Macroinvertebrate Forage in the Smith River Tailwater

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

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

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

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

Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies 55:116–125

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

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

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

The authors thank A. Black, M. Chan, and C. Krause for their assistance in collecting macroinvertebrate samples; the Biological Sciences Initiative and the Minority Academic Opportunities Program, both at Virginia Tech, for funding this work; the Department of Fisheries and Wildlife Sciences at Virginia Tech for providing laboratory space and equipment, and 4 anonymous reviewers for their comments on this manuscript.

Methods

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

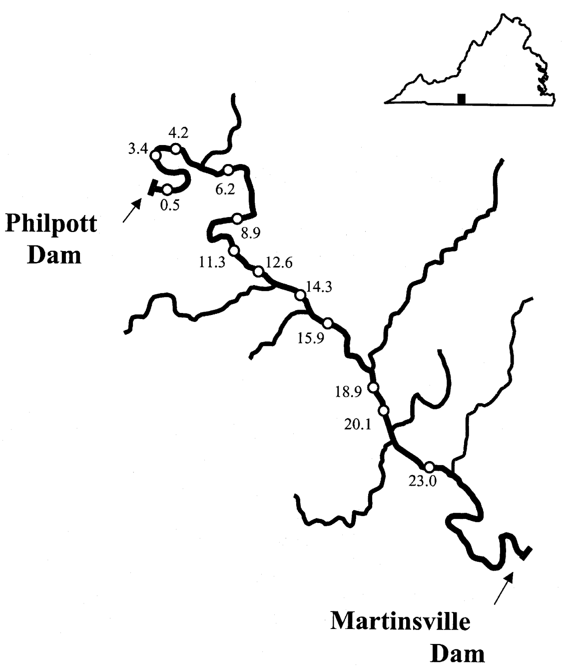

Figure 1. Location of sites (km below dam) for sampling aquatic macroinvertebrates below Philpott Dam in the Smith River, Virginia, in July 2000 and April 2001.

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

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

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

Results

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

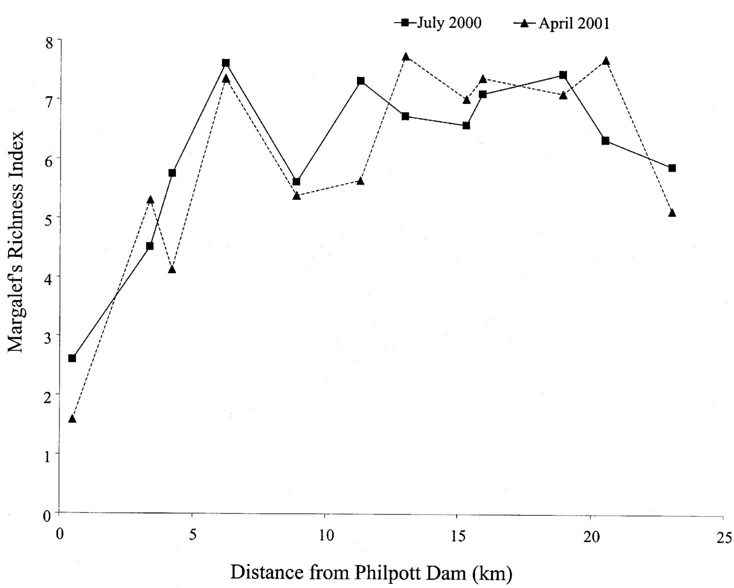

Figure 2. Margalef's Richness Index (based on family) of invertebrate taxa in the Smith River, Virginia, July 2000 and April 2001.

