Virginia Polytechnic Institute and State University

Development and Application of an Energy Flow Model to Analyze Impacts of Navigation Changes on the Kanawha River in West Virginia

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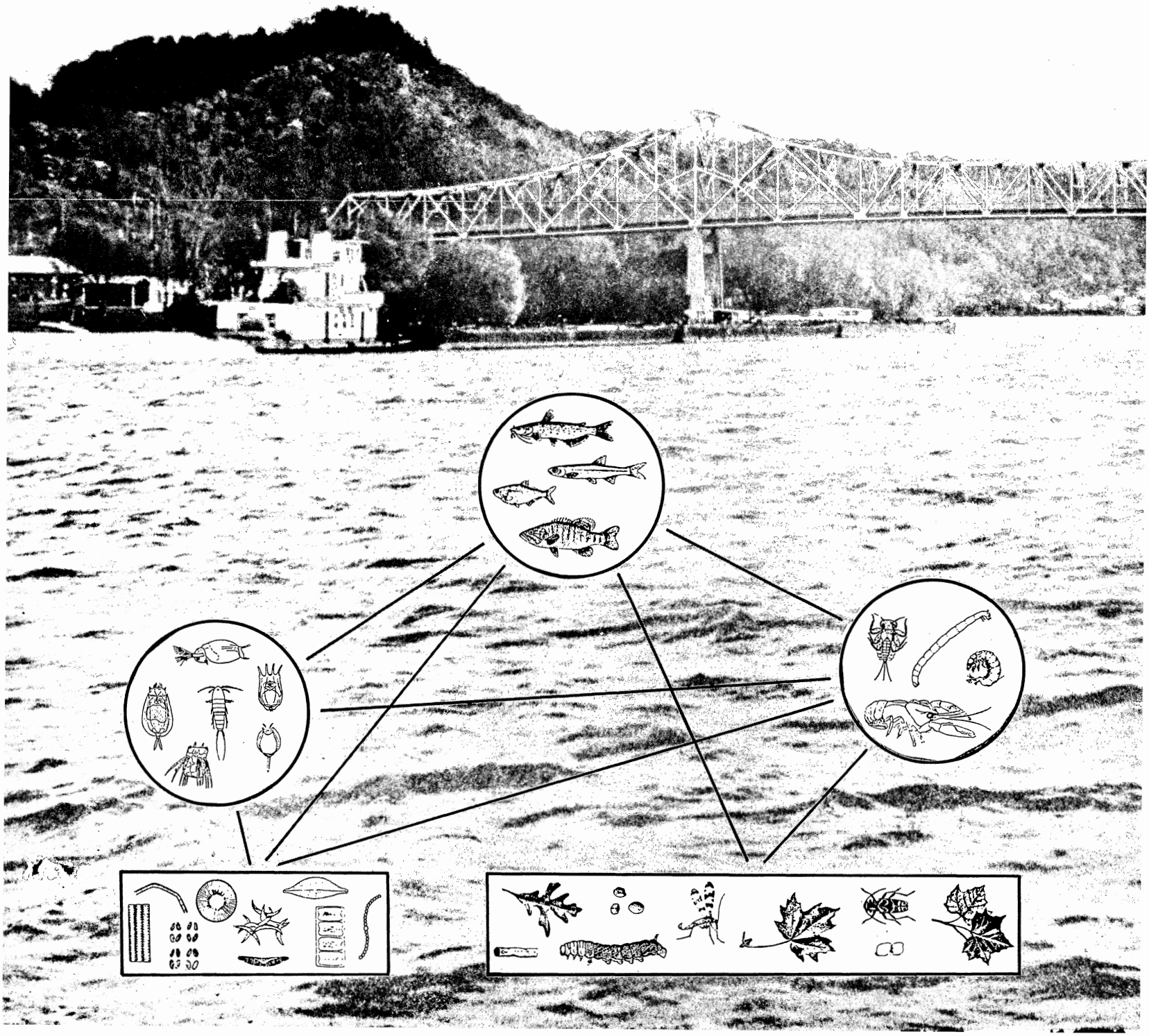
Huntington District



Development and Application of an Energy Flow Model to Analyze Impacts of Navigation Changes on the Kanawha River in West Virginia

Report Submitted to:

Huntington District
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by

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EXECUTIVE SUMMARY

A study was conducted by an interdisciplinary research group at Virginia Polytechnic Institute and State University to develop an energy flow model for the Winfield Pool of the Kanawha River. The model was used to analyze the possible impacts of an increased increment of tow traffic that might be brought about by improvements to the locks at Winfield. Data necessary to develop the model were collected for one year at two sites: Upper Winfield (UW) located at River Mile 66-67 and Lower Winfield (LW) located at River Mile 33-34. In the Winfield Pool the habitats and biota approximate a continuum from more riverine types at the upper end to more lacustrine at the lower end. There is always perceptible current in the upper reaches, whereas, there is slack water downstream during low flow conditions. The bottom is composed of cobble and pebbles at the upper end with sand at the lower end. The riverine section ("UW-like") was demarcated from the lacustrine section ("LW-like") at River Mile 57 for the purposes of the energy flow model. However, the Winfield Dam does not produce a habitat that is typical of a reservoir. Even in the lower reaches of the pool, the Kanawha continues to behave as a large river because there is little accumulation of organic detritus on the bottom and the sediments are primarily coarse materials.

The Kanawha River has suffered from very poor water quality; however, since the 1960's water quality appears to have improved. During 1982-1983, the water quality at the ends of the Winfield Pool did not present conditions that would be deleterious to aquatic biota. As the water quality has improved, the biological communities have become more normal for a river the size of the Kanawha. Phytoplankton diversity is reasonably good, but the density of organisms is somewhat low in comparison to other similar rivers. Although water quality has improved in the Kanawha River, the industries and municipalities throughout the valley still have some impact on the biota. Rotifer concentrations have increased markedly since the 1960's. The greater current and availability of coarse bottom sediments, lead to higher diversity and productivity of benthic macroinvertebrates at the upper end of the pool. The diversity and productivity of benthic macroinvertebrates is within the lower part of the range reported for similar rivers. The fish community of the Kanawha River appears typical of a large river, both in terms of total biomass per unit area and distribution among species and trophic groups.

The energy required for the respiration of the organisms in the Kanawha River exceeds the amount of energy that is captured by the photosynthetic activity of the algae, hence, the river functions as a heterotrophic system. The amount of energy that is available from allocthonous sources (leaves, terrestrial

insects, seston from upstream) into the Winfield Pool far exceeds the amount of energy that is produced autochthonously (phytoplankton, periphyton) within the pool. The assemblage of consumer organisms in the pool utilizes only a very small fraction of the energy that is available to them from the detritus passing through the system. Within the primary consumers (zooplankton, benthic macroinvertebrates), 0.2 - 45 % of the energy they consume is dependent upon primary producers, whereas 29-161 % of their consumption is dependent upon detrital components (> 100% possible because of materials cycling). Within most of the fishes only 2-30 % of their consumption is dependent upon primary producers, but 16-127 % is dependent upon consumption of detrital components. Heterotrophy and dependence upon energy from allocthonous detritus are typical of large rivers. Overall, the levels of productivity and the flow of supporting energy in the Kanawha River are what would be expected in the physical environment of a large river that has been altered by a series of low dams to facilitate commercial navigation.

Several different studies during the project did not document any direct adverse effects to selected faunal components from the passage of tows. There were no more damaged rotifers collected directly behind a passsing tow than in areas out of the sailing line. Benthic macroinvertebrates exhibit a different community structure in the sailing line in shallow water, but the

standing stock biomass is actually higher in the sailing line. Laboratory studies indicated that larval fishes can withstand very high levels of turbulence, although mortality increases significantly with the suspension of inorganic sediments. Field studies indicated that tow boats probably have little effect on larval fishes, because the larvae are most abundant in a narrow zone near shore where tows do not pass. All indications are that the fauna of the Kanawha River has become adjusted to the present levels of traffic, and an increased increment of traffic would have little effect.

The most likely impacts from an increased impact of tow traffic would be indirect trophic effects brought about by an increase in turbidity and an accompanying reduction in photosynthesis. The energy flow model was used to simulate several scenarios of projected levels of tow traffic in combination with the physical changes that were described for passing tows. In the worst case scenario (year 2050, 45 million tons, improvements at both Winfield and Marmet), assuming that all primary production is eliminated for 20 minutes after each passing tow, the annual average standing stock of primary producers (phytoplankton, periphyton) could be reduced by as much as 15%. However, because of the heterotrophic nature of the system and the dependence on allocthonous detritus for energy, the reduction in primary production would have little impact on consumer components of the ecosystem. Zooplankton and scraper -

grazer invertebrates that feed heavily on algae would only be reduced by < 6%, and other benthic macroinvertebrates would exhibit practically no reduction in annual average standing stock (< 1.9%) The reduction in primary production would cause negligible reductions in the annual average standing of the fish components (0.02 - 1.43 %). The energy flow model indicates that with any projected increase in tow traffic the Winfield Pool ecosystem would continue to function as it does with the present levels of traffic.

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PERSONNEL

This study was accomplished by an interdisciplinary research group at Virginia Polytechnic Institute and State University that included members of three departments (Biology, Entomology, Fisheries and Wildlife Sciences) in two colleges (Agriculture and Life Sciences, Arts and Sciences). The following 25 persons composed the "Kanawha River Team".

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I. INTRODUCTION

The Huntington District of the U.S. Army Corps of Engineers is considering making improvements to the Winfield Locks and Dam in order to make transportation on the Kanawha River more efficient. As part of the planning process, it is necessary to consider the environmental impacts that might result from an increased increment of tow traffic on the river.

Large rivers, such as the Kanawha, are perplexing to aquatic scientists because so little is known about them. The lack of knowledge about large rivers is due to the difficulty of sampling a diverse biota in a challenging physical environment as well as the difficulty in interpreting the data from such complex systems. Environmental impact studies, in general, have suffered from an inability to integrate biological information in a manner suitable for making decisions. The traditional method for environmental impact evaluations of construction projects on rivers involves developing lists of the biota and making sporadic measurements of physical-chemical properties, population densities, and community structure. Impacts are then predicted by determining how each individual component will be affected by the proposed action. Many impact statements of this type are inadequate because they focus on the species level or physical-chemical level. By themselves these measurements provide very little information, because decisions should be based on overall

ecosystem function. Therefore, environmental impacts can be better assessed by a holistic approach that quantifies the simultaneous effects on all components according to their interrelationships in the ecosystem. Energy flow modeling was chosen as the best method for accomplishing such a holistic analysis.

The overall objective of the project was to develop an energy flow model that describes the function of the Winfield Pool ecosystem under the present conditions and then to apply the model to predict the changes that would occur from an increased increment of traffic. The ecosystem boundaries were considered to be the shoreline of the Winfield pool and Winfield and Marmet dams. Energy flows were tracked from the allocthonous and autochthonous inputs to the pool, through 19 major components within the pool, and then accounted for as exports from the pool. The energy flow model was developed on the basis of extensive data collected during 1982-83, including: terrestrial inputs (leaves and insects), benthic detritus, organic seston, bacteria, primary production (phytoplankton and periphyton), zooplankton, benthic macroinvertebrates, and fish. For the secondary producers (zooplankton, benthic macroinvertebrates, fish) it was necessary to describe the functional composition of the biological communities, then measure the production of each component and the sources of energy accounting for the observed levels of production. An important advantage of the energetics

approach is that the traditional ecological data are obtained in the course of developing an energy flow model.

In order to apply the model to predict ecosystem level impacts, it was necessary to have information on the physical effects of tow passage and a description of the future fleet. The Huntington District conducted intensive studies at representative sites and retained a consultant to quantify the field of motion generated by tows in transit. The Huntington District conducted economic modeling to provide the ecosystem model with the necessary projections for density and frequency of tow passage. It was also necessary to know the biological effects of tow passage; therefore, experiments were conducted to determine the degree of mortality to zooplankton from entrainment in towboat propellers and any differences in the biomass of benthic macroinvertebrates that reside in the sailing line. It was determined that the major impact on the ecosystem would result from primary production being reduced or even eliminated by the increased turbidity which would persist for a period of time after tow passage. The energy flow model was then used to determine the cumulative impacts on the components of each trophic level, according to five scenarios for future traffic. The model output was the annual average standing stock for each ecosystem component under the conditions of each scenario. Environmental impacts were determined by calculating the difference in annual average standing stock between present traffic levels and those projected in each scenario.

Other ecological studies on water quality and entrainment of larval fishes were conducted, but the results of those studies were outside the scope of the energy flow model. In the following sections the previous knowledge of navigation impacts, Kanawha River biota, and modeling of large river ecosystems are reviewed, then the methods and results of the present study are reported and discussed to draw conclusions on the impacts of increased increments of tow traffic. The components of the model are treated individually first, then an integrative section on modeling and ecosystem impacts follows.

II. DESCRIPTION OF THE KANAWHA RIVER

II.0. HISTORICAL PERSPECTIVE

The Kanawha River flows approximately 117 river miles (188 km) through southern West Virginia. It is formed by the confluence of the Gauley and New Rivers at Gauley Bridge, West Virginia and flows to Point Pleasant, West Virginia where it joins the Ohio River (Fig. II.2.1.)

The Kanawha River was initially used for navigational purposes in its unimproved condition in 1774. Subsequently, the salt industry, centered just above what was to become Charleston, relied on the river for transport of its product on flatboats to the Ohio River in the early 1800's. However, prior to the construction of navigation aids, rapids hindered shipping. The steamboat, Robert Thompson, tried for two days to ascend a white-water stretch at Red-House Shoals, just above present-day Winfield, but failed to reach Charleston in 1819. A year later, a steamboat belonging to the owner of the Kanawha salt furnaces successfully negotiated the shoals and reached the Charleston area. The completion of a system of wing dams and chutes in 1828 provided a 3-foot navigational channel, and much of the Kanawha River became open to shipping. The steamboat Enterprise, circa 1830, pulled tows laden with salt over the still-treacherous shoals and signaled the beginning of a long history of barge traffic on the river.

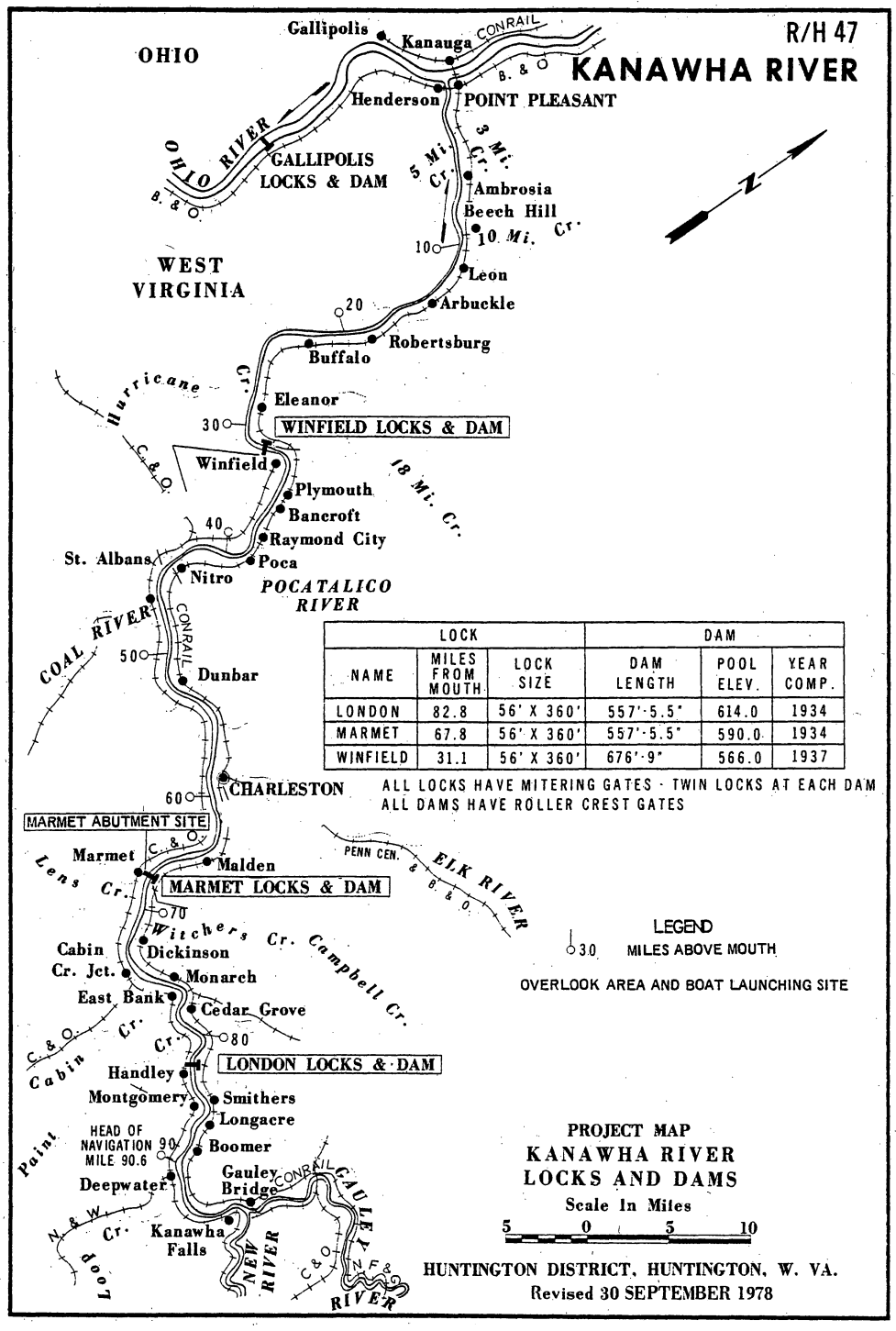

Fig. II.2.1. Map of the Kanawha River.

Further modifications of the Kanawha River were begun in the late 1800's to facilitate the downriver barge shipment of coal from the Kanawha coalfield. Also, it was envisioned that the Kanawha River would comprise a segment of the Central Water Line, that was proposed as a trans-Allegheny water route linking the Middle West with the Atlantic Coast at Hampton Roads. The completion of a system of 10 low-lift locks and dams in 1898, which created a 6-foot channel for 88.8 miles upstream from the mouth, came well after the idea of a trans-mountain canal had been abandoned. The current navigational system on the Kanawha River results from a replacement of the ten-structure low-lift system during the period 1931-1937 with four high-lift structures: Gallipolis Locks and Dam on the Ohio River below the mouth of the Kanawha, and Winfield, Marmet, and London Locks and Dams on the Kanawha River. This system creates a 9-foot navigable channel.

II.1. PRESENT NAVIGATION SYSTEM

Commercially navigable for 91 river miles (146 km) upstream from its confluence with the Ohio, the Kanawha is divided into five sections: (1) the Gallipolis Pool which extends from the Gallipolis Locks and Dam on the Ohio River to River Mile 31.1 on the Kanawha River; (2) the Winfield Pool which extends from River Mile 31.1 (Winfield Locks and Dam) to River Mile 67.7 (Marmet Locks and Dam); (3) the Marmet Pool which extends from River Mile

67.7 to River Mile 82.8 (London Locks and Dam); (4) the London Pool extending from River Mile 82.8 to the "head of navigation" at River Mile 90.6; and (5) the reach extending from the head of navigation to Gauley Bridge. The "head of navigation" represents the limit of channel maintenance by the U.S. Army Corps of Engineers and not a definite ecological boundary.

In addition to the influences of the three high-lift lock and dam structures, Kanawha River flows are also influenced by three large impoundments upstream on the New and Gauley Rivers. On the New River, there are Claytor Lake near Radford, Virginia, and Bluestone Lake near Hinton, West Virginia. Claytor Lake is used for hydroelectric power by the American Electric Power Company System while Bluestone Lake is operated by the U.S. Army Corps of Engineers as a flood control project. Summersville Lake is located on the Gauley River and is operated as a flood control project by the U.S. Army Corps of Engineers. Bluestone, Summersville, and Sutton (Elk River) are capable of providing consistent flow regimes in downstream reaches. Other Kanawha tributaries include the Elk River (River Mile 58), the Coal River (River Mile 45.5) and the Pocatelico River (River Mile 39).

Barge traffic is greatest below Charleston and gradually decreases upstream. Winfield Locks and Dam is the most heavily used of the three Kanawha navigation structures. Total annual barge traffic tonnage on the river increased from 4.5 million in 1940 to 15.3 million in 1980. Industrial expansion in the

Kanawha Valley, particularly the coal and chemical industries, has stimulated this growth of water-borne traffic.

The Winfield Locks and Dam structure consists of two chambers measuring 56 by 360 ft adjacent to the right descending bank, a non-navigable roller gate type dam with six 100 ft adjustable roller gates, and a privately operated hydro-electric plant located adjacent to the left descending bank. A lift of 28 ft is provided between normal pools by the Winfield Locks. When the Winfield Locks were built, the chambers allowed efficient passage of most tows because tows were composed of only two or three "standard" barges (175 x 26 ft, 1000 tons). With newer, more powerful towboats, new barges, called "jumbos" (195 x 35 ft, 1500 tons) came into regular use and barge strings increased in length. Most of the lock chambers on the Ohio River were rebuilt to measure 1200 x 110 ft to allow more efficient passage of these jumbo barges and longer tows. On the Kanawha River a tow may consist of up to ten standard barges or five jumbo barges. Whereas a tow with ten standard barges would require three lockings to be lifted between pools, a tow with five jumbo barges would require six lockings, one for each barge and one for the tow boat. Breaking apart a tow into sections which will fit into the smaller Kanawha chambers is not only time consuming, but in the case of chemical and petroleum cargoes may be dangerous.

Projections by Robert R. Nathan Associates, assuming the use of current operating practices, predict that traffic demands will

exceed physical lock capacity at Winfield Locks and Dam by about 1990. Predicted traffic demand at Winfield for the year 2040 is in excess of 150% of lockage capacity (U.S. Army Corps of Engineers 1983).

II.2. STUDY SITES

The Kanawha River is a 6th order stream with average discharge ranging from 12,560 cfs (106 years at Kanawha Falls) to 15,000 cfs (44 years at Charleston). Maximum discharges that have been recorded are 320,000 cfs on Sept. 14, 1878 at Kanawha Falls and 216,000 cfs on Aug. 15, 1940 at Charleston. There is a reasonable amount of riparian vegetation, consisting of a mixture of deciduous trees and shrubs. The riparian vegetation is sometimes restricted to a narrow zone because of commercial development in the valley. Only in the industrial section of Charleston is the river devoid of riparian vegetation because of solid concrete or rip-rap banks. There are no macrophytes except for a very few beds of water willow (Justicia americana) around some islands and near shore areas.

Over the 91 miles that are made navigable by locks and dams (Fig.II.2.1.) the Kanawha River is an ecological hybrid of a river and a reservoir. In each pool the habitats and biota approximate a continuum from more riverine types at the upper end to more lacustrine at the lower end (Voshell et al. 1982). In the Winfield Pool, the riverine section was demarcated from the

lacustrine section at River Mile 57 (11 miles below Marmet Dam) because of the predominance of sand instead of pebbles at that point (Huntington District Corps of Engineers, unpublished data). In order to develop an energy flow model for the Winfield Pool two study sites were chosen: one representative of the riverine section and the other representative of the lacustrine section. These were designated Upper Winfield (UW), River Mile 66-67, and Lower Winfield (LW), River Mile 33-34 (Fig. II.2.2). A third site, Upper Gallipolis (UG), River Mile 29-30, was also sampled, but less intensively to provide baseline data for the Gallipolis Pool. Specific locations within these study sites were used for sampling certain parameters, and these are described in detail in the Methods and Materials sections.

At UW, the Kanawha River was lotic in character, with perceptible current throughout the year. The river bottom was composed of cobble and pebble, interspersed with sand. The depth at mid-river during periods of low flow was approximately 3 m, but during periods of high flows the depth increased to at least 5 m. These increases could be very sudden due to releases of water from reservoirs upstream. A 2 m increase occurred overnight at UW in December 1983. A depth profile at River Mile 66.7 on July 25, 1984 indicated that depth varied from 3 - 4 m over most of the width. The right descending bank was much steeper than the left (45° and 20°, respectively). The width of the river at UW was about 170 m. During the period from Oct. 1,

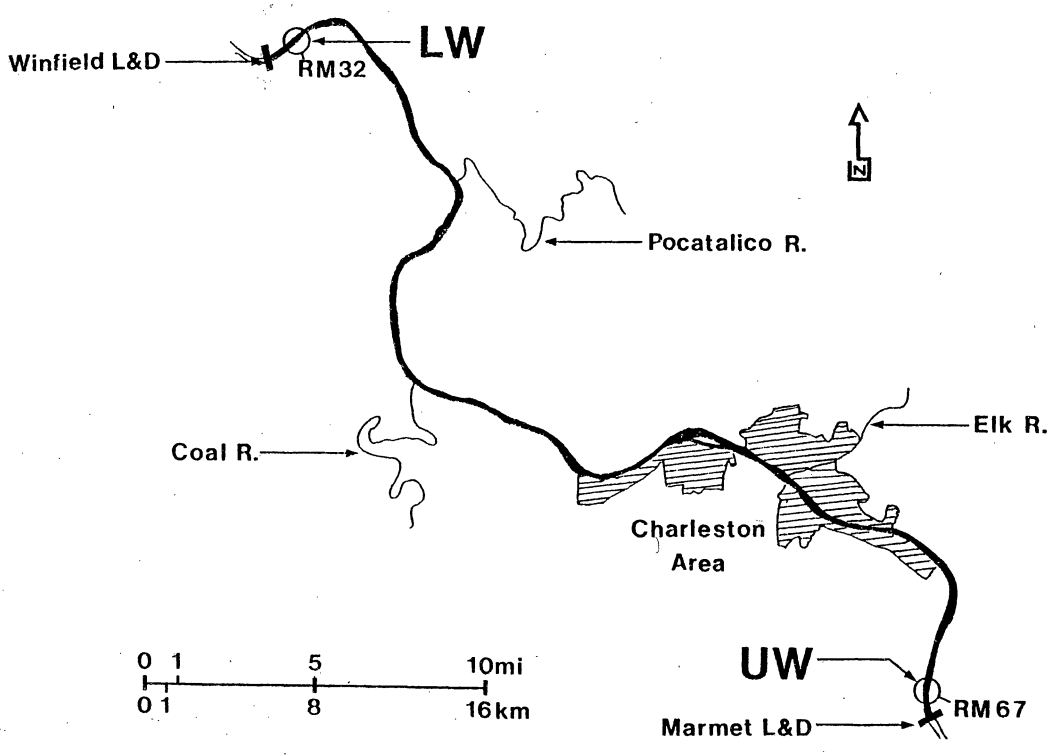

Fig. II.2.2. Map of Winfield Pool of Kanawha River illustrating the study sites. UW = Upper Winfield; LW = Lower Winfield.

1982 to Sept. 30, 1983 the average discharge was 13,050 cfs with a maximum of 81,200 cfs and minimum of 1,850 cfs (values from gauging station at Kanawha Falls). During 1982, 2,696 tows passed through this study site.

At LW, the Kanawha River was usually lentic in character, with barely perceptible current during periods of low discharge. The bottom consisted of finer materials, particularly sand. A depth profile at River Mile 31.7 on July 23, 1984 indicated that depth varied from 9 - 10 m over most of the river. Both banks were very steep (70° each). The width at LW was approximately 207 m. The average discharge during the study was 17,048 cfs with a maximum of 117,000 cfs and minimum of 1,918 cfs (sum of values from gauging stations on Kanawha River at Charleston and Coal River at Tornado). A total of 3,852 tows passed through this study site during 1982.

III. LITERATURE REVIEW

III.0 Effects of Navigation

It has been said that virtually every biological component of the river ecosystem is affected by physical and chemical changes caused by barge traffic (Upper Mississsippi River Basin Commission 1981). Conversely, it is held that navigation-use-related impacts are imperceptible when compared to changes in streambed, streambank, islands, bars, habitat structure, and biota that occur during storms and major flood flows (U.S. Army Corps of Engineers 1980). The sentiment of these two conflicting positions on the ecological effects of navigation can hardly be mistaken, yet research documenting said effects (or lack thereof) is nearly nonexistent. There is good reason for the paucity of information relating barge traffic to aquatic ecosystems. Ideally, this question would be examined directly through carefully controlled ecological investigations wherein navigational waters would be analyzed in the presence and absence of barge traffic over extended periods of time. Alternatively, the level of barge traffic activity could be systematically manipulated, thus making it possible to quantitatively relate ecological responses to traffic levels. The socio-economic hardships that would be imposed render such methodology infeasible.

Workers have had the opportunity to examine navigational waters during temporary cessations of barge traffic due to lock and dam repair (Sparks et al. 1980, Waldron 1979). However, physico-chemical and biological comparisons between conditions during the presence and absence of barge traffic were at least somewhat confounded by the effects of seasonality on river discharge and characteristics. Also, these studies were not of sufficient length to detect changes in biological communities that might occur over the long term.

Most efforts have been directed at examining the effects of single barge or recreational boat passage events. Aside from the difficulty in detecting what may be subtle changes in a hydraulically dynamic system, it is a very perplexing proposition, fraught with assumptions, to evaluate the effects of a single passage at some place in time and space on a river, and then extrapolate to those effects occurring along an entire segment of a navigation system. An additional difficulty arises from the fact that rivers lack uniformity along their length. The contemporary view holds that physical variables and biological communities should be regarded as continua from river headwaters to mouth (Vannote et al. 1980). Environmental perturbations associated with a single barge passage event, which may be insignificant and below thresholds of detection, may (or may not) become palpable when all barge passage events throughout the seasons and throughout a segment of river are considered in toto.

While there is little agreement about or solid information concerning navigation related biological or chemical impacts, there is at least qualitative agreement in regard to the physical effects of barge passage on rivers. Towboats and barges displace water during passage, and boat propellers transfer kinetic energy to the river. The effects of boat-induced wave action and turbulence on lotic habitats and biota have not been thoroughly examined. The increase in suspended solids concentration due to boat-induced wave action and turbulence is well documented, but the magnitude and persistence of increases in suspended solids is dependent on a number of variables. The ecological significance of elevations in suspended solids due to boat traffic has yet to be investigated in lotic systems. Due to the aforementioned information gaps, the following discourse outlines both realized and potential impacts of boat traffic on the biotic and abiotic components of lotic ecosystems.

Suspended Solids and Turbidity

One of the more obvious effects of boat passage is on sediment resuspension, and therefore on suspended solids concentration. Propeller rotation imparts vorticity to the water, and the dissipation of vorticity and other eddy effects lead to quasi-turbulent motion in the propeller wake. While this disturbance does not propagate outward as does the boat-generated surface wave system, locally high particle velocities are capable

of considerable sediment stirring, and therefore capable of resuspending solids (Gucinski 1982). Suspended solids can be either inorganic, such as sand or silt, or organic, such as detritus, plankton or other microscopic organisms (American Public Health Association et al. 1976). The potential impacts of suspended sediments on fish can be described as direct or indirect. At high concentrations, suspended sediments can be lethal. At lesser concentrations, suspended sediments can elicit physiological, anatomical, or behavioral responses. The resuspension of sediments can change sedimentation patterns, and sedimentation can directly impact fish by smothering eggs or indirectly by altering habitat. Suspended sediments can also affect water chemistry, the availability of nutrients, and the transport of toxic substances in aquatic systems.

The primary environmental concern regarding the resuspension of sediments is the optical quality that suspended solids impart to the water, known as turbidity. Increased turbidity is thought to result in decreased primary productivity (King and Ball 1964, Hart and Fuller 1972, Stefan et al. 1982). Ruttner (1974) stated that the decrease in light intensity across depth due to turbidity can be especially critical in flowing waters because the compensation point for photosynthesis can be reached at a shallow depth, and therefore plant life may not develop on the bottom of a stream.

