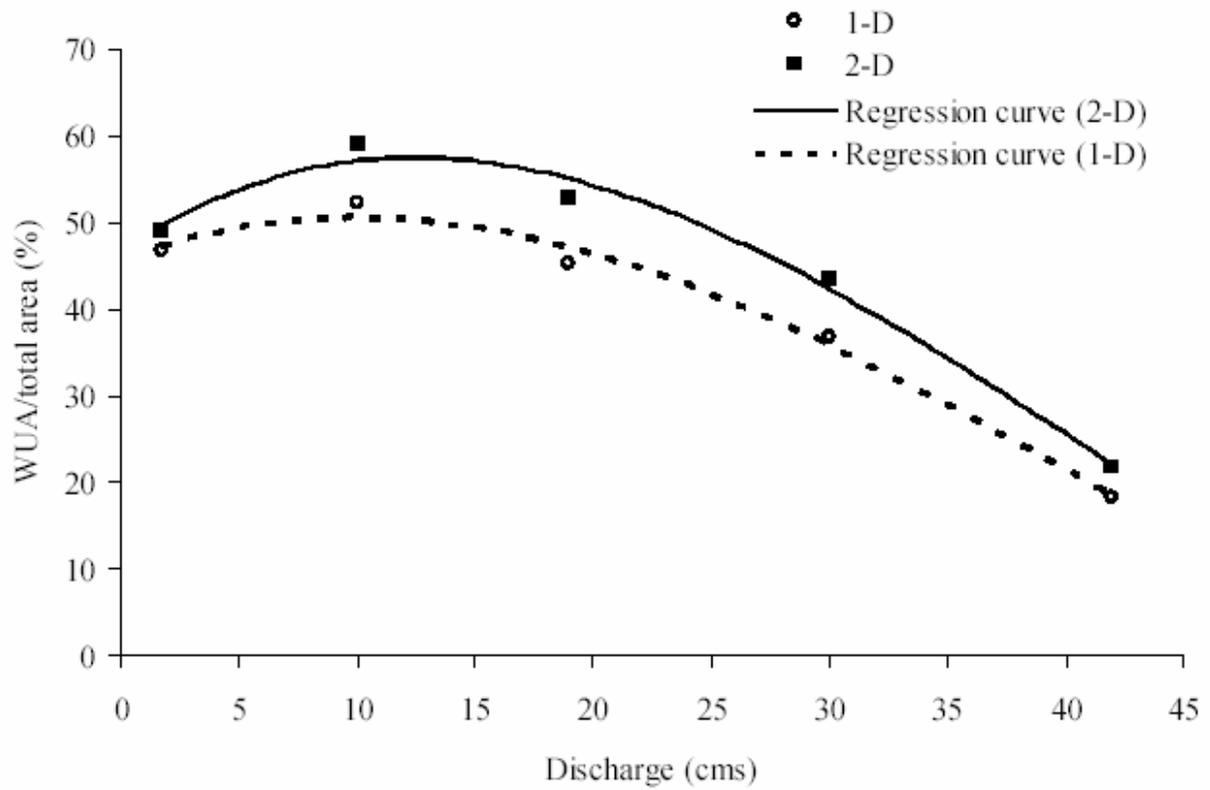


Figure 3.15. The WUA/Total Area ratio (%) predicted by both the 1-D and 2-D models at multiple flows. Best fit curves are created through polynomial regression analysis by Microsoft Excel.

Management Recommendations

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

Dam operation has changed the daily flow to one with base flows and peak flows with less magnitude, shorter duration and more frequent pre-dam flows. The flow regimes during this study were significantly different from normal in that 2003 had more peak flows and 2002 had fewer. Water temperature is directly affected by hypolimnetic release from Philpott dam and is a key factor to cue brown trout spawning, control incubation of eggs, and determine early growth.

A combination of peaking flow, temperature, sedimentation, and gravel permeability influence spawning and recruitment success. An improvement in first-year growth and survival of brown trout could be achieved via warmer water temperatures upstream, peak flows of less magnitude and occurrence, and a reduction in fine sediments. In particular, peak flow restrictions during the spawning and incubation periods for brown trout would result in enhanced recruitment success.

Job 2, Part A. Determinants of Brown Trout Growth and Abundance

Water temperature failed to explain observed variation in growth in age-0 and age-1 brown trout, whereas consumption rates were better predictors of observed growth. Fish contributed little to the diet of brown trout except where relative abundance exceeded 100 fish per 100 meters of stream. Comparisons of predicted growth of brown trout under three alternative flow regimes showed that none of the alternatives would likely increase growth rates in all four reaches of the tailwater. The lack of predicted response to temperature change is due to the predominance of small invertebrates in the diet. Increased growth of brown trout is possible only if the diet is shifted to include more fish or crayfish. Therefore, management efforts should be directed towards enhancing the productivity of the Smith River for insects, crayfish, and prey fishes.

Mean daily flows in April through June were strongly related to the abundance of age -0 brown trout and the relationship persisted to the next year. High flows during this period, therefore, have a lasting influence on year class strength in brown trout. Adaptive management of peaking flows during April through June would provide an approach for balancing peak power demands with management targets for brown trout population abundance.

While this project provided an overview of the brown trout population in the Smith River, there are aspects that still have not been assessed. Competition between rainbow trout and brown trout in the Dam Reach should be assessed and holdover rates of stocked rainbow trout should be determined. If holdover rates are high and diet studies show that there is significant diet overlap between brown trout and rainbow trout, then stocking rates of rainbow trout should be reduced to help decrease the competition.

Consumption by brown trout decreased with increasing downstream distance from the dam for age-0 brown trout, which was the inverse of what was observed in age-1 brown trout. Although it is unclear why this trend in age-0 consumption occurred, regression analysis

indicated that higher P-values occurred with lower relative abundances of non-game fish. With nongame fish, such as Roanoke darters *Percina roanoka* and bluehead chubs *Nocomis leptocephalus*, consuming aquatic insects (Jenkins and Burkhead 1993), there is potential for diet overlap and competition between nongame fish and age-0 brown trout in the downstream reaches because of the high abundances of nongame fish. Understanding of potential diet interactions between trout and nongame fish is needed to fully analyze potential changes in population dynamics due to alternative thermal regimes.

One difficulty during the assessment was the identification of age classes for trout. While length-frequency analysis allowed for the identification of age-0 and age-1 trout, age-2 and older trout were undistinguishable through length-frequency analysis. Identification of the older year classes would provide a greater understanding of mortality and survival of trout in the Smith River. Additional tagging studies to determine harvest rates would provide insight into the fate of brown trout.

Fecundity of brown trout was not examined during this study. Assessment of fecundity would provide additional insight into spawning potential of brown trout. It is also unknown at what size brown trout in the Smith River become mature. Information on maturity combined with energy requirements could be used in bioenergetics modeling to evaluate potential changes in growth rates due to reproductive costs.

Job 2, Part B. Longitudinal Patterns of Community Structure for Stream Fishes in a Virginia Tailwater

Numbers of non-salmonid fishes are depressed in the Smith River and controlled by temperature and flow variation. Tributaries appear to serve as refuges for these fishes and supplement the mainstream populations. Enhancing the non-salmonid community in the Smith River could depend on several factors. Changing the operation of the dam to a discharge schedule with lower magnitude and duration releases could increase the productivity of the fishery by increasing numbers of individual species. Improving the water quality in tributaries (i.e. enforcing water quality standards, decreasing sources of sedimentation, educating the public about watershed dynamics to help decrease litter input) could enhance the non-salmonid fish community since tributaries appear to moderate the effects of flow regulation. Decreasing the input of sediment into the river could aid reproductive efforts of benthic animals, such as *Etheostoma flabellare*, and could benefit benthic feeders such as *Percina rex* (the federally endangered Roanoke logperch). Because *Nocomis leptocephalus* mounds in the Smith River were found adjacent to cover objects almost without exception, planting cover objects (i.e. small boulders) throughout the mainstem could provide velocity shelter for this species to utilize for mound-building.