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

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

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

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

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

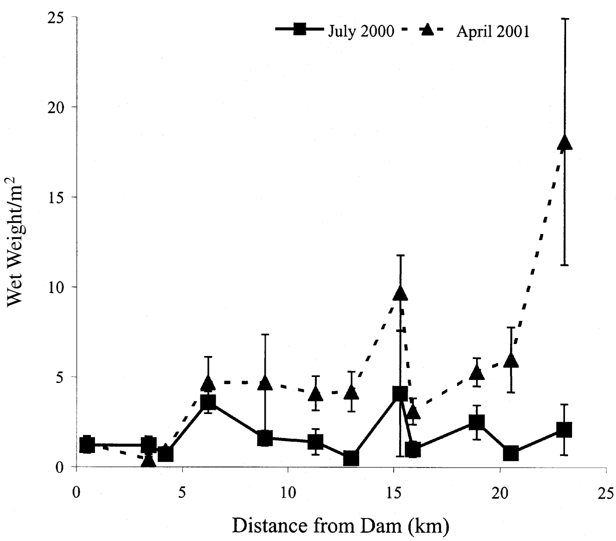

Figure 3. Biomass of aquatic macroinvertebrates (mean wet weight (g)/m^2 ± 1 SE) and distance from Philpott Dam at 12 sites in the Smith River, Virginia, in July 2000 and April 2001.

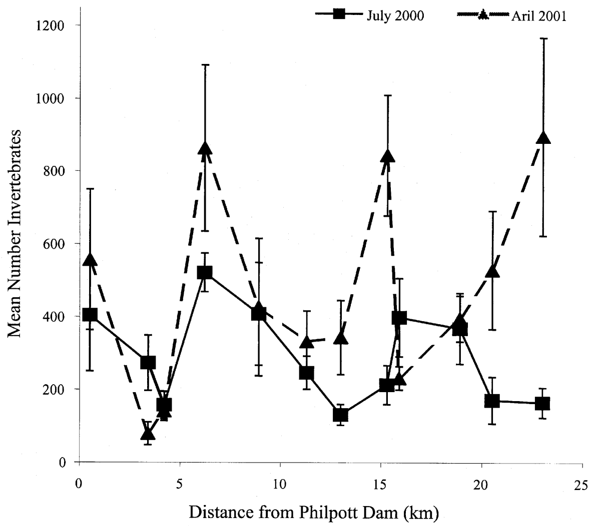

Figure 4. Mean number of invertebrates /m^2 (±1 SE) sampled at 12 sites below Philpott Dam, Smith River, Virginia, in July 2000 and April 2001.

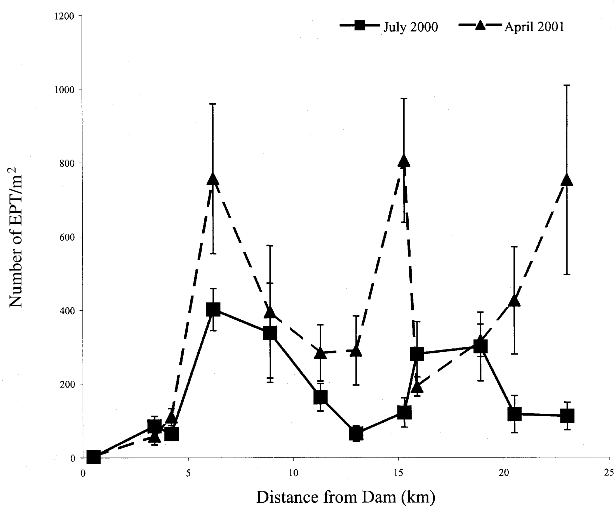

Figure 5. Mean number of Ephemeroptera/Plecoptera/Trichoptera/m^2 (± 1 SE) sampled at 12 sites below Philpott Dam, Smith River, Virginia, in July 2000 and April 2001.

Discussion

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

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

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

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

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

Literature Cited

Anderson, N. H. and J. B. Wallace. 1984. Habitat, life history, and behavioral adaptations of aquatic insects. Pages 38–58 in R. W. Merritt and K. W. Cummins, eds. An introduction to the aquatic insects of North America, 2nd ed. Kendall Hunt Publ. Co., Dubuque, Iowa.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: Periphyton, benthic macroinvertebrates and fish, 2nd ed. EPA 841-B-99-002. U.S. Environ. Protection Agency: Office of Water; Washington D.C.

Blinn, D. W., R. Tuitt, and A. Pickart. 1989. Response of epiphytic diatom communities from the tailwaters of Glen Canyon Dam, Arizona, to elevated water temperature. Regulated Rivers: Res. and Manage. 4:91–96.