Historically, the standard method for the determination of turbidity has been based on the Jackson candle turbidimeter which employs visual methods. The lowest turbidity that can be measured on this device is 25 units or JTUs. The nephelometric method (reported in nephelometric turbidity units or NTUs) is preferable to visual methods because of its greater precision, sensitivity, and applicability over a wide range of turbidities. However, the presence of "true color," that is the color of the water which is due to dissolved substances which absorb light, will cause measured turbidities to be low. Measurements from different types of instruments used to measure turbidity are not likely to show close agreement owing to differences in optical properties, and no instrument duplicates the results obtained with the Jackson candle turbidimeter (American Public Health Association et al. 1976).

According to Ruttner (1974), the extinction of light radiation in water is controlled by two factors: 1) absorption by water itself and by the substances dissolved in it, and 2) scattering by suspended solids. Ruttner refers to this second factor as turbidity. He comments that the opacity of turbid waters does not necessarily mean that they are not well-lighted. According to Ruttner's 1974 treatment, turbidity affects the path of light but does not affect the amount of light in water through absorption. The more universally accepted concept of turbidity holds that it is the optical property of water which causes light

to be scattered and absorbed rather than transmitted in straight lines (American Public Health Association et al. 1976). Sparks et al. (1980) reported that virtually all the turbidity in the Illinois River was attributable to suspended matter rather than to color from dissolved substances. In essence, dissolved substances did not reduce the transmission of light.

Although suspended solids concentration affects turbidity, efforts directed at correlating these two variables have resulted in mixed findings. Suspended solids are measured as the residue in a well-mixed sample of water which will not pass a standard glass fiber filter (American Public Health Association et al. 1976). It is believed that turbidity cannot be related to suspended solids concentration because of the effects of size, shape, and refractive index of the particles (American Public Health Association et al. 1976, Sorensen et al. 1977). Sorensen et al. (1977) believed that turbidity does give a good indication of the relative abundance of suspended material. Reed et al. (1983) found that turbidity did not correlate well with suspended solids concentration (r^2=0.24) due to the variability associated with dissolved solids. However, when the turbidity of the filtrate was subtracted from initial turbidity values, suspended solids accounted for 68% of the variability in turbidity.

It appears that suspended clay, perhaps because of the relative uniformity of the particles in size, shape, and refractive index, can be correlated well with turbidity. Sigler

et al. (1984) found turbidity significantly correlated with suspended clay filtered from water (NTU = 10.0 + 1.178 (mg/l); r^2=0.764) and with bentonite clay added to the water (NTU=5.49 + 0.162 (mg/l); r^2=0.926). Swenson (1978) found that suspended red clay sediments correlated well with turbidity (mg/l=0.1552 + 0.6089 (NTU); r^2=0.98). He also showed that turbidity could be significantly related to light transmittance: NTU=84.5413 - 1.5894 (T) + 0.0077 $(T)^2$; r^2=0.95 for light transmittance (T) readings less than 90%, and NTU=27.7976 - 0.2733 (T) for light transmittance readings between 90 and 100%. Reed et al. (1983) used the terms turbidity and suspended solids interchangeably. This appears to be somewhat inappropriate since turbidity is not always dependent on suspended solids concentration alone. However, for the sake of simplicity, usage of these terms herein will follow that of Reed et al. (1983) except where noted.

Sediment resuspension by boat passage is dependent on a number of factors. Water depth, sediment particle size and composition, and the energy imparted to the water by boat movement all play roles in the resuspension of sediments (Yousef 1974). The energy imparted to the water by boat movement is in turn dependent on boat design, weight, engine speed, type of propeller, and especially in lotic systems, direction of travel. Thereafter, the cross sectional shape of the channel will influence turbidity, as will the settling rate of the suspended material. Settling rates of solids are affected by water

currents, water density, and particle size, shape and composition (Reed et al. 1983). Local sediment concentration also affects settling time (Einstein 1972). Turbulent forces in river flow retard the deposition of suspended materials.

Sediment is quantitatively the greatest single pollutant in the nation's water (Ritchie 1972). Erosion in the watershed is normally the primary source of stream sediments. Much of the eroded material entering streams is due to agricultural and riparian land management practices. Construction activities in the catchment area can also contribute significantly to stream sediments. Some sediments are generated within the stream through erosion of the stream channel (Apmann and Otis 1955). Rivers with erodable beds tend toward an equilibrium between their sediment carrying capacity and the amount of sediment they carry; if the flow is overburdened with sediment, some is deposited, and if the sediment load is less than the flow is able to carry, the river seeks out additional material from the channel. Most streams ultimately tend to degrade, although they may go through agradation/degradation cycles (Apmann and Otis 1955). Fine materials usually do not accumulate in the river bed because of the flushing action of river flow. Impounded rivers tend to accumulate fine materials at faster rates than natural rivers because of the attenuating effects of dams on river flow velocities. Boat traffic may somewhat compensate for the decreased flushing capacity of impounded rivers through the

resuspension of sediments. Resuspended sediments would be transported through navigational waters by saltation. Nakai (1978) provided relationships between soil particle diameters and critical resuspension velocities.

Attempts have been made to document both temporary increases in suspended solids concentration over ambient due to boat passage, and long-term increases due to boat traffic (referred to by Sparks et al. 1980 as increases in "back-ground" or "ambient" turbidity levels). The latter effect would be related to the frequency of boat traffic and to the settling times of the resuspended sediments. The availability of sediments for resuspension would be affected by increases in background turbidity; if all the sediments available for resuspension are already in the water column (i.e., part of the background turbidity) then increases in boat traffic activity could not further affect turbidity levels. Following this line of reasoning, one would expect to see decrements in the rate of increase of suspended solids concentration due to increasing boat traffic levels.

It is possible that boat traffic may have some qualitative effect on sediments pertinent to increases in background turbidity. The energy imparted to the water by boats could affect the morphology of sedimentary particles. Sediments could become more finely divided due to mechanical abrasion, and settling times would become extended. Conceivably, sediments

might become incorporated into physical, chemical, or biological complexes, altering the likelihood of their resuspension. Frequent mixing of sediments by boat traffic could inhibit the formation of such complexes.

Sparks et al. (1980) studied the Illinois River in August and September of 1978 during the absence of commercial boat traffic due to the closure of three adjacent locks for repairs. They monitored the contributions of tributaries to the suspended solids concentrations in the river, and compared turbidity levels during the presence and absence of barge traffic. They found that both tributary discharge and tow traffic had separable, significant effects on turbidity. The return of tow traffic to the river increased suspended solids by an average of 19 mg/l. This represented an approximate 30-40% average increase in turbidity attributable to boat traffic at all sites. The effects of tow traffic on suspended solids were transmitted uniformly across all sites, including side channel stations. Sparks et al. (1980) reported that the passage of a single commercial boat and tow could raise the turbidity of the water by 100-200 JTU, and turbidity did not return to background levels for up to 2.5 hours. Davis and Eckenfelder (1971) found that although the passage of deep barges in a Texas estuary raised the suspended solids concentration in excess of 150 mg/l, background levels returned in 1 hour.

Water quality was examined in the Des Plaines River, Illinois, from July through October, 1978 (Waldron 1979). Barge traffic was suspended during August and September due to lock repairs. Total suspended solids were significantly greater in July and October (mean suspended solids 52.1 and 32.1 mg/l, respectively) than they were in August and September when no barges were present (24.4 and 22.9 mg/l). The differences in suspended solids concentration appeared to be chiefly due to changes in fixed suspended solids levels (inorganic solids). Fixed suspended solids were significantly less during August and September (13.9 and 16.7 mg/l) than they were when barges were present in July and October (44.0 and 22.1 mg/l). Turbidity levels did not follow the same pattern (19.9 and 26.5 NTU during August and September, and 42.2 and 22.6 NTU in July and October). The relatively high turbidity level during a month where suspended solids were relatively low appeared to be due to either the size, shape, or refractive index of particles in suspension in September.

Studies of individual barge passage events indicated that a number of factors influenced the resuspension of sediments in the Des Plains River (Waldron 1979). Little or no increases in suspended solids were noted in a majority of barge passage events. However, the majority of "full" barge passages (versus empty barges) resulted in increased suspended solids levels. Approximately 30% of the full barge passages resulted in a

doubling of suspended solids concentration, and in agreement with Davis and Eckenfelder (1971) suspended solids increased by more than 100 mg/l in a number of cases. About 40% of the full-upstream barge passages resulted in a doubling of suspended solids, whereas only 20% of the full-downstream passages in July and none in October resulted in a doubling of suspended solids.

Gucinski (1982) found that the effective depth of mixing (i.e., the maximum depth from which sediments can be resuspended) was increased when boat velocity relative to propeller rpm decreased. This supports the findings that suspended solids were increased the most be boats moving upstream with fully loaded tows (Waldron 1979); the increased weight of the tows and the opposing flow of water would decrease boat speed relative to propeller rpm. However, Gucinski (1982) found only modest increases in suspended solids (1.3-2.7 mg/l) in Chesapeake Bay waters at depths of 2.0-2.5 m following the passage of a 6.2 m runabout, powered by a 135 HP motor. Similarly, Moss (1977) reported that the effects of boats on turbidity in the Norfolk Broads and rivers of East Anglia, England, were minimal. Hilton and Phillips (1982) suggested that Moss' 1977 approach, which used simple correlative techniques, did not provide conclusive evidence. Hilton and Phillips (1982) modeled the effects of boat movement on turbidity in the River Ant, England. According to their calculations, turbidity generated by maximum boat activity (up to 100 boat passages per hour) returned to background levels

within 5.5 hours after the cessation of boat movement. They found that algae was the major source of turbidity on the River Ant, and boat-induced turbidity ranged from 8.2-44.0% of the total turbidity. This corresponded to a maximum increase in turbidity due to boat activity of about 10.0 NTU. Hilton and Phillips (1982) concluded that boating activity was not likely to lead to a long-term buildup of turbidity on the River Ant. Yousef (1974) reported that a 50 HP outboard motor raised the turbidity in shallow waters of Wekino Springs, Florida, from 2.3 JTU to levels of 4.5-6.5 JTU. The depth of effective mixing was found to be related to the horsepower of the motor. The effective mixing depths of 10 and 50 HP motors were 6 and 15 feet respectively. Boat induced turbidity did not persist as long as 1 hour.

Turbidity and Suspended Solids -- Primary Production

There are three general ways according to Farnworth et al. (1979) that sediment can impact primary productivity: (1) by decreasing light penetration, (2) by destroying photosynthetic organisms by abrasion, and (3) by changing species composition or relative abundance through changes in sediment type or composition. Although it seems intuitively obvious that increased turbidity from suspended sediment will decrease light penetration and therefore primary productivity, it must be remembered that the underlying assumption of this thesis is that

primary production is light limited. Field observations indicate that in many instances this view is probably too simplistic (see review by Farnworth et al. 1979). Essential plant nutrients are often limiting in aquatic ecosystems (Ruttner 1974), and it appears that the resuspension of sediments increases the availability of nutrients (Nakai 1978, Farnworth et al. 1979). Also, primary production may be inhibited at high light intensities; many algae appear to be adapted to or require low light conditions (McIntire and Phinney 1965, Tilzer et al. 1976).

Ruttner (1974) presented methods for calculating the percentile transmittance of light. The intensity of light at a given depth (I_h) can be calculated by:

$$I_h = I_o \cdot e^{Eh}$$

Where I_o = the light intensity at the surface,
e = the base of natural logarithm,
E = the light extinction coefficient, and
h = the length of the light path in the water column.

Gucinski (1982) discussed the relationship between light transmittance and photosynthetic rate. Reed et al. (1983) found that light transmittance was directly influenced by turbidity; each increase in NTU produced a 0.06 unit increase in the light extinction coefficient. Each 1.0 mg/l increase in suspended solids (calcium hydroxide and plankton) produced a 0.07 unit

increase in the extinction coefficient. This agreed well with the estimation by Stefan et al. (1982) of a 0.06 unit increase in the extinction coefficient for every 1.0 mg/l increase in suspended solids.

Direct evidence linking boat-induced increases in suspended solids to reductions in primary productivity is limited, but there is evidence that turbidity can influence primary productivity. King and Ball (1964) reported that a doubling of inorganic sediment load in a stream reduced primary production by about 54%. Persistent high levels of turbidity were thought to be responsible for the loss of macrophytes from the Patuxent River (Hart and Fuller 1972) and in Cedar Key, Florida (Strawn 1961). However, it is not clear whether these reductions in primary productivity or in the abundance of primary producers were due to decreases in light transmittance or to other factors such as those noted by Farnsworth et al. (1979). An analysis of data collected by Waldron (1979) on the Des Plains River showed that diel dissolved oxygen concentrations were indicative of a significant degree of primary productivity in two months when barges were absent from the river, but overall DO was lower, and late afternoon peaks in DO were diminished during two months of barge traffic activity.

Net plankton in the surface of farm ponds with less than 25 mg/l suspended solids was 8.0 and 12.8 times greater than those with 44-86 and 116-214 mg/l, respectively (Buck 1956). Buck did

not report the mesh size of the net used, but these values may only represent zooplankton densities, since he indicated that this was a measure of the abundance of microscopic fish food. It should be noted, however, that zooplankton density and primary production of phytoplankton are usually strongly linked, and thus it is germane that Williams (1966) showed reductions in zooplankton associated with increases in suspended solids in major U.S. waterways.

Suspended Solids -- Toxicity to Fish

Sorensen et al (1979), Farnworth et al. (1977), and Reed et al. (1983) reviewed the effects of suspended solids on fish. The direct toxicity of suspended solids to fish appears to be influenced by a number of factors, such as size, shape, concentration, and composition of particles, species, life stage, previous life history, and race of fish. Also, other environmental parameters seem to influence the toxicity of suspended solids. Wallen (1951) examined the acute toxicity of clay turbidity to 16 species of fish. Overt behavioral reactions did not appear until concentrations neared 20,000 mg/l. Most species endured concentrations as high as 100,000 mg/l, but 175,000-225,000 mg/l produced mortalities. Smith et al. (1965) found the toxicity of suspended wood fibers to be influenced by the type of wood fiber, processing method, DO, and water temperature. Suspended solids (natural sediment) at

concentrations of 1,000-2,500 mg/l were reported to be lethal in 20 days, and gypsum at a concentration of 4,250 mg/l produced 50% mortality in rainbow trout within 3.5 weeks (European Inland Fisheries Advisory Committee 1965). Also, spruce fibers at 200 mg/l produced 50% mortality in 16 weeks, and cellulose fibers at 1,000 mg/l elicited 20% mortality in rainbow trout in 37 days.

Toxic levels of suspended solids appear to cause gill disfunction (Neumann et al. 1982). Wallen (1951) reported gill damage to fish held at high suspended solids concentrations, and Herbert and Merkens (1961) found that 270-810 mg/l of diatomaceous earth elicited gill epithelial thickening in rainbow trout (Salmo giardneri). Ritchie (1972) felt that suspended sediments clog the gills, and that fish die from a combination of anoxemia and carbon dioxide retention. Heimstra et al. (1969) found that silt turbidities of 14-16 JTU increased the frequency of the cough reflex in largemouth bass (Micropterus salmoidea) and green sunfish (Lepomis cyanellus). They attributed the coughing behavior to attempts to free the gill lamellae of silt. Horkel and Pearson (1976) postulated that the observed increases in ventilation rate of green sunfish in response to bentonite clay suspensions (3,000-27,000 mg/l) compensated for reduced respiratory efficiency. Neumann et al. (1982) believed that this increase in ventilation rate may have been a short-term response to an acute stress, and that the increased oxygen carrying capacity of the blood in response to suspended solids reported by

other workers (increased hematocrit, erythrocyte count, and hemoglobin concentration) may be a long-term response to suspended particle stress.

Virtually no information is available concerning the toxicity of suspended solids to larval fish. Swenson and Matson (1976) reported that red clay turbidity over the ranges tested (6-46 NTU) was not lethal to larval lake herring (or ciscos), Coregonus artedii. Fish secrete mucus over the gill surface in response to stressful levels of suspended solids (Ritchie 1972). Everhart and Duchrow (1970) felt that because larval fish lack the ability to shed solids from their gills through mucous secretions, they would be more susceptible to the adverse effects of suspended solids. Many species of fish require extensive metamorphosis before larvae can achieve the juvenile stage. Prior to the development of functional gills, larvae may utilize a variety of respiratory surfaces (such as yolk sacs and highly vascularized fins or fin folds) to satisfy their gaseous exchange requirements (Bond 1979). It is not known whether the use of such respiratory surfaces render larval fish more or less susceptible to suspended solids toxicity.

Trautman (1957) summed up the anthropogenic influences on the fish fauna of Ohio during the period 1750 to 1950. He believed that the modification of aquatic ecosystems had drastically changed the fish fauna from species complexes requiring clear and/or vegetated waters to species tolerant of

turbidity and of bottoms composed of clay and silt. Trautman felt that there had been a shift from species with food value to smaller fishes or fishes of little food value. However, it appears that certain species are capable of some degree of adaptation to silty waters. Hubbs (1940) found that races of a given species from silty streams had smaller eyes, yet better developed acoustico-lateralis systems than races inhabiting clear waters.

There is a wealth of information documenting the deleterious effects of siltation on salmonid reproduction (see reviews by Cordone and Kelly 1961, Sorensen et al. 1977; Farnworth et al. 1979). A number of factors affect the survival of salmonid eggs and fry in gravel. Small particles (especially those less than 300 µm) reduce the flow of water through the redd. Water flow removes metabolic wastes and provides DO for developing embryos. Sedimentation can reduce the hydraulic exchange rate in the redd or directly smother eggs. In 1968, the Forest Service estimated that a clear stream with spawning gravel in Alaska was valued at about $131,000 per acre for salmon production, whereas a sediment-polluted stream was worth about $26,000 per acre (Ritchie 1972).

There is very little published information regarding the effects of suspended solids or sedimentation on the reproductive success of the so-called "coolwater" or "warmwater" fishes. However, Schubel et al. (1974) found that suspended fine-grained

Chesapeake Bay sediments did not affect hatching success or embryonic development of yellow perch (Perca flavescens) or striped bass (Morone saxatilis) eggs at concentrations as high as 500 mg/l; 1000 mg/l reduced hatching success. Longear sunfish (Lepomis megalotis) spawned over clean gravel substrates, but avoided areas with a thin white layer of limestone quarry sediment (Farnmworth et al. 1979). Schneider (1977) reported that circumstantial evidence indicated that poor recruitment of walleye (Stizostedion vitreum) in Lake Huron was primarily related to increased turbidity which resulted in the smothering of eggs. A sedimentation rate of 1.0 mm per day due to wave action and bank slumping resulted in mortality rates of northern pike eggs (Esox lucius) in excess of 97% (Hassler 1970). Siltation had much less of an effect once hatching occurred. Hassler (1970) suggested that year-class strength was related to the availability of suitable spawning habitat; weak year classes of pike were associated with low water levels during which only mud or rubble substrates were available for spawning.

In a survey of Oklahoma farm ponds, Buck (1956) found that young largemouth bass were not evident in ponds with greater than 84 mg/l, redear sunfish (Lepomis microlophus) in greater than 174 mg/l, and bluegill (Lepomis magalotis) in greater than 185 mg/l suspended solids. Buck (1956) also reported that the standing crops of fish in Oklahoma farm ponds was 1.7 greater in low turbidity ponds (less than 35 mg/l suspended solids) than in

ponds with intermediate levels of turbidity (25-100 mg/l suspended solids) and 5.5 times greater than in highly turbid ponds (100-388 mg/l suspended solids). Buck attributed these differences to better growth and reproduction is the less turbid ponds.

The tendency for fish larvae to be transported downstream is well established in the scientific literature (recently reviewed by Muth and Schmulbach 1984). These same workers estimated that in excess of 10 million fish larvae were recruited from the lower James River (South Dakota) to the Missouri River in one month. It appears likely that any changes in the reproductive success of fishes in the Winfield Pool are likely to be reflected in recruitment in the lower portion of the Kanawha River and the Ohio River.

Suspended Solids -- Behavior and Predator-Prey Relationships

The effects of turbidity on the behavior of fishes and predator-prey relationships are influenced by a variety of biotic and abiotic factors; consequences of, and responses to turbidity are likely to be dependent on species, life stage, fish species complex, type of forage, surface light conditions, and quantity and qualities of suspended solids. Waters with stressful levels of suspended solids are likely to be avoided by fish (Sigler et al. 1984). But, a common response to turbidity by larval, juvenile, and adult fishes appears to be vertical migration to surface waters (Netsch et al. 1971, Swensen and Matson 1976,

Swenson 1978, Matthews 1984). Swenson (1978) found that inflows of red clay turbidity in Lake Superior were responsible for the concentration of larval lake herring (Coregonus artedii) and zooplankton in surface waters. Thus, turbidity was believed to stimulate lake herring production by increasing the availability of high densities of food. However, a similar effect of turbidity on the behavior of subsequently introduced rainbow smelt (Osmerus esperlanus) was thought to lead to the decline of lake herring through predation on the larvae.

Matthews (1984) stated that increased turbidity can have an important effect on fish larvae with potentially important direct or indirect consequences for survival. Mortalities of larval fish are often associated with starvation because of their relatively limited energy reserves. Since most larval fish are sight feeders (Hunter 1981) and are handicapped by their size which limits them to small prey-search volumes (Johnston and Wildish 1982), interest has arisen regarding the influence of turbidity on the feeding of larval fishes. Matthews (1984) found that larval shad (Dorosoma cepedianum and D. petenense) responded to turbidity by seeking surface waters in a manner similar to that of larval ciscos (Swenson and Matson 1976, Swenson 1978). Matthews (1984) concluded that low prey availability, exacerbated by the concentrating effect of turbidity on the larvae, led to starvation, and that this phenomenon was responsible for the population dynamics of shad in Lake Texoma (Oklahoma-Texas).

However, red clay turbidity over the range of 0-48 NTU (0-28 mg/l) did not affect the survival and growth of larval ciscos in laboratory studies (Swenson and Matson 1976). The feeding rate of herring larvae (Clupea harengus) on brine shrimp (Artemia sp.) was significantly reduced at suspended natural sediment concentrations of 20 mg/l (Johnston and Wildish 1982). These workers empirically linked the suppressive effect of turbidity on feeding to the light-attenuating effects of suspended solids. They also found that turbidity-induced reductions in feeding were more pronounced in smaller larvae; this observation may indicate that the influence of turbidity on predator-prey relationships is related to the relative swimming abilities of predator and prey. Blaxter (1966) found the minimum amount of light necessary for herring larvae to capture Artemia sp. was 0.3 lux, whereas for faster-moving prey such as Balanus sp. nauplii, it was 13.0 lux. Gardner (1981) showed that the feeding rate of bluegill (Lepomis macrochirus) fingerlings on Daphnia pulex significantly decreased across turbidities ranging from 0-190 NTU. He believed that the influence of turbidity on feeding rate would have been greater with faster-moving prey that could have escaped by swimming out of the predator's field of vision. This view was supported by the findings of Moore and Moore (1981). They found that the fast-moving Crangon vulgaris escaped 55% of the time following encounters with flounder (Platichthys flexus) fingerlings in clear water and 100% of the time in turbid water. Moore and

Moore (1976) also showed that suspended fine sand (85-90 JTU) reduced the reactive distance to prey by 46%, and that the minimum detectable prey size increased in turbid water. Sigler et al. (1984) reported that clay turbidity at levels as low as 25 NTU significantly reduced the growth of steelhead (Salmo gairdneri) and coho (Oncorhynchus kisutch) salmon fingerlings.

Heimstra et al. (1969) reported that silt turbidities of 4-6 JTU reduced the activity of, and disturbed the social hierarchies of largemouth bass and green sunfish. However, the decrease in activity at higher turbidity may not reflect a deleterious effect. Matthews and Hill (1979) found in gradient tests that red shiners (Notropis lutrensis) selected for higher turbidities (90-155 JTU) in the laboratory environment, but selected clear waters in their natural environment. They believed that the selection for turbid waters in the laboratory reflected a "shelter-seeking" behavior. The higher activity levels reported for centrarchids in clear water by Heimstra et al. (1969) may have been due to "restlessness" arising from the lack of cover in a non-natural environment. Gradall and Swenson (1982) commented on the importance of, and use of turbidity as cover; they concluded that turbidity may serve to physically or visually isolate creek chubs (Semotilus atromaculatus) from predators, and that creek chub abundance can be expected to be high in turbid waters when other forms of cover do not provide adequate protection from predation.

Cerri (1983) examined the hypothesis that light intensity plays a major role in the outcome of predator-prey relationships. It is believed that prey risk increases at lower light levels in many such relationships, since schooling behaviors break down, and early detection of predators is lessened under low light conditions. Cerri concluded that prey risk is closely related to predatory activity, and that other factors play secondary roles. However, predatory activity seems to be closely related to light intensity since as noted by Lagler et al. (1962) many species exhibit peaks in feeding during periods of low light (dawn and dusk). Shafland (1974) argued that there should be an optimum level of turbidity for sight-feeding predators since the maximum visual acuity of most species is at rather low light intensities. It appears that since surface light intensity varies through the day, turbidity might affect the time of day that sight-feeders are most efficient. It remains to be seen how this might alter predatory activity, and thereby prey risk, but Swenson (1978) thought walleye (an active feeder under conditions of low illumination; Pfleiger 1975) developed a feeding pattern which positively correlated with low light conditions that periodically resulted from turbid inflows in Lake Superior.

Sediment Resuspension -- Nutrients and Toxic Substances

Much of the plant nutrients enter streams in particulate form. Phosphorus is normally the most limiting plant nutrient in aquatic systems, and it is thought that nearly all the phosphate

entering streams in agricultural areas is due to erosion (Hynes 1970). Sediments serve as a trap or sink for nutrients (Robinson 1971); nutrients can then be lost to the ecosystem through further sedimentation, or utilized by benthic organisms, or released to the overlying water through aerobic or anaerobic processes. Until recently, anaerobic release pathways were thought to be dominant, but now it is recognized that aerobic phosphorus release from sediments (which would probably be primary in lotic systems) is of the same order of magnitude (Bostrom et al. 1982). The immediate effect of sediment resuspension in lake waters is an increase in particulate-bound phosphorus. It appears that these particles can serve as a source of nutrients for planktonic algae, since studies have shown that algae are capable of extracting phosphorus from suspended sediment. However, it is possible that phosphorus availability is decreased by adsorption of dissolved inorganic phosphorus to suspended sediment particles. The dynamics of sedimentary adsorption and release of phosphorus are complex and dependent on a variety of factors (recently reviewed by Bostrom et al. 1982). Whether or not the resuspension of sediments results in an overall increase in biologically available phosphorus, as well as other nutrients, is likely to be case specific. Even when such increases have been found, corresponding increases in primary production have not always ensued (see review by Sorenson et al. 1977). The concomitant

attenuation of light transmittance due to the particles themselves has been believed to offset the increased availability of nutrients (Peterson 1981). Also the release of phosphorus from resuspended sediments in lotic systems is likely to be of less importance than in the lacustrine environment. Phosphorus release is pronouncedly enhanced by mechanical mixing (Bostrom et al. 1982), and turbulent river flow is likely to exhaust much of the phosphorus that is absorbed to, or complexed with sedimentary particles and available for release.

Toxic substances associated with sediments may be withdrawn from aquatic ecosystems through burial by further deposition or through in situ detoxification. Heavy metals are absorbed by, coprecipitated with, or complexed by suspended sediments, and thus are transported and deposited with the sediment load (Sorensen et al. 1977). Onishi and Wise (1982) felt that adsorption of pesticides to sediments may reduce their immediate biological availability, but that contaminated sediments become a long-term source of pollution through resuspension and desorption. Fish and invertebrates have been shown to accumulate significant amounts of PCB's from contaminated sediments, but none of the species tested accumulated zinc (Great Lakes Fishery Laboratory 1983). These findings point out the potential problems associated with the transport of contaminated sediments, but for the most part the subsequent release of, or incorporation of sediment-bound toxic substances into food webs is not well

documented (see reviews by Sorensen et al. 1977, Peterson 1981). Indeed it is arguable that the resuspension of sediments could remove contaminants from solution, and thereby possibly lessen their ecological impact. The reverse is possible also, as pointed out by Onishi and Wise (1982).

Barge traffic facilitates the downstream saltatory transport of contaminated sediments, and therefore decreases the withdrawal of toxic substances from the ecosystem through burial; but impoundment of the river for navigation would seem to favor sediment deposition. It cannot be said what hydrodynamic equilibrium in regard to sediment transport has been reached by these two opposing forces, nor do we know how it compares to the pristine condition. Also, it is debatable as to which way the equilibrium should be shifted to secure the greatest ecological benefits. Burial of contaminated sediments would inhibit detoxification mechanisms relying on sunlight or aerobic microbial pathways, and resuspension would presumably enhance the immediate bioavailability of toxic substances.

Turbidity -- Quality of Light and Water Temperature

The greatest effect of turbidity on light transmittance is that it acts as an opaque screen regardless of the color of light. Ellis (1936) measured the "millionth intensity depths" (the depth at which light was reduced to $1/10^6$ of its surface intensity) at 202 locations in streams across the United States and found that values ranged from 84-53,887 mm. The depth to

which light is absorbed and converted to heat is decreased as the number of particles in the water increases, and heating can become concentrated in the surface of turbid waters. Since particles absorb light and convert the energy to heat, water temperatures will tend to increase faster in turbid waters than in clear waters. However, at some concentration light will tend to be reflected out of the water, and increases in suspended solids will not result in further increases in water temperature (Reed et al. 1983).

Suspended solids have been found to preferentially absorb some wavelengths of light (Ellis 1936, Swenson 1978); blue light (shorter wavelengths) is absorbed more so than red light (longer wavelengths). Distilled water is most transparent to light in the violet-blue region of the spectrum (Ruttner 1974). Pacific salmon (Oncorhynchus) were able to feed at light levels of 1/300 that of moonlight (Brett and Groot 1963), yet Sigler et al. (1984) reported that turbidity levels as low as 25 NTU reduced the feeding of steelhead and coho salmon. Sigler et al. (1984) suggested that large suspended particles may intercept the wavelengths of light used by fish to see food, and that the effect of turbidity on the quality of light may be the factor reducing feeding.

Although red light is normally absorbed in distilled or natural water at depths of less than 1 meter (Ruttner 1974), suspended solids have been shown to further reduce the

transparency of water to wavelengths in this region of the spectrum (Buck 1956). Only 24.9% of the incident red light was visible at a depth of 4 inches at a suspended solids concentration of 25 mg/l, and at 50 mg/l only 6.3%. No light of any visible wavelength penetrated to a depth of three inches at a suspended solids concentration of 150 mg/l.

Turbidity -- Aesthetics and Angler Success

For many human uses (swimming, bathing, drinking, etc.), clear water is preferable, but for others (navigational, industrial, agricultural, etc.), turbidity may not be a consideration. One factor determining the preference of clear to turbid water for some uses lies in the realm of human visual perception; i.e., clear waters are generally considered to be more aesthetically pleasing than turbid waters (Sorenson et al. 1977, Sparks et al. 1980). Masteller et al. (1976) reviewed methodologies assessing aesthetic values of streams and landscapes, and concluded that aesthetic measuring techniques were generally inadequate.

Little quantified information is available indicating that water clarity directly impacts human uses such as fishing, but Buck (1956) implied that fewer anglers visited a turbid reservoir for this reason. Oschwald (1972) concluded that angler success for most species was inversely related to turbidity.