The Smith River represents a complex environment for non-salmonid species with no single mechanism driving community dynamics. Based on this study, efforts to enhance the fish community via decreased fine sediment for peak flow moderations would likely be effective. Numbers of non-salmonid fishes dramatically increased under a lower magnitude and duration flow release, similar to the response seen among young brown trout. Therefore, changing the operation of Philpott Dam in low flow years to have lower peak flows or longer periods without

generation would prove to have the greatest impact on non-salmonid fish productivity in the Smith River.

The fish community does not appear to be stabilized in terms of constancy in numbers of fish due to flow variability (Ross et al., 1985). Therefore, long-term monitoring is needed in the Smith River to evaluate the full extent of changes in fish community characteristics over time. This research indicates that tributaries play a major role in the structure of fish communities. Thus, it follows that tributary junctions should be considered as sampling stations for similar studies. Finally, fish-habitat models developed to predict functionally significant habitat requirements such as spawning microhabitat are transferable and represent a viable tool for management. Key microhabitat features, such as mixture of boulders, cobbles, and gravel, appear to enhance the abilities of non-salmonid fishes to persist and spawn despite the daily flow pulses.

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

Optimum flow range for brown trout spawning -- Hydraulic models were used in predicting physical habitat for brown trout spawning under alternative flow scenarios. Results from the habitat simulation indicates significantly positive relationship between redd density and habitat quality predicated by the model. Based on our regression analysis, current base flow appears to be below the optimal reservoir release range (9-15 m³/s), while the peak flow is too high to support suitable brown trout spawning environment. A 12 m³/s reservoir release scenario predicted the best suitable habitat availability in our study site. However, because the highly fluctuating flow causes temporal changes in the locations of suitable habitat, a adaptive monitor program is needed to correctly evaluate the fish habitat under such a flow scenario.

Shear stress effect of unsteady flow increase -- Our analysis from hydraulic models shows that gravels, which are important to redd construction, may be removed under current reservoir flow within its initial release period. This fluctuating flow also causes a higher stress to fish and may displace young brown trout as well. Our recommendation is to use a two-step flow release scenario. It is better to use one turbine to release flow for half an hour and then add another turbine to release additional flow if necessary. Study illustrates the shear stress acting on gravel and drag force exerted on fish may be greatly reduced under such a hypothetical reservoir release scenario, and hence a healthier stream can be maintained without affecting power generation requirement.

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Appendix A. Number of brown trout for each diet item in each percent diet composition category for four months and four reaches in the Smith River, Virginia, in 2002.

Month	Reach	Diet Item	Percent Diet Composition				
			0.00- 0.11	0.12- 24.99	25.00- 49.99	50.00- 74.99	75.00- 100
February	Dam	Ephemeroptera	3	5	3	4	5
February	Sp Regs	Ephemeroptera	0	4	2	5	9
February	Bassett	Ephemeroptera	1	1	2	2	13
February	Koehler	Ephemeroptera	2	1	3	4	10
May	Dam	Ephemeroptera	8	3	3	0	6
May	Sp Regs	Ephemeroptera	6	9	2	2	0
May	Bassett	Ephemeroptera	3	10	2	1	3
May	Koehler	Ephemeroptera	9	9	1	1	0
September	Dam	Ephemeroptera	13	7	0	0	0
September	Sp Regs	Ephemeroptera	4	16	0	0	0
September	Bassett	Ephemeroptera	11	9	0	0	0
September	Koehler	Ephemeroptera	10	7	2	0	1
December	Dam	Ephemeroptera	18	2	0	0	0
December	Sp Regs	Ephemeroptera	5	11	2	2	0
December	Bassett	Ephemeroptera	8	11	1	0	0
December	Koehler	Ephemeroptera	14	4	1	0	1
February	Dam	Plecoptera	14	5	0	0	1
February	Sp Regs	Plecoptera	8	8	3	1	0
February	Bassett	Plecoptera	12	4	3	0	0
February	Koehler	Plecoptera	10	4	4	2	0
May	Dam	Plecoptera	20	0	0	0	0
May	Sp Regs	Plecoptera	17	1	1	0	0
May	Bassett	Plecoptera	18	1	0	0	0
May	Koehler	Plecoptera	19	1	0	0	0
September	Dam	Plecoptera	19	1	0	0	0
September	Sp Regs	Plecoptera	9	10	0	1	0
September	Bassett	Plecoptera	8	9	2	1	0
September	Koehler	Plecoptera	16	3	0	0	1
December	Dam	Plecoptera	11	8	1	0	0
December	Sp Regs	Plecoptera	6	9	3	1	1
December	Bassett	Plecoptera	10	7	1	2	0
December	Koehler	Plecoptera	13	4	0	0	3
February	Dam	Trichoptera	18	1	1	0	0
February	Sp Regs	Trichoptera	12	6	1	0	1
February	Bassett	Trichoptera	10	7	2	0	0
February	Koehler	Trichoptera	10	8	2	0	0
May	Dam	Trichoptera	20	0	0	0	0
May	Sp Regs	Trichoptera	8	2	3	2	4
May	Bassett	Trichoptera	4	7	6	0	2

Appendix A. Continued.

Month	Reach	Diet Item	Percent Diet Composition				
			0.00- 0.11	0.12- 24.99	25.00- 49.99	50.00- 74.99	75.00- 100
May	Koehler	Trichoptera	8	6	2	2	2
September	Dam	Trichoptera	16	4	0	0	0
September	Sp Regs	Trichoptera	5	8	0	3	4
September	Bassett	Trichoptera	5	6	1	2	6
September	Koehler	Trichoptera	13	3	0	1	3
December	Dam	Trichoptera	17	1	2	0	0
December	Sp Regs	Trichoptera	4	11	3	0	2
December	Bassett	Trichoptera	6	9	2	2	1
December	Koehler	Trichoptera	11	4	1	2	2
February	Dam	Diptera	9	9	2	0	0
February	Sp Regs	Diptera	12	6	1	1	0
February	Bassett	Diptera	13	5	1	0	0
February	Koehler	Diptera	8	11	1	0	0
May	Dam	Diptera	8	9	2	0	1
May	Sp Regs	Diptera	8	10	1	0	0
May	Bassett	Diptera	9	7	2	1	0
May	Koehler	Diptera	7	8	2	3	0
September	Dam	Diptera	4	14	2	0	0
September	Sp Regs	Diptera	4	13	2	1	0
September	Bassett	Diptera	8	12	0	0	0
September	Koehler	Diptera	20	0	0	0	0
December	Dam	Diptera	4	13	2	1	0
December	Sp Regs	Diptera	11	9	0	0	0
December	Bassett	Diptera	15	4	0	0	1
December	Koehler	Diptera	15	4	1	0	0
February	Dam	Fish	20	0	0	0	0
February	Sp Regs	Fish	20	0	0	0	0
February	Bassett	Fish	19	0	0	0	0
February	Koehler	Fish	19	1	0	0	0
May	Dam	Fish	20	0	0	0	0
May	Sp Regs	Fish	19	0	0	0	0
May	Bassett	Fish	18	0	0	1	0
May	Koehler	Fish	13	4	0	2	1
September	Dam	Fish	20	0	0	0	0
September	Sp Regs	Fish	20	0	0	0	0
September	Bassett	Fish	18	1	0	0	1
September	Koehler	Fish	14	3	2	0	1
December	Dam	Fish	18	0	0	0	2
December	Sp Regs	Fish	19	1	0	0	0
December	Bassett	Fish	17	2	0	0	1
December	Koehler	Fish	12	0	2	3	3

Appendix A. Continued.