______, J. P. Shannon, L. E. Stevens, and J. P. Carder. 1995. Consequences of fluctuating discharge for lotic communities. J. North Am. Benthol. Soc. 14:233–248.

Brower, J. E. and J. H. Zar. 1977. Field and laboratory methods for general ecology, 2nd ed. Wm. C. Brown Co. Publ., Dubuque, Iowa. 226pp.

Cada, G. F., J. M. Loar, and M. J. Sale. 1987. Evidence of food limitation of rainbow and brown trout in southern Appalachian soft-water streams. Trans. Am. Fish. Soc. 116:692–702.

Cummins, K. W. 1979. The natural stream ecosystem. Pages 7–24 *in* J. V. Ward and J. A. Stanford, eds. The ecology of regulated streams. Plenum Press, New York. 398pp.

Cushman, R. M. 1985. Review of ecological effects of rapidly varying flows downstream from hydroelectric facilities. North Am. J. Fish. Manage. 5:330–339.

Death, R. G. and M. J. Winterbourn. 1995. Diversity patterns in stream benthic invertebrate communities: The influence of habitat stability. Ecology 76:1446–1460.

Gislason, J. C. 1985. Aquatic insect abundance in a regulated stream under fluctuating and stable diel flow patterns. North Am. J. Fish. Manage. 5:39–46.

Hartwig, J. J. 1998. Recreational use, social and economic characteristics of the Smith River and Philpott reservoir fisheries, Virginia. M.S. Thesis, Va. Polytech. Inst. and State Univ., Blacksburg. 205pp.

Hawkins, C. P., J. N. Hogue, L. M. Decker, and J. W. Feminella. 1997. Channel morphology, water temperature, and assemblage structure of stream insects. J. North Am. Benthol. Soc. 16:728–749.

Krause, C. W. 2002. Evaluation and use of stream temperature prediction models for instream flow and fish habitat management. M.S. Thesis, Va. Polytech. Inst. and State Univ., Blacksburg. 144pp.

Lagler, K. F. 1956. Freshwater fishery biology. Wm. C. Brown Co. Publ., Dubuque, Iowa. 421pp.

Lauters, F., P. Lavender, P. Lim, C. Sabaton, and A. Belaud. 1996. Influence of hydropeaking on invertebrates and their relationship with fish feeding habits in a Pyrenean River. Regulated Rivers: Res. and Manage. 12:563–573.

Merritt, R. W. and K. W. Cummins. 1996. An introduction to the aquatic insects of North America, 3rd ed. Kendall/Hunt Publ. Co., Dubuque, Iowa. 722pp.

Moog, O. 1993. Quantification of daily peak hydropower effects on aquatic fauna and management to minimize environmental impacts. Regulated Rivers: Res. and Manage. 8:5–14.

Ney, J. J. 1996. Oligotrophication and its discontents: Effects of reduced nutrient loading on reservoir fisheries. Am. Fish. Soc. Symp 16:285–295.

Orth, D. J., T. J. Newcomb, P. Diplas, A. Dolloff. 2001. Influences of fluctuating releases on stream habitats for brown trout in the Smith River below Philpott Dam. Annu. Rep. Fed. Aid for Sport Fish. Contract No. 08220203. Va. Dep. Game and Inland Fish., Richmond. 109pp.

Sweeney, B. W. and R. L. Vannote. 1986. Growth and production of a stream stonefly: Influences of diet and temperature. Ecology 67:1396–1410.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37:130–137.

______ and B. W. Sweeney. 1980. Geographic analysis of thermal equilibria: A conceptual model for evaluating the effect of natural and modified thermal regimes on aquatic insect communities. Am. Nat. 115:667–695.

Voelz, N. J. and J. V. Ward. 1990. Macroinvertebrate responses along a complex regulated stream environmental gradient. Regulated Rivers: Res. and Manage. 5:365–374.

Ward, J. V. and J. A. Stanford. 1982. Thermal responses in the evolutionary ecology of aquatic insects. Annu. Rev. Entomol. 27:97–117.

Wieland and Hayward. 1997. Cause for the decline of large rainbow trout in a tailwater fishery: Too much putting or too much taking? Trans. Am. Fish. Soc. 126:758–773.