Water Quality Standards for Suspended Solids and Turbidity

The National Academy of Sciences, National Academy of Engineering (1974), recommended water quality criteria for the protection of aquatic communities. They concluded that concentrations of suspended solids of 25 mg/l afforded a high level of protection, 25-80 mg/l a moderate level, 80-400 mg/l a low level of protection, and concentrations in excess of 400 mg/l provided a very low level of protection. Alabaster and Lloyd (1980) stated that there is probably no sharply defined concentration of a particular solid above which a fishery is harmed, yet they believed than any increase in the normally prevailing suspended solid concentration above a low level may cause some decline in the value and status of a freshwater fishery, and that the risk of damage increases with concentration. Their conclusions regarding suspended solids concentration and the maintenance of freshwater fisheries were as follows:

(a) There is no evidence that suspended solids concentrations less than 25 mg/l have any harmful effects on fisheries.

(b) It should usually be possible to maintain good or moderate fisheries in waters which normally contain 25-80 mg/l suspended solids. Other factors being equal, however, the yield of fish from such waters might be somewhat lower than from those in category (a).

(c) Waters normally containing 80-400 mg/l suspended solids are unlikely to support good freshwater fisheries, although fisheries may sometimes be found at the lower concentrations in this range.

(d) At the best, only poor fisheries are likely to be found in waters which normally contain more than 400 mg/l suspended solids.

The U. S. Environmental Protection Agency (1976) recommended the following criterion for the protection of freshwater fish and aquatic life:

> Settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than ten percent from the seasonably established norm for aquatic life.

A summary of state standards (U.S. Environmental Protection Agency 1980) showed that thirteen states had not established water quality criteria for turbidity in waters similar to that of the Kanawha River. However, ten states required no visible or substantial increase in turbidity, or no increase resulting in any loss of beneficial use. Six states prohibited any unnatural increases in turbidity of 10-50 JTU (or NTU), and eleven states set maxima for turbidity of 10-50 JTU (or NTU). The remaining states either established guidelines for specific bodies of water, or varied permissible turbidity levels according to existing background levels. West Virginia has not established

regulations regarding turbidity in waters such as the Kanawha River (West Virginia Department of Natural Resources 1980).

Boat-induced Wave Action

Although most of the sediment that is resuspended by boat passage originates from the watershed, some portion may be generated through erosional forces exerted by the boats themselves. Any physical impact that the movement of a boat has on the environment is initiated by the transfer of kinetic energy from the boat engine. Energy is transmitted to the river by the boat propeller and through the displacement of water by the vessel as it travels. Both of these actions generate waves. Waves produce currents, shoreline drawdown, and hydrostatic pressure and water velocity changes. Physical impacts of boat-generated waves are dependent on the size and shape of the boat, boat speed and draft, water depth, location of the boat in relation to the shoreline, direction of travel, and the width of the channel (Schnick et al. 1982). A boat traveling fast in shallow water close to the shoreline generates the highest waves (Sorenson 1973). Bhowmik et al. (1981) found that the maximum wave heights generated by boats in the Illinois and Upper Mississippi rivers ranged from 0.1-1.05 ft. Recreational boats generated waves that were of greater amplitude but shorter wavelength than those due to commercial tows. Schnick et al. (1982) concluded that the wave heights and energies imparted to the water by both recreational and commercial boats were sufficient to cause erosion.

Lubinski et al. (1981) reported criteria developed by Hurst and Brebner to account for the proportion of shoreline erosion attributable to navigation on certain portions of the St. Clair and St. Lawrence rivers. The criteria were as follows:

(1) If the center of the navigation channel is 2000 ft or less from the bank, 50% or more of the bank erosion is due to navigation.

(2) If the center of the navigation channel is between 2000 and 3000 ft from the bank, less than 50% of the erosion is due to navigation.

(3) If the center of the navigation channel is more than 3000 ft from the bank, erosion is essentially due to natural causes.

Barges produce three successive effects as they pass a given point on shore. There is a slight rise in water level as the bow wave passes followed by a drawing away of water from the shore. The vertical drop in water level can be substantial (on the order of 1.5 ft), and if the shore has a shallow slope a considerable portion of the bottom may be exposed. Finally, the water rushes back in a series of waves as the stern of the towboat passes, and the water level may continue to oscillate for many minutes. These effects are most pronounced in the narrowest portions of the river and where shorelines are gently sloping (Schnick et al. 1982).

Studies on the Upper Mississippi River showed that barges traveling upstream doubled or tripled ambient water velocities for up to three minutes, and downstream barges reversed the flow of the river. Current velocities in backwaters were increased by 0.5-1.0 fps (Schnick et al 1982).

Boat traffic may affect sedimentation patterns. Sparks et al. (1980) believed that tow traffic increased the transport of sediment to backwaters, and that these areas may be lost to accretion. Schnick et al. (1982) reported that the rate of accumulation of fine sedimentary particles in the borders of navigation channels was greater after barge passage. They felt that tow passage can cause small sediment particles to be deposited along the banks since relative increases in water velocities decline as distance from the tow increases, and thus fine particles are transported farther than larger ones.

Wave Action -- Fish Reproduction

The negative effects of waves on reproductive success is well documented in the fisheries literature (Eipper 1975, Aggus 1979). Kramer and Smith (1962) concluded that the year-class strength of largemouth bass was established in the first two weeks after hatching, and the deleterious effects of wind-induced wave action on survival of embryos was the single most significant factor in establishing year-class strength. They determined that the extent of damage was affected by the location and depth of the nests, and by the velocity, direction, and

duration of waves. Largemouth bass nests as deep as 1.5 m can be destroyed by waves (Miller and Kramer 1971). Similar findings were reported for rainbow smelt, Osmerus mordax, (Rupp 1965) and yellow perch (Clady and Hutchinson 1975, Clady 1976). Von Geldern (1971) noted that brush protected nests from wave action, and therefore afforded greater reproductive success.

The energy imparted to navigation waters by boats may be negligible when compared to natural forces (U.S. Army Corps of Engineers 1980), but barge passage events occur a number of times each day, 365 days of the year. Thus, a primary dissimilarity between natural phenomena, such as storms and floods, and barge traffic is in the temporal distribution of impacts. As noted in the previous section, barge and recreational boat passage results in physical impacts such as shoreline drawdown, wave action, and hydrostatic pressure and water velocity changes. Spawning periods for most warm- and coolwater species are seasonal and usually extend for several weeks, yet embryonic development times, though dependent on water temperature, are often very short; e.g., about 4 days for largemouth bass (Miller and Kramer 1971). Thus, Aggus (1979) concluded that the possibility of a single storm decimating a year class is very small, and that frequent periods of high winds occurring throughout a spawning period would most often be required to significantly alter the reproductive success of a particular species.

Simulations of drawdowns due to navigation showed that walleye and northern pike (Esox lucius) eggs did not exhibit increased mortality at dewatering rates as high as once per hour, but mortality of larvae increased significantly when dewatered at frequencies higher than once every 6 hours (La Crosse National Fishery Research Laboratory 1984a). It appears that above some barge traffic level, the mortality of larval fishes due to dewatering may increase.

The percent of damaged eggs remained consistently higher for up to 90 minutes after passage of a barge in the Upper Mississippi River, and downstream passage of loaded barges had a greater effect than upstream passages of unloaded barges. Barge passage only rarely damaged larval fish, and did not appear to cause mortality among juveniles (La Crosse National Fishery Research Laboratory 1984b).

Holland and Sylvester (1983) concluded that increases in commercial traffic in Pool 7 of the Mississippi River can be expected to directly affect survival of fish larvae in the main channel and main-channel border more than in areas removed from the main channel, and that catches of larval fish were greater in the main channel border and surface waters of the main channel than in backwaters. However, it should be noted that factors associated with barge traffic which may impact larval fish have yet to be fully delineated, and this conclusion may be premature. For example, if dewatering due to barge passage is the most

significant lethal action of boat traffic, then larvae in shallow, low-slope backwaters may be at greatest risk. Holland and Sylvester (1983) also concluded that night passage of commercial barges may have greatest direct impact on larval fish communities, because more larvae were collected at dusk, dawn, and midnight than during the day. However, this conclusion suggests inadequacies in their sampling design. In that backwaters, main channel, and border areas were sampled, the question must be asked: where were the larvae that appeared at night during the day? The answer may lie in the fact that they sampled only the uppermost and lowermost meters of the 4-5 meter deep channel.

The potential impact of barge passage on fishes during their early life history is apparent. The available evidence suggest that the magnitude of the impact may be directly related to barge traffic level. It might be expected that increased early-stage mortality may be somewhat offset by compensatory survival and growth of remaining members of cohorts. How much mortality could be withstood by species of the Kanawha River before significant effects on populations are realized is not known.

Tow Passage -- Benthic Macroinvertebrates

During the last several years, there have been several reviews of potential biological impacts from tow traffic (Academy of Natural Sciences of Philadelphia 1980, Wright 1982), but there have been no direct studies of actual impacts. Chambers et al.

(1983) studied aquatic insect drift in relation to tow passage, and Beckett and Miller (1982) investigated macroinvertebrate colonization on artificial substrates in the Ohio River in relation to navigation structures. This review is based primarily on the work by the Academy of Natural Sciences of Philadelphia (1980) and Wright (1982).

The effects of navigation on a large river can be divided into two categories: those which are associated directly with the passage of a tow boat and those which are associated with the operation of a navigation system. In both categories there are physical, chemical and biological effects. Because no comprehensive environmental impact studies were conducted on large rivers before the introduction of commercial navigation, it is impossible to describe the pre-navigaton biological communities. It is also hard to separate long-term tow traffic effects from the effects of industrial and municipal pollution which are common in large, navigated rivers.

Tow passage effects that are directly associated with the passage of a vessel are primarily the result of the turbulence produced by the propeller. The amount of turbulence is related to the horsepower of the engine, diameter of the propeller, whether or not the propeller is open or enclosed, the direction of travel (upstream or downstream), and the depth of the river. The turbulence from a passing tow may affect organisms which are drifting in the water column as well as those on the bottom. The

Academy of Natural Sciences of Philadelphia (1980) suggested that tow induced turbulence may be severe enough to injure drifting invertebrates. No comprehensive drift studies have been conducted in the Kanawha River, so the magnitude of this problem is difficult to estimate. Larval fish have been shown to be tolerant of high levels of turbulence; however, their tolerance decreases significantly with the addition of suspended sediments (this report Section V. 10.) In the case of drifting invertebrates those high in the water column may not be hurt by the turbulence created by a passing tow, while those closer to the bottom where sediments may be thrown into suspension are more likely to be affected.

Turbulence could also directly affect macroinvertebrates on the river bottom in several ways. In areas of extremely high turbulence, direct mortality could occur from abrasion by sediment particles. In areas of less turbulence, benthic organisms might be inconvenienced by habitat displacement, which could result in lower production rather than mortality. Aquatic insects might be induced to drift downstream (Chambers et al. 1983) or seek refugia which are more protected from the currents. This could be in response to an increase of suspended solids (Gammon 1970), physical dislodgement from the substrate (Academy of Natural Sciences of Philadelphia 1980), or the movement of net-spinning caddisflies after their nets have been clogged by increased suspended sediments or burst by excessive current

(Academy of Natural Sciences of Philadelphia 1980). The organisms in the Ohio River which were thought to be most likely to be dislodged from the substrate by a passing tow were Tricorythodes spp., Caenis spp., Stenonema spp. and the Odonata. The Academy of Natural Sciences of Philadelphia estimated that effects would not be significant at depths greater than 6.7 m, but that fauna residing in shallower water would be affected. Many areas of the Kanawha River, especially towards the upper reaches of the navigational pools, are less than 6.7 m deep, and organisms in these locations could be adversely affected. The magnitude of impact is also related to the duration of the increased turbulence. The Academy of Natural Sciences of Philadelphia (1980) reported that the duration of the effects caused by a passing tow may vary between 4 and 34 minutes in the Ohio River, depending on the river depth and particles disturbed. It was thought that most disturbances, however, would last less than 15 minutes.

During July-August 1983, a study was conducted by the Huntington District Corps of Engineers (Hochstein and Adams 1985) in the Kanawha River to see how much energy from a passing tow reaches the bottom. In a 6 m deep river section it was found that tows could possibly change ambient flow conditions on the river bottom for several minutes. The length of these disturbances generally were thought to be less than 11 minutes each (Hochstein and Adams 1985).

A tow which is moving downstream during low flow conditions can push water upstream, a phenomenon called backflow. How an aquatic insect might respond to backflow is not known, although a drift response is highly probable. Filter-feeding caddisflies build their nets in areas of constant, unidirectional flow. Current reversals and turbulence could destroy the nets. Any action by the aquatic macroinvertebrate to avoid the turbulence or rebuild burst nets would be an unnatural energy drain and would result in lower production.

A third negative effect of tow passage is the formation of standing waves. These waves are caused by both the passage of the bow of the tow and the wake and may be especially important in very near shore areas (< .5 m). Two different impacts may occur. First, an alternating watering and dewatering of substrates may occur within the wave affected area. A second impact would be the scouring of organisms present with sediment suspended by the orbital motion of the waves.

It is conceivable that tow traffic could have beneficial effects for some aquatic macroinvertebrates. If cobble/pebble substrates are swept clean of fine sediments by turbulence reaching the bottom, this would assure more habitat for macroinvertebrates requiring firm substrates. Organisms which require currents for feeding or respiration could inhabit near-shore areas which receive waves from passing tows much the way that rheophilic organisms inhabit the wave swept shores of lakes.

Turbulence may also have other indirect effects on macroinvertebrates such as altering the available food. As a tow passes, it has the potential to suspend sediments in the water column. The Academy of Natural Sciences of Philadelphia (1980) attempted to predict the loss of photosynthesis which might occur in the Ohio River because of increased navigation. They indicate that at center channel the loss might be as great as 6 to 17% during the summer months, while total productivity might only decrease 1% when the entire cross sectional area is taken into account. A reduction in the amount of primary productivity would affect collector-filterers and scrapers which depend upon phytoplankton and periphyton production for energy. A lesser effect might also be felt by collector-gatherers because the amount of detritus resulting from dead algae cells would be decreased. Organisms which live on the river bottom may also be affected by the turbulence as organic materials are suspended in the water column. During periods of low flow these suspended materials could be utilized by collector-filterers. However, at the same time firm substrates could be swept clean of detritus, removing a food source for collector-gatherers.

III.1 EFFECTS OF IMPOUNDMENT FOR NAVIGATION

The locks and dams of the Kanawha River are operated to maintain constant water depth for navigation. The lock and dam system converts the river from a free-flowing condition to a

series of four impoundments or pools. Aside from the historically dominant effects of water pollution, the Kanawha River probably has been altered from its pristine state to its current condition as much by the construction of the lock and dam system as any other factor.

No pre-impoundment physical, chemical, or biological data from the Kanawha River are available. However, generalizations about the effects of the lock and dam system can be inferred from information garnered from other river systems. One of the long term effects of locks and dams on the Upper Mississippi River has been to increase the river width directly upstream from a lock and dam and to slightly narrow the river immediately downstream (Simons et al. 1981). In general, locks and dams increase and stabilize river depths. Degradation of the river channel will occur immediately below dams. Aggradation of the channel in the lower reaches of the pool will result in shallower water depths. Simons et al. (1981) reported that degradation occurs below locks and dams in the Upper Mississippi River because much of the sediment that normally would have been delivered was being trapped or dredged in upstream pools. Impoundment increases the amount of surface area of the river due to inundation of the flood plain, and surface area remains more constant throughout the year (Kittrell 1959). Channel configuration becomes more uniform, and riffle habitats become eliminated. Shallow water zones and total edge habitat increase (Leopold et al. 1964).

Water velocities become lower for all river stages and all points on the river following impoundment except in the immediate vicinity of the dam, where structures restrict the size of the channel. The greatest relative decrease in water velocity occurs at times of low discharge. Peak flow rates generally become only slightly lower than those before lock and dam construction. Velocities below the dam can be greater than any occurring before impoundment due to localized constrictions of the channel (Koryak 1976).

Navigation pools attenuate sediment movement by reducing the hydraulic gradient. Coarse bed materials settle out in the lower portions of navigation pools, leaving only fine sand to move as bedload. This fine bedload is relatively unstable (Yorke 1978) and, therefore, poor substrate for aquatic organisms. Clay and silt normally pass through navigation pools relatively unimpeded (Yorke 1978).

The influence of impoundment on bedload type and sedimentation can, in the long-term, alter river morphology considerably from that which was originally established by the locks and dams. The lower reaches of each pool of the Upper Mississippi River increased greatly due to inundation of marshlands after construction of the lock and dam system. However, Simons et al. (1981b) reported that water areas outside the main channel are rapidly being lost due to sediment inflow, deposition, and accumulation. Mueller (1977) noted that the

Mississippi River could soon consist only of a main channel, and that off-channel areas could be entirely lost. Aggradation can be expected to be most serious in the downstream reaches of navigation pools and backwater areas, where water velocities are lowest. Such decreases in velocity usually result in lower concentrations of suspended solids in the main channel (Lubinski et al. 1981).

Navigation pools have little effect on flow variability because of their relatively small storage capacities (Koryak 1976). The Mississippi River navigation pools do not exhibit thermal stratification because they are shallow and have enough flow-through to maintain adequate mixing (Yorke 1978). Given the hydraulic geometry of the Kanawha River, its navigation pools probably resemble those of the Mississippi River in this respect.

River water temperatures can be affected somewhat following impoundment (Kittrell 1959). An impounded river will receive more sunlight and provide a larger interface with the atmosphere than does the natural river, due to the increase in surface area. The attenuation of flow velocity in navigation pools leads to greater residence times. Hence, river water temperatures would be expected to increase following impoundment.

Koryak (1976) believed that lower DO may be a long-term effect of impoundment due to decreases in aeration. However, decreases in atmospheric contributions to DO may be somewhat offset by increased primary productivity. Reductions in river

velocity following impoundment would favor the establishment of true potamon phytoplankton populations (Hynes 1970, Whitton 1975). Also, photosynthesis would increase since the river would receive more sunlight following impoundment due to the increase in surface area. DO levels found in the Kanawha River in recent surveys do not indicate any long-term decrease in DO due to the lock and dam system (West Virginia Department of Natural Resources, unpublished data, U.S. Army Corps of Engineers 1983).

Little information is available for delineating the effects of locks and dams on river biota. However, Lubinski et al. (1981) concluded that locks and dams affected the ichthyofauna of the Upper Mississippi River in three general ways: (1) lentic species were thought to be selected for, and lotic species may be eliminated, (2) the number of fish probably increased because the locks and dams broadened the river, increased its surface area, corrugated it, and supplied nutrient to it, and (3) the effects on fish migration patterns would be variable, and be dependent upon the timing, magnitude, and duration of floods that enable fish to circumvent these barriers. Given the reported effects on hydraulic geometry, locks and dams could also adversely impact coldwater stenotherms, rheophilic species, and riffle species.

III.2. STUDIES OF THE KANAWHA RIVER

Water Quality

The Kanawha Valley lies amidst one of the richest bituminous coal and natural gas regions in the world. The river provides water, an inexpensive means of shipping, and recreation. High quality ground water is plentiful in the valley. The hydroelectric generating stations situated at each of the Kanawha River navigation dams furnish relatively inexpensive energy to the region. The West Virginia state capitol, Charleston, lies on the banks of the Winfield Pool. This amalgamation of abundant natural and human resources has fostered a high degree of urbanization and industrialization in the Kanawha Valley. As a result, much of the Kanawha River ecosystem, especially below the London Dam, was seriously degraded by pollution through the early 1970's, largely due to a lack of environmental awareness and/or interest.

The Kanawha River has been referred to as the Ruhr of the U.S. chemical industry. The industrial center of the Kanawha Valley extends roughly 40 miles below the London Dam. Principal industries in the valley include chemical and glass manufacturing and oil refining (Burmeister 1978). Several major industries and seven municipalities discharge wastewaters into the Kanawha River. There was little treatment of wastewater effluents before 1948 (U.S Army Corps of Engineers 1983), and the Kanawha River was once considered one of the most polluted streams in the United States (U.S. Army Corps of Engineers 1975). The Kanawha

River received wastewater effluents high in nitrogenous and organic matter from the huge organic chemical industrial complex below Charleston, from the large synthetic rubber plant at Nitro, and from 40 outfalls from the Blaine Island Plant of Union Carbide.

Kanawha River water quality began to show improvement in the 1960's due to the enforcement of wastewater treatment requirements by the Kanawha River Industrial Advisory Committee, formed in 1958, and the U.S. Department of the Interior Federal Water Pollution Control Administration, founded in 1963 (U.S. Army Corps of Engineers 1983). However, as late as 1970, river water BOD was extremely high, and DO ranged to 0.0 mg/l. Ammonia concentrations were elevated, and prevailing turbidity levels were 20-35 JTU in the Winfield Pool, with maximum values of 180 JTU recorded at Winfield Locks and Dam. Lead and zinc levels up to 1500 and 164 mg/l, respectively, were measured in the Winfield Pool. Mercury levels up to 1 µg/l were found. Winfield Pool pH ranged from 9.5 to 4.2 (U.S. Army Corps of Engineers 1975).

There have been several environmental studies of the Kanawha River that included data on water quality. Mason et al. (1971) summarized the water quality monitoring of the Ohio River Basin by the U.S. Environmental Protection Agency for the years 1963-1967. There were two stations on the Kanawha: one at London Dam (River Mile 82.6) and one at Winfield Dam (River Mile 31.1). The station at London Dam was established in 1966 as a

"control" to monitor the effects of the industrial complex at Charleston. Temperature and flow data were available for 1963, 1966, and 1967. There were a limited number of chemical analyses for June through August at River Mile 74 in 1964 and 1965. The minimum dissolved oxygen concentration was 6.3 mg/l, and the average monthly value approximately 7 mg/l. The temperature did not exceed 31° C, and the minimum pH was 7.1. Mason et al. (1971) described very poor water quality in the vicinity of Winfield Dam, which is 30 mi below the Charleston industrial complex. The maximum water temperature was 32.5° C, and the pH seldom dropped below 6.5 during the summer. The mean monthly values for dissolved oxygen were usually below 1 mg/l during the summer, and minimum values of 0.0 mg/l were recorded during most summer months. Odors of hydrogen sulfide and other chemicals were noticeable at the dam. Undecomposed leaves, coal fines, and tar-like deposits covered the bottom. Quite often the surface of the water at the dam was black with coal dust, and the water had an overall brown color.

Water quality in the upper reaches above the Winfield Pool was less affected. The 1975 ten-year average of dissolved oxygen was > 7 mg/l at Marmet Pool, but < 2 mg/l at Winfield Pool during low flow (Dames and Moore 1975).

The U.S. Army Corps of Engineers (1975) prepared an environmental impact statement for the continuation of operation and maintenance of the Kanawha River Navigation System. In this

report they reviewed the history of the water quality in the Kanawha River with emphasis on the years 1967-1974. The data sources that were reviewed for the EIS included: U.S. Environmental Protection Agency, West Virginia Department of Natural Resources, Ohio River Sanitation Commission, and Huntington District U.S. Army Corps of Engineers. The review indicated that abatement programs had resulted in significant improvements in water quality, but as late as 1974 the remaining pollution problems continued to dominate the water quality in the area below Charleston. Violations of several state water quality standards continued, particularly during the low flow conditions of late summer. The U.S. Army Corps of Engineers (1975) concluded that "In view of the dominant effects caused by the pollutants discharged in the Charleston area, the effects of navigation, dredging, lock and dam operations and other uses of the river in this area are not likely to be measurable, and were not discernible in the data base available at the time of this review".

From 1973-1977, FMC Corporation in Charleston contracted a broad ecological reconnaissance of the Kanawha River for the purpose of determining the impact of effluents from its plants. The results of the studies pertaining to water quality and plankton were presented in several reports (Benfield et al. 1974, Hendricks et al. 1977, Hendricks and Seagle 1978). Benfield et al. (1974) conducted chemical analyses of river water and

specific effluents at 19 sites from River Mile 89.9-42.9. They found the chemical properties of the water to be highly variable near different effluents, but the effluents appeared to become diluted a short distance into the river. Nine transects between River Mile 55.0 and 42.9 were sampled in August and September for dissolved oxygen, temperature, and specific conductance. The lowest concentration of dissolved oxygen (4.6 mg/l) occurred at River Mile 43.3, but most values were 6-7 mg/l. The highest water temperature (34.0° C) occurred near the left bank at River Mile 54.95 in August. The high temperature at River Mile 54.95 may have been caused by an effluent because the temperature was only 25.5° C at the right bank on the same date. The maximum temperature at most sites ranged form 25-27° C.

Recent water chemistry data collected by the West Virginia Department of Natural Resources (unpublished data) revealed that only fecal coliforms and iron levels violated water quality standards in the Kanawha River basin. Kanawha mainstem DO levels averaged 8.6 mg/l, and DO minima during the study period 1979-1981 did not fall below the West Virginia standard of 4.0 mg/l in the Kanawha River. The trend of improving water quality in the Kanawha River appears to be continuing. DO concentrations in the Winfield Pool in 1982 ranged from 7.3 to 10.0 mg/l (U.S. Army Corps of Engineers 1983).

Phytoplankton

The phytoplankton of the Kanawha River, WV have been studied infrequently as a part of regional or national surveys. Brinley and Katzin (1942) observed large numbers (> 100/ml) of Melosira spp., Cryptomonas erosa, Pteromonas aculeata, and Peridinium spp. in samples collected form 1939-40. Published reports of the National Water Quality Network surveys of 1960-62 included the diatoms Cyclotella pseudostelligera, Stephanodiscus hantzschii, Nitzschia (lanceolate group) (Williams and Scott 1962), Achnanthes minutissima, Cyclotella meneghiniana, C.stelligera, and Synedra ulna (Williams 1964) as important taxa. Palmer (1967) reported that green flagellates were also present in large numbers.

The U.S. Army Corps of Engineers (1975) presented limited data on phytoplankton in the Kanawha River. They listed the common genera and noted that shifts in community structure occur along the river according to changes in water quality. The density of phytoplankton in the Kanawha was reported as being twice that in the Ohio and nine times that in the Potomac. The authors of the report speculated that algae were probably a more important component of the ecosystem in the Kanawha than in some other large rivers because of reduced turbidity, nutrients from municipal sewage discharges, and relatively slow velocities.

The National Stream Quality Accounting Network survey in 1974-75 included the Kanawha River in a comparison of annual phytoplankton densities (Briggs and Ficke 1977). In comparison,

the present study found only species of Melosira, Cyclotella, and green flagellates to still be dominating, with cell concentrations not very different from those seen in 1975.

Hendricks et al. (1977) reported on a study of the phytoplankton of the Kanawha River that was conducted during the summer of 1976 in the vicinity of South Charleston and Nitro. From that study they concluded that: (1) the phytoplankton changed from a diatom dominated community at London Dam to one dominated by green and blue-green algae at Winfield Dam; (2) the water coming from the channel behind Blaines Island affected the phytoplankton for some distance downstream; (3) the water from behind Blaines Island spread across the river rather quickly; (4) the problems created by the FMC outfall at Nitro were localized and dissipated quickly. Hendricks and Seagle (1978) reported on a study conducted during the summer of 1977. The purpose of this study was to determine the limits of the plume generated by flow from the channel behind Blaines Island under ambient flow conditions. Sampling efforts were confined to the area between River Mile 59.9 and 47.1. They found that the phytoplankton were severely stressed in the surface water for approximately one mile below FMC's effluents in the channel behind Blaines Island. The effluent from the back-channel moved across the river within 0.1 mi downstream, but carbon assimilation was not affected deeper than 1 m. Hendricks and Seagle (1978) suggested that the damage to the phytoplankton occurred as the organisms passed through the

FMC plant. By August 1977, new treatment facilities installed on the outfall appeared to have improved water quality in the plume. The effects of FMC's effluents could not be detected below River Mile 50.1 because other perturbations obscured the interpretation of the data.

Periphyton

Apparently, the periphyton community was not considered in any of the earlier environmental studies of the river. Only the recent studies of the Kanawha River and its environs by Marshall University (1982) and VPI&SU (Voshell et al. 1982) have included data on periphyton. Both studies were contracted by the Corps and were included in the Kanawha River Navigation Study Reconnaissance Report (U.S. Army Corps of Engineers 1983).

In the studies conducted by VPI&SU (Voshell et al. 1982), Simmons found approximately 50 taxa from River Mile 82 to River Mile 29. Diatoms tended to dominate the community in regard to biomass, whereas blue-green algae were numerically most abundant. A significant reduction in density was observed in the lower portion of the Winfield Pool.

In the studies reported by Marshall University (1982), Weaks described the communities of periphyton and estimated production in three tributaries of the Kanawha River between River Mile 28-31: Buffalo Creek, Hurricane Creek, and Little Hurricane Creek. He considered these tributaries to be "unpolluted to slightly polluted."

Zooplankton

Water quality in the lower reaches of the Kanawha River in the early 1960's greatly reduced rotifer populations (Williams 1966). Average rotifer concentrations in the Winfield Pool were 6/l in 1959-1961 (Palmer 1967) and 34/l in 1961-1962 (Williams 1966). The U.S. Army Corps of Engineers (1975) presented limited data on zooplankton in the Kanawha River, primarily a list of common genera.

Numerous studies have been conducted on seasonal distribution of rotifers in other river systems (Kofoid 1908, Carlin 1943, Beach 1960, Green 1960, Holden and Green 1960). Fewer studies have been done on the longitudinal distribution of rotifers in river systems (Hutchison 1939, Beach 1960, Rai 1974).

Previous thought on the importance of Cladocera in lotic systems is summarized well by Hynes (1970): "In great contrast to the condition in lakes, crustaceans are always unimportant (in rivers) and the animals are represented mainly by rotifers." Hynes (1970) also states that Cladocera in rivers are generally considered to be strays from other bodies of water. However, when dams reduce the rate of flow as in an impoundment, crustacean numbers and importance increase. It has been demonstrated that Cladocera can be present in large quantities in the plankton of lotic systems. Kofoid (1908) showed this in the Illinois River, USA, and Green (1960) showed the same phenomenon

in the Sokoto River, Nigeria. Many of the Cladocera present in these systems were characteristically from mud or mud and vegetative habitats.

Benthic Macroinvertebrates

There have been several environmental studies of the Kanawha River that included information on benthic macroinvertebrates. Mason et al. (1971) reported the data that had been collected by the U.S. Environmental Protection Agency during 1963-1967. The EPA attached floating rock-basket samplers to the upstream side of the London Lock and Dam (River Mile 82.6) and the Winfield Lock and Dam (River Mile 31.1). They collected Petersen dredge samples below London Lock and Dam (River Mile 79.6), above Winfield Lock and Dam (River Miles 32.2 and 31.6), and below Winfield Lock and Dam (River Mile 30.6). The sampling site near London was considered to be suitable for a "control"; the fauna was indicative of mild to moderate organic enrichment. Mason et al. (1971) reported a very unbalanced population of 3-8 taxa at Winfield contrasting with 33 taxa uniformly distributed in composition at London. They concluded that the nearly complete elimination of benthic macroinvertebrates at Winfield was indicative of gross toxic pollution from the Charleston industrial complex.

The U.S. Army Corps of Engineers (1975) prepared an environmental impact statement that included the results of a survey of benthic macroinvertebrates. Benthic organisms were

collected at several sites on one occasion (April 25-28, 1975) with a Ponar grab. The sampling sites on the main channel of the Kanawha were: (1) just above Winfield Dam, (2) approximately 5 mi below Marmet Dam, (3) just above Marmet Dam, (4) approximately 6 mi below London Dam, (5) just above London Dam, and (6) approximately 5 mi above London Dam. The U.S. Army Corps of Engineers (1975) concluded that the nearly complete dominance of oligochaetes in the lower Winfield pool was due to industrial and domestic waste discharges from Charleston. They also concluded that benthic organisms in the upper Kanawha were limited more by lack of suitable substrate rather than degraded water quality. They found that a pollution intolerant midge-mayfly community was present on the rock substrate in the upper Kanawha.