Month	Reach	Diet Item	Percent Diet Composition				
			0.00- 0.11	0.12- 24.99	25.00- 49.99	50.00- 74.99	75.00- 100
February	Dam	Decapoda	20	0	0	0	0
February	Sp Regs	Decapoda	19	0	0	1	0
February	Bassett	Decapoda	19	0	0	0	0
February	Koehler	Decapoda	20	0	0	0	0
May	Dam	Decapoda	20	0	0	0	0
May	Sp Regs	Decapoda	16	1	0	0	2
May	Bassett	Decapoda	16	2	0	0	1
May	Koehler	Decapoda	16	2	1	1	0
September	Dam	Decapoda	20	0	0	0	0
September	Sp Regs	Decapoda	17	2	0	0	1
September	Bassett	Decapoda	15	3	1	1	0
September	Koehler	Decapoda	13	3	3	0	1
December	Dam	Decapoda	20	0	0	0	0
December	Sp Regs	Decapoda	17	1	2	0	0
December	Bassett	Decapoda	11	4	3	2	0
December	Koehler	Decapoda	17	0	1	2	0
February	Dam	Isopoda	0	12	2	4	2
February	Sp Regs	Isopoda	14	6	0	0	0
February	Bassett	Isopoda	14	4	0	0	1
February	Koehler	Isopoda	18	2	0	0	0
May	Dam	Isopoda	6	4	3	5	2
May	Sp Regs	Isopoda	16	3	0	0	0
May	Bassett	Isopoda	18	1	0	0	0
May	Koehler	Isopoda	20	0	0	0	0
September	Dam	Isopoda	2	11	4	0	3
September	Sp Regs	Isopoda	20	0	0	0	0
September	Bassett	Isopoda	20	0	0	0	0
September	Koehler	Isopoda	20	0	0	0	0
December	Dam	Isopoda	1	8	1	7	3
December	Sp Regs	Isopoda	12	7	1	0	0
December	Bassett	Isopoda	15	5	0	0	0
December	Koehler	Isopoda	19	1	0	0	0
February	Dam	Gastropoda	16	3	1	0	0
February	Sp Regs	Gastropoda	15	5	0	0	0
February	Bassett	Gastropoda	18	1	0	0	0
February	Koehler	Gastropoda	18	2	0	0	0
May	Dam	Gastropoda	10	8	2	0	0
May	Sp Regs	Gastropoda	7	8	1	3	0
May	Bassett	Gastropoda	14	5	0	0	0
May	Koehler	Gastropoda	19	0	1	0	0

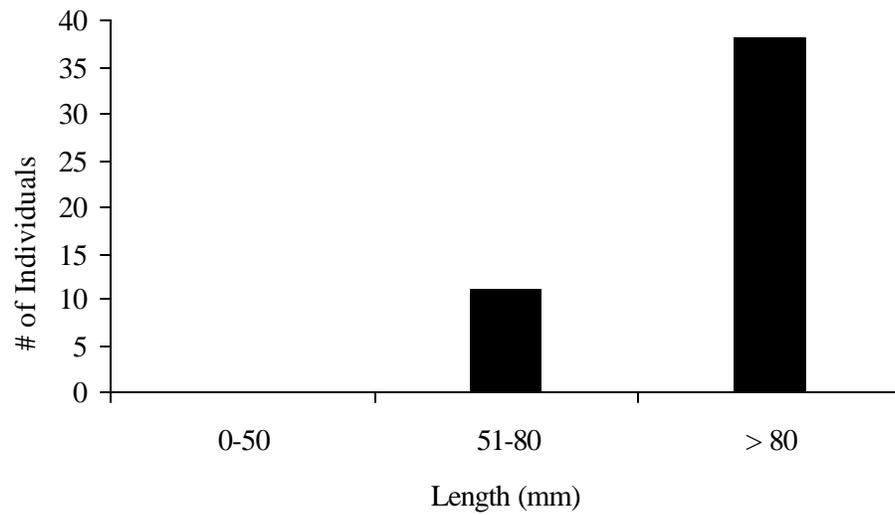
Appendix A. Continued.

Month	Reach	Diet Item	Percent Diet Composition				
			0.00- 0.11	0.12- 24.99	25.00- 49.99	50.00- 74.99	75.00- 100
September	Dam	Gastropoda	7	8	3	2	0
September	Sp Regs	Gastropoda	14	5	1	0	0
September	Bassett	Gastropoda	7	11	2	0	0
September	Koehler	Gastropoda	12	7	1	0	0
December	Dam	Gastropoda	6	9	3	1	1
December	Sp Regs	Gastropoda	11	6	1	2	0
December	Bassett	Gastropoda	7	6	3	1	3
December	Koehler	Gastropoda	9	6	3	1	1
February	Dam	Terrestrial	18	1	1	0	0
February	Sp Regs	Terrestrial	17	3	0	0	0
February	Bassett	Terrestrial	18	1	0	0	0
February	Koehler	Terrestrial	20	0	0	0	0
May	Dam	Terrestrial	16	4	0	0	0
May	Sp Regs	Terrestrial	16	3	0	0	0
May	Bassett	Terrestrial	9	6	2	2	0
May	Koehler	Terrestrial	10	3	5	1	1
September	Dam	Terrestrial	3	5	2	4	6
September	Sp Regs	Terrestrial	2	9	2	3	4
September	Bassett	Terrestrial	7	7	3	3	0
September	Koehler	Terrestrial	9	6	1	1	3
December	Dam	Terrestrial	19	1	0	0	0
December	Sp Regs	Terrestrial	20	0	0	0	0
December	Bassett	Terrestrial	20	0	0	0	0
December	Koehler	Terrestrial	20	0	0	0	0

Appendix B. Roanoke Logperch Data

Appendix A, Table 1. Roanoke Logperch, *Percina rex*, occurrence in 12 sites below Philpott Dam in the Smith River. Sampling periods include: June and October 2000, April, June, and October 2001, April, June, and October 2002.

Sampling Date	Location (KM below dam)	Number Caught
June 2000	13	2
June 2000	20.5	3
April 2001	8.9	1
April 2001	11.3	1
April 2001	15.3	1
April 2001	18.9	1
April 2001	20.5	1
April 2001	23	1
June 2001	20.5	3
June 2001	23	7
October 2001	18.9	2
October 2001	20.5	4
April 2002	8.9	1
April 2002	15.9	1
April 2002	18.9	2
April 2002	20.5	2
April 2002	23	1
June 2002	15.9	1
June 2002	18.9	3
June 2002	20.5	1
June 2002	23	1
October 2002	18.9	3
October 2002	20.5	2
October 2002	23	4



Appendix B, Figure 1. Roanoke Logperch, *Percina rex*, length frequency of all individuals caught during sampling periods June and October 2000, April, June, and October 2001, April, June, and October 2002. A length of greater than 80 mm represents an adult (Jenkins and Burkhead 1993).

Appendix C. Chlorophyll A

Chlorophyll a Substrate Sampling

I chose 20 sample sites corresponding with the 12 fish sampling sites and 8 additional sites, one above and below each of the 4 main tributary junctions. A sample of 5 rocks was taken from a riffle in each site the last week of both June and July 2002. Small cobble rocks were selected with a diameter of 100-150 mm from the middle of the channel. June and July represent a time of year when photoperiod is high and were chosen because I expected to sample the highest rate of primary production during this time of year. The rocks were covered with aluminum foil, placed on ice, and transported to a lab facility at Virginia Tech where they were kept in a freezer.

Chlorophyll a Extraction

The upper surface of the rock (surface not embedded but exposed to light) was scrubbed with a metal brush to remove any plant or algae growth. Distilled water was minimally used to wash the rock surface while scrubbing. A subsample (at least 10 ml) of this solution with known volume was filtered onto a glass fiber filter. To estimate the surface area, each rock was wrapped with aluminum foil (of known weight cm^2), the foil was trimmed to cover the upper surface of the rock, and the final foil hat was weighed. A single filter was prepared for each rock sample, labeled, and frozen. I was unable to use this technique for rocks with high levels of filamentous plant growth.

Filters were frozen at least 24 hours before inserting them into separate Falcon tubes each with 10 ml of basic acetone solution. The tubes were covered in aluminum foil to minimize exposure to light and kept refrigerated for at least 20 hours, but no more than 24 hours. Afterwards, the samples were centrifuged for 5 minutes at 1000 RPM before the final step in pigment analysis. I transferred 3 ml of each sample to a 1-cm cuvette and read optical densities at 750, 664, and 665 nm with a spectrophotometer. Then, 0.1 ml of 0.1 N HCl was mixed with each sample to acidify it, and after 90 s the same optical densities were read.

Chlorophyll a Calculation

$$\text{Chlorophyll } a \text{ (ug/cm}^2\text{)} = 26.7 (E_{664b} - E_{665a}) * \text{volume of acetone for extraction (ml)} / \text{rock area (cm}^2\text{)} * \text{length of path light through cuvette (cm)}$$

Where,

Rock area (cm^2) =

$$[\text{known area of foil (cm}^2\text{)} / \text{known weight of foil (g)}] * \text{weight of rock foil hat}$$

E_{664b} =

$$[\text{absorbance of sample at 664 nm} - \text{absorbance of sample at 750 nm}] \\ \text{before acidification}$$

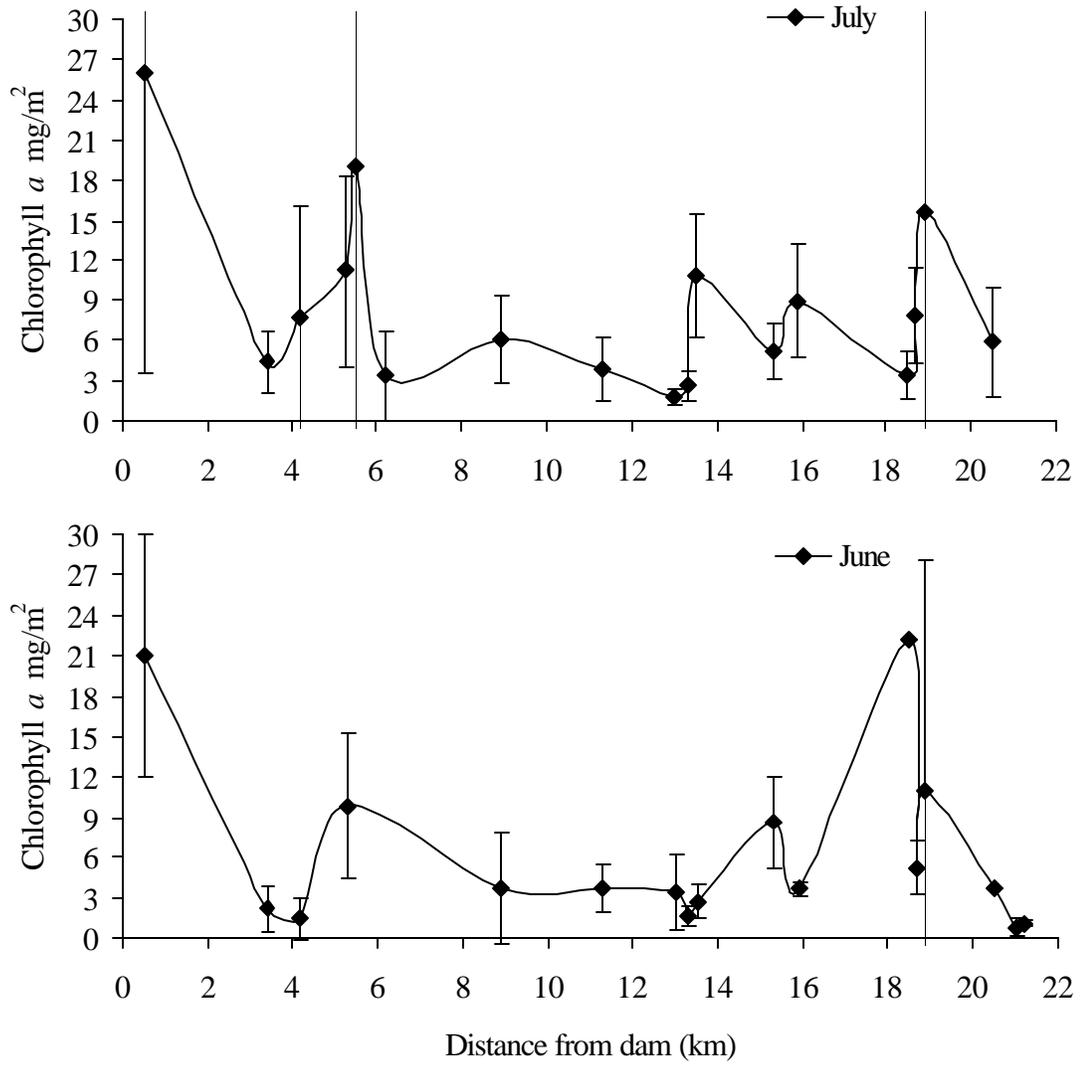
E_{665a} =

$$[\text{absorbance of sample at 664 nm} - \text{absorbance of sample at 750 nm}] \\ \text{after acidification}$$

Procedures to extract and calculate chlorophyll *a* were modified from Hauer and Lamberti (1996). Absorbance readings were converted into mg/m^2 of chlorophyll *a* present. Due to high quantities of filamentous plant growth on a portion of the rock samples, chlorophyll *a* was not extracted for some sites including: below Town Creek (5.5 km), site 4 (6.2 km), above Jordan Creek (21 km), below Jordan Creek (21.2), and site 12 (23 km) (Table A.1, Figure A.1).

Appendix C, Table 1. Chlorophyll *a* content for two sampling periods, June and July 2002, for 20 sites in the Smith River.

Site	Distance from dam (km)	Chlorophyll <i>a</i> (mg/m ³) June, 2002	Chlorophyll <i>a</i> (mg/m ³) July, 2002
1	0.5	21.04	25.99
2	3.4	2.16	4.39
3	4.2	1.43	7.63
Upstream Town Creek	5.3	9.86	11.16
Downstream Town Creek	5.5	-	19.05
4	6.2	-	3.33
5	8.9	3.67	6.04
6	11.3	3.67	3.81
7	13	3.36	1.77
Upstream Blackberry Creek	13.3	1.63	2.63
Downstream Blackberry Creek	13.5	2.71	10.86
8	15.3	8.64	5.18
9	15.9	3.61	8.93
Upstream Reed Creek	18.5	22.28	3.37
Downstream Reed Creek	18.7	5.20	7.86
10	18.9	11.06	15.70
11	20.5	3.70	5.88
Upstream Jordan Creek	21	0.80	-
Downstream Jordan Creek	21.2	1.07	-



Appendix C, Figure 1. Chlorophyll *a* content for sampling periods in June and July 2002 with distance from the dam (95% confidence intervals).

Appendix D. Nongame Species Relative Abundance Data Archive

Appendix C, Table 1. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during June 2000.

Species	Sites												Total
	1	2	3	4	5	6	7	8	9	10	11	12	
<u>Catostomidae</u>													
White sucker	2			32	1	14	10	34	42	1	6	101	243
Northern hogsucker				1		2			1	4			8
Roanoke hogsucker										2			2
Golden redhorse				12			3		1			3	19
V-lip redhorse													
Black jumprock						1	1			6	1		9
<u>Cyprinidae</u>													
Bluehead chub				6	1	12	40	47	7	84	34	35	266
Creek chub				2				2			4		8
Central stoneroller							25			7			32
Cutlips minnow						1						2	3
Rosyside dace							4	27		3	3		37
Crescent shiner				4			2	7	1	7			21
Mountain redbelly dace								9					9
Spottail shiner													
Swallowtail shiner													
Redlips shiner												1	1
Golden shiner													
White shiner													
Whitetail shiner										2			2
Rosefin shiner					1								1
<u>Centrarchidae</u>													
Largemouth bass													
Smallmouth bass							1	1				1	3
Redbreast sunfish							1			1	2		4
Green sunfish	2			2		1				3			8
Buegill	2			1			1			3			7
Roanoke bass													
Black crappie													
<u>Percidae</u>													
Fantail darter				63	9	30	58	62	6	46	55	3	332
Glassy darter													
Riverweed darter				1				4	6	1	5		17
Roanoke darter						2		3		20	28		53
Roanoke logperch							2				3		5
<u>Ictaluridae</u>													
Brown bullhead													
Margined madtom				4		1	1			3	12		21

Appendix D, Table 2. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during June 2001.