Benfield et al. (1974), Benfield et al. (1975), and Hendricks et al. (1979) reported on the results of benthic macroinvertebrate studies that were part of an ecological reconnaissance of the Kanawha conducted for FMC Corporation in Charleston. The information on the oligochaetes encountered during this study was also published in a refereed journal article (Maciorowski et al. 1977). Benfield et al. (1974) and Benfield et al. (1975) took Ponar grab samples at 14 sites from River Mile 42 to River Mile 89.9. Most of these sites were located at the effluents from particular industries, but 6 of the sites were considered to be suitable for "reference" sites. The

reference sites were located at River Miles 43.3, 49.9, 55.0, 63.4, 83.2, and 89.9. The investigators found that the Ponar grab sampler could only be used in near-shore areas (10-50 m from bank), but they took 30 samples at each study site. Artificial substrates were used at 9 of the study sites. Benfield et al. (1975) concluded that the fauna of the London pool was typical of that expected in a relatively undisturbed segment of a "river like the Kanawha." The area between London Lock and Dam and just upstream from Charleston showed some evidence of degradation. Benthic macroinvertebrates in the area of South Charleston were severely depressed as compared to upstream sites. They attributed the depression of benthic macroinvertebrates to the heavy use of the river by industry and municipalities. Between South Charleston and Nitro the fauna recovered markedly in areas where heavy industry was absent. Some of the effluents severely depressed the fauna due to toxic factors, but the fibers effluent from FMC caused a very high density of oligochaetes. The investigators speculated that the fibers provide a suitable substrate and/or food supply.

Hendricks et al. (1979) reported on a second study of the Kanawha River conducted for FMC Corporation. Samples were collected at 10 sites between River Mile 32.3 and 63.4 during the summer of 1977. Five of the sampling sites in the 1977 study corresponded with sites sampled in the 1973 study. Nine Ponar grab samples were collected at each site (3 left, 3 center, 3

right). Hendricks et al. (1979) concluded that the river had improved since the 1973 survey, with an increase in numbers of individuals and diversity at all sites and particularly those below the outfalls at South Charleston. However, they also concluded that the fauna of the study area was indicative of organic enrichment.

Recent studies of benthic macroinvertebrates in the Kanawha River have been reported by Marshall University (1982) and Voshell et al. (1982). Both studies were contracted by the Corps and were included in the Kanawha River Navigation Study Reconnaissance Report (U.S. Army Corps of Engineers 1983).

Voshell et al. (1982) found that grab sampling proved to be unreliable for benthic macroinvertebrates because of the hard river bottom covered with stones and debris. Artificial substrates proved to be effective for collecting a diverse assemblage of organisms that prefer a firm substrate. A total of 32 taxa were collected at five sites: Upper Marmet - River Mile 32.3, Lower Marmet - River Mile 70.4, Upper Winfield - River Mile 67.0, Lower Winfield - river Mile 31.6, and Upper Gallipolis - River Mile 30.5. The highest number of taxa (21) occurred at Upper Winfield, and the lowest number (14) occurred at Lower Winfield. However, Lower Winfield had the second highest density on the artificial substrates (935/sampler) and the highest density in the grab samples (963/m^2).

On the artificial substrates, Diptera (primarily Chironomidae) were the dominant members of the community at all sites. The sites in the upper ends of the pools, particularly Upper Marmet and Upper Winfield, contained a fauna that was indicative of riverine conditions. The rheophilic organisms that were abundant at these sites included several general of Ephemeroptera and filter-feeding Trichoptera. The fauna at the sites in the lower ends of the pools (Lower Marmet and Lower Winfield) were characteristic of lentic environments. Other organisms that were commonly collected on the artificial substrates were two genera of amphipods (Gammarus and Hyalella) and the mollusc Corbicula. The grab samples contained a very different assemblage of organisms, primarily those adapted for inhabiting soft sediments. Oligochaetes were very abundant at Lower Winfield. Analysis of the benthic community according to functional groups, indicated that the community was dominated by collectors, those organisms which ingest fine particulate organic matter.

In the studies reported by Marshall University (1982), Tarter described the community structure of benthic macroinvertebrates in three tributaries of the Kanawha River between River Miles 28-31: Buffalo Creek, Hurricane Creek, and Little Hurricane Creek. Sixty taxa, comprising 39 families, were encountered. The dominant taxa according to numerical percentage were Chironomidae (37.4%), Stenonema (9.8%), Cheumatopsyche

(9.1%), Isonychia (8.9%), Hydropsychidae (8.3%), and Corbicula (7.8%).

The mussel fauna of the Kanawha River was surveyed by Taylor (1983) and included in the Reconnaissance Report by the U.S. Army Corps of Engineers (1983). Taylor searched for mussels throughout the entire length of the Kanawha River from its head at Gauley Bridge to its mouth at Point Pleasant. Only the introduced pest species, the Asiatic clam, Corbicula fluminea (Muller), occurred below River Mile 81. He considered the headwaters above the reach of navigation (River Mile 90.6) to be the "richest single stream site in the State of West Virginia" because of the 34 species that occur there. Taylor attributed the absence of mussels in the lower Kanawha River to the lentic conditions produced by impoundment and other environmental degradation.

Welch and Joy (1984) have recently studied the growth of the Asiatic clam in the Kanawha River. They found that summer growth rates were higher than winter growth, and that smaller clams had higher growth rates than larger clams. In addition, the growth of C. fluminea in the Kanawha River compared favorably with rates reported from the west and southwest.

Crayfish (Decapoda: Cambariidae) have been inadequately represented in studies of benthic macroinvertebrates in the Kanawha River. Because of their greater mobility and size, they are not captured by the typical sampling methods. In a statewide

survey of the crayfishes, Lawton (1979) found 4 species that occur in the counties through which the Winfield pool of the Kanawha River flows (Mason, Putnam, and Kanawha). Three of these species (Cambarus robustus, Cambarus bartonii, and Orconectes sanbornii) are common throughout West Virginia and are likely to occur in the main stem of the Kanawha River. The fourth, Cambarus rusticus, has a limited distribution in West Virginia. It was collected by Lawton (1979) from a Kanawha River embayment area at river mile 37. Orconectes sanbornii was collected from Little Hurricane Creek, near Winfield Locks and Dam by Marshall University (1982).

Crayfish represent an important link in the food web of the Kanawha River; Voshell et al. (1983) reported that crayfish were a major component of the diet of smallmouth bass (Micropterus dolomieui), largemouth bass (M. salmoides), spotted bass (M. punctulatus), and channel catfish (Ictalurus punctatus). Crayfish have been described as opportunistic scavengers since they display both herbivorous and carnivorous feeding habits. Capelli (1980) reported that stomachs of Orconectes propinquus contained primarily diatoms and other algae, along with midge larvae, mayfly nymphs, and other crayfish, and smaller amounts of detritus, fish, rotifers, copepods, cladocerans, and amphipods.

Standing crop of four species of crayfish in the New River, West Virginia, was 2.85 g dry wt./m^2 (or 9.573 kcal/m^2) (Nielsen and Orth 1984). This estimate may be a reasonable approximation

for hard substrates in the Kanawha River. Annual production-to-biomass ratios for crayfish generally range from 0.9 to 1.5 (Momot and Gowing 1977).

Fish

The results of ichthyological studies on the Kanawha River are reviewed in Appendix A, Table A9.1.1. The earliest extensive survey of the fishes of the Kanawha River was conducted by Addair (1944). He found a total of 86 species and subspecies of fish in the Kanawha River system in West Virginia. Of these, 39 occurred in the Kanawha mainstem below the falls, and 30 were found in the Winfield Pool. Southern Appalachian regional ichthyological data, compiled from museum records, literature references and field observations, were summarized by Jenkins et al. (1972). They reported 86 native freshwater forms, four forms classified as marine, and 8 introduced or possibly introduced forms (Dorosoma petenense, Salmo gairdneri, S. trutta, Salvelinus fontinalis, Esox niger, Carassius auratus, Cyprinus carpio, and Pimephales promelas) in the Kanawha River drainage below Kanawha Falls. No endemic species were reported to occur in the river below the falls. The ichthyofauna of the Kanawha River are similar to other major drainages of the Ohio River Basin; the Kanawha River drainage system below the falls shares 75 species with both the Cumberland River below its falls and the Tennessee River (Jenkins et al. 1972).

Fish have been sampled since 1968 at the London and Winfield Locks in a cooperative effort by several state and federal agencies (Ohio River Sanitation Commission 1981). Sampling protocol involves leaving the downstream gates of one lock chamber open overnight while maintaining some flow through the chamber to attract fish. The gates are then closed, rotenone is applied to kill the fish, and all fish dying and surfacing the first day are collected. The efficiency of the one-day pick up is known to vary from about 50 to 90% and the species composition of the sample does not truly reflect relative abundances of all species in the river. However, the 12 samples taken since 1968 do allow some conclusions to be drawn. Lock rotenone samples taken at Winfield and London during the period 1968-1983 document the recent occurrence of at least 58 species (as compared to the 39 found by Addair 1944) in the Kanawha mainstem below the falls (Ohio River Sanitation Commision 1981, West Virginia Department of Natural Resources, unpublished data). This is a conservative estimate of species richness in the Kanawha River mainstem because only two locations were sampled. The increase in species diversity since Addair's work is most likely primarily attributable to improved water quality.

From the period 1968 through 1983 the number of species collected per sample ranged from 8 to 25 (Table III.2.1). At both sites there has been an increase in the number of species collected per sample. At London, the number of species collected

averaged 11.5 for 1968-73 and 22 for 1976-83. At Winfield, the increase was significant but not as dramatic (10.8 for 1968-73 and 15.6 for 1976-83). Over the entire period more species were collected at London Locks than Winfield Locks in 9 out of 12 years. Since 1976, an average of 6.4 more species were collected from London Locks than from Winfield Locks. The apparent recovery and reestablishment of fish populations at both sites is evident; for example, sauger and walleye have occurred regularly in lock samples since 1976 (Table III.2.1). However, it appears that the recovery has occurred more rapidly in the reaches near London and improvement is still possible near Winfield.

Numbers and weight of fish collected in lock samples were much higher at Winfield Locks than at London Locks (Table III.2.1). Mean numbers per hectare averaged 4,021 at London and 13,412 at Winfield. Number of fish collected was highly variable and no trends were apparent at either lock. Biomass of fish collected averaged 148.18 kg/ha at London and 230.81 kg/ha at Winfield. Biomass was less variable than numbers through time. Biomass at London has increased from 72.77 kg/ha for 1968-73 to 185.86 kg/ha for 1976-83. No comparable increase in biomass was apparent at Winfield.

Although species richness was greater at London than at Winfield, the dominant fish species were the same at both sites (Table III.2.2). The emerald shiner was the dominant species numerically at both sites. At London, six species (emerald

Table III.2.1. Summary of rotenone samples of fish at the London and Winfield locks, Kanawha River. Area of lock chambers was 0.187 ha. Data from 1968-1980 extracted from Ohio River Sanitation Commission (1981) and data since 1981 West Virginia Dept. Natural Resources unpublished stream survey forms.

	London			Winfield		
Year	No. of Species	Number	Weight(kg)	No. of Species	Number	Weight(kg)
1968	9	219	12.62	10	2055	45.52
1969	13	446	9.66	8	1653	35.67
1970	14	1461	10.16	12	14716	51.72
1973	10	79	21.99	13	1055	25.32
1976	25	827	17.96	16	1278	27.55
1977	25[b]	238	16.89	13[a]	166	19.57
1978	25[b]	2614	29.44	15[a]	1208	53.17
1979	18[a]	470	31.03	13[a]	252	35.97
1980	21[c]	644	75.13	15[b]	423	47.89
1981	22[b]	757	12.63	20[b]	2532	96.66
1982	17[a]	203	67.20	17[b]	3708	34.99
1983	23	1060	27.77	16[b]	1043	45.92
mean		752	27.71		2508	43.16
mean/ha		4021	148.18		13412	230.81

[a] sauger collected

[b] sauger and walleye collected

[c] walleye collected

shiner, mimic shiner, channel catfish, gizzard shad, flathead catfish, and freshwater drum) made up 91.9 and 86.3% of the total numbers in 1968-73 and 1976-83, respectively. At Winfield, five species (emerald shiner, gizzard shad, channel catfish, mimic shiner, and freshwater drum) made up 96.3 and 95.0% of the total numbers in 1968-73 and 1976-83, respectively. The gizzard shad and carp were the dominant species by weight at both sites. At London, seven species (gizzard shad, carp, channel catfish, flathead catfish, freshwater drum, silver redhorse, and longnose gar) made up 74.7 and 86.0% of the total weight in 1968-73 and 1976-83, respectively. At Winfield, five species (carp, gizzard shad, channel catfish, emerald shiner, and freshwater drum) made up 96.9 and 87.3% of the total weight in 1968-73 and 1976-83, respectively. A notable difference in the species composition between sites was the lower abundance and fewer species of the redhorse (Moxostoma spp.) at Winfield. Redhorses are intolerant of industrial pollutants and siltation (Trautman 1981) and may not yet have reestablished large populations in the vicinity of Winfield.

III.3. PERTINENT STUDIES OF BIOTA IN OTHER LARGE RIVERS

Benthic Macroinvertebrates

Studies of macroinvertebrate production have been conducted on the Satilla, Savannah, and Thames Rivers. These rivers

Table III.2.2 Percent composition of common fishes (70.1% of total weight or number at either site over all years) in lock-rotenone samples taken from London and Winfield Locks, Kanawha River, from years 1968-1973 and 1973-83. Data from 1968-1980 extracted from Ohio River Sanitation Committee (1981) and data since 1981 are from West Virginia Department of Natural Resources stream survey forms.

Species	London				Winfield			
	Numbers		Weight		Numbers		Weight	
	1968-73	1976-83	1968-73	1976 83	1968-73	1976-83	1968-73	1976-83
Lepisosteus osseus (Longnose gar)	0.3	0.5	3.5	4.4	-	-	-	-
Anguilla rostrata (American eel)	0.1	T	0.3	0.1	-	T	-	-
Alosa chrysochloris (Skipjack herring)	-	T	-	0.3	T	1.2	T	1.0
Dorosoma cepedianum (Gizzard shad)	0.4	6.8	3.8	24.3	5.5	24.1	23.6	30.7
Esox masquinongy (Muskellunge)	T	T	11.2	1.0	-	-	-	-
Campostoma anomalum (Stoneroller)	-	T	0.1	T	-	-	-	-
Cyprinus carpio (Carp)	0.4	0.5	4.7	26.6	1.2	0.4	44.2	25.8
Hybopsis aestivalis (Speckled chub)	-	0.2	-	T	-	T	-	T
Hybopsis amblops (Bigeye chub)	-	1.0	-	T	T	-	T	-
Notropis atherinoides (Emerald shiner)	62.3	55.3	4.1	1.5	75.5	50.4	13.1	4.9
Notropis buchanani (Ghost shiner)	4.4	6.8	0.3	0.1	T	0.6	T	T
Notropis stramineus (Sand shiner)	T	0.4	T	T	1.8	-	0.2	0.1
Notropis volucellus (Mimic shiner)	12.2	16.5	0.3	0.3	2.3	9.2	0.2	0.2

Species	London				Winfield			
	Numbers		Weight		Numbers		Weight	
	1968-73	1976-83	1968-73	1976 83	1968-73	1976-83	1968-73	1976-83
Pimephales notatus (Bluntnose minnow)	0.3	0.2	T	T	T	0.5	T	-
Ictiobus bubalus (Smallmouth buffalo)	T	T	0.1	T	T	T	0.2	0.7
Ictiobus niger (Black buffalo)	-	T	-	2.4	-	T	3.6	1.6
Moxostoma anisurum (Silver redhorse)	0.2	0.7	4.4	6.7	-	-	-	-
Moxostoma carinatum (River redhorse)	-	0.1	-	0.7	-	-	-	-
Moxostoma erythrurum (Golden redhorse)	-	T	-	0.3	-	-	-	-
Moxostoma macrolepidotum (Shorthead redhorse)	-	0.2	-	0.7	-	T	-	0.1
Ictalurus melas (Black bullhead)	-	-	-	-	0.1	-	0.7	-
Ictalurus nebulosus (Brown bullhead)	-	-	-	-	0.2	-	1.3	-
Ictalurus punctatus (Channel catfish)	10.8	5.1	30.7	10.1	13.0	7.2	15.9	25.4
Pylodictis olivaris (Flathead catfish)	4.9	0.4	19.5	7.8	-	0.1	-	1.3
Morone chrysops (White bass)	0.4	0.3	1.0	0.9	-	0.8	-	2.3
Ambloplites rupestris (Rock bass)	-	0.1	-	0.2	-	T	-	T
Lepomis macrochirus (Bluegill)	0.2	0.3	T	0.3	T	0.2	0.1	0.1
Lepomis megalotis (Longear sunfish)	0.2	0.1	0.1	0.1	-	0.1	-	T

Species	London				Winfield			
	Numbers		Weight		Numbers		Weight	
	1968-73	1976-83	1968-73	1976 83	1968-73	1976-83	1968-73	1976-83
Micropterus dolomieui (Smallmouth bass)	T	0.2	0.3	0.4	-	-	-	-
Micropterus punctulatus (Spotted bass)	0.8	0.6	6.0	1.2	T	0.1	0.2	0.2
Micropterus salmoides (Largemouth bass)	-	0.1	-	0.2	-	T	-	0.1
Pomoxis annularis (White crappie)	0.7	T	1.2	0.2	T	T	0.1	0.1
Stizostedion canadense (Sauger)	-	0.2	-	1.2	-	0.5	-	2.2
Stizostedion vitreum (Walleye)	-	0.1	-	1.9	-	0.1	-	0.4
Aplodinotus grunniens (Freshwater drum)	1.3	2.2	8.1	6.1	T	4.1	0.1	10.5

T = < 0.1%

represent distinctly different environments from the Kanawha, but they are the only other large rivers for which the production of the macroinvertebrate communities have been estimated.

Benke et al. (1979) and Benke et al. (1984) estimated production of invertebrates in the Satilla River which they described as a "subtropical backwater river." They indicated that submerged wood substrates, or snags, were heavily colonized by aquatic insects, especially Hydropsyche spp. and Simulium spp. Several species of Chironomidae, Ephemeroptera, and Coleoptera were also important on the snag habitats. Various Chironomidae accounted for 32 out of a total of 67 taxa collected. Annual production was estimated at 51,900 and 67,100 mgDW/m^2/yr (dry mass per surface area of snag, or effective habitat) at the two study sites. Sand and silt substrates were also sampled. These substrates had high densities of small aquatic insects but biomass values were rather low (100 mgDW/m^2). Production in sandy substrates was estimated at 11,000 mgDW/m^2/yr. Production in silt habitats was estimated at 7,000-10,000 mgDW/m^2/yr. On the basis of these production estimates and a survey of available snag habitat, Benke et al. (1979) predicted that over half of the invertebrate biomass and 15-16% of the annual production occurred on the snag habitat, which made up only about 6% of the total substrate. Production was partititioned among trophic groups as: 12% collector-filterers, 71% collector-gatherers, and 17% predators. Based on the amount of the different types of

substrate found in the river, an estimate can be made of total benthic macroinvertebrate production in a "typical" or "average" square meter of river bottom. Combining the two Satilla River study sites, and calculating production in this manner indicate that production in the Satilla averages about 22,000 mgDW/m^2/yr.

Production was estimated for six caddisfly taxa in the lower Savannah River, Georgia by Cudney and Wallace (1980): Hydropsyche incommoda, H. rossi, Cheumatopsyche passella, Macrostemum carolina (as Macronema carolina), and Neureclipsis crepuscularis. This study concentrated on snag or submerged woody plant (stems and roots) habitat, and standing stock and production estimates were based on the amount of snag surface area preseent in the river at various stages. Samples were taken in areas with different current velocities. Total annual caddisfly production was 12,600, 41,400, and 22,700 mgDW/m^2 of snag surface area at low, medium, and high river velocities. Reported values have been converted to mg/DW/m^2/yr according to conversion factors suggested by Cummins and Wuycheck (1971). Total production of these taxa was not extrapolated for the entire river, although it was suggested that the ratio of snag surface area in the Savannah is about the same as that estimated by Benke et al. (1979) in the Satilla River. This would mean that annual caddisfly production in the Savannah would be about 600-900 mg/DW in a generalized square meter of river bottom. Winter cohorts were estimated to have high production due to

higher terminal instar weights. It was suggested on the basis of this study that space, or suitable habitat, is apparently the limiting resource for production in the Savannah River.

Mann et al. (1972) studied trophic relationships and energy flows in the Thames River at two sites, above and below the Kennet River (near Reading, England). Both the Thames and Kennet were eutrophic at the time of this study. Production was estimated for two aquatic insect taxa. Production of Caenis sp. and Cyrnus sp., which were considered together, was 147 $mgDW/m^2/yr$ above Kennet River and 110 $mgDW/m^2/yr$ below. Chironomid production was estimated to be 2,829 $mgDW/m^2/yr$ above Kennet River and 570 $mgDW/m^2/yr$ below. Production of gastropods, crustacea, bivalves, sponge and bryozoans was also estimated with total benthic community production estimated at 26,549 $mgDW/m^2/yr$ above Kennet River and 25,171 below. These estimates for chironomid production were considered by the authors to greatly underestimate actual production. The "best estimate" of chironomid production was 36,195 $mgDW/m^2/yr$, based on consumption of adult chironomids by fish in the Thames. This increased the estimate of total benthic community production to 62,600 $mgDW/m^2/yr$.

Fish

In this section we review the species composition, standing stocks, production, and patterns of energy flow for fish in other large rivers. Information on species composition was entirely from the Mississippi River drainage, while information on standing stocks and production was from a wider geographical range.

Pearson and Krumholz (1984) summarized extensive sources of information regarding species composition for the Ohio River. The Ohio is a larger stream than the Kanawha, but information gathered there provides a frame of reference with respect to the species one would expect to find in a large river habitat. Pearson and Krumholz (1984) reported 100, 94, and 91 species for the upper, middle, and lower third of the river, respectively, over the period 1970 - 83. This level of diversity exceeds the 68 species found in the present study on the Kanawha, but represents the integrated findings of numerous impact assessment projects as well as several long term monitoring programs.

Numerically, the 10 most abundant species of the Ohio River (determined from lock chamber rotenone studies during the period 1957-1980) were (in decreasing order) emerald shiner, gizzard shad, freshwater drum, mimic shiner, channel catfish, common carp, bullheads, skipjack herring, white crappie, and threadfin shad (Dorosoma petenense). By weight the most abundant species reported were (in decreasing order) gizzard shad, common carp,

freshwater drum, channel catfish, bigmouth buffalo (Ictiobus cyprinellus), smallmouth buffalo, emerald shiner, paddlefish (Polyodon spathula), flathead catfish, and various species of bullhead (Ictalurus spp.)

Sylvester and Broughton (1983) sampled main channel, side channel, and backwater habitats of the upper Mississippi River and found 36 fish species. By number the most abundant species were the common carp, silver redhorse, shorthead redhorse, black crappie (Pomoxis nigromaculatus), and northern pike (Esox lucius). The greatest diversity of species occurred in backwaters, a habitat type not as common in the Kanawha River as in the upper Mississippi River. In a study of natural and artificial banks conducted on the lower Mississippi River near Greenville, Mississippi, Pennington et al. (1983) listed the dominant fish species collected with electroshocking and hoop nets. Twenty-four species were captured from natural bank habitats, while 27 species occurred along revetted banks. Gizzard shad, flathead catfish, blue catfish (Ictalurus furcatus), and freshwater drum accounted for more than 75% of the samples from the natural banks. Flathead catfish, gizzard shad, blue suckers (Cycleptus elongatus), channel catfish, and freshwater drum were the most abundant species taken from the revetted bank habitats.

Mills et al. (1966), reporting on the fishery of the Illinois River, listed the common carp, black bullhead, emerald shiner, and goldfish (Carassius auratus) as dominant species. This stream has a history of severe pollution disturbances similar to that which occurred on the the Kanawha River before the 1970's.

Hesse et al. (1982) conducted studies of the channelized middle Missouri river using 6 different sampling methods and found 57 species of fish. The 10 most abundant species collected were gizzard shad, goldeye (Hiodon alosoides), common carp, river carpsucker (Carpiodes carpio), blue sucker, smallmouth buffalo, shorthead redhorse, flathead catfish, sauger, and freshwater drum. These species together accounted for approximately 90% of the fish taken (by electroshocking) for the entire study.

Fish abundance and production are seldom estimated in large river systems due to sampling difficulties. Mann (1965) estimated that the abundance of all fish in River Thames, England, was 659 kg/ha. This was viewed as an exceptionally high fish biomass estimate and one of the unique characteristics of the Thames. Standing stock estimates from Belgian rivers range from 130 to 300 kg/ha (Timmermans 1961, Huet and Timmermans 1963; cited in Mann 1965). North American streams have been reported to support up to 471 kg/ha (Needham, Moffett, and Slater 1945, McFadden and Cooper 1962). In comparison, Carlander (1955)

reported standing stocks of about 440 kg/ha for reservoirs and 550 kg/ha for river backwaters.

A number of studies on tropical, floodplain rivers demonstrate quite high variabiity in standing stock, even for the same river at different times. Welcomme (1979) suggests that standing stock is closely linked to both the current flood stage as well as recent flood history in these systems. Ichthyomass from main river channels of tropical rivers ranged from approximately 100 to 600 kg/ha, which is well within the range (130-1100 kg/ha) reported for temperate systems (Table III.3.1). Standing stock estimates (Welcomme 1979) for river backwater habitats are generally higher than for main channels (Table III.3.2). Pearson and Krumholz (1984) analyzed the results of lock chamber fish surveys conducted along the Ohio River since 1957. Biomass estimates for all species combined averaged 381 kg/ha and ranged from 205 to 470 kg/ha (Table III.3.3). When the data were partitioned by river mile, standing stocks were lowest in the upper reaches of the Ohio River and increased in downstream reaches (Table III.3.4).

Estimates of fish production by entire fish communities are scarce for large rivers. Watson and Balon (1984) studied the fish communities of the Baram River (Malaysia) tributaries, and estimated that total production in these systems was estimated at 76.71 kg/ha per year. These estimates are much lower than

Table III.3.1. Main channel estimates for standing stocks of fish from various rivers.

(kg/ha)	River	Source
130- 300	Belgian rivers	Timmmermans (1961), Huet & Timmermans (1963)
143	Coosa R. (deep)	Swingle (1954)
154	Coosa R. (shallow)	Swingle (1954)
200-1100	Vistula R.	Backiel (1971)
311	Horokiwi Stream	Allen (1951)
457	Tombigbee R. (deep)	Swingle (1954)
471	No. Amer. trout streams	McFadden & Cooper (1962)
570	Tombigbee R. (shallow)	Swingle (1954)
659	R. Thames	Mann (1965)
1730	Tenson R.	Swingle (1954)

Table III.3.2. Backwater estimates for standing stocks of fish from various rivers.

(kg/ha)	River	Source
149-350	Bandama R.	Daget et al. (1973)
219.8	Mekong R.	Sidthimunka (1970)
300-500	Danube R.	Chitravadivelu (1974)
369-5616	Chari R.	Loubens (1969)
2084	Tombigbee R.	Swingle (1954)

Table III.3.3. Total fish biomass from the Ohio River over time (Pearson and Krumholz 1984).

(kg/ha)	Standard Error	Period
205	33	1957 - 1960
391	218	1967 - 1970
470	76	1974 - 1977
458	82	1978 - 1980

Table III.3.4. Total fish biomass from the Ohio River by location (Pearson and Krumholz 1984).

(kg/ha)	Standard Error	Ohio River Mile[a]
251	28	0 - 300
258	28	301 - 700
485	85	701 - 981
458	82	1978 - 1980

[a]River Mile O is at confluence of Allegheny and Monongahela Rivers at Pittsburgh.

estimates for other rivers, which range up to 2,000 kg/ha/yr for the River Thames, England (Table III.3.5).

Since the Kanawha River had been modified with locks and dams it has characteristics that are intermediate between rivers and reservoirs. Therefore, we include reference to reservoir investigations in this review. Long term studies (Rainwater and Houser 1982) of Beaver Lake, Arkansas, fish population have provided a time series of standing stock estimates by treating coves of known area with rotenone. The mean for the years 1969-1980 was 325 kg/ha. However, the data demonstrate considerable variability through time. The highest estimate of standing stock, 498 kg/ha, was made the year following achievement of full pool (1968). Since that time estimates have varied from a low of 209.8 kg/ha to a more recent high of 485.9 kg/ha. For 173 reservoirs throughout the U.S. total standing stocks averaged 200 kg/ha (Jenkins 1975), based also on cove rotenone samples. Jenkins (1982) reviewed standing stock data from over 290 U.S. reservoirs to assess the relation between fish abundance with the morphoedaphic index (MEI = total dissolved solids/mean depth). Total fish standing stocks averaged approximately 40 for low MEI (0.5) to 300-500 kg/ha for MEI greater than 10. However, the total quantity of nutrients, and not their concentration alone, might be the primary determinant of fish standing stocks in reservoirs (Aggus and Lewis 1978). Consequently, water exchange rates are of fundamental importance

Table III.3.5. Estimates of fish production from various rivers.

(kg/ha/yr)	P/B ratio	River	Source
77	1.4	Baram R., Malaysia	Watson and Balon (1984)
121	0.7	Nida R., Poland	Watson and Balon (1984)
279	1.2	Grand R., Canada	Watson and Balon (1984)
426	0.64	R. Thames	Mann (1965)
533	1.7	Horokiwi stream	Allen (1951)
630-870	0.75 - 1.63	Kafue R.	Kapetsky (1974b)
2000	1.12 - 1.92	R. Thames	Mann et al.(1972)

in characterizing different reservoir ecosystems. In this regard the Winfield pool of the Kanawha River is most similar to a mainstem hydroelectric impoundment.

Only 1 total ecosystem investigation concerning the pathways of energy flow on a large river has been conducted. Mann et al. (1972) concluded that fish production in the River Thames, England, estimated at 198 kcal/m^2/yr, could not be supported from total production by macroinvertebrates (330 kcal/m^2/yr). The food web is described as "exceedingly complex", with the adult fish obtaining their energy primarily from allochthonous origins. Major sources of energy to adult fish include insects at the surface, organic detritus, and algae. Young fish depend primarily on rotifers and cladocerans, which in turn rely on a very rich source of phytoplankton supported by nutrients derived from sewage effluents.

The significance of energy flow pathways and their influence on fish biomass was confirmed in an application of Ploskey and Jenkins' (1982) biomass model of fish and fish-food interactions. Ploskey and Jenkins developed a model of 5 functional groupings of reservoir fishes based on trophic relationships. The model predicted the food requirements of each group from input data consisting of the biomass of the species present and their food habits. Food requirements were then projected through time and compared with changes in the abundance or productivity of lower

trophic levels. In a test of the model using information from DeGray Lake, Arkansas, fish biomass responded (both positively and negatively) when available prey differed from the level required to sustain the standing stock of each group. The authors concluded that reservoir fish populations are regulated by changes in the seasonal availability of forage.