Species	Sites												Total
	1	2	3	4	5	6	7	8	9	10	11	12	
<u>Catostomidae</u>													
White sucker				7			37	2	1		7	22	76
Northern hogsucker							5				1	1	7
Roanoke hogsucker								2					2
Golden redhorse													
V-lip redhorse													
Black jumprock										1		1	2
<u>Cyprinidae</u>													
Bluehead chub				6	1	4	27	30	9	43	40	70	230
Creek chub			1					4					5
Central stoneroller							1					2	3
Cutlips minnow				1			4					1	6
Rosyside dace						5	6	44	5	15	5	5	85
Crescent shiner				1				6	1	2		2	12
Mountain redbelly dace								9					9
Spottail shiner						1	2	66	5	35	4	183	296
Swallowtail shiner				2								3	5
Redlips shiner													
Golden shiner													
White shiner													
Whitetail shiner										1			1
Rosefin shiner													
<u>Centrarchidae</u>													
Largemouth bass													
Smallmouth bass							1						1
Redbreast sunfish				2		1							3
Green sunfish	1	1	2	4						2			10
Buegill					1			2	2	2			7
Roanoke bass													
Black crappie													
<u>Percidae</u>													
Fantail darter				2	20	29	38	60	37	24	59	78	347
Glassy darter							1	1			4		6
Riverweed darter				1	2		5	16	4	1	18	1	48
Roanoke darter									2	2	27	11	42
Roanoke logperch											1	6	7
<u>Ictaluridae</u>													
Brown bullhead				1							1		2
Margined madtom				1			1		1		3	1	7

Appendix D, Table 3. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during June 2002.

Species	Sites												Total
	1	2	3	4	5	6	7	8	9	10	11	12	
<u>Catostomidae</u>													
White sucker	4			1		2	31	21	11	5	7	146	228
Northern hogsucker					1		6	1		8		2	18
Roanoke hogsucker						1							1
Golden redhorse								1				26	27
V-lip redhorse													
Black jumprock										2			2
<u>Cyprinidae</u>													
Bluehead chub				7		6	18	16	22	141	33	36	279
Creek chub				2						1			3
Central stoneroller								1		1			2
Cutlips minnow							3		1			4	8
Rosyside dace						1	26	13	3	18	1		62
Crescent shiner							1		1	3			5
Mountain redbelly dace				1			1	11					13
Spottail shiner				1			2	8	5	68	4	26	114
Swallowtail shiner				1									1
Redlips shiner													
Golden shiner													
White shiner				5			12			2			19
Whitetail shiner													
Rosefin shiner													
<u>Centrarchidae</u>													
Largemouth bass							1			1			2
Smallmouth bass					1								1
Redbreast sunfish													
Green sunfish		1	3	1									5
Buegill			1					1	3	6	2		13
Roanoke bass													
Black crappie													
<u>Percidae</u>													
Fantail darter				63	22	26	23	122	119	85	168	21	649
Glassy darter				3						2	2		7
Riverweed darter				4	1			12	16	9	5		47
Roanoke darter								1		31	40	2	74
Roanoke logperch								1		2			3
<u>Ictaluridae</u>													
Brown bullhead													
Margined madtom				1						5	2		8

Appendix D, Table 4. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during April 2001.

Species	Sites												Total
	1	2	3	4	5	6	7	8	9	10	11	12	
<u>Catostomidae</u>													
White sucker	5			7	5	8	20	13	13	17	19	44	150
Northern hogsucker						1	1	1		2	5		9
Roanoke hogsucker													
Golden redhorse	1			3			1		5		10	1	21
V-lip redhorse													
Black jumprock													
<u>Cyprinidae</u>													
Bluehead chub				15	3	2	26	22	7	26	25	25	151
Creek chub													
Central stoneroller			1			1						1	2
Cutlips minnow							2					1	2
Rosyside dace				2		1	8	1		3	6	3	24
Crescent shiner								1		2	1	1	4
Mountain redbelly dace				1	1		2	1			1		5
Spottail shiner				2						66	15	89	173
Swallowtail shiner													
Redlips shiner													
Golden shiner													
White shiner				1							1		3
Whitetail shiner					1		1						1
Rosefin shiner													
<u>Centrarchidae</u>													
Largemouth bass													
Smallmouth bass													
Redbreast sunfish													
Green sunfish							1	1	1	1		1	5
Buegill													
Roanoke bass													
Black crappie													
<u>Percidae</u>													
Fantail darter				78	7	31	31	34	21	4	4	19	227
Glassy darter													
Riverweed darter				5					1	1	1	1	9
Roanoke darter							1		1	2	1	4	9
Roanoke logperch													
<u>Ictaluridae</u>													
Brown bullhead													
Margined madtom				2			1	1		1	1		5

Appendix D, Table 5. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during April 2002.

Species	Sites												Total
	1	2	3	4	5	6	7	8	9	10	11	12	
<u>Catostomidae</u>													
White sucker	5	1		9	9	5	12	11	10	13	16	38	130
Northern hogsucker							2	2	1	3	1		7
Roanoke hogsucker										2			2
Golden redhorse							2		4		4	4	14
V-lip redhorse													
Black jumprock													
<u>Cyprinidae</u>													
Bluehead chub				5	5	5	6	5	3	17	7	9	63
Creek chub													
Central stoneroller						1						1	3
Cutlips minnow							1						1
Rosyside dace						4	6	1	4	3	1		18
Crescent shiner				2						1			2
Mountain redbelly dace				1			1					1	2
Spottail shiner							3		1	49	45	117	215
Swallowtail shiner													
Redlips shiner													
Golden shiner													
White shiner	1				1	1	2	1		5		3	13
Whitetail shiner													
Rosefin shiner													
<u>Centrarchidae</u>													
Largemouth bass													
Smallmouth bass													
Redbreast sunfish													
Green sunfish									1			1	1
Buegill										1			1
Roanoke bass													
Black crappie													
<u>Percidae</u>													
Fantail darter				26	10	14	23	10	31	6	9	2	132
Glassy darter											3		3
Riverweed darter				1	1	1			5	2	8	2	20
Roanoke darter											3	3	6
Roanoke logperch										1	1		1
<u>Ictaluridae</u>													
Brown bullhead													
Margined madtom						1							1

Appendix D, Table 6. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during October 2000.

Species	Sites												Total
	1	2	3	4	5	6	7	8	9	10	11	12	
<u>Catostomidae</u>													
White sucker	2	2	2	3		4	11	19	33	6	76	70	228
Northern hogsucker		1		3		3	1		2	10	4	3	27
Roanoke hogsucker													
Golden redhorse		2	3	7	2		3		1	1		2	21
V-lip redhorse													
Black jumprock													
<u>Cyprinidae</u>													
Bluehead chub				8	6	10	26	31	6	55	9	73	224
Creek chub								2					2
Central stoneroller		1		1			2	7		2		8	21
Cutlips minnow							1	1				1	3
Rosyside dace			1			4	5	22	1			16	49
Crescent shiner				3			2	4	1	3	1	3	17
Mountain redbelly dace							2	4	2			1	9
Spottail shiner											12	36	48
Swallowtail shiner													
Redlips shiner													
Golden shiner													
White shiner										1	1	1	3
Whitetail shiner		1		12	2		2						17
Rosefin shiner													
<u>Centrarchidae</u>													
Largemouth bass				1		1							2
Smallmouth bass				1						2	1	1	5
Redbreast sunfish												1	1
Green sunfish				1				1		1		1	4
Buegill			2	3						1	1	1	8
Roanoke bass													
Black crappie													
<u>Percidae</u>													
Fantail darter			1	13	15	10	4	11	25	6	3	29	117
Glassy darter										1	1	1	3
Riverweed darter								8	13	16	4	18	59
Roanoke darter								2	3	6		8	19
Roanoke logperch													
<u>Ictaluridae</u>													
Brown bullhead													
Margined madtom				1									1

Appendix D, Table 7. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during October 2001.