III.4. MODELS OF AQUATIC ECOSYSTEMS

The study of the structure and function of aquatic ecosystems has lagged behind studies focusing on the individual organism or population levels of organization. Consequently, environmental impact studies have tended to focus on developing taxonomic inventories of biological communities. These inventories have proven to be inadequate for studying ecosystem-level impacts to aquatic environments (Odum 1977). It was not until investigators began to recognize that functional roles in ecosystems were filled by various taxa that generalizations about ecosystem structure and function were possible. Lindeman (1942) categorized biota in aquatic ecosystems based on nutritional habits or trophic levels, making it possible to quantify energy flow through ecosystems. Odum (1957) was the first to quantify energy flows between trophic levels in lotic ecosystems. Odum's (1957) investigation, in Silver Springs, Florida, focused on five trophic levels: producers, herbivores, carnivores, top

carnivores, and decomposers. The estimates of efficiencies derived from the Silver Springs study provided some of the first hypotheses of ecosystem structure and function to be tested in later research. The concept of energy flow as a unifying concept in ecology is now well established as evidenced by recent ecology text books (Odum 1971). In this section we review briefly the current understanding of the structure and function of stream ecosystems. We then review the more detailed models that have been developed for simulating the dynamics of aquatic ecosystems.

The conceptual model of stream ecosystem structure and function that most stream modelers currently adhere to was developed by Cummins (1973, 1974). This model was based on studies on small streams (3rd order and less). These streams derive the majority of their energy supply from imports of organic matter elaborated in the terrestrial system through which the stream flows and a great deal of this organic input is utilized during the fall and winter period. (Cummins 1974). Cummins' (1974) model emphasized the processing of particulate organic matter and dissolved organic matter by microbes (bacteria, fungi) and invertebrates. In this model, invertebrates were categorized into functional groups based on the mode of obtaining food (i.e., shredders, collector-gatherers, collector-filterers, predators).

Investigators of stream fishes recognized quite early that the assemblages of species change as the physical structure of

river systems change from headwaters to mouth (Shelford 1911, Thompson and Hunt 1930). Recently, Vannote et al. (1980) proposed that the structure and function of the stream ecosystem also varies in a predictable manner from the headwaters to the mouth of a river. This river continuum concept is based upon the geomorphological concept that watersheds are in a dynamic equilibrium in which stream channel characteristics, such as width, depth, velocity, and sediment load, although temporally variable, do tend toward a mean state. Biological organization will then conform to the dynamic physical conditions of the channel and hence also be in dynamic equilibrium. The concept further assumes that the biota develop to maximize energy utilization and minimize the variation in its use over the year.

The river continuum concept predicts that river systems show a shift from strongly heterotrophic headwater reaches (Orders 1-3) to autotrophic mid-reaches (Orders 4-6) and then a gradual return to heterotrophic processes in larger downstream reaches (>6). Headwater streams are strongly influenced by riparian vegetation which shades out most primary producers and contributes great quantities of organic matter by means of leaf fall. In the mid-size reaches the algal and rooted vascular plant production increases in importance as the degree of shading by riparian vegetation decreases. Consequently, the ratio of gross primary productivity to community respiration may exceed 1 in the mid-reaches. Large rivers depend to a lesser extent on

primary production and receive most energy in the form of fine particulate organic matter from upstream processing of leaves and debritus. Primary production in large rivers may be limited by depth and turbidity and consequently, are heterotrophic (Vannote et al. 1980).

The biota of rivers reflect the changing sources and characteristics of available energy according to the river continuum concept (Vannote et al. 1980). Headwaters are dominated by invertebrates in the shredder and collector groups and invertivore fishes. In the mid-reaches, grazers become more important and shredders less so. Invertivorous and piscivorous species of fish dominate in these reaches. As stream size increases further collectors dominate the macroinvertebrate assemblages and some planktivorous fishes may be found.

The Kanawha River is a large river and we would expect the ecosystem to be primarily heterotrophic. However, the relative importance of primary production to fine particulate organic matter inputs to the higher trophic levels has not been established in any large river system. Furthermore, the river continuum concept was developed specifically in reference to unperturbed river ecosystems. Consequently, one would expect impoundment, municipal and industrial effluents, and barge traffic to affect the quantity and quality of transport and the relative degree of autotrophy-heterotrophy, thereby shifting the overall continuum.

Compartment models are most frequently employed for ecosystem modeling efforts. Groups of species or ecosystem components with similar ecological functions are aggregated into compartments to keep the number of state variables to a reasonable number. At the crux of this aggregation problem is the selection of system components to simplify the interactions and still maintain ecological realism and minimize error due to aggregation (Cale and Odell 1979, O'Neill and Rust 1979). Once the components are identified, the next step is to identify the interactions between system components and factors that control the interactions. The mathematical form of compartment models is a system of differential equations, representing rates of energy inputs and outputs for compartments. The explicit formulation of rates for energy flow between compartments can be simple or quite detailed and complex. The use of linear donor-dependent flows, in which energy flows are a function of the quantity of the donor compartment, is appropriate for ecosystem level modeling (Patten 1975) and provides for a stable, well behaved model (Mulholland and Keener 1974).

Considerable advancements have been made in the area of water quality modeling for stream sanitation purposes (James 1984). However, these models are usually developed for evaluating dissolved oxygen levels, biochemical oxygen demand, ammonia and other parameters for the purpose of managing waste land allocations in rivers. Few of these models include detailed representatives of higher trophic groups.

Another component of aquatic ecosystems that has received considerable attention has been nutrient cycling and plankton productivity in lakes (Canale 1976). The purpose of this area of research is to develop better tools to predict effects of eutrophication in lake environments. In most of these models only the phytoplankton and zooplankton components are included in the model. Therefore, they are not total ecosystem models.

A total ecosystem model has been developed for a generalized lake ecosystem (Park et al. 1974; Scavia and Park 1976; Park et al. 1978). The original model, CLEAN (Comprehensive Lake Ecosystem Analyzer), was developed by a group of 25 scientists supported by the Eastern Decidous Forest Biome, U. S. International Biological Program (Park et al. 1974). CLEAN consists of twenty-eight differential equations which represent sixteen compartments of lake ecosystems, including attached aquatic plants, phytoplankton, zooplankton, bottom-dwelling aquatic insects, fish, suspended organic matter, decomposers, sediments, and nutrients. Fish are represented by three functional groups, benthos by two, zooplankton by three, and phytoplankton by two. Driving variables in the model include incident solar radiation, temperature, nutrient loadings, wind, and inputs of organic matter. The differential equations were defined to be as realistic as possible, therefore, the model includes many nonlinearities and feedback effects. The CLEAN model has been modified in succeeding versions (CLEANER, Scavia

and Park 1976; MS. CLEANER, Park et al. 1978) to incorporate new findings of research on processes in lake ecoystems. The original version was calibrated and verified with data from Lake George, New York, and subsequent versions have been applied to several diverse lakes in the U.S. and Europe to address environmental problems such as nutrient enrichment, thermal pollution, siltation, impoundment, and fish removal (Park et al. 1978).

Although the CLEAN model can be applied to reservoirs, it is not totally appropriate since reservoirs have greater water exchange rates and water level fluctuation than natural lakes. Patten (1975) described a computer simulation model of a reservoir cove ecosystem. This model, formulated by 42 investigators, describes the ecosystem of a 4.6 - ha cove in Lake Texoma, Texas - Oklahoma. The model describes the biomass or element concentrations of 33 compartments in 5 submodels, 9 compartments in the decomposer submodel, 9 primary producers, 2 zooplankton, 9 vertebrates, and 4 macroinvertebrates. The major driving functions were solar radiation, temperature, precipitation, wind, and water level. Energy and material flows were modeled in a donor-dependent fashion. The reservoir cove model provided a realistic and reasonable simulation of both state and output variables under normal conditions. The model was also used to describe the magnitude of changes that may occur in the cases of thermal pollution, phosphorus enrichment, and

introduction of a new piscivorous predator. Patten (1975) emphasized that no theory or method other than total ecosystem modeling is capable of predicting the far-reaching effects of these perturbations.

Total ecosystem models of reservoirs have not yet been routinely applied in environmental impact assessments on fisheries management because of the detailed data requirements. Ploskey and Jenkins (1982) developed a model that calculates production of food organisms required to produce and maintain biomass of fish. The biomass of fish is partitioned into six types; biomass supported by (1) plants (phytoplankton, macrophytes, periphyton), (2) detritus, (3) benthos, (4) zooplankton, (5) fish, (6) and terrestrial insects. Although this model cannot be used to simulate changes that may occur under different environmental conditions, it can be used to assess whether particular food types may be limiting fish production.

Most lotic ecosystem models and research have focused on small headwater streams and few detailed studies have been carried out on larger systems. Since the Kanawha River has been impounded for navigation, the lower reaches of the Winfield pool may be more similar to a reservoir ecosystem while the upper end of the pool is definitely riverine. Since the modeling framework and techniques utilized in small streams are relevant to our modeling efforts, these studies will be reviewed here. Small

stream models necessarily emphasize benthic processes and large river modeling will require consideration of water column processes as well.

Research and modeling efforts on small streams have concentrated on detrital processing (Boling et al. 1975) and periphyton dynamics (McIntire 1973), and annual energy budgets (Fisher and Likens 1973). Fisher and Likens (1973) estimated that 97% of the annual input of energy to a small second-order stream was allochthonous while primary production by mosses accounted for less than 1% of the annual energy input. Warren et al. (1964), and Warren et al. (1971) estimated energy flows in a small stream and Ruttledge et al. (1976) developed a linear donor-controlled model based on these data. McIntire (1973) developed a detailed model to simulate periphyton dynamics in flowing waters; the model included nonlinear effects of substrate, nutrients, temperature, current velocity, and light intensity, and silt load. Based on the periphyton model and the functional group concepts (Cummins 1974), McIntire and Colby (1978) developed a total stream model to simulate dynamics of small lotic ecosystems in the coniferous forest biome, northwestern United States. This model consists of 14 state variables, and the model structure is based on 7 basic processes: periphyton dynamics, grazing, shredding, collecting, invertebrate predation, vertebrate predation, and detrital conditioning. Although the goal of the McIntire and Colby (1978) model was to

increase understanding of lotic ecosystem processes and behavior, the model was also used to simulate the effects of clear-cut logging and slash burning on stream dynamics. Several new hypotheses about small stream processes were also developed through the model building process. Functions expressing light limitation due to suspended load could be adapted to a large river ecosystem model and are especially relevant to navigation impacts.

Large river ecosystem modeling is understandably a difficult task due to the logistics of studying large rivers, the upstream influences on quality and quantity of seston transported, the intensive level of effort required to obtain sufficient precision in estimates of production, and the ubiquitous influences of man-induced perturbations. Consequently, few modeling studies have been conducted on large rivers and those that have usually emphasized one component or a few major components. Webster et al. (1979) developed a model of particulate organic matter dynamics to simulate transport and utilization along the entire length of a river. This transport model was also used to assess the effects of impoundment and flooding on suspended particulate organic matter. The functions used to model transport, deposition, and resuspension of particulate organic matter are particularly relevant to the Kanawha River ecosystem model.

Chen and Orlob (1975) developed many fundamental concepts, mathematical formulations, and solution techniques for modeling

reservoir and riverine ecosystems. As an example, Chen and Wells (1976) developed a model of the Boise River, Idaho, for investigating waste water management alternatives to protect and enhance the river's water quality. The model simulates 24 water quality and biological parameters including temperature, toxicity, total suspended solids, coliform bacteria, BOD, dissolved oxygen, ammonia, nitrite, nitrate, phosphate, alkalinity, pH, floating algae (two types), benthic algae (two types), zooplankton, insects, detritus, organic sediment, benthos, and fish (cold-water game fish, warm-water game fish, and benthic feeders). This model was used to analyze ecosystem responses to pollution, reduced river flows, and improved irrigation return water. Although the model functions for water quality parameters are beyond the scope of the Kanawha River modeling efforts, the approach to modeling transport and deposition of suspended organic matter is relevant.

The collection of detailed data on sources and fates of energy and productivity at all trophic levels in large rivers has limited progress in modeling. In fact, the advances in theoretical formulations and technical computing methods have underscored the inadequacies in our data bases on large rivers. The studies by Mann (1964, 1965) and Mann et al. (1972) are the only published reports on productivity and patterns of energy flow at all trophic levels in a large river. These studies were done on the River Thames near Reading, England; this lowland

river is modified with small locks and low dams and is subject to considerable sewage enrichment. Several conclusions from Mann et al. (1972) are relevant to our modeling efforts. (1.) The two most numerous fish species, the roach Rutilus rutilus and the bleak Alburnus alburnus), rely heavily on benthic detritus. In their first year of life, these fish rely heavily on zooplankton (rotifers and cladocerans) and chironomid larvae, while older roach and bleak rely on organic detritus, terrestrial insects taken at the surface, and algae. (2.) Substantial periphyton production occurred at depths up to 1 m, beyond which light was severely limiting. (3.) Production of benthic animals was 331 kcal/m^2/year. (4.) Fish production was 198 kcal/m^2/year, of which roach and bleak accounted for 137 kcal/m^2/year; 69% of roach and bleak production occurred during their first year of life. (5.) A large biomass of bivalve molluscs was very important in reducing the load of suspended solids. (6.) Nutrients derived directly from sewage effluents supported a very rich phytoplankton production (1,907-4,388 kcal/m^2/year). (7.) The system as a whole was heterotrophic.

Recently, a large research program was initiated to develop a river ecosystem model for navigation pools on the Upper Mississippi and Illinois River (Sparks 1984). The conceptual biological model developed consists of 7 habitat compartments and 22 state variables. In addition a mathematical model for water and sediment transport is also being developed. Output from the

water and sediment model will be used as input to the biological model.

IV. METHODS AND MATERIALS

IV.0. WATER QUALITY

Temperature and Specific Conductance

Vertical profiles were taken at depths of 0, 0.5, 1, 2, 3 m, and every 2 m deeper where appropriate. These data were recorded monthly at UW and LW from October 1982 - September 1983. Measurements were made in the field using a Hydrolab® temperature/conductivity meter (Model TC-2) with submersible probe.

pH and Alkalinity

Samples were collected from Top and Bottom at UW and LW with a Niskin water bottle, placed in dark polyethylene 500 ml bottles and stored on ice for 24-48 hr. In the laboratory, pH was measured with an Altex ChemMate® pH meter on water samples from April - September 1983. Alkalinity was measured in the laboratory on samples from October 1982 - September 1983 by the Titrimetric Method of Standard Methods (American Public Health Association et al. 1981), using methyl purple as the indicator.

Dissolved Oxygen

Vertical profiles were taken at depths of 0, 0.5, 1, 2, 3 m and every 2 m deeper where appropriate, at UW and LW, each month between October 1982 and September 1983. Samples were collected with a Niskin water bottle, fixed through the acidification step

of the Winkler Dissolved Oxygen Method (American Public Health Association et al. 1981), and stored in the dark on ice for 24-48 hr. In the lab, the titration of the samples was completed as described in Standard Methods (American Public Health Association et al. 1981) for the Winkler Method.

Transparency and Light Penetration

Measurements of transparency were determined with the Secchi disc at UW and LW. The Secchi disc is a circular plate, 20 cm in diameter, with alternating black and white quadrants. As it is lowered through the water column, the depth at which it is no longer visible is determined (Lind 1979). Vertical profiles of light intensity were recorded to a depth of approximately 1% of the surface light intensity, where that depth was less than the maximum depth. A Montedero-Whitney® light meter (Model LMD-8A) was used monthly from October 1982 - February 1983, and a LiCor® photometer (Model LI-185B) was used monthly from March - September 1983 (with the exception of July, when equipment malfunctioned). Vertical light intensity was expressed as a percentage of the incident light at the surface, and a vertical extinction coefficient (n) was determined by the following equation (Lind 1979):

$$n = \frac{\ln I_o (Z) - (z \ [\ln I_z 0])}{z^2}$$

where the surface light intensity (I_o) and the light intensity (I_z) for each depth (z) was measured.

IV.1. TERRESTRIAL INPUTS

Leaf Fall

The input of leaves falling from riparian vegetation was estimated by placing traps on the shore in the vicinity of the river where the aquatic samples were collected. Leaf-fall samples were collected monthly from April - December 1983. Originally, four traps were deployed at each of four locations: Upper Winfield - Left Descending Bank (UW-LD), Upper Winfield - Right Descending Bank (UW-RD), Lower Winfield - Left Descending Bank (LW-LD), and Lower Winfield - Right Descending Bank (LW-RD). We adhered to this design from April - June 1983, but there were problems with vandalism at two locations frequented by the public (UW-RD and LW-LD). Therefore, we attempted to collect eight replicate samples at UW-LD and LW-RD from July - December 1983. However, all of the leaf-fall traps were destroyed by highway workers at UW-RD in November and December 1983.

The leaf-fall traps were constructed by fencing a 0.5 x 0.5 m area with 1-in mesh poultry cloth. The poultry cloth was 61.0 cm high, and two layers were wrapped around four stakes at the corners of the 0.25 m^2 area. Black polyethylene covered the

bottom of the trap and extended approximately 15 cm beyond all sides of the trap to prevent the intrusion of ground vegetation. The traps were placed in areas of trees and shrubs that were representative of the riparian vegetation.

At monthly intervals the leaves (and other vegetative material from the overstory) were collected from each trap, placed in individual plastic bags, and returned to the laboratory. The samples were then dried at 60° C for 24 hr and weighed to the nearest 0.1 g on an Ohaus Dial-O-Gram® triple-beam balance.

In order to extrapolate the data from the leaf-fall traps to the entire Winfield Pool, the riparian vegetation was surveyed along the entire length of the pool in the summer of 1983. The purposes of the survey were: (1) to estimate the area of canopy that extends over the surface of water; and (2) to estimate the area of flood plain that would be inundated during the average annual peak discharge. These parameters were estimated on both banks at every river mile from Marmet Locks and Dam to Winfield Locks and Dam. The total input of leaves from the overhanging canopy and the potential input from the flood plain on each bank of each river mile were then estimated by multiplying the leaf-fall/m^2 by the estimated areas (m^2).

The dry weights of leaves were converted to caloric equivalents by using values from the literature (Cummins and Wuycheck 1971). The caloric equivalents (Kcal/gDW) of two

families of trees that were common on the banks were averaged to obtain a conversion factor of 4.61 Kcal/gDW. The values used to obtain this average were: 4.54, Magnoliaceae and 4.68, Leguminosae.

Leaf Blow In

The input of leaves that fall on the ground and then are blown into the water was also estimated by placing traps on the river banks in the vicinity of the aquatic sampling. Blow-in samples were collected monthly from February - May and October - December 1983. Sampling was discontinued between June and September because blow-in was negligible, due to the heavy brush that grew along the river and the lack of leaves falling from the trees during this period. Originally, three traps were placed at the same four sites as the leaf-fall traps: UW-LD, UW-RD, LW-LD, and LW-RD. One trap that was placed in a meadow at LW-RD never collected any leaves and was omitted from all analyses. The same design was used from February - May 1983, but some samples were lost because of vandalism. From October - December 1983, we attempted to use three traps at UW-LD and two traps at LW-RD. However, the traps at UW-LD were destroyed by highway workers in November and December.

The blow-in traps were constructed by placing 2-m lengths of 1-in mesh poultry cloth parallel to the river. The poultry cloth was 30 cm tall and was secured by wood stakes. In order to retain the leaves more effectively, there were 0.5 m sections of

poultry cloth attached to each end of the trap. These "wings" extended shoreward at right angles to the trap. The bottom of the enclosure was covered with black polyethylene so that the blow-in sample could be distinguished from the vegetation growing on the ground. The leaves were collected from the blow-in traps and analyzed like the leaf-fall samples.

The results from the blow-in traps were expressed as gDW/m. These data were extrapolated to the entire Winfield Pool by the same methods as leaf fall. Dry weights were converted to caloric equivalents in the same manner as the leaf-fall samples.

Insect Fall

The amount of terrestrial insects that fall into the water from the trees that overhang the water was estimated by traps placed on the shore in the vicinity of the aquatic sampling. Insect fall was sampled from April - November 1983. Originally, four insect-fall traps were placed at the same four sites as the leaf-fall and blow-in traps: UW-LD, UW-RD, LW-LD, and LW-RD. Because of vandalism, this design had to be modified like that for the leaf-fall sampling.

Insect-fall traps were made from plastic dish pans that measured 30 x 35 x 15 cm. The pans were placed under trees that were representative of the riparian canopy, and approximately 2 l of ethylene glycol (automobile antifreeze) was added to each pan. At monthly intervals, the contents of each pan were poured through a No. 45 sieve (106 µm mesh), and the insects were placed

in 5% formalin and returned to the laboratory. In the laboratory, the samples were sorted, and the insects that are known to dwell in trees were dried at 60° C for 24 hr and weighed to the nearest 0.01 g on a Mettler® Model AE163 electronic balance. The ground-dwelling insects were discarded under the assumptions that they crawled into the traps, and they would not fall into the water from trees.

The data from the insect-fall traps (gDW/m^2) were extrapolated to the entire Winfield Pool in the same manner as the leaf fall from overhanging tree canopy. The dry weights of insects were converted to caloric equivalents by using the value of 5.82 Kcal/gDW reported for mixed insects by Cummins and Wuycheck (1971).

IV.2. BENTHIC DETRITUS

Samples of bottom sediments were collected with a coring device in order to estimate the availability of fine particulate organic matter to detritivores and to provide information on detritus dynamics in the pool. Five samples were collected monthly at LW and UW from January - December 1983. The sampling device was a Phleger corer manufactured by Kahl Scientific Instrument Corporation (Model 217WA200). The sampler was dropped randomly in the vicinity of the benthic macroinvertebrate artificial substrate samplers until five cores of sediment were

retrieved. The Phleger corer had to be dropped and retrieved many times because the bottom proved to be too hard for penetration (hardpan clay, gravel, cobble) or too loose to remain in the tube (coarse sand) in many locations.

When a successful core sample was brought to the surface, the contents of the tube were washed with a squeeze bottle into a plastic container and sealed for transportation. In the laboratory the samples were transferred to beakers and allowed to settle for 48 hr. Most of the water above the sediment was removed by pipette, and the sediment was dried at 60° C for 24 hr. A subsample of the dry sediment was weighed in a tared aluminum pan to the nearest 0.00001 g and then ashed by standard procedures (see Seston section of Methods & Materials).

Results could only be reliably reported as percent organic content. Because of the sandy sediment and the effect of current on the sampler as it descended, the material was not retained in the coring tube in a fixed vertical profile as it occurred on the river bottom. It would have been inaccurate to report results as weight of organic matter per unit volume of sediment because the deep and shallow sediments usually became mixed.

IV.3. SESTON

Seston was sampled monthly at UW and LW from October 1982 - September 1983. At each study site, seston was collected from

one location (at macroinvertebrate artificial substrate samplers). Our previous reconnaissance of the Kanawha River (Voshell et al. 1982) indicated that there were no significant differences in seston across a transect of the river. Seston was collected at two depths: near surface, to represent the photic zone, and a deeper sample, to represent the water column below the photic zone. At both sites the near-surface samples (designated Top) were collected 1m below the surface. At UW the deep samples (designated as Bottom) were taken 1m above the bottom. In December 1982 the current was too swift at UW to collect Bottom samples, but the Top samples were representative of the entire water column under those flow conditions. At LW the Bottom samples were always collected at a depth of 9 m. In March 1983, Bottom samples could not be collected at LW because of high flow conditions.

Sufficient seston samples were collected at each site and depth to analyze the following components: (1) weight of organic and inorganic matter, (2) biomass of phytoplankton, and (3) density of bacteria. Throughout the study we collected three replicate seston samples for weight determinations and a single sample for bacterial densities. From October 1982 - February 1983 we collected a single sample for determining phytoplankton biomass, but from March - September 1983 we collected three replicate samples. In order to analyze the size distribution of the suspended particles, seston was separated into six size

classes: > 1000 µm (large - L), 234-1000 µm (medium large - ML), 105-234 µm (small - S), 43-105 µm (fine - F), 25-43 µm (very fine - VF), and 0.45-25 µm (ultrafine - UF).

Seston was collected by means of a gasoline-powered pump (Homelite Waterbug®). The volume of water was measured by calibrating the discharge of the pump and then pumping water for a specified length of time. The volumes of water that had to be filtered to obtain measurable weights of seston varied according to the discharge of the river. The volumes required for measurable weights of seston in the VF-L size classes varied from 1 - 1000 l and for the UF size class from 0.250 - 1.500 l (Table IV.3.1).

The following methods for concentrating seston, separating it into size classes, and weighing the quantities in each size class were derived from Gurtz et al. (1980) and Voshell and Parker (1984). The methods for determining the biomass of phytoplankton and density of bacteria are explained in Sections IV.4 and IV.5, respectively. Seston in the VF-L size classes was first concentrated by pumping the desired volume of water into a 25 µm plankton net. The seston that was retained in the plankton net was then filtered by vacuum (15 psi at pump, 5-7 psi effective at filter) through a series of five stainless screens of successively finer mesh to separate it into the desired size classes. The screens were replaced with duplicates as necessary during the filtering process to prevent them from clogging and

Table IV.3.1. Volumes of water filtered for determining weights of seston.

Date	Site	Depth	Size Class L-VF	Size Class UF
19 Oct 1982	UW	Top	100	0.675
	UW	Top	100	0.675
	UW	Top	100	0.675
	UW	Bot	100	0.750
	UW	Bot	100	0.750
	UW	Bot	100	0.750
18 Oct 1982	LW	Top	100	0.650
	LW	Top	100	0.650
	LW	Top	100	0.650
	LW	Bot	100	0.570
	LW	Bot	100	0.570
	LW	Bot	100	0.570
19 Nov 1982	UW	Top	400	0.650
	UW	Top	400	0.670
	UW	Top	400	0.650
	UW	Bot	400	
	UW	Bot	400	0.500
	UW	Bot	400	0.500
18 Nov 1982	LW	Top	1000	1.000
	LW	Top	1000	1.000
	LW	Top	1000	1.000
	LW	Bot	1000	0.900
	LW	Bot	1000	0.800
	LW	Bot	1000	0.800
17 Dec 1982	UW	Top	10	0.250
	UW	Top	10	0.250
	UW	Top	10	0.250
	UW	Bot		
	UW	Bot		
	UW	Bot		
16 Dec 1982	LW	Top	800	0.900
	LW	Top	800	0.900
	LW	Top	800	0.900
	LW	Bot	800	1.000
	LW	Bot	800	1.000
	LW	Bot	800	1.000

Date	Site	Depth	Size Class L-VF	Size Class UF
21 Jan 1983	UW	Top	500	1.500
	UW	Top	500	1.500
	UW	Top	500	1.250
	UW	Bot	500	1.000
	UW	Bot	500	1.000
	UW	Bot	500	1.000
20 Jan 1983	LW	Top	1000	1.000
	LW	Top	1000	1.000
	LW	Top	1000	1.000
	LW	Bot	1000	1.000
	LW	Bot	1000	1.000
	LW	Bot	1000	1.000
18 Feb 1983	UW	Top	50	1.000
	UW	Top	50	1.000
	UW	Top	50	1.000
	UW	Bot	50	0.700
	UW	Bot	50	0.700
	UW	Bot	50	0.700
17 Feb 1983	LW	Top	500	1.000
	LW	Top	500	1.000
	LW	Top	500	1.000
	LW	Bot	500	1.000
	LW	Bot	500	1.000
	LW	Bot	500	1.000
25 Mar 1983	UW	Top	100	0.400
	UW	Top	100	0.400
	UW	Top	100	0.400
	UW	Bot	100	0.700
	UW	Bot	100	0.500
	UW	Bot	100	0.600
24 Mar 1983	LW	Top	10	0.300
	LW	Top	10	0.300
	LW	Top	10	0.300
	LW	Top	10	0.300
	LW	Bot		
	LW	Bot		
	LW	Bot		

Date	Site	Depth	Size Class L-VF	Size Class UF
22 Apr 1983	UW	Top	50	1.000
	UW	Top	50	1.000
	UW	Top	50	1.000
	UW	Bot	50	1.000
	UW	Bot	50	1.000
	UW	Bot	50	1.000
21 Apr 1983	LW	Top	150	1.000
	LW	Top	150	1.000
	LW	Top	150	1.000
	LW	Bot	150	1.000
	LW	Bot	150	1.000
	LW	Bot	150	0.850
19 May 1983	UW	Top	15	0.500
	UW	Top	15	0.500
	UW	Top	15	0.500
	UW	Bot	15	0.500
	UW	Bot	15	0.500
	UW	Bot	15	0.500
20 May 1983	LW	Top	25	0.500
	LW	Top	25	0.500
	LW	Top	25	0.500
	LW	Bot	2*	0.250
	LW	Bot	2*	0.250
	LW	Bot	2*	0.250
22 June 1983	UW	Top	50	0.500
	UW	Top	50	0.500
	UW	Top	50	0.500
	UW	Bot	50	0.500
	UW	Bot	50	0.500
	UW	Bot	50	0.500
23 June 1983	LW	Top	50	0.400
	LW	Top	50	0.400
	LW	Top	50	0.400
	LW	Bot	50	0.400
	LW	Bot	50	0.400
	LW	Bot	50	0.400
20 July 1983	UW	Top	150	0.900
	UW	Top	150	0.900

Date	Site	Depth	Size Class L-VF	Size Class UF
	UW	Top	150	0.900
	UW	Bot	150	0.900
	UW	Bot	150	0.900
	UW	Bot	150	0.900
21 July 1983	LW	Top	100	0.500
	LW	Top	100	0.700
	LW	Top	100	0.600
	LW	Bot	100	0.350
	LW	Bot	100	0.350
	LW	Bot	100	0.350
24 Aug 1983	UW	Top	100	0.700
	UW	Top	100	0.700
	UW	Top	100	0.700
	UW	Bot	100	0.700
	UW	Bot	100	0.700
	UW	Bot	100	0.700
25 Aug 1983	LW	Top	150	0.750
	LW	Top	150	0.750
	LW	Top	150	0.750
	LW	Bot	150	0.750
	LW	Bot	150	0.750
	LW	Bot	150	0.750
14 Sept 1983	UW	Top	75	0.500
	UW	Top	75	0.500
	UW	Top	75	0.500
	UW	Bot	75	0.500
	UW	Bot	75	0.500
	UW	Bot	75	0.500
15 Sept 1983	LW	Top	100	0.750
	LW	Top	100	0.750
	LW	Top	100	0.750
	LW	Bot	100	0.750
	LW	Bot	100	0.750
	LW	Bot	100	0.750

* Only 1 liter filtered for VF.

changing the effective pore size. In order to determine the weight of seston in each size class, the material collected on each screen was washed with distilled water onto separate preashed, preweighed Gelman® 47 mm type A/E glass fiber filters. In order to obtain seston in the UF size class, a measured volume of water was taken directly from the river, passed through a VF screen, and then filtered through a glass fiber filter. All seston samples were filtered at the study sites within a few hours after collection. The glass fiber filters were kept in the aluminum weighing pans in which they were ashed and tared, and transported to and from the study sites by placing the aluminum pans in small metal cans with tight-fitting lids.

In the laboratory, the glass fiber filters were dried at 60° C for 24 hr, cooled in a dessicator, and weighed to the nearest 0.00001 g on a Mettler® Model AE163 electronic micro balance to determine the dry weight. The filters were then ashed at 500° C for 1 hr, dampened with distilled water to restore the water of hydration, dried and cooled again, and reweighed to determine the ash weight. The original dry weight was considered to be the total seston, the ash weight was considered to be the inorganic seston, and the ash-free dry weight (loss on ignition) was considered to be the organic seston. The final values were reported as seston weight per unit volume of water (gDW/l, gASH/l, gAFDW/l, respectively).

IV.4. BACTERIA

Sampling

Water samples were collected and separated into size classes by the same methods used for seston analyses (Section IV.3). The filtration column was rinsed with 70% ethanol prior to use. The screens were backwashed and the collected material was suspended in 50 or 100 ml of distilled water. The UF size fraction was collected from unconcentrated water that passed the 25 µm screen. The 50 or 100 ml samples were put in acid washed, autoclaved glass bottles, and fixed with formalin to a final concentration of 2% (7 ml of formalin in a 100 ml sample, and 3.5 ml of formalin in a 50 ml sample).