Species	Sites												Total
	1	2	3	4	5	6	7	8	9	10	11	12	
<u>Catostomidae</u>													
White sucker		1		1		1	3	6	8	9	16	4	49
Northern hogsucker							3			4	2	1	10
Roanoke hogsucker								1		3	1		5
Golden redhorse										3	5	1	8
V-lip redhorse													
Black jumprock													
<u>Cyprinidae</u>													
Bluehead chub				9	3	4	18	13	8	16	50	33	155
Creek chub					1			1	2	1	1		6
Central stoneroller					1	1		1		1	1	6	11
Cutlips minnow				1			3				1	2	7
Rosyside dace				1	3	1	7	8	4	13	12	1	51
Crescent shiner				11	1			2		6	3	8	32
Mountain redbelly dace				2	1			7	1		1		13
Spottail shiner					1		9	2	48	14	65	29	167
Swallowtail shiner				1									1
Redlips shiner													
Golden shiner													
White shiner							8	4		4	4		20
Whitetail shiner											6		6
Rosefin shiner													
<u>Centrarchidae</u>													
Largemouth bass											1	1	1
Smallmouth bass											1		1
Redbreast sunfish													
Green sunfish		1		1							5	1	8
Buegill									1	2	8		11
Roanoke bass													
Black crappie													
<u>Percidae</u>													
Fantail darter				26	9	22	13	20	78	12	70	19	268
Glassy darter							1	1		1	15		17
Riverweed darter				1		1	1	6	34	12	22	3	81
Roanoke darter									1	3	4	6	14
Roanoke logperch										1	1		2
<u>Ictaluridae</u>													
Brown bullhead													
Margined madtom				1			1			1			2

Appendix D, Table 8. Relative abundance (number of fish per 100 m) within sites 1-12 for individual species during October 2002.

Species	Sites												Total
	1	2	3	4	5	6	7	8	9	10	11	12	
<u>Catostomidae</u>													
White sucker	1			3		3	18	1	66	42	159	94	386
Northern hogsucker			1				1			18	4	9	32
Roanoke hogsucker							1			3	6	6	15
Golden redhorse												1	1
V-lip redhorse													
Black jumprock								3		2			4
<u>Cyprinidae</u>													
Bluehead chub		1			7	18	27	36	27	66	44	84	309
Creek chub							1	10	5		7		23
Central stoneroller						2	2	2	1	6	2	20	35
Cutlips minnow							3	1		10	12	9	34
Rosyside dace				2	11	13	24	2	7	6	23	5	92
Crescent shiner							2	2		3	3	15	24
Mountain redbelly dace						3	15	21	1				39
Spottail shiner					1		45	3	657	32	316	78	1132
Swallowtail shiner													
Redlips shiner													
Golden shiner													
White shiner						1	7	1		16	26	14	64
Whitetail shiner													
Rosefin shiner													
<u>Centrarchidae</u>													
Largemouth bass										1	14		16
Smallmouth bass													
Redbreast sunfish											1		1
Green sunfish	1		1	2									3
Buegill			1						3		1		5
Roanoke bass													
Black crappie													
<u>Percidae</u>													
Fantail darter				10	15	22	19	188	74	26	24	89	466
Glassy darter			1							7	4	1	13
Riverweed darter					1		1		24	125	76	53	280
Roanoke darter						1	3		1	34	34	49	121
Roanoke logperch										1	1	2	4
<u>Ictaluridae</u>													
Brown bullhead													
Margined madtom										1	4		6

Appendix E. Presentations and Publications by Project Researchers

1. Anderson, M. R., and A. K. Hunter. 2001. The Smith River tailwater: feast or famine? Roanoke Chapter of Trout Unlimited, September 19, Roanoke, Virginia.
2. Anderson, M.R., D.C. Novinger, D.J. Orth, T.J. Newcomb, A.K. Holloway, and C.A. Dolloff. 2001. Longitudinal Patterns in the Smith River, Virginia, Tailwater Fishery. Poster presentation delivered at 2001 Annual Meeting of the Southern Division American Fisheries Society, Jacksonville, FL.

Abstract: The Smith River tailwater below Philpott Dam provides an important trout fishery in Virginia. However, abiotic conditions, including flow fluctuations from hydroelectric power generation (30-1,400 cfs daily), consistently cold temperatures upstream ($<16^{\circ}\text{C}$), and rapidly fluxing downstream temperatures during summer hydropeaking (10°C decrease 60 min^{-1}) may be limiting the fisheries' potential. Brown trout in the tailwater rarely attain lengths desired by managers and anglers (>14 in). Our study, in cooperation with the Virginia Department of Game and Inland Fisheries, aims to assess spawning and rearing habitats for brown trout, determine limitations on brown trout growth and abundance, and describe distribution and abundance of nongame fishes. Initial results indicate longitudinal patterns with increasing distance downstream from Philpott Dam: brown trout redd density and brown trout abundance decrease, whereas brown trout size increase. Nongame species increase in abundance and diversity. Temporal patterns in abundance of age-0 brown trout suggest that downstream displacement occurs. Notable changes in substrate composition also occur with increasing distance downstream; the percentage of boulder and bedrock decreases, whereas the percentage of sand and silt increases. Results of the study will allow managers to recommend habitat and flow alterations to enhance brown trout production.

3. Anderson, M. R., C.W. Krause, and G. M. Buyoff. 2002. Fish Sampling in the Smith River, Virginia. Bassett Middle School, June 28, Bassett, Virginia.
4. 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.
5. 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 ($>356\text{ mm}$) 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.

6. 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.
7. Anderson, M. R., and C. W. Krause. 2003. Macro-invertebrate Mayhem; a Project Wet activity. Bassett River School program, Bassett Middle School, June 19th.
8. Anderson, M. R., D. J. Orth, and S. M. Smith. 2003. Historical change in the brown trout fishery in the Smith River Tailwater, Virginia. Proceedings of the Southeastern Association of Fish and Wildlife Agencies 57:150-159.

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.

9. 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.

10. Anderson, M. R. completed the Coosa Valley Chapter of Trout Unlimited progress report for the research grant on the Smith River brown trout diet study.
11. Black, A.T. and T.J. Newcomb. 2000. Availability of Forage for Brown Trout in the Smith River: Feast or Famine? Virginia Tech Minority Academic Opportunities Program, Summer (poster).

Abstract: The Smith River, below Philpott Dam in southwestern Virginia, supports a valuable and popular trout fishery valued at over \$400,000. The United States Army Corps of Engineers operates the dam as a hydropeaking facility by quickly releasing high flows of cold water. The cold water released from the dam provides good quality trout habitat. However, recent observations have indicated that the trout are slow growing, possibly because food is limited. Our objective was to measure longitudinal trends in the trout forage base in the Smith River below Philpott Dam. We selected twelve sites with well-developed riffles for collecting macroinvertebrates. Macroinvertebrates were collected with Surber samplers (929 cm²) at random locations in the upstream, middle, and downstream parts of the riffle. Each sample was weighed (wet weight mg) and identified to order with a taxonomic key and dissecting microscope. Ephemeroptera, Trichoptera, and Plecoptera, a food source commonly utilized by brown trout, were prevalent at all sites. The number of taxonomic orders in our samples increased at downstream sites and included; Diptera, Coleoptera, and Hemiptera. Simpson's Index of Diversity did not indicate large differences between the sites and values ranged between 0.6 and 0.8. Using an Analysis of Variance, we did not find significant differences ($p = 0.05$) in mean wet weight among individual sites. However, wet weights of the invertebrate samples generally decreased downstream. Although we found no statistically significant differences in invertebrate biomass or diversity, further identification to family may yield additional insights. The trend for decreasing invertebrate biomass downstream may be related to urban effects such as pollution, erosion, and sedimentation. We recommend further investigation of the availability of forage for brown trout in the Smith River.