Density and Relative Abundance

Before sample analysis, all pipet tips, small petri dishes, scintillation vials, glass parts of the filtration apparatus, and a large flask of distilled, deionized water were autoclaved. The Gelman 25mm type A-E glass fiber filters were ashed for 1 hr at 500°C, and the scintered glass support was rinsed in chromic acid and dried for 1 hr at 70°C. The staining procedure generally followed the method of Porter and Feig (1980). The acridine orange (AO) working solution was prepared by filtering 4.9 ml of distilled, deionized, autoclaved (d/d/a) water and 0.1 ml of AO stock (10 mg/ml in 2% formalin), through a 25mm cellulose fiber nuclepore filter (pore size 0.2µm). The working solution was

kept in a scintillation vial in a beaker of ice while in use, and discarded within 24 hours. The stock solution was kept at 10°C in a foil wrapped container. The nuclepore filters were soaked in Iragalan Black (2 g/l in 2% acetic acid) for a period of at least ten minutes, but not exceeding 24 hours. The Iragalan Black was also stored in a foil wrapped container at 10°C. After the nuclepore filters were soaked for the appropriate length of time, a blank was prepare by placing a nuclepore filter, pre-soaked in Iragalan Black and rinsed twice in d/d/a pre-soaked water, on top of a glass fiber filter to avoid clumping. After the filtration chimney had been placed on the apparatus, 1 ml of d/d/a water and 0.1 ml of AO were added. The AO and water were allowed to stand for two minutes before filtering. After filtering, the nuclepore filter was briefly dried. One drop of Cargille type B immersion oil was then placed on a clean acetone rinsed slide on top of which the air dried filter was placed. Another drop of oil was placed on the filter and then covered by a clean acetone rinsed No. 1 cover slip. The samples were processed in the same way as the blanks. A recorded volume of formalin fixed sample was placed on the filtration apparatus and 0.1 ml of AO was added. After two minutes, the sample was filtered and rinsed with two ml of d/d/a water and the filter was air dried. The filter was mounted in the same way as described for the blank. The volume of sample filtered was determined by the appearance of the samples. Volumes ranged from 1-30 ml:

Larger volumes were usually required for the large to small size fractions. After the slides were made they were numbered according to size fraction and stored flat at 10°C.

The slides were viewed on an epifluorescent microscope unthe 100X, oil immersion objective. Bacteria were counted using a Whipple grid with a total area of 4900μm^2. The grid was divided into large squares of 7x7μm^2 and smaller squares of 1.4x1.4μm^2. The size of the bacteria were estimated by using the smallest grid. Usually, ten fields were counted per slide; however, if the number of bacteria exceeded 50 per field, only five fields were counted. The organisms were divided into Bacteria, Nonliving, and Living (including organisms other than procaryotes). The number of bacteria, size of the bacteria, area of nonliving matter, and area of living matter were recorded for each field counted.

The percentages of each group were derived by finding the area in μm^2 that the bacteria and nonliving particles covered. For samples filtered through the 25 µm mesh net in the field (filter sizes L, ML, S, F, VF), the calculation for density was:

$$\text{No. Bacteria/ml} = A * B * 1/C * D * 1/E$$

Where: A = x no. bacteria counted per field from a given filter

B = Total number of fields per filter

C = Total volume of filtered river water

D = Formalin fixed volume

E = Volume from D passed through nucleopore filter.

For unfiltered field samples, the equation used was:

No. Bacteria/ml = A * B * 1/E

Respiration and Production

An estimate of the contribution of bacteria to oxygen consumption in the plankton was calculated from experiments where the bacteria were physically separated from the larger plankton. These experiments were conducted at LW during the sampling trips for June - September 1983. One experiment was conducted during the night, and one experiment was conducted the following day. Water was collected from a depth of 1 m with a Niskin sampler. Six sterile 60 ml BOD bottles were filled immediately, and a duplicate set was filled with water which had been passed through a 3µm Nuclepore® filter. Samples of filtered and unfiltered water were collected to check filtering efficiency. Phytoplankton samples were preserved with Lugol's Solution (American Public Health Association 1981), and for bacterial counts, water was placed in a sterile bottle and preserved with formalin. Bottles were sterilized by autoclaving, and other equipment was sterilized in the field by rinsing with 70% ethanol

followed by 10 rinses with distilled water. For each set of 60 ml BOD bottles, 3 were fixed as initial dissolved oxygen samples; 0.5 ml of a 116 mg/l suspension of DCMU was added to each of the other 3 samples (Golterman 1971), and the latter bottles were replaced at the 1 m depth. Following an incubation of 7 to 9 hr, these samples were fixed for dissolved oxygen analyses to the acidification step of the Azide Modification of the Winkler Method (American Public Health Association et al. 1981). All samples were stored in the dark on ice for 12 - 36 hr. In the lab, the dissolved oxygen samples were titrated with 0.0125 N sodium triosulphate directly into the sample bottle after removing 10.00 ml. The endpoint was monitored photometrically (Carpenter 1965) with a Perkin-Elmer UV-VIS® Spectrophotometer (Model Lambda 1) at a wavelength of 350 nm (Bryan et al. 1976).

Phytoplankton samples were concentrated by settling and the cells were counted with an inverted compound microscope (see section IV.5). Bacterial samples were stained with acridine orange, filtered onto 0.2µm Nuclepore® filters, and counted with a Zeiss aus Jena® epifluorescent compound microscope (see section IV.4).

The daily oxygen consumption (CONS) by the total plankton and by the fraction smaller than 3 µm was calculated as:

$$\text{Daily CONS } (mgO_2/m^3/day) = \frac{(I - D,\ mgO_2/l)(1000/m^3)(24\ hrs/day)}{(\text{incubation period, hrs})}$$

where

I = mg O_2/l initial

D = mg O_2/l DCMU-treated bottle.

The contribution of the fraction smaller than 3µm was calculated as a percentage of the total community oxygen uptake. The average percent contribution from all experimental trials was applied to the annual community oxygen consumption rates derived by the light-dark bottle method (section IV.5) to estimate the annual bacterial production.

Annual bacterial production was derived from the estimate of annual bacterial oxygen uptake by the following relationship (derived from Sorokin and Kadota 1972):

Bacterial Production, mgC = (0.8)(Bacterial Oxygen uptake, mgO_2)

Annual bacterial respiration was estimated as follows (Winberg 1980):

Bacterial Respiration = 2 (Bacterial Production).

For inclusion in the model, results were converted to caloric equivalents by multiplying the mg carbon value by 2 mg dry weight/mg carbon (Sorokin and Kadota 1972) and again by 0.004414 Kcal/mg dry weight (Cummins and Wuycheck 1971).

IV.5. PHYTOPLANKTON

Taxonomic Composition and Cell Density

Samples were collected from Top and Bottom at UW and LW each month from October 1982 - September 1983. Samples were collected from Top and Bottom at UW station in November 1982, March, and August 1983. Water was collected with a Niskin water bottle, 50 ml samples were preserved with Lugol's solution (American Public Health Association et al. 1981), and 100-400 ml of the sample were concentrated on 0.45 µm Millipore® filters and stored in vials. The 50 ml samples were concentrated by sedimentation and the cells counted on a Leitz Diavert® inverted compound microscope (American Public Health Association et al. 1981). The Millipore filters were soaked in concentrated nitric acid to dissolve them and partially destroy the organic matter; potassium dichromate crystals completed the digestion and left clean diatom frustules. The diatoms were mounted in Hyrax® for identification slides.

General taxonomic references used for algae were: Huber-Pestalozzi (1938-1968), Kudo (1966), Prescott (1962), Smith (1950), Tiffany and Britton (1971), Whitford and Schumacher (1973). Cocke (1967), was used for blue-green algae and Hustedt (1930), Patrick and Reimer (1966, 1975), and Weber (1971) for diatoms.

Biomass

Biomass estimates for the phytoplankton community were based on chlorophyll a (chl a) measurements. Samples were collected from Top and Bottom at UW and LW each month from October 1982 - September 1983. The biomass of phytoplankton was determined for the same size classes as the seston weight analysis. Water samples were collected, concentrated, and fractionated by the same methods used for the seston (see section IV.3) and finally filtered onto Gelman® type A/E 47 mm glass fiber filters. In January and February 1983, one sample was collected at each site and depth, frozen, and later ground and extracted for chl a analyses in the laboratory. From March - September 1983, three replicate samples were collected at each site and depth, the filters ground and extracted in the field, and the samples stored on ice for 24-48 hr for transport to VPI & SU.

The Fluorometric Method (Wetzel and Likens 1979) was used to determine chl a concentration. The glass fiber filters were ground in a 10 ml tissue grinder with 5 ml 90% alkaline-buffered acetone. The sample, was rinsed with 5 ml of 90% alkaline-buffered acetone, poured into a 20 ml vial, and refrigerated in the dark for at least 24 hr. The extracted sample was centrifuged to remove particulate matter and glass fibers. The fluorescence of the supernatant was then determined on a Turner Designs® Fluorometer (Model 10-000R). Samples were acidified with 4N HCl and the pheophytin concentration was measured on the fluorometer. The fluorometer was calibrated with standard chl a

solutions. The concentration of chl *a* was determined by the following equation:

$$\text{chl } a \text{ } (mg/m^3) = \frac{(F)(\text{Fluorometer reading})(v)}{(V)}$$

where:

F = calibration factor

v = volume of extract in ml

V = volume of water filtered in liters.

The sum of the chl *a* concentrations for the different size fractions was used for total biomass estimates.

Estimates of the biomass for the months of October, November, and December 1982 were necessary to estimate the annual average for the model. Total cell counts of the phytoplankton were available for all three months, and a conversion factor to chl *a* was needed. The average ratio of (mg chl *a*/m^3): (total # cells/ml) for February - June 1983 was calculated to be 0.00865 at UW and 0.00645 at LW. These dates were chosen because the cell counts were in the same range as those for the months when the samples were lost and the relationships of chl *a* concentrations to cell counts were similar. Estimates of Top and Bottom chl *a* concentrations were made for October, November, and December 1982 by multiplying the total cell counts by these conversion factors.

For inclusion to the model, the concentrations of chl a had to be converted to kilocalories per unit area. Concentrations per cubic meter for Top and Bottom were each multiplied by half the maximum depth in meters for that particular site and date, and the sum of these two estimates represented the chl a per square meter. Conversion to kilocalories was made by multiplying by three constants: 17.05 mg Carbon/mg chl a (Paerl et al. 1976); 1.98 mg dry weight/mg Carbon (Lind 1979); and 0.004135 Kcal/mg dry weight (Cummins and Wuycheck 1971).

Gross Primary Production and Respiration

The Light-Dark Bottle Oxygen Method (American Public Health Association et al. 1981) was used for estimating community primary production and respiration rates. Water collected from a selected depth was enclosed in clear and opaque bottles and returned to the same depth for an incubation period. The theory is that photosynthetic organisms will produce oxygen in the light bottle, but not in the dark, while the entire community will consume oxygen for respiration in both the light and dark bottles. The gross oxygen production, or gross primary production, was then determined as the differences in dissolved oxygen concentration between the light bottle and dark bottle. Community respiration was the difference between the initial dissolved oxygen concentration and the dissolved oxygen concentration in the dark bottle.

Duplicate samples for light, dark and initial bottles (300 ml BOD bottles) were taken from depths of 5, 1, 2, and 3 meters with a Niskin water bottle, and the light and dark bottles were hung at their original depths. During periods of high discharge, only surface samples could be incubated because the bottles would not hang vertically in the strong current. The initial samples were fixed immediately, and the light and dark samples were fixed at the end of the incubation period, which lasted from 4.5 to 8 hr. Dissolved oxygen was determined by the Azide Modification of the Winkler Method. Samples were fixed in the field through the acidification step, stored in the dark on ice for 24-48 hrs, and titrated upon returning to the lab.

The results from duplicate samples were averaged for each depth and each treatment. The mean of all initial dissolved oxygen concentrations was taken if there was no stratification with depth. The results from the dark bottles were treated similarly if significant difference were not apparent. The deepest initial and dark bottle values were extrapolated to the rest of the water column below that depth.

Estimates of gross primary productivity (GPP) were calculated for each depth by the equation:

$$\text{Daily GPP } (mgO_2/m^3/day) = (\text{L-D, mg } O_2/1)(1000\ 1/m^3)/(R)$$

where:

L = mg O_2/l in light bottle

D = mg O_2/l in dark bottle

R = ratio of solar radiation for the incubation period/total for day.

The ratio of solar radiation available during the incubation period to the total solar radiation for the day was estimated from measurements taken the same day with a Belmont pyrheliometer. The GPP was calculated on an areal basis by integrating the GPP per cubic meter over the depth (in meters), assuming the GPP to equal zero at the depth to which only 1% of the light at the surface penetrated (as estimated from vertical light extinction). Negative values for GPP were assumed to be zero. The annual GPP per square meter was estimated by multiplying the mean of the daily GPP values from each month by 365 days.

Estimates of community respiration (RESP) were calculated for each depth by the equation:

$$\text{Daily RESP } (mgO_2/m^3/day) = \frac{(L\text{-}D,\ mgO_2/l)(1000\ l/m^3)/(R)}{(\text{Incubation period, hrs})}$$

where:

I = mg O_2/l initial

D = mg O_2/l dark bottle

R = ratio of solar radiation for the incubation period/total for day

Community respiration was calculated on an areal basis by integrating the respiration per cubic meter over the depth (in meters), assuming the rate of the deepest incubation to be the rate through the rest of the deeper water column. When all initial and all dark bottle results were averaged, the respiration per cubic meter multiplied by the maximum depth equaled the respiration per square meter. Negative values for respiration rate were assumed to be zero, except for May 1983, when positive results from an experiment on the relative contribution of bacteria to plankton respiration were substituted (see section V.4). The annual respiration per square meter was estimated by multiplying the mean of the daily respiration values from each month by 365 days.

Community respiration in the plankton included oxygen consumption by bacteria, zooplankton, and phytoplankton. Respiration was not partitioned among these three components each month, but was derived from relationships found in other studies (see section V.4). Bacterial oxygen consumption was estimated to be approximately 50% of the total plankton oxygen consumption, therefore, the annual phytoplankton respiration would be 50% of the annual oxygen consumption, expressed as dry weight, minus the annual zooplankton respiration, expressed as dry weight. The annual zooplankton respiration was calculated by dividing our

caloric estimate (section IV.7) by 0.004713 Kcal/mg dry weight, a general conversion factor for aquatic microconsumers (Cummins and Wuycheck 1971).

Phytoplankton metabolism and total oxygen uptake were converted to dry weight and caloric values as follows. Oxygen values were converted to carbon by multiplying by a factor of 0.375, the ratio of the atomic weight of carbon to the molecular weight of O_2. The carbon value was then multiplied by 1.98 mg dry weight/mg carbon (Lind 1979) and again by 0.004135 Kcal/mg dry weight (Cummins and Wuycheck 1971) which is a conversion factor listed for primary producers.

Extracellular Release

The extracellular release of photosynthetic products can be an important loss in algae. The loss of extracellular products in the phytoplankton of eutrophic canal reservoirs was reported to range from 0.2 to 20 percent of the total primary production and was correlated with chl *a* concentration (Fogg and Watt 1965). Based on the annual average chl *a* concentrations for the Kanawha River of approximately 3-4 mg/m^3 for both UW and LW, and the correlations of Fogg and Watt, extracellular release would be approximately 10% of the primary production.

IV.6 PERIPHYTON

Taxonomic Composition and Cell Density

Glass slides (25 x 75 mm) were used as artificial substrates for periphyton colonization. The slides were submerged for 3-5 weeks by placing them in a floating plastic rack which was attached by a cable to one of the buoys for the macroinvertabrate artificial substrates. Slides were collected, and clean ones replaced monthly at UW and LW from November 1982 - October 1983. Occasionally, the glass slides were stolen or damaged by river traffic or large debris. For these reasons, the periphyton were not sampled at UW in January, May, or August 1983, or at LW in September 1982.

Samples of natural substrates (sticks and stones) were also collected at UW in Novermber 1982, January, June, and August 1983, and at LW in November, December 1982, February, June, and August 1983. The substrates were placed in slide containers or small tubs and preserved with Lugol's solution (American Public Health Association et al. 1981).

In the laboratory, the entire glass slides were scraped and washed; an area of 10 cm^2 was scraped and washed from the natural substrates. The resulting cell suspension (approximately 100 ml) was brought to 250 ml with 0.2 M dibasic potassium phosphate, to prevent clumping of cells and detritus, and "whipped" in a blender for 30 seconds. An appropriate volume (0.2 to 25 ml) of this cell suspension was diluted to 50 ml with 0.2 M dibasic potassium phosphate and placed in a settling chamber as was used for phytoplankton. The settled cells were identified and counted

with a Leitz Diavert® inverted compound microscope. Half of the blended sample was centrifuged to remove the Lugol's solution and potassium phosphate, washed with distilled water, and digested with 30% hydrogen peroxide for 1 hour, followed by addition of potassium dichromat crystals. The diatom frustules cleaned by this treatment were mounted in Hyrax® for identification.

Periphyton species were identified by the same taxonomic references as mentioned previously for the phytoplankton (Section IV.5).

Biomass

Biomass was estimated as the chl *a* concentration on glass slides. Several additional glass slides were collected from the plastic racks on the same dates as those used for determination of taxonomic composition. The glass slides were scraped and washed, and the film was collected on a glass fiber filter (47 mm Gelman® Type A/E). The chl *a* content of the filter was determined by the same methods used for phytoplankton biomass (section IV.5), with a final calculation of

$$\text{chl } a \text{ (mg/m}^2\text{)} = \frac{(F)(\text{fluorometer reading})(v)(1000)}{A}$$

where

F = calibration factor

v = volume of extract in ml

A = area of glass slide in square meters.

Samples collected in November and December 1982 were placed on filters, packaged in dry ice, and sent to the U.S. Army Corps of Engineers Water Quality Laboratory in Huntington, WV for chl a analyses. Extraction in 90% acetone was followed by determination of chl a and phaeophytin by the Spectrophotometric Method (American Public Health Association et al. 1981).

The initial results were expressed as mg chl a per square meter of glass slide. To interpret the contribution of the periphyton to the river, the results had to be multiplied by the fraction of the water surface area overlying the available natural substrate (this would be equivalent to spreading the periphyton evenly across the width of the river). This surface area fraction was difficult to estimate due to the various natural substrates available, such as river bottom which receives enough light (probably less than 3 m most of the year), trees and vegetation submerged along the bank, and the man-made structures placed in the river. An estimate of 10 square meters of substrate per meter of river bank was made for the Winfield Pool. A conversion factor applicable to both UW and LW was calculated as follows. The pool has a length of approximately 58,900 m, an average width of 210 m, and, therefore, an area of approximately 1.2×10^7 m^2. The pool area available for substrate was determined as the pool length (58,900 m) multiplied by 2 banks, and by 10 m^2/m, or approximately 1.2×10^6 m^2 square meters. The ratio of

available substrate to total pool area was approximately 0.1. This factor was multiplied by all results for periphyton on glass slides for inclusion into the model.

Conversion from chl a to Kilocalories was made in the same manner as described for phytoplankton (section IV.5).

Gross Primary Production and Respiration

The Light-Dark Bottle Oxygen Method (American Public Health Association 1981) was modified for use with the glass-slide artificial substrates. Colonized half-slides (12.5x75 mm) were placed into 300 ml BOD bottles, which had been filled with water that had passed through a glass fiber filter (47 mm Gelman® type A/E). The light and dark bottles were hung just below the surface of the river for an incubation period of 3.25 to 8.25 hr. Duplicates of each treatment were used. Experiments were done monthly from November 1982 - September 1983, except when the half-slides were lost at UW in January, February, May and August 1983, and at LW in September 1983. After incubation, the samples were treated in the same manner as the dissolved oxygen samples phytoplankton production (section IV.5). The results from replicate samples were averaged.

Daily gross primary production (GPP) was calculated by the equation:

$$\text{Daily GPP } (mgO_2/m^3/day = \frac{(L - D,\ mg\ O_2/l)(0.3\ l)}{(A)(R)}$$

where

L = mg O_2/l in light bottle

D = mg O_2/l in dark bottle

A = area of half slide in square meters

R = ratio of solar radiation, incubation period: total radiation for the day.

The ratio of solar radiation available during the incubation period to the total solar radiation for the day was estimated from measurements taken the same day with a Belmont pyrhelimeter. The annual GPP was estimated by multiplying the mean of the daily GPP values from each month by 365 days.

Daily respiration (RESP) was calculated by the equation:

$$\text{Daily RESP } (mgO_2/m^3/day) = \frac{(I - D,\ mg\ O_2/l)(0.3\ l)(24\ hrs/day)}{(A)(\text{Incubation period, hrs})}$$

where

I = mg O_2/l initial

D = mg O_2/l dark bottle

A = area of half slide, in square meters.

The annual respiration was estimated by multiplying the mean of the daily respiration values from each month by 365 days.

The same conversion factor that was used for biomass (0.1) was used to extrapolate from these values based on substrate area

to values for the width of the river. Oxygen values were converted to Kilocalories for use in the model by the same calculation used for phytoplankton.

Extracellular Release

The extracellular release of photosynthetic products was assumed to be on the same order of magnitude as for the phytoplankton of the river, i.e., 10% of the primary production.

IV.7. ZOOPLANKTON

Design

Seasonal samples were collected monthly, October 1982 - September 1983, from UW and LW. Longitudinal samples were collected monthly at eight sites (Fig. IV.7.1) during peak rotifer abundance, July - October 1983. These sites were six miles apart, with site one at River Mile 32 (LW) and site eight at River Mile 74.

Seasonal and longitudinal samples were collected from mid-channel only, because a preliminary study conducted in June 1982, established that samples obtained from mid-channel were representative of sites left and right of mid-channel when sampled at the same depth (Voshell et al. 1983). Samples were collected in duplicate October 1982 - February 1983, and in triplicate March - October 1983 (Green 1977). Seasonal samples were collected 1 m below surface and 1 m above the sediment-water

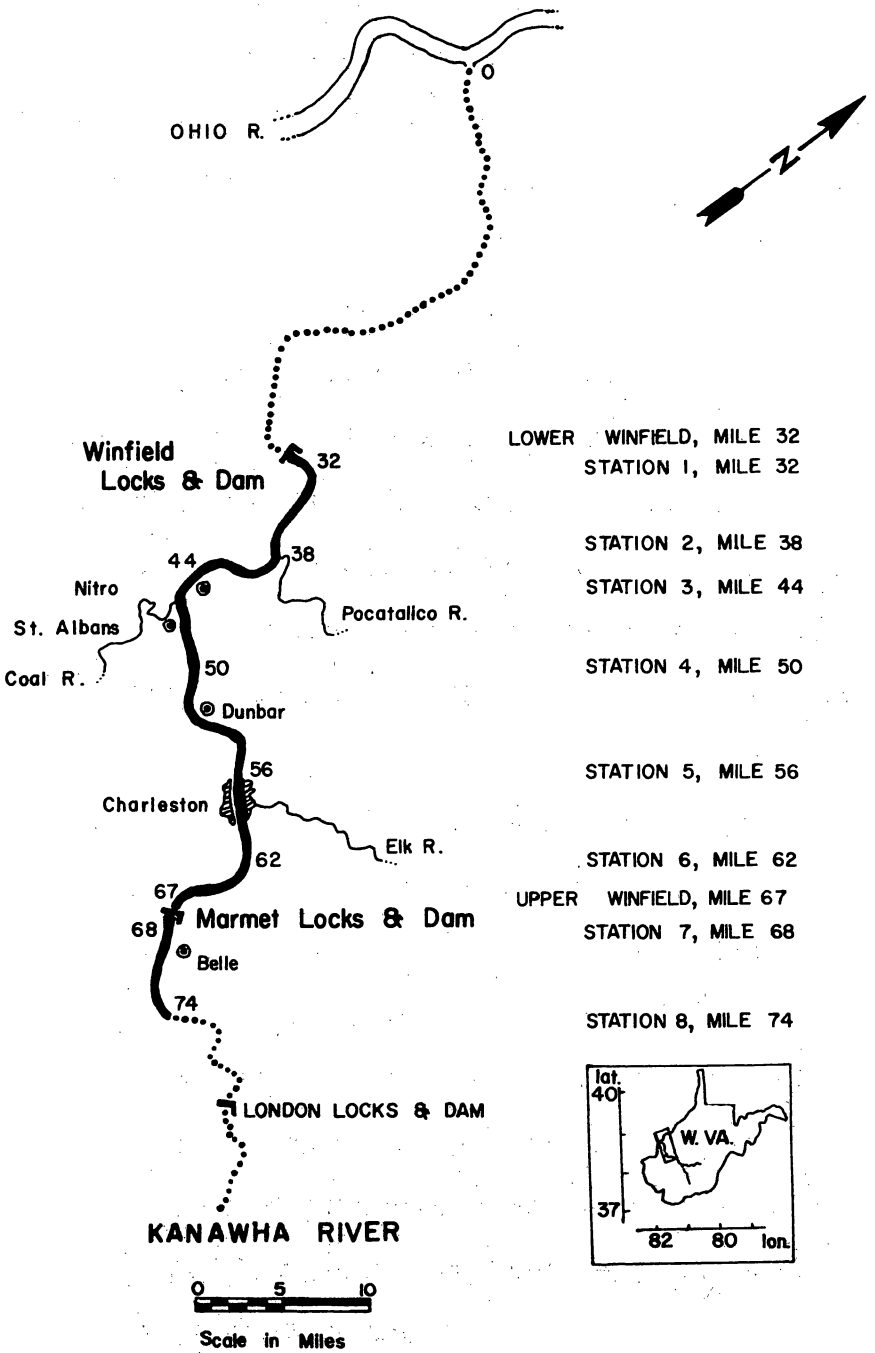

Fig. IV.7.1 Map of Kanawha River illustrating longitudinal sampling sites for zooplankton.

interface (referred to as Top and Bottom, respectively). Longitudinal samples were collected from 1 m, 4 m, and (where depth permitted) 8 m below surface. Rotifers were collected with a 5 l Juday trap (Lind 1979) equipped with 35 µm mesh net on the Wisconsin bucket (Likens and Gilbert 1970). Trap avoidance by rotifers was assumed negligible (Green 1977). Rotifers were narcotized with carbonated water and preserved with 10% formalin in the field (Gannon and Gannon 1975). In the laboratory, each rotifer sample was concentrated to a known volume, a 1 ml subsample was placed in a Sedgewick-Rafter cell, and the entire cell was counted under 100X (American Public Health Association et al. 1981).

A study was done to analyze the immediate impacts on zooplankters when they are entrained in the propeller jet of tow boats. Samples were collected July 20, 1983, behind the towboat Interstate (running on 1 engine), and on September 14, 1983, behind the towboat James K. Ellis. Samples behind the Interstate were collected at River Mile 32 and samples behind the James K. Ellis were collected near River Mile 52. Samples were collected within 1 m of the surface with a 20 l bucket. Samples were concentrated through a Wisconsin bucket with 10 µm mesh net on the bucket.

Biomass and Production Estimates

Biomass values were calculated as follows. Rotifer dimensions were measured with an ocular micrometer in a compound microscope, and volumes of rotifers were calculated according to formulas given by Ruttner-Kolisko (1977). From volumes, dry weights were calculated from average dry weight per cubic micron values given by Dumont et al. (1975). Dry weights were converted into Kcal values by a conversion factor given by Winburg-Duncan (1971). Cladocera and Protozoa estimates were also calculated from values cited in Dumont et al. (1975), and Kcal conversion factors were obtained from Winburg-Duncan (1971). The Kcal value for each organism was then multiplied by the number of those organisms present in the monthly plankton counts to obtain biomass.

Production estimates were calculated as follows. Cren and Lowe-McConnel (1980) explained that production of zooplankton (P_z) is related to respiration of zooplankton (R_z) by the formula:

$$2.33P_z = R_z \qquad (1)$$

Respiration of zooplankton was accounted for as described in Section IV. 5.

Consumption (C_z) was calculated according to formulas 2 (Pedros-Allio and Brock 1983) and 3 (Comita 1972) below:

$$C_z = (0.1)(P_b + P_p) \qquad (2)$$

$$C_z = (R_z + P_z)/0.61 \qquad (3)$$

where

P_b = bacterial production

P_p = primary production

C_z = zooplankton consumption

It was assumed that on the average, populations of zooplankton did not differentiate between bacterial and primary production for their consumption. P_b and P_p were measured by the methods described in Section IV.4.

Excretion and egestion values are known to equal the total consumption of the individual minus the energy lost in respiration as shown below:

$$E_z = C_z - (R_z + P_z) \qquad (4)$$

Excretion and egestion values were calculated using formula (5) derived from formulas (3) and (4):

$$E_z = .39\ C_z \qquad (5)$$

Statistical analyses (Duncans Multiple Range Test, Multivariate Analysis of Variance, and Pearson Correlation) were carried out by means of the Statistical Analysis System (SAS User's Guide 1979). Pinkham-Pearson similarity index cluster analysis was performed using a fortran program described by Pinkham and Pearson (1976).

IV.8. BENTHIC MACROINVERTEBRATES

Design

Benthic macroinvertebrates were sampled monthly from January - December 1983 at UW and LW for the purposes of: (1) determining population densities and life cycles, (2) analyzing community structure and function, (3) estimating secondary production, (4) determining the trophic basis of production, and (5) analyzing the flow of energy through the macroinvertebrate components for the model of the Winfield Pool. A preliminary survey (Voshell et al. 1982) had indicated two dominant types of bottom materials: soft sand/silt and hard cobble/pebble. In order to obtain consistent macroinvertebrate samples which could be used to estimate production on each substrate type, two different sampling methods were used: (1) artificial substrates to simulate the hard bottom and (2) Ponar grabs to take direct samples of the soft bottom. Five replicate samples of each type were taken at both study sites, except for UW and LW in March and UW in May 1983, when the current was too strong to use the Ponar grab. Additional organisms were collected at monthly intervals from May - October 1983 in order to analyze the diets of the dominant taxa. The results from the benthic macroinvertebrate sampling were extrapolated to the entire pool by considering the proportion of hard and soft bottom in the pool.

As part of the contract we also analyzed the community structure of benthic macroinvertebarates at one site in the Gallipolis Pool (River Mile 30.5). Two replicate artificial substrate samplers were used quarterly for this purpose.

Sampling

Rosenburg and Resh (1982) classified artificial substrates into two groups: representative artificial substrates (RAS) and standardized artificial substrates (SAS). RAS samplers are similar to the natural substrates that are found in the study area (cobbles, cement objects, etc.), while SAS samplers are made of materials which are completely foreign to the study environment (hardwood plates, webbing, etc.). In this study, we constructed RAS to mimic the firm components of the natural river bottom. These artificial substrates had the advantagaes of being quantitative and easily retrievable during periods of cold water and/or high river stage.

The artificial substrates consisted of a 30.5 x 30.5 x 5.1 cm concrete slab (commercial patio block) with five river cobbles laid on top, all enclosed in a wire basket with lid (Fig. IV.8.1.). A harness of wire cable was then laced through the basket and attached to a buoy which was anchored in the river (Fig.IV.8.2). The baskets were hung approximately 1.5m below the water surface so that they would be below the photic zone and would not accumulate heavy periphyton growth. The artificial substrate samplers were placed in a row, approximately 150m

apart, along the edge of the navigation channel. At UW the artificial substrate samplers were placed at River Mile 67.0 on the right (descending) side of the channel. At LW the artificial substrates were placed at River Mile 32.0 on the left (descending) side of the channel.

At monthly intervals the artificial substrate baskets were pulled from the river and placed immediately in large metal washtubs to prevent loss of organisms. The artificial substrates were kept moist until the organisms could be removed. River water was pumped into each washtub to a depth of several cm. The cobbles and slab were removed from each basket and placed in the washtup as the empty basket was removed. A soft bristle brush was used to dislodge all organisms into the water contained in the washtub. The water was then poured through a sieve (No. 140, 106µm) to collect the organisms and debris. After rinsing the tub several times into the sieve, the sample was placed in a plastic bag and preserved with 5% formalin. The concrete slab and cobbles were returned to their original wire baskets, which were then reattached to the same buoys.

At the UG site, smaller standardized artificial substrates (Rosenberg and Resh 1982) were used. These were the same samplers that had been used in the previous reconnaissance (Voshell et al. 1982). The artificial substrates consisted of PVC pipe with holes cut in it. The pipe was filled with leaves, sticks, and rocks and then placed in a plastic-mesh bag. The

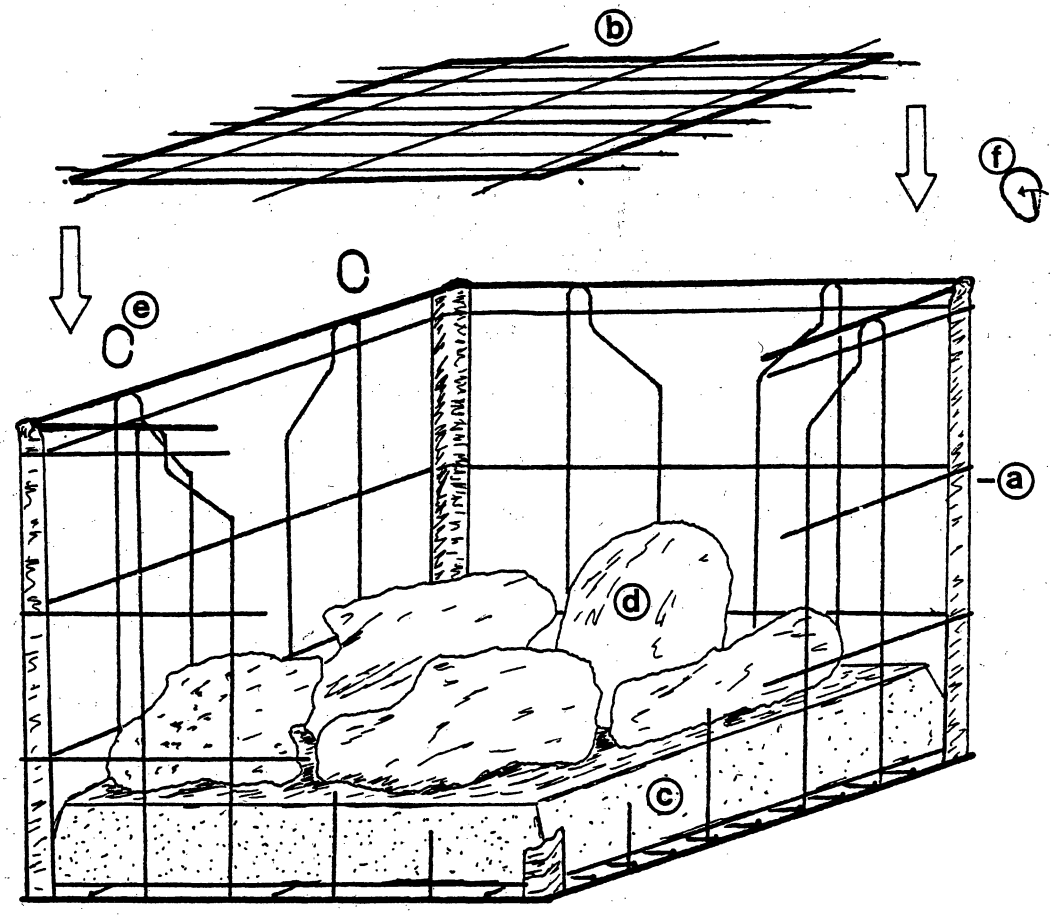

Fig. IV.8.1. Artificial substrate sampler for benthic macroinvertebrates. a = wire basket; b = lid; c = 12 x 12 inch concrete slab; d = river cobbles; e = hinge clips; f = closure clip.

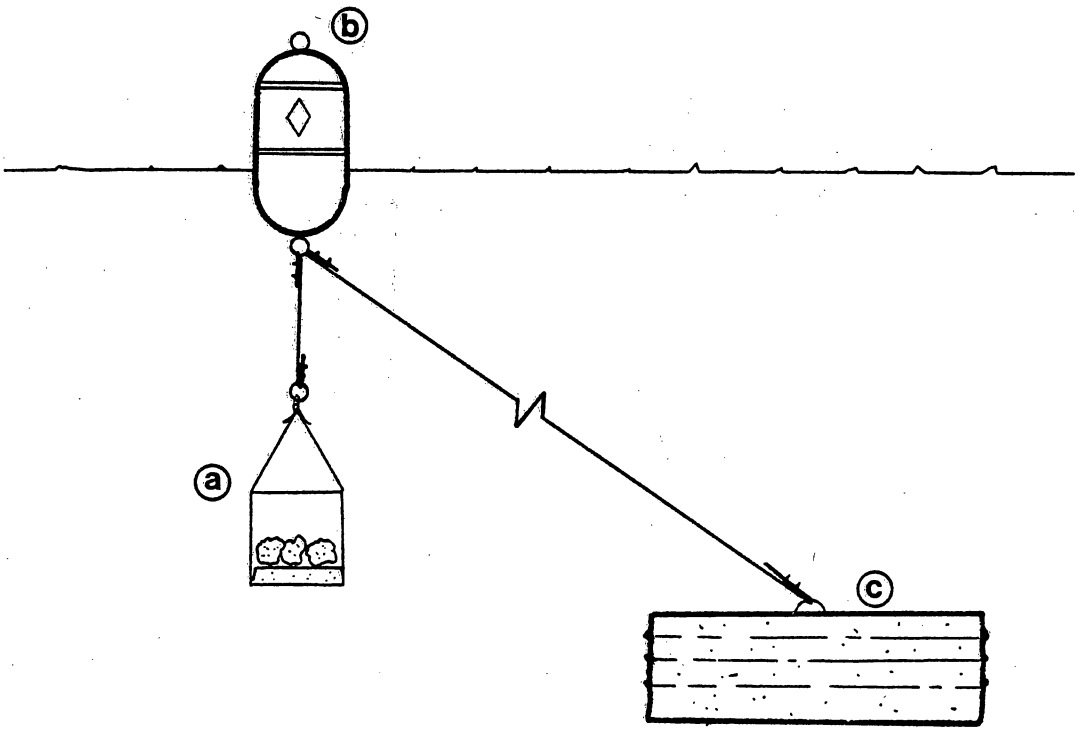

Fig. IV.8.2. Artificial substrate deployment scheme. a = artificial substrate basket; b = buoy with navigation markings; c = concrete anchor. Not drawn to scale; artificial substrate sampler hung approximately 1.5 m below the surface.

artificial substrates were suspended in the water by attaching cables to trees overhanging the water. They were left in the water for 4 weeks before removing them.

In the laboratory, all artificial substrate samples were washed with tap water over a 106µm sieve, and the organisms were removed by hand under a stereomicroscope at 4-10X magnification. All organisms were placed in 70% ethyl alcohol and later identified to the lowest possible taxonomic level.

A Ponar grab was used to sample the soft substrates in the river. The Ponar grab samples an area of 22.9 x 22.9 cm. Samples were pulled to the surface and emptied into metal buckets for transport to shore. On shore the samples were sealed in plastic containers with 5% formalin. In the laboratory, the Ponar grab samples were elutriated from the inorganic sand and silt with a device described by Magdych (1981). Rinse water from the elutriator flowed through a 106 µm sieve to collect the organisms. Sediments left in the elutriator were examined to make sure that all organisms had been removed from the sample. Specimens were stored in 70% ethyl alcohol and later identified to the lowest possible taxonomic level.

Organisms were collected for diet analysis by using the PVC-pipe artificial substrates. These were attached by a lightweight cable to the wire baskets containing the concrete slab and cobble and were retrieved in conjunction with the concrete/cobble artificial substrates. The organisms were dislodged by washing

and brushing techniques, identical to the methods used for the concrete/cobble artificial substrates. However, the organisms destined for diet analysis were first anesthetized in a saturated carbon dioxide solution (commercial club soda), to prevent the regurgitation of gut contents, before they were preserved in 10% formalin. When some taxa were not represented in sufficient numbers on the PVC-pipe artificial substrates, additional specimens were taken from the concrete/cobble artificial substrates (after measuring and enumerating for production estimate).

Production Estimates

Secondary production of benthic macroinvertebrataes was estimated by the Size Frequency Method (Hynes 1980, Waters and Hokenstrom 1980). The method was originally proposed by Hynes (1961) to estimate production of groups of several species which could not be separated into distinct species or cohorts. The Size Frequency Method has undergone several revisions (Hynes and Coleman 1968, Hamilton 1969, Benke 1979, and Benke and Wallace 1980), and is now used commonly because species of benthic macroinvertebrates often cannot be distinguished in the immature stages and many species exhibit overlapping cohorts. In the Size Frequency Method the organisms collected on each date are separated into appropriate size groupings according to the measurement of some body structure and enumerated. A weight is then obtained for each size grouping representing the average

individual dry weight or equivalent for each individual in that size group. The weight used in this study was calculated by taking all of the average individual dry weight values for individuals in a given size class and computing a geometric mean weight. The number of organisms "lost" (presumably due to natural mortality or predation) between one size group and the next largest size group is then calculated by subtraction. The weight of those organisms "at loss" is then calculated by taking the geometric mean weight (Waters 1980) of individuals in the two size groups. This mean weight is then multiplied by the number of individuals lost to equal the weight loss between groups. The weight loss is multiplied by the total number of size groups (the "times loss" referred to by Waters and Crawford 1973) for a size class production value. All size class production values are summed to equal the "uncorrected production" (Benke 1984) estimate. The uncorrected production estimate is then multiplied by the inverse of the fraction of a year that the organism actually spends in production (365/Cohort Production Interval) to estimate the annual production value for that taxa. These steps are summarized in the following equation:

$$P = \sum_{i=1}^{k} (N_i - N_{i+1})(\sqrt{W_i * W_{i+1}})k(365/CPI)$$

where:

P = annual production (mg/m^2)

k = number of size groups

N_i = average annual density of size group i ($number/m_2$)

N_{i+1} = average annual density of size group i + 1 ($number/m^2$)

w_i = average individual dry weight for size group i

w_{i+1} = average individual dry weight for size group i + 1 (mg/m^2)

CPI = cohort production interval (number of days between egg hatch and pupation or emergence).

An example of the calculation of secondary production by the Size Frequency Method is shown in Table IV.8.1.

There are several assumptions that must be met to use the Size Frequency Method (Waters 1977): (1) all of the organisms grouped together can grow to the same maximum size, (2) all organisms in the group have the same CPI, or spend the same amount of time in productive stages, and (3) the organisms grow linearly or nearly so. Of these three assumptions, the assumption of linear growth is probably the most easily relaxed, and non-linear growth is thought not to lead to great errors in the estimate of production (Hamilton 1969, Benke and Waide 1977). Another assumption which has been frequently made in the past few years when using this method is that, since negative losses are theoretically impossible between size groups (the next largest size group should always have fewer individuals than the

Table IV.8.1. Calculation of annual production (mgDW/m2/yr), Stenacron spp., LW, artificial substrate, using the size frequency technique. Negative production values are dropped.

Size Class	Avg. Ann. Density (N/m2)	Avg. Ind. Dry Wt. (mg)	Avg. Ann. St. Stck. (mg/m2)	No. Lost per m2	Ind. Wt. at Loss (mg)	St. Stck. Loss (mg/m2)	Correction Factor n= # sizes
	a	b	a * b	c=ai-aj	d=$\sqrt{bi*bj}$	e=c*d	f= e * n
1	72.8	0.04	2.91				
2	100.3	0.03	4.01	-27.5	.04	-1.10	-9.90
3	60.5	0.07	5.45	39.8	.06	2.39	21.49
4	34.0	0.19	6.46	26.5	.13	3.47	31.24
5	32.9	0.44	14.15	1.1	.29	.31	2.83
6	25.4	0.66	20.07	7.5	.58	4.37	39.35
7	16.6	2.56	20.92	8.8	1.00	8.78	79.04
8	10.8	2.97	27.32	5.8	1.79	10.35	93.18
9	4.7	4.74	22.14	6.1	3.45	21.06	189.51
				4.7	4.71	22.14	199.23

AAD		
358.0	Production estimate (not CPI corrected)	655.90
BBAR	CPI correction factor (365/CPI)	2.40
123.4	Estimated total annual production (CPI corr.) mg/m2 .	1574.20

preceeding one), that they should be dropped when calculating production (Benke and Wallace 1980).

The Size Frequency Method requires that each organism be measured and the number of organisms in each size class be totaled. Measurements were made using either a standard calibrated ocular micrometer or a Zeiss/-Boeckeler® filar eyepiece system interfaced with an IBM Personal Computer® for recording data. With a standard ocular micrometer, the distance in ocular units was recorded for each individual and all values rounded to the nearest ocular micrometer unit. The filar eyepiece system produced values in multiples of one micrometer. For the purpose of production calculations these values were rounded to 0.1mm. The distance across the widest point of the head capsule in dorsal view was measured on specimens of Ephemeroptera, Odonata, Plecoptera, and Trichoptera. At first each individual caddisfly was measured, but after several hundred measurements and the plotting of size frequency curves on each species to show instars, individuals could be placed into instar groups without having to be measured under the microscope. Chironomidae and Simuliidae specimens were measured from the posterior to the anterior edge of the head capsule in lateral view.

Some samples contained extremely large numbers of chironomids (> 9000). In these cases we subsampled by the following method. Four lines were inscribed on the bottom of a

15 cm diameter Petri dish, dividing the dish into eight, equally-sized pie-shaped sections. The chironomids were poured into the Petri dish and the dish was shaken in order to evenly distribute the chironomids. Sufficient sections to provide at least 200 larvae were randomly chosen, and all chironomid larvae were carefully removed by using a pipette. In production calculations each individual from the subsampled population was valued as the inverse of the fraction of the total sample actually measured. The reliability of the subsampling method was examined by counting all of the chironomids in each of the eight sections and then using a t-test to compare the counts. There were no significant differences ($p < 0.05$) between the different sections.

The Size Frequency Method also requires the estimation of an individual weight value or equivalent for the organisms in each size class. After each individual organism was measured, it was placed in a vial of 70% ethyl alcohol with other individuals of the same type and size class. The size classes used for weighing were either ocular micrometer units (for those taxa measured on a standard stereomicroscope), 0.1 mm units (for those measured with filar eyepiece), or instars (Trichoptera). Individuals from a size class were dried (60°C, 24 hr) in a preashed (500°C, 1 hr), tared aluminum weighing pan and then weighed to the nearest 0.00001 g on a Mettler® AE163 electronic microbalance. Only individuals which appeared to have all body parts present were

used in the weight determinations. The weights of the organisms were corrected for loss in preserving fluids by assuming a 20% loss for Odonata and Trichoptera, a 30% loss for Ephemeroptera and Plecoptera, and a 40% loss for Chironomidae (Howmiller 1972, Stanford 1973). This procedure produced a series of weight versus size relationships which could be used for production. The size classses used for production calculations were usually combinations of several of the size classes used to develop the weight/size relationships (except Trichoptera). In the case of all organisms except the Cheumatoysyche spp., all organisms from throughout the year were combined for weighing purposes. Cheumatopsyche spp. individuals were divided into two generations for weighing, individuals from instars 1-3 of June, 1-5 of July and August, and 4-5 of September, were combined as representative of the summer generation. All other individuals were considered part of the overwintering generation.

A further requirement of the Size Frequency Method is the knowledge of how much of the year is spent in actual biomass elaboration or producton (Cohort Production Interval). Only part of the insect life cycle is spent in actual production; the egg, pupae and adult stages are times of non-production. By plotting the presence and abundance of the different size classes of each taxa over time, the life cycles of some taxa could be elucidated and an estimate made as to the CPI. Information from the literature was also used to delimit CPI's of some taxa.

Trophic Basis of Production

Benthic macroinvertebrates are consumer organisms that occupy intermediate trophic levels. To facilitate the analysis of energy flow, benthic macroinvertebrates were classified into several feeding-functional groups (Cummins 1973, 1974, Cummins and Klug 1979, Merritt and Cummins 1978): predators, collector-gatherers, collector-filterers, scraper-grazers, and shredders. Each functional group was treated as a separate compartment in the energy flow model, with each compartment representing the annual production (net) for that group. The flow of energy from allocthonous inputs and autochthonous production into and among the various benthic macroinvertebrate compartments was analyzed by a combination of direct analytical procedures and indirect manipulation of literature values. The overall approach was to determine the types of food that each organism consumed, and then calculate the amounts of those food types that had to be consumed to account for the annual production that was measured for that organism (Benke and Wallace 1980, Parker and Voshell 1983).

In order to determine the diets of the dominant taxa, the methods of Shapas (1973), McCullough (1975), Shapas and Hilsenhoff (1976), and Parker and Voshell (1983) were used to analyze gut contents. Specimens were placed in distilled water and the digestive tract removed. Intact foreguts were placed on a microscope slide containing several drops of 20% sucrose (by volume). The gut contents were emptied onto the slide, while the gut was removed and discarded. Several guts were emptied onto

the slide and spread around evenly with fine forceps and dissecting needles, so that the density of material was sufficient for analysis. After a 24 hr drying period, a drop of 80% sucrose solution was added and a coverslip placed on the gut contents.

The gut contents were viewed with a phase contrast microscope at either 100X or 200X magnification. The areal standard unit method (Welch 1948) was used to estimate the amount of material in each of five food types: animal matter, diatoms, other algae, fine detritus, and vascular plant detritus. Five fields were examined on each slide, and the results were averaged and expressed as % volume of diet.

The relative contribution of each food type to the annual production was calculated by:

$$RAP_{ij} = FT_{ij} * AE_{ij} * NPE_{i}$$

where

RAP = relative contribution of food type j to production of functional group i

FT = percentage of each food type (j) in the diet of functional group i

AE = assimation efficiency of functional group i for each food type j

NPE = net production efficiency (tissue growth efficiency).

The net production efficiency (production/assimilation) was obtained from the literature for each functional group and was assumed to be the same for all food types of a given species. Assimilation efficiencies for each food type were obtained from the literature. The amount of each food type that must have been consumed to account for the measured production was calculated by:

$$CFT_{ij} = P_i * PFT_{ij}/GPE_{ij}$$

where

CFT_{ij} = consumption by group i of each food type j ($mg/m^2/yr$)

P_i = annual production ($mg/m^2/yr$) of functional group i

PFT_{ij} = production by group i attributed to each food type j (% based upon relative amount to production)

GPE_{ij} = gross production efficiency of functional group i feeding on food type j (assimilation efficiency X net production efficiency).

The values for each food type were summed to calculate the total consumption for an organism or the entire functional group. All values were converted to Kcal by literature conversion factors (Cummins and Klug 1979).

Effects of Tow Passage

In order to look at the effects of tow passage on macroinvertebrate colonization of substrates, ten artificial substrates were deployed on August 23, 1983 on the river bottom approximately one kilometer downstream from the Marmet Lock and Dam (UW). This location was chosen because as the tows leave or approach the locks they stay within a relatively narrow sailing line. Artificial substrates then could be placed within the sailing line as well as in areas which would have similar ambient depths and flows but no tow passage. The artificial substrates used in this experiment were of the same design as those used to collect macroinvertebrates thoughout the year. Five artificial substrate baskets were anchored by a team of divers directly in the sailing line. Five other artificial substrates were placed in an area where tows would not pass, however, ambient conditions would be similar. Guidelines made of polypropylene rope were anchored using concrete block and run from selected positions on the shore out to the anchored artificial substrate baskets.

The artificial substrate baskets were retrieved on September 25, 1983 by the divers. The substrates were located by use of the guide ropes. Guide ropes for the artificial substrate baskets placed in the center channel had to be located using a grappling hook. The artificial substrate baskets were covered with a muslin bag by the divers before the substrate was lifted from the river bottom. This was done to prevent the loss of organisms which might occur. The materials within each basket

were then processed using the same methods as the artificial substrate baskets used throughout the year.

IV.9. FISH

Field Work

The ichthyofauna of the Winfield Pool of the Kanawha River were sampled monthly (excluding January) from October 1982 through September 1983 at UW and LW. The objective of this phase of the study was to provide data for determinations of species composition, estimates of relative species abundance, age and growth analyses, and food habit studies of the fishes of the Winfield Pool. A third site, Upper Gallipolis (UG), was also sampled once during October 1982 and in March and July 1983 to provide baseline data for the Gallipolis Pool (Table IV.9.1).

Specific locations for each of eight electrofishing and four gill net samples were selected at all three river sites (Tables IV.9.2-4). Sampling sites were chosen to encompass the various habitats that were evident--a secondary consideration in site selection was suitability for the use of a particular sampling gear type.

Electrofishing was done by boat during daylight hours. A 3500-watt, 220-volt generator and Coffelt VVP-15 variable voltage pulsator was used to produce pulsed, direct current. The electrode system consisted of two Wisconsin hoop anode arrays and

Table IV.9.1. Sampling dates and methods used to collect fish from the three study sites on the Kanawha River.

Date	Upper Winfield	Lower Winfield	Upper Gallipolis
Oct. 21-24, 1982	Electrofishing Gill netting Seining	Electrofishing Gill netting Seining	Electrofishing Gill netting Seining*
Nov. 18-20, 1982	Electrofishing Gill netting	Electofishing Gill netting	
Dec. 16-18, 1982	Electrofishing Gill netting	Electrofishing Gill netting**	
Feb. 17-18, 1983	Electrofishing Gill netting	Electrofishing Gill netting	
Mar. 16-18, 1983	Electrofishing Gill netting	Electrofishing Gill netting	Electrofishing Gill netting
Apr. 20-22, 1983	Electrofishing Gill netting	Electrofishing Gill netting	
May 17-19, 1983	Electrofishing ***	Electrofishing Gill netting	
Jun. 21-23, 1983	Electrofishing Gill netting	Electrofishing Gill netting	
Jul. 20-22, 1983	Electrofishing Gill netting	Electrofishing Gill netting	Electrofishing Gill netting
Aug. 17-19, 1983	Electrofishing Gill netting	Electrofishing Gill netting	
Sep. 21-23, 1983	Electrofishing Gill netting	Electrofishing Gill netting	

* Seining subsequently discontinued.
** Only two nets set.
*** No nets set due to high water.

Table IV.9.2. Description of specific sampling locations at the Upper Winfield site (RM 66 - 67).

Method	Location	Description
Electrofishing	1	River left, starting at McCorkins Light.
	2	River right, starting about 3,200 ft downstream from the arrival point marker.
	3	River right, starting at a concrete discharge located about 200 ft downstream from the arrival point marker.
	4	River left, starting approximately 2000 ft. downstream from the mouth of Rush Creek.
Gill netting	1	River right, at the upstream side of the mouth of Burning Springs Branch.
	2	River left, at the downstream side of the mouth of Rush Creek.
	3	River right, even with the black bouy located about 800 ft below the arrival point marker.
	4	River left, approximately 2500 ft below the mouth of Rush Creek (about 300 ft above the submerged cable).
Seining	1	River right, at the discharge pipe about 2000 ft below McCorkins Light.
	2	River right, about 3000 ft below the arrival point marker (slightly upstream from the black bouy).
	3	River right, about 800 ft. downstream from the mouth of Burning Spring Branch.
	4	River left, immediately upstream from the mouth of Rush Creek.

Table IV.9.3. Description of specific sampling locations at the Lower Winfield site (RM 33 - 34).

Method	Location	Description
Electrofishing	1	River right, starting about 1000 ft downstream from the mouth of Little Guano Creek
	2	River left, starting about 500 ft below the mouth of an unnamed creek (directly across the river from site 1).
	3	River right, starting about 1000 ft downstream from the Red House Light.
	4	River left, directly across the river from site 3.
Gill netting	1	River right, at the upstream side of the mouth of Little Guano Creek.
	2	River left, at the upstream side of the mouth of an unnamed creek (slightly downstream and across the river from site 1).
	3	River right, at the upstream side of the mouth of an unnamed creek (about 1500 ft upstream from the Red House Light).
	4	River left, and directly across the river from site 1.
Seining	1	River left, across the mouth of Bear Creek.
	2	River left, across the mouth of an unnamed creek (slightly downstream and across the river from Little Guano Creek).
	3	River right, across the mouth of an unnamed creek (about 300 ft upstream from the Red House Light).
	4	River right, across the mouth of an unnamed creek (about 1800 ft upstream from the Route 34 bridge).

Table IV.9.4. Description of specific sampling locations at the Upper Gallipolis site (RM 29 - 30).

Method	Location	Description
Electrofishing	1	River left, starting about 1000 ft upstream from the mouth of Hurricane Creek (even with the red bouy).
	2	River right, starting directly across the river from site 1.
	3	River left, starting at the upstream side of the mouth of Tucker Brook.
	4	River right, and starting at the upstream side of an unnamed creek (across the river and about 500 ft downstream from the Winfield Dam Lower Daymark).
Gill netting	1	River left, at the upstream side of the mouth of Little Hurricane Creek.
	2	River left, at the Winfield Dam Light.
	3	River left, at the mouth of Tucker Brook.
	4	River right, across the river and about 500 ft below site 3.
Seining	1	River left, about 200 ft downstream from the mouth of Tucker Brook.
	2	River left, about 800 ft upstream from the mouth of Hurricane Creek.
	3	River right, just downstream from the mouth of the unnamed creek described for electroshock site 4.
	4	River left, at the downstream side of the mouth of Little Hurricane Creek.

10 flexible conduit droppers for cathodes. Generated amperages ranged from 6 to 8. The outboard motor used on the electrofishing boat was equipped with a jet unit (Specialty Manufacturing, Inc., San Leandro, Calif.) instead of the conventional propeller-drive lower unit. This feature provided a high degree of maneuverability around structures where fish appeared to be concentrated, such as brush piles and over-hanging vegetation, and permitted electro-fishing in very shallow water. Two people captured stunned fish from the front of the boat using 4.8 mm bar-mesh dip nets at four of the eight electrofishing locations at each river reach.

It became apparent during the October 1982 sampling period that shore-line seining for smaller fish was not feasible due to a scarcity of areas compatible to the use of this sampling gear type. Subsequently, young-of-the-year and smaller species (principally cyprinids) were sampled during separate electrofishing runs at the four electrofishing locations at each of the three sites, using 1.6 mm bar-mesh dip nets. The use of the smaller mesh dip nets reduced escapement of smaller cyprinids. All electrofishing was conducted by proceeding upstream for fifteen minutes along the shoreline. The distance covered at each location (150-200m) varied slightly over the course of the study due to the effects of differences in the velocity of river flow on boat-handling.

Experimental gill nets were used to sample species that frequented deeper water. The gill nets were constructed of monofilament nylon line, and were 38.1 m long by 1.8 m deep. Each net was composed of five 7.6 m panels, each panel of a different mesh size (1.9, 2.5, 3.2, 3.8, or 5.1 cm bar-mesh). Gill nets were fished overnight with the smallest mesh inshore and increasing mesh sizes extending out into the channel. The nets were set at an approximate 45 degree angle to the downstream shoreline. The lead lines of the gill nets, for the most part, rested on the river bottom along their entire lengths.

Larger fish collected by electrofishing or gill-netting were identified, weighed, and measured in the field. Scales (or spines from ictalurids) were collected. Stomachs were removed and preserved in 10% formalin for future analyses. Smaller fish were preserved in their entirety, in 10% formalin, and returned to the lab for identification and other analyses.

Species Composition and Relative Species Abundance

All fish were identified in accordance with descriptions given by Eddy (1969), Pflieger (1975), or Trautman (1981), and subsequently enumerated by species. Weights of all individuals collected were recorded with the following exceptions: when gizzard shad or small cyprinids appeared in abundance, weights were recorded for individuals from subsamples. At least forty randomly selected members of each species per month per site were

weighed. The remaining fish of each species were weighed en masse for the relative species abundance estimates. Length-weight relationships for gizzard shad and smaller cyprinids were developed from the subsample data. Fish from which lengths, but not weights, were taken could then be included in the growth analyses. Relative species abundance was estimated both by numbers and by weight.

Age Determinations

Most species were aged by counting scale annuli. Scale impressions were made in acetate slides, and subsequently projected at 40X to facilitate reading. Catfish were aged from pectoral spine sections. Pectoral spines were decalcified, sectioned, stained and mounted on glass slides using methods of Ashley and Garling (1980). Each fish collected was aged independently by two people. A third person resolved any disagreements. Scale impressions that lacked clarity were discarded. Subsamples of 40 randomly selected gizzard shad per site were aged in May and June.

Attempts to age emerald shiners using scales were unsuccessful. The Petersen method (inspection of the length frequency distribution) was used to age this species (Jearld 1983).

Longnose gar, members of the family Cyprinidae (other than emerald shiners and common carp), and other species of which only a few specimens were collected were not aged.

Growth

Von Bertalanffy growth models were fitted for the eighteen most common species of fish. Fish were assigned ages in months by adding to the annular age (expressed in months) the number of months elapsed between the month of collection and January of the same year. Thus an Age 0 fish taken in October was assigned an age of 10 months (0x12 + 10); an Age I, 22 months, etc. Ages were subsequently converted back to years for input to the von Bertalanffy growth models. Age/length and age/weight relationships were established using:

$$W_t = W_\infty(1 - e^{-K_w(t-t_0)})^3$$

$$L_t = L_\infty(1 - e^{-K_l(t-t_0)})$$

where:

W_t = predicted weight of fish at time t

L_t = predicted length of fish at time t

W_∞ = asymptotic maximum weight

L_∞ = asymptotic maximum length

t_0 = time intercept (t for W = 0 or L = 0)

K_w = coefficient for growth in weight

K_l = coefficient for growth in length

The parameters of these relationships were obtained by least-squares estimation using a nonlinear regression (SAS User's Guide, 1979 Edition). Fitted growth models were then used as predictors of fish size (length or weight) at age.

Instantaneous growth rates (G) for each species and age group (t) were then calculated as follows:

$$G_{t,t+1} = \ln (W_{t+1}) - \ln(W_t)$$

where:

W_t = weight (g) predicted from the fitted von Bertalanffy equation

Mortality

In the absence of suitable data for application of catch-curve analysis, mortality rates (Z) were estimated from median lengths. Hoenig et al. (1983) reviewed methods of estimating Z from mean length, concluding that these possessed undesirable statistical properties. He proposed an estimator requiring an exponential transformation of fish lengths:

$$y_i = -\ln(1 - (l_i/L_\infty))$$

where:

l_i = length of the i^{th} fish in the sample

L_∞ = asymptotic maximum length

The median of the transformed lengths, along with a transformation of the length at which the fish is first vulnerable to the gear (y_c), is used to estimate mortality as follows:

$$Z = K (\ln(2)/(y_m - y_c)$$

where:

K = von Bertalanffy growth coefficient

y_m = median of transformed lengths

Other estimators based on mean length (Beverton and Holt 1956, Ssentongo and Larkin 1973) and age-composition (Robson and Chapman 1961) were also used for comparison. Hoenig's estimator, which is relatively insensitive to variability in year class strength, was preferred in cases where the four methods differed considerably. In all other cases the average of the four mortality estimates was used.