12. Buhyoff, G.M., C.W. Krause, M.R. Anderson, and D.J. Orth. 2004. Managing a trout tailwater in the presence of a warmwater endangered species. Wild Trout VIII Symposium. Working Together to Ensure the Future of Wild Trout, 8 pp.
13. Crowder, D.C. and P. Diplas. 2002. Vorticity and circulation: spatial metrics for evaluating flow complexity in stream habitats, Canadian Journal of Fisheries and Aquatic Sciences, 59(4):633-645.

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

15. Hanna, K.M, T.J. Newcomb, and M.R. Anderson. 2001. Influence of Hydropeaking on the Abundance and Distribution of Invertebrates in the Smith River, Virginia. Virginia Academy of Science Meeting, 2001 (slide presentation).

Abstract: Benthic macroinvertebrates were sampled at 12 fish sampling stations in the tailwater of Philpott Dam in July 2000 and April 2001. A riffle within each site was stratified into top, middle, and bottom sections and surber samples were collected at two randomly selected locations within each section. Samples were preserved in 70% ethanol and returned to the lab for identification to family and measures of wet weight. Species richness, ANOVA, and linear regression were used to evaluate longitudinal trends with increasing distance from the dam and to determine significant differences between sites and years. Richness was low near the dam but increased at site 4.2 km and remained high downstream. Both wet weight and abundance were significantly greater in April than July and Ephemerellidae dominated the samples in April. Abundance of aquatic invertebrates in this tailwater was similar to abundances found in Appalachian streams where trout growth was limited. All sites, with the exception of sites 4, 8, and 12 in April, had lower densities than what is commonly found in trout streams in Virginia.

16. Hanna, K. A., T. J. Newcomb, and M. R. Anderson. 2001. Macroinvertebrate Forage for Brown Trout in the Smith River: Feast or Famine? Southeastern Association of Fish and Wildlife Agencies, Louisville, Kentucky.

Abstract: Benthic macroinvertebrates were sampled at 12 sites in the Smith River below Philpott Dam in July 2000 and April 2001. One riffle in each site was stratified into upstream, middle, and downstream transects and Surber samples were collected at two 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.

17. Hewitt, D.A. and T.J. Newcomb. 2000. An evaluation of brown trout spawning habitat in a hydropeaking tailwater. Virginia Academy of Science Meeting.

Abstract: Brown trout *Salmo trutta* provide an economically significant fishery in the Smith River tailwater below Philpott Dam, VA. Brown trout recruitment may be limited and could detrimentally impact the fishery. Using habitat suitability indices (HSIs), we calculated the amount of available spawning habitat in three sites below the dam. Sites were chosen based on relative abundances of age-0 brown trout because age-0 fish have been shown to be correlated with spawning areas. Microhabitat in the three sites was characterized by water depth, velocity, dominant substrate, and percent fines (substrate < 2 mm). High quality spawning habitat was limited, comprising no more than 6% of the total area of any of the sites. The most available spawning habitat was at the site closest to the dam. The least available spawning habitat was in site two, where the greatest abundance of age-0 brown trout were captured. Age-0 brown trout distribution was not correlated with high quality spawning areas. We suspect that daily releasing flows may affect their distribution by washing young fish downstream and affecting other important habitat features such as food availability and water quality.

18. Hunter, A. K. 2003. Fishing for trout, but what about those rough fish?, The Complex Nature of Managing a Tailwater. Fisheries 28(2):29. *Student essay 2nd place winner.* American Fisheries Society.
19. Hunter, A. K., and C. A. Dolloff. 2002. Longitudinal Patterns of Community Structure for Stream Fishes in a Virginia Tailwater. Southern Division American Fisheries Society Midyear Meeting, Little Rock, Arkansas.

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 hypolimnetic releases that create flows fluctuating from 30 to 1400 cfs. A primary objective of our research is to describe nongame species distribution, abundance, and diversity and to relate these patterns to environmental conditions. Preliminary results indicate that temperature regimes and tributaries influence nongame fish community patterns. Species distributions show a general trend of increasing abundance as distance increases from the dam and as temperature increases along the longitudinal gradient. However, peaks in fish abundance and diversity occur at tributary junctions. Tributary junctions may provide localized benefits to mainstem biotic communities, thereby increasing the likelihood that nongame species will persist in the Smith River tailwater. Understanding such mechanisms behind fish community structure in regulated rivers will improve efforts to manage streamflow and preserve aquatic life.

20. 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.

21. 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/>

22. 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.

23. Hunter, A. K., and A. C. Dolloff. Manuscript in review. Longitudinal Patterns of Stream Fishes in a Virginia Tailwater. Intended for 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.

24. Krause, C. W., T. J. Newcomb, and D. J. Orth. 2001. Ability of Three Models to Predict Water Temperature in an Unregulated Stream and Hydro-peaking River, 2001 Virginia Water Research Symposium, Charlottesville, Virginia.

Abstract: Stream temperature models can be used to predict thermal regimes following changes in watershed hydrology, land use, and riparian conditions. To produce accurate predictions and answer chosen management questions the appropriate model must be used. We evaluated the performance of three software packages that model stream temperature (SNTEMP, QUAL2E, and TVA River Modeling System) for use on two stream networks (a third-order stream and hydropeaking tailwater). We assessed model predictive ability, parameter sensitivity, data collection requirements, and user friendliness. Steady-state models, SNTEMP and QUAL2E, predicted similarly for both the third-order stream where daily flow was relatively constant, as well as for the hydropeaking tailwater where flow fluctuated daily. Though SNTEMP required more

collection of data than QUAL2E, SNTMP had fewer limitations, which makes it better for evaluating alternate shade and flow scenarios. The TVA model, a dynamic model, was better suited to model the rapidly changing flows of the tailwater by predicting hourly rather than daily temperature. The TVA model required the most intensive data collection therefore it was less efficient for use in the third-order stream. Each model had sensitive parameters, air temperature, relative humidity, and starting water temperature, which required accurate collection for optimal predictive ability. Consideration of stream type and modeling objectives are imperative factors for choosing a stream temperature modeling approach.

25. Krause, C. W., T. J. Newcomb, and D. J. Orth. 2001. Choosing the Appropriate Stream Temperature Prediction Model, American Fisheries Society 131st Annual Meeting, Phoenix, Arizona.

Abstract: Stream temperature models can be used to predict thermal regimes following changes in watershed hydrology, land use, and riparian conditions. To produce accurate predictions the appropriate model must be chosen. We evaluated the performance of three software packages that model stream temperature (SNTMP, QUAL2E, and TVA River Modeling System) for use on two stream networks (a third-order stream and hydropeaking tailwater). We assessed model predictive ability, parameter sensitivity, data collection requirements, and user friendliness. Steady-state models, SNTMP and QUAL2E, predicted better for the third-order stream where daily flow was relatively constant. Though SNTMP required more collection of data than QUAL2E, SNTMP predicted more days correctly and had fewer limitations, which makes it better for evaluating alternate shade and flow scenarios. The TVA model, a dynamic model, was better suited to model the rapidly changing flows of the tailwater. The TVA model required the most intensive data collection therefore it was less efficient for use in the third-order stream. Each model had sensitive parameters, air temperature, relative humidity, and starting water temperature, which required accurate collection for optimal predictive ability. Consideration of stream type and modeling objectives are imperative factors for choosing a stream temperature modeling approach.

26. Krause, C. W. 2002. Evaluation and Use of Stream Temperature Prediction Models for Instream Flow and Fish Habitat Management, Department of Fisheries and Wildlife Sciences Thesis Seminar, Virginia Tech.

Abstract: The SNTMP (U.S. Fish and Wildlife Service), QUAL2E (U.S. Environmental Protection Agency), and RQUAL (Tennessee Valley Authority) stream temperature prediction models were evaluated. All models had high predictive ability with the majority of predictions, >80% for Back Creek (Roanoke County, VA) and >90% for the Smith River tailwater (SRT) (Patrick County, VA), within 3°C of the measured water temperature. Sensitivity of model input parameters was found to differ between model, 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. All three models predicted well, therefore, selecting a model to assess alternative water management scenarios was based on model capabilities. The RQUAL model, used to

predict SRT temperatures under alternative hydropower release regimes, illustrated potential thermal habitat improvement for brown trout (*Salmo trutta*) compared to existing conditions. A 7-day/week morning 1 hr release was determined to best concurrently increase occurrence of brown trout optimal growth temperatures (+10.2% mean), decrease 21°C (state standard) exceedances (99% prevention), and decrease hourly changes in temperature (-1.6°C mean) compared to existing thermal conditions. The SNTTEMP model was used to assess thermal habitat under flow, shade, and channel width changes occurring from future urbanization within the Back Creek watershed. Predictions reveal that additional urban development could limit thermal habitat for present fish species by elevating summer mean daily temperature up to 1°C and cause 31°C (state standard) exceedances compared to existing conditions. Temperature impacts were lessened by single rather than cumulative changes suggesting mitigation measures may maintain suitable thermal habitat.

27. Krause, C. W., T. J. Newcomb, and D. J. Orth. 2002. Modeling Optimum Growth Temperatures for Trout in a Tailwater Fishery, 2002 Conference on Water Resources Planning and Management Proceedings, Roanoke, Virginia.

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. 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 15 alternative hydropower generation flows released from Philpott dam were assessed with a one-dimensional hydrodynamic model coupled with a water temperature model. Simulated temperatures at 13 river locations under each flow scenario were assessed for occurrence of optimal growth temperatures. Increased occurrence of optimal growth temperatures resulted from releasing water in the morning, decreasing the duration of release, and maintaining existing baseflow. A 7-day/week, morning, one hour release regime caused the greatest increase in occurrence of brown trout optimal growth temperatures compared to existing conditions.

28. 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.

29. 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 SNTMP, 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 SNTMP's ability to assess alternative shade levels.

30. Krause, C. W., and Y. Shen. 2003. Measuring Water-Velocity Profiles with Acoustic Doppler Technology in a Virginia Tailwater. Virginia Water Research Symposium, p.87-91. (available at www.vwrrc.vt.edu/pdf/2003%20symposium.pdf)

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.

31. Krause, C. W., T. J. Newcomb, and D. J. Orth. 2004 . 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.

32. 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

Abstract: Benthic macroinvertebrates were sampled at 12 fish sampling stations in the tailwater of Philpott Dam, Smith River, Virginia in July 2000 and April 2001. A riffle within each site was stratified into top, middle, and bottom sections and surber samples were collected at two randomly selected locations on each transect within each section. Insects were preserved in 70% ethanol and returned to the lab for identification to family and measures of wet weight. Species richness, analysis of variance, and linear regression were used to evaluate longitudinal trends with increasing distance from the dam and to determine significant differences between sites and years. We found that richness increased 4.2 km from the dam and remained high throughout the rest of the reach. Both wet weight and abundance was significantly higher in April than in July and Ephemerellidae proportionately dominated the samples in April. Abundance of aquatic invertebrates in this tailwater was similar to abundances found in Appalachian streams where trout growth was limited and all sites, with the exception of sites 4, 8, and 12 in April, had lower densities than what is commonly found in trout streams in Virginia.

33. Newcomb, T. J., C. W. Krause, and D. J. Orth. 2002. Evaluation of Alternative Flow Regimes in a Hydropeaking Tailwater: What to do when the fog doesn't lift until noon. Invited Presentation. National Instream Flow Council meeting. Linville, North Carolina.

Abstract: Southeastern tailwaters can provide high quality, unique trout angling experiences and they can be economically important to the communities that surround them. However, tailwater management operations can produce challenges to fish growth resulting from limited food resources, highly variable physical habitat, and quickly fluxing stream temperatures. We assessed the potential for thermal habitat improvement

for a naturalized brown trout population in a Virginia tailwater by use of 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 flow scenarios that released water in the morning and by decreasing the duration of the release. Maximum daily temperatures in the summer were decreased by changing the daily and diurnal schedule of releases and by increasing baseflow. Hourly temperature flux was decreased by increasing baseflows, releasing in the morning, and reducing the 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. It appears possible to improve conditions in this tailwater by altering the thermal regime through flow releases, however other biological limitations should be considered to prioritize alternative modes of enhancement. Would altering the thermal regime be worth the cost? 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.

34. Novinger, D.C. and D.J. Orth. 2000. Growth, abundance, and displacement of young-of-year brown trout related to hydropower generation and environmental conditions in a Virginia tailwater. Midwest Fish and Wildlife Conference. Minneapolis, MN Dec 2003.

Abstract: Young-of-year brown trout in the Smith River, Virginia, demonstrate intriguing patterns of growth and abundance correlated with environmental conditions and distance downstream of a hydroelectric dam. Releases for power generation increase discharge from 50 to 1400 cfs in 30 min on a daily basis. Corresponding upstream to downstream gradients in temperature and water chemistry correlate with trends in emergence dates, food availability, and growth. Consistently cool temperatures, low productivity, and streambed scour in upstream reaches result in later emergence dates, a paucity of invertebrate food sources, and slow growth rates for young-of-year trout. Trout in downstream reaches with warmer temperatures, higher productivity, and less scour have earlier emergence dates and high growth rates. A history of peaking flows has also created distinct trends in the availability of spawning and nursery habitats that affect abundance of young-of-the-year trout. Scour in upstream reaches has depleted gravel substrates necessary for spawning whereas deposition of fine sediments in downstream reaches may also limit spawning habitat. Sloughing of stream banks from fluctuating flows may result in a loss of low-velocity, lateral refuge habitat for trout during peak flow events. This could lead to high energetic costs, downstream displacement, and high mortality of young-of-year brown trout.

35. Novinger, D.C. 2001. Abundance, growth, and displacement of age-0 brown trout related to hydropower generation and environmental conditions in the Smith River tailwater. Annual Meeting of the Virginia Chapter American Fisheries Society. February, 2001. Luray, VA.

36. Novinger, D. C., and D. J. Orth. 2001. Spatial and Temporal Patterns in Spawning/Growth of Brown Trout in Relation to Environmental Conditions. American Fisheries Society 131st Annual Meeting, Phoenix, Arizona

Abstract: Brown trout demonstrate spatial and temporal patterns in spawning, abundance, and growth in relation to temperature and flow conditions below a hydropower dam in the Smith River, Virginia. Longitudinal trends in temperature and physical habitat correlated with spawning site location, spawning and emergence dates, and characteristics of redds. Most redd construction (78%) took place in the upper 3-8 km of the tailwater where spawning substrates were retained in bends in the river channel. Estimated median spawning dates occurred in early November downstream and late December upstream. However, mean temperatures during the week preceding median spawning dates varied within a tight range regardless of calendar date or distance downstream (mean=7.9°C, Se=0.2, n=19). We also found that redds were larger and located in faster water in downstream spawning sites. Earlier spawning and emergence of age-0 downstream allowed for a broader window for growth and attainment of larger sizes compared to age-0 upstream. This size difference is apparently maintained into maturity. Our conclusion is that temperature plays a strong role in determining the timing of spawning. Differences in spawning dates between upstream and downstream sites ultimately have important implications for survival and growth of age-0 brown trout in this tailwater.

37. Orth, D. J., M. R. Anderson, A. K. Hunter, C. W. Krause, and T. Smith. 2001. Summary of nongame species and life history in the Smith River. Martinsville Naturalists Club Meeting, November 27.
38. Orth, D. J. 2002. When will we start adaptive management? Update on the Smith River-Philpott dam tailwater study. Instream Flow Council Biennial Meeting, Crossnore, North Carolina. May 19-22.
39. 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.
40. 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.

41. Orth, D. J. 2003. Closing Remarks. International IFIM Users Workshop, Fort Collins, Colorado, June 2-5th.
42. Rummel, M. 2003. What Lies Beneath... The surface of the Smith River. Virginia Wildlife, June Issue.
43. 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).

44. 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.

45. 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.

46. 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.
47. 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).