Production-Biomass Ratios

The approach taken in estimating production follows Ricker's (1975) derivation of yield from an arbitrary recruitment. Ricker's method is a tabular computation which divides a fish population of arbitrary initial weight into age intervals. One-thousand kilograms of recruits were assumed at the beginning of the first age interval. Each age interval is characterized by an instantaneous growth rate obtained, and a mortality rate. These rates define a weight change factor (e^{G-Z}) used to predict the weight of recruits remaining at the end of the interval. Weights at the beginning and end of an interval are averaged and the result multiplied by the growth rate to estimate production during the interval. Production over all intervals was summed to give total production per recruit. Similarly, biomass was summed over all intervals to give total biomass per recruit. The sum of production was then divided by the sum of biomass to yield the

P:B ratios. Young of the year and juveniles of most species were not vulnerable to the collection gear, and hence this period of growth is not reflected in the above method.

Production estimates reflect only production by adults, and as such underestimate total production for each species. However, since the diet of young-of-year fish differs considerably from older fish of the same species they do not belong in the same functional groups as adults. Several investigators cited by Neves (1981) gave estimates of the contribution of young of the year fishes to total species production. These estimates range from 22 to 84 percent. Burgis and Dunn (1975) used 70 percent in partitioning production of roach. Consequently, our estimates of fish production and our ecosystem model structure does not include young-of-year fish.

Actual Production

Estimates of actual production per unit area were calculated as the product of the production-biomass ratios and biomass density factor for each of the eighteen most common species. Annual lock rotenone sampling by the West Virginia Department of Natural Resources at Winfield Locks and Dam provided estimates of total biomass per unit area. Biomass of individual species varied considerably among years in these collections, whereas biomass of all species combined was much more consistent among years. Therefore total biomass estimates were averaged, and partitioned according to the relative weights of each fish

species collected during 1982-83 field collections at UW and LW. This approach is less sensitive to temporal and behavioral biases inherent in the annual lock rotenone surveys. Biomass per unit area was estimated by:

$$B_i = (W_G / A) * (W_i / W_T)$$

where:

B_i = biomass per unit area

W_G = average of total weights in lock samples

A = area of lock chamber

W_i = weight of species i in samples

W_T = total weight of VPI&SU sample

Wet weights were converted to dry weights using a factor of 25 percent. Caloric densities (kcal/g dry weight) were obtained for the Centrarchidae, Cyprinidae, and Clupeidae from Cummins and Wuycheck (1971).

Production-biomass ratios were then multiplied by the biomass estimate for each species to estimate their production per unit area. Production by each trophic group was calculated by summing the available production estimates of species in a group and then multipying this by a correction factor to account for the contribution of species for which production estimates were not available.

Food Habits

The food habits of the fishes of the three Kanawha River sites under study were investigated by examining gut contents. A number of different techniques were required to gain this information, due to anatomical and dietary differences between species.

The stomach contents of species exhibiting a distinct stomach (centrarchids, ictalurids, etc.) were examined. Stomachs were emptied, food items blotted dry, and the volume of the stomach contents was measured by displacement. Individual food items were identified, often microscopically, and counted. The volume of each food type was measured by displacement whenever possible. The volumes of food items too small to be measured were estimated relative to the volume of other food types too small to be measured by displacement. Absolute volume estimates for these smaller food items were then calculated based on the difference between measurable food items and the total volume of the stomach contents.

Catostomids and common carp food habits were also examined using these methods. However, the contents of the first loop of the intestine was considered, since these species lack true stomachs.

Foreguts were collected from 20 gizzard shad per site per month, when available. The contents of the esophagus and gizzard were extracted. Pooled samples from all 20 gizzard shad were examined. These were stained with rose bengal to help

distinguish between inorganic and organic matter. A subsample of the pooled sample was placed in the Sedgewick-Rafter counting chamber, and viewed microscopically at a magnification of 200X. Ten fields of view per subsample were observed. Mean values for each pooled sample were derived from three subsamples. The area covered by each food type in the Sedgewick-Rafter chamber was estimated by using an ocular micrometer grid. Area was considered to be proportional to volume. Estimating the volume of stomach contents from area has been used for threadfin shad (Gerdes and McConnell 1963) and gizzard shad (Baker and Schmitz 1971). Gerdes and McConnell (1963) developed "relative thickness factors" for various taxa found in the diet, and calculated relative volume by multiplying area by relative thickness. The abundance of amorphous matter found in the gut contents of gizzard shad in this study prevented the development of relative thickness factors. The volume of a particular food item in the gut contents of gizzard shad was based on area covered and a visual estimate of relative thickness.

The food habits of the emerald shiner were considered to be representative of all the cyprinid species with the exception of common carp. Methods used to examine the gut contents of emerald shiners were similar to those used for gizzard shad. However, the entire contents of the first intestinal loop were scrutinized, rather than subsamples taken from pooled samples of the foregut of 20 fish.

Longnose gar lack a distinct stomach. The contents of the complete digestive tract were examined. Techniques for quantification of the gut contents were the same as those for species with true stomachs.

Food habits for several species were not analyzed because very few individuals were collected, and they represented a small proportion of the fish biomass. The food habits of the Ohio lamprey, brook silversides, Etheostoma spp., and Percina spp. were not examined for the above reasons. The food habits of the American brook lamprey were not studied, since this species does not feed during the adult stage of its life. Descriptions in Pennak (1953) and in Merritt and Cummins (1978) were used to identify invertebrates to the lowest feasible taxon.

Proportional annual consumption (PAC), the contribution of a given food item to the total annual consumption of a given species of fish (or trophic group of fishes), was calculated from gut contents observations. These observations were corrected for estimated changes in rates of food intake, and for the proportion of fish of a given species that had empty guts when collected each month from the Winfield Pool. Thus, the PAC for each food item type was obtained for each species of fish through a series of calculations.

The average percent of total volume (V_{ai}), in the gut contents of a species of fish (a), comprised of an individual type of food item (i), was computed from each month's field

samples. Monthly V_{ai} values were then weighted to take into account two factors: one weight corrected for the percent of fish collected each month that had empty guts, and the other weight corrected for assumed temperature dependent changes in the rate of food intake. First, each monthly V_{ai} was weighted according to the proportion of fish examined with non-empty guts relative to that proportion for the entire sampling season. Second, monthly V_{ai} values were weighted according to estimated digestive tract evacuation rates derived from water temperatures measured each month in the Winfield Pool and from egestion times for sauger in the Gallipolis Pool of the Ohio River as reported by Wahl (1982). This weighting factor was applied under the assumption that evacuation time is inversely proportional to the rate of food intake. Hence, diet during periods of faster evacuation contributed more to the annual ration of a species than did diet during slower periods of evacuation. The two weighing factors, therefore, were used to compute adjusted diet composition volumes for each food item type (VI_i) according to the formula:

$$VI_{aim} = V_{ai} \ (N_{am}/N_{at})E_m$$

where:

VI_{aim} = adjusted diet composition volume for diet item i in species a during month m

V_{ai} = average percent, of total volume, for diet item i in the gut contents of species a during month m

N_{am} = number of fish of species a captured in month m with non-empty guts

N_{at} = total number of fish of species a with non-empty guts captured throughout the study

E_m = the weighting factor for evacuation time (Table IV.9.5)

The proportional annual consumption of a particular type of food item for a given species of fish was then calculated by summing the adjusted diet composition volumes (VI_{aim}), and dividing by the total of such sums for all diet items:

$$PAC_{ai} = \sum_{m=1}^{11} VI_{aim} / \sum_{i=1}^{n} \sum_{m=1}^{11} VI_{aim}$$

where:

PAC_{ai} = Proportional annual comsumption of diet item i by species a

m = index for months

Table IV.9.5. Weighting factors used to adjust fish consumption data for variable digestion rates.

Sampling Month	Kanawha River water temp. (° C.)	Evacuation time(hours)[1]	Weighting factor[2]
October	20.0	16.0	0.41
November	14.0	16.5	0.40
December	6.2	38.5	0.17
February	3.0	186.5	0.04
March	4.7	112.5	0.06
April	9.0	39.8	0.16
May	15.0	16.5	0.40
June	22.6	8.2	0.80
July	26.0	7.1	0.93
August	26.6	6.6	1.00
September	25.6	7.6	0.87

[1] based on Wahl and Nielsen (1985)
[2] calculated as (6.6 hrs/evacuation time in hrs)

n = number of diet items

Proportional annual consumptions were calculated for the 17 most abundant species of fish in the Winfield Pool. The PACs were then used to assign each species to one of five trophic groups, namely, piscivores, crayfish/piscivores, invertivores, omnivores, or detritivores. Proportional annual consumptions for each trophic group of fishes and for each diet item were then determined for UW LW by weighting the proportional annual consumption indices according to the relative species abundance estimates (by weight) obtained at each of the two Winfield sites. Accordingly, proportional annual consumption of a given diet item was computed for each trophic group for UW and LW as follows:

$$PAC_{ti} = \sum_{a=1}^{x} PAC_{ai} \, (B_a / \sum_{a=1}^{x} B_a)$$

where:

PAC_{ti} = proportional annual consumption of diet item i by trophic group t

PAC_{ai} = proportional annual consumption of diet item i by species a

B_a = total biomass of species a captured at a given site (UW or LW) during the study

x = the number of species in the trophic group

Consumption

Annual consumption by each trophic group was derived from total annual production by applying a series of general bioenergetics relationships involving consumption (C), assimilation (A), production (P), respiration (R), and wastes (W). Consumption is first partitioned into its various fates:

$$C = P + R + W$$

Assimilation, defined as consumption less wastes, was assumed to occur at approximately 80% efficiency for pure carnivores and 50% efficiency for pure herbivores (Webb 1978). Intermediate efficiencies were assumed for species demonstrating diet habits between these two extremes (Table IV.9.6). Net production efficiencies in the range of 30% have been estimated for fish (Kozlovsky 1968), implying that respiration requires approximately 70% of assimilated energy. Using these relationships it was possible to back-calculate the consumption necessary to support observed production. The overall relation between consumption and production was obtained as follows:

Let E_a = assimilation efficiency (A/C)

E_r = respiration efficiency (R/A)

E_p = production efficiency (P/A)

then

$$R = P\ (E_r\ /\ E_p)$$

and $C = (P + R)/E_a$

Consumption by trophic groups was determined in the same manner as was production, by summing available estimates for species within a group and then proportionally increasing this to account for species for which estimates were not available.

IV.10. LARVAL FISHES

Design

To assess the direct physical impacts of increased navigation traffic on larval fish, studies were conducted with the following objectives: (1) determine spatial and temporal distribution of ichthyoplankton in the Kanawha River; (2) determine (on a spatial and temporal basis) direct mortality caused by towboat passage; and (3) determine the relative mortality rates of several species of larval fish when subjected to standardized levels of physical disturbance.

Ichthyoplankton sampling was conducted at two sites in the Winfield Pool. Two sampling locations were chosen because the Winfield Pool is not homogeneous throughout its length. The lower portion of the pool has greater depth, slower water velocity, and more embayments than the upper end. A site was selected in the lower end of the pool at river mile 36.5 (Fig. IV.10.1). The width of the river at this point is 225 m, and the

Table IV.9.6. Conversion efficiencies used to estimate consumption. E_r = respiration efficiency; E_p = production efficiency; E_a = assimilation efficiency.

Species	E_r	E_p	E_a
Gizzard shad	0.8	0.2	0.5
Common carp	0.7	0.2	0.5
Channel catfish	0.6	0.4	0.7
Smallmouth buffalo	0.7	0.3	0.5
Sauger	0.8	0.2	0.8
Silver redhorse	0.7	0.3	0.7
Golden redhorse	0.7	0.3	0.7
Shorthead redhorse	0.7	0.3	0.5
Freshwater drum	0.7	0.3	0.8
Smallmouth bass	0.7	0.3	0.8
Emerald shiner	0.8	0.2	0.7
Spotted bass	0.7	0.3	0.8
Largemouth bass	0.7	0.3	0.8
White bass	0.8	0.2	0.8
Bluegill sunfish	0.7	0.3	0.7
Spotted sucker	0.7	0.3	0.6
Longear sunfish	0.7	0.3	0.7
White crappie	0.7	0.3	0.8

midchannel depth is approximately 9.4 m. An upper sampling site was established at river mile 64.5 (Fig. IV.10.2). The river's width is 200 m at this location, and midchannel depth is approximately four meters. The selection criteria for the sites were a symmetrical bottom profile (desirable for sampling regime described in methods) and lack of major underwater obstructions.

Distribution

Knowledge of larval fish distribution patterns is of primary importance in determining which species may be more vulnerable to barge-associated impacts and when. The majority of the fish species present in the Kanawha River spawn from April to August (Breder and Rosen 1966, Pflieger 1975). Walleye and sauger may spawn earlier (February or March) depending on water conditions. Sampling was originally scheduled to begin in April 1983; however, difficulties in procuring necessary equipment required that sampling begin in late May 1983. The two Winfield sites were sampled a total of six times (Table IV.10.1). Five sets of samples were taken from late May until August of 1983, and an additional set in early May 1984.

A total of 40 distribution samples were collected with bongo nets on each of the six sampling trips (Table IV.10.1). At the Upper Winfield site (UW), two samples were collected during daylight hours at each of six points in the river: near surface

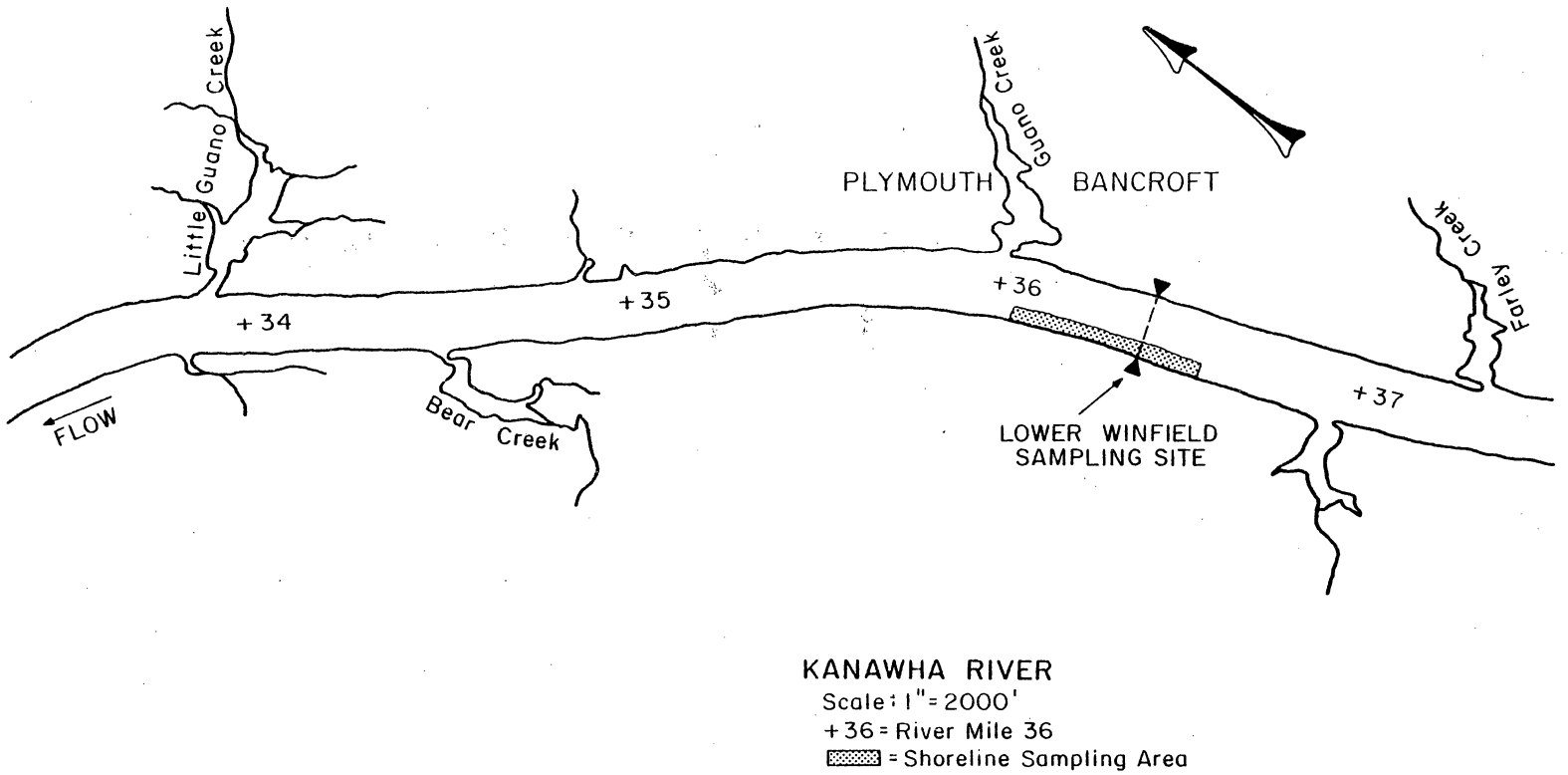

Figure IV.10.1. Location of lower Winfield larval fish sampling sites.

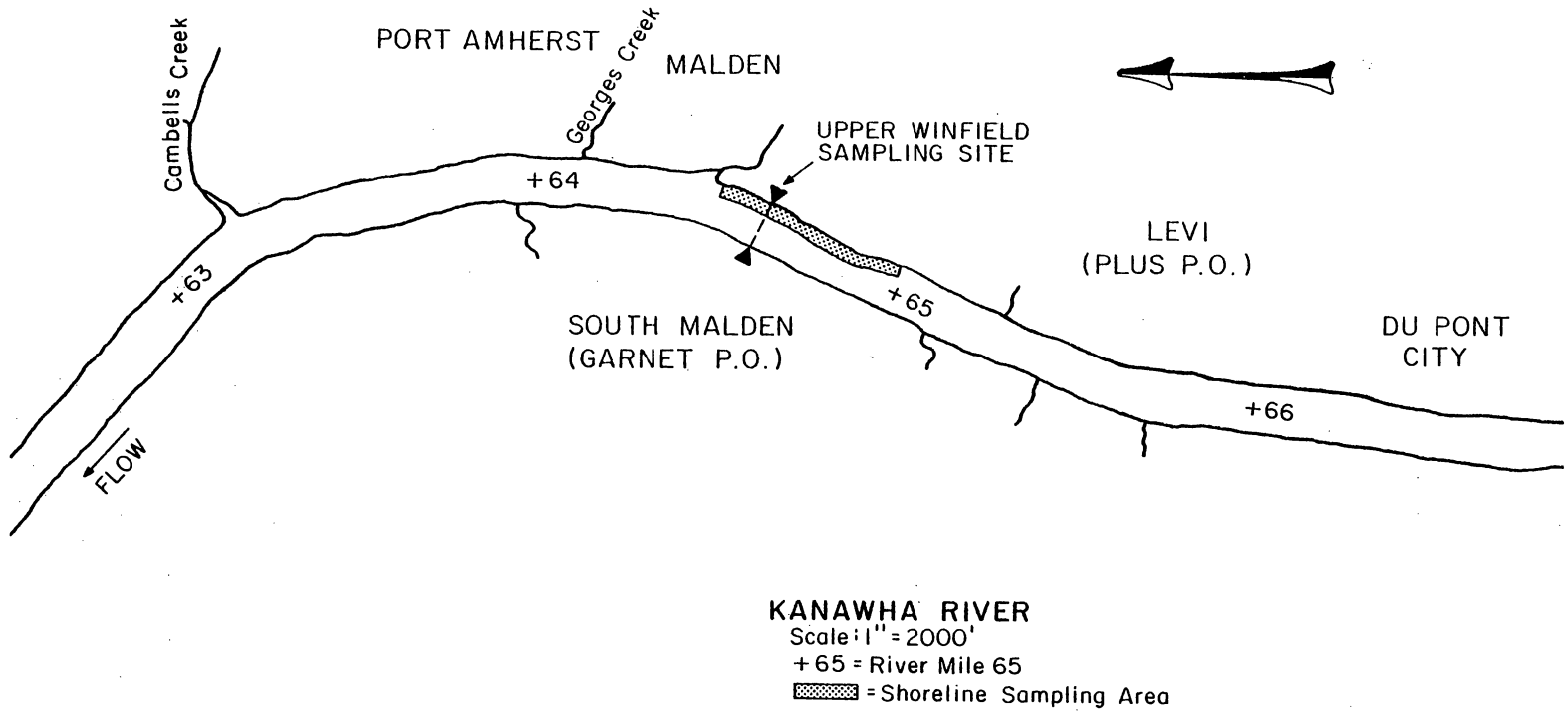

Figure IV.10.2. Location of upper Winfield larval fish sampling sites.

Table IV.10.1. Summary of distribution sampling activities for larval fish.

Date	Sampling site	Number of samples: Bongo nets	Number of samples: Push nets
5/24/83	Lower Winfield Day samples	14	-
	Night samples	14	-
5/25/83	Upper Winfield Day samples	12	-
6/14/83	Lower Winfield Day samples	14	-
	Night samples	14	-
6/15/83	Upper Winfield Day samples	12	-
7/5/83	Lower Winfield Day samples	14	4
	Night samples	14	4
7/6/83	Upper Winfield Day samples	12	4
7/27/83	Lower Winfield Day samples	14	4
	Night samples	14	4
7/28/83	Upper Winfield Day samples	12	4
8/9/83	Lower Winfield Day samples	14	4
	Night samples	14	4
8/10/83	Upper Winfield Day samples	12	4
5/1/84	Lower Winfield Day samples	14	4
	Night samples	14	4
5/2/84	Upper Winfield Day samples	12	4
Total distribution samples		240	48

and near bottom at 20, 40, and 50% of distances across river from one bank (Fig. IV.10.3). At Lower Winfield (LW), two samples were collected during daylight hours at each of seven points in the river: near surface and near bottom at 20 and 40% of distances across river from one bank, and near surface, near bottom, and mid-depth (4.5 m) at 50% of distance across river (Fig. IV.10.4). An identical set of nocturnal samples were collected at LW to identify differences in spatial distribution between day and night (Table IV.10.1). The mid-river sampling points (50% distance across river) at both UW and LW are in the approximate sailing line for navigation traffic.

The two samples from each of the six sampling points at UW, and seven sampling points at LW were obtained by making two tows in an upstream direction at each sampling point with twin bridleless, 0.5-m diameter bongo nets (500 micron mesh). The contents of the paired nets were pooled to constitute one individual sample for each tow. This is required as the contents of each bongo net are not independent samples (J. B. Birch, Department of Statistics, Virginia Polytechnic Institute and State University, personal communication).

Bridleless bongo nets were selected based on the large volume of water that can be sampled in a short time period (Bowles et al. 1978). They are more efficient at catching ichthyoplankton than bridled plankton nets of either 0.5-m or 1-m

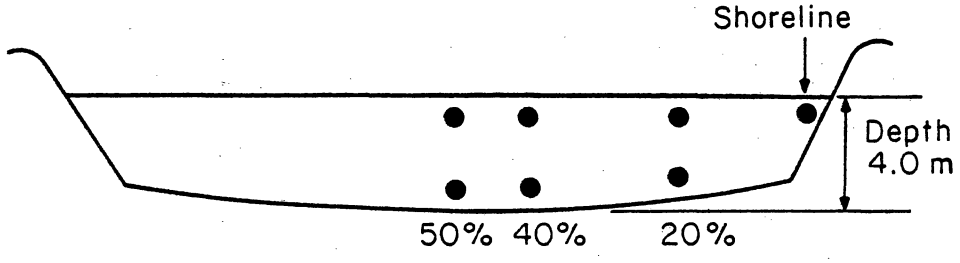

Figure IV.10.3. Design of sampling points in river cross section at upper Winfield site. (Descending view).

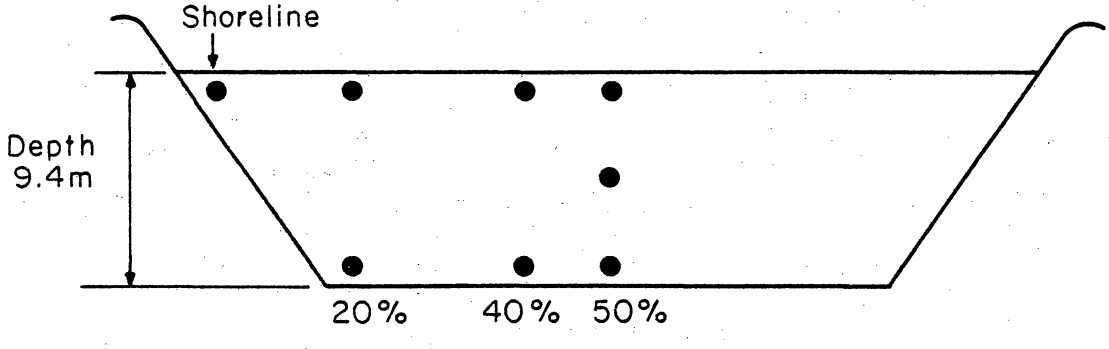

Figure IV.10.4. Design of sampling points in river cross section at lower Winfield site. (Descending view).

diameter (Marine Research Incorporated 1975, cited in Bowles et al. 1978). A flow meter (General Oceanics Model 2030) mounted in one of the bongo nets recorded the volume of water sampled for each tow. This did not appear to alter the capture efficiency of the net it was installed in (Table IV.10.2). A mean of 121.9 cubic meters of water were sampled by each tow.

Towing durations were five minutes, and discrete towing depths of the nets were determined by use of a calibrated cable gauge (to measure the length of cable let out) and a clinometer mounted on the cable to measure its angle. Depth of the towed nets (d) was determined from:

$$d = (\cos b) \times l$$

where l = length of towing cable

b = angle of towing cable relative to the vertical axis.

The towing velocity of the nets was approximately one meter per second for all distribution samples taken at both UW and LW to allow for comparisons. Because these samples were to represent "natural" ichthyoplankton distribution and densities, it was desirable to avoid any potential bias resulting from passage of tow boats. Therefore, all of the distribution samples were collected at least 40 minutes after the passage of a barge.

Table IV.10.2. Results of paired comparisons (Wilcoxon Signed Ranks test) between the catch of Net 1 (containing flow meter) and Net 2 (without flow meter).

Dates	Number of Tows	P-value
6/14-15/83	28	.26
7/5-6/83	28	.21

All samples were preserved in 5-10% formalin buffered with calcium carbonate (Taylor 1977) for subsequent counting, measurement, and identification in the laboratory. All fish were identified to the lowest practical taxonomic level using information and keys by Auer (1982) and Hogue et al. (1976). Due to their scarcity in the samples and extreme difficulty in identification, eggs were not considered in this study. Juveniles (full complement of adult fin rays and absorbed finfold) were similarly excluded.

Total lengths were measured with an ocular micrometer mounted in a stereo dissecting scope. If large numbers of a given species/taxon were present in a sample, not all larvae were measured. A subsample was taken by placing the larvae in a water-filled tray marked with a numbered grid pattern. The tray was agitated to disperse the larvae and then allowed to settle. Grids were selected with the aid of a random number table. The larva nearest the upper left corner and within each selected grid was taken for the subsample. Subsample size varied depending on the range of lengths within the original sample.

During the mid-June 1983 sampling, large numbers of larvae were observed (visually) in proximity of the shoreline. The scheduled sampling did not include the shoreline areas, but the apparent abundance of larval fish made sampling of this habitat desirable. Seining was impractical due to numerous submerged

obstructions; bongo nets were similarly excluded because of the shallow water and obstructions. An alternative sampling gear (push nets) was developed and employed beginning in early July 1983 (Table IV.10.1).

Two WILDCO stream drift nets (mouth diameter 45 x 30 cm, 363 micron mesh) were mounted on a metal frame which extended out in front of a 17-foot aluminum boat. This push net arrangement allowed sampling in water as shallow as 55 cm, and the small outboard-propelled craft could maneuver the nets around obstructions. A WILDCO Model 39 A10 flowmeter mounted in one of the nets (Net 1) recorded the volume of water sampled by each "push." However, Net 1 was noticeably less efficient at capturing ichthyoplankton than the net without a flowmeter (Net 2). Therefore, for each "push", the contents of Net 2 only were used to constitute a given sample; the contents of both nets were not combined as with the bongo nets.

A total of 12 distribution samples were collected with push nets on each sampling trip from early July 1983 to May 1984 (Table IV.10.1). While the scheduled bongo net sampling was being carried out at a given site, a second crew simultaneously collected four push net samples along one shoreline of the site (Figs. IV.10.1-2). Beginning at a selected point along the shoreline in water as shallow as possible, the push nets mounted on the metal frame were lowered into the water and pushed

upstream parallel to shore at a velocity of approximately one meter per second for a duration of five minutes. The nets were then immediately raised, rinsed out, and the captured larvae preserved in 5-10% buffered formalin, providing the first sample. The remaining three samples were similarly collected, each beginning at the point along the shoreline where the previous sample terminated. Approximately 200 m of shoreline and a mean of 40.3 cubic meters of water were sampled with each "push" sample.

Seven diurnal push net samples were collected at the surface in the river concurrent with bongo net sampling (two in early July, three in late July, and two in August) to investigate differences in gear selectivity between bongo and push nets. These samples were preserved as described above for subsequent laboratory processing.

Densities of larvae were computed and a log transformation was used to normalize the data:

$$\text{Ln } (A + 0.5)$$

where A = # of larvae/100 cubic meters of water samples

Analysis of variance (ANOVA) procedures were used to test if larval fish densities in the river samples varied with site (UW vs. LW), time (day vs. night), depth, and distance across river. T-tests were used to test if shoreline densities varied with site

and time. To test if shoreline densities differed from river densities, oneway ANOVA and Duncan's Multiple Range tests were used; for these analyses, surface and bottom samples were pooled to give four replicates at each level (shore, 20, 40 and 50% distance across river). This pooling made rejection more difficult for those instances when surface and bottom samples differed significantly.

The above tests were run on each taxonomic group and sampling date, if larvae were abundant enough. Decision level for all tests was 0.01.

Barge-induced Mortality

Mortality resulting from direct physical damage of fish larvae in the vicinity of moving barges was to have been investigated by comparison of recent mortality in samples collected at the seven sampling points at LW, before and immediately after the passage of tow boats. Sampling was scheduled to be conducted every three weeks from May until August of 1983, provided ichthyoplankton densities were high enough (a minimum of 15 larvae per 100 cubic meters of water sampled). Densities were lower than expected and mortality sampling could be conducted on only two of the five scheduled trips: mid-June and early-July 1983.

Samples were collected using twin bridleless 0.5-m diameter bongo nets (500 micron mesh) towed for five minutes at a speed of 85 cm/sec. This towing speed was selected during pre-sampling field reconnaissance, and was uniform for all samples to allow for valid comparisons. Because mortality levels were being compared, capture-related mortalities had to be kept to a minimum. As towing velocity increases, net-induced mortalities increase (New York University Medical Center 1976, Cada and Hergenrader 1978). However, net velocity may influence length frequency distributions of ichthyoplankton collections (Aron and Collard 1969 cited in Graser 1977). As towing velocity decreases, avoidance capability of larval fish, especially larger individuals, may increase. R. G. King (Ecological Analysts, Inc., personal communication) advised that sampling velocities between 25-50 cm/sec might be an acceptable compromise for this investigation. However, the boat used to tow the nets had a minimum speed of 70 cm/sec. Several reconnaissance tows were made at each of the following speeds: 70, 76, 85, and 104 cm/sec. The percent living larvae in each sample was determined, and based on the results, 85 cm/sec appeared to yield adequate numbers of larvae per sample as well as relatively high percentage of living larvae.

Using the criteria described in King (1977) and Hergenrader et al. (1982), the larvae in the samples were to be separated immediately after collection into three groups: live, recently

killed, and previously dead. King (1977) observed that sacrificed transparent larvae from the Missouri River became opaque within one to two hours following death. G. F. Cada (Oak Ridge National Laboratory, personal communication) recalled that freshly killed larvae began to turn opaque within 15-30 minutes, usually starting at the head region; it took at least one hour for the body to turn completely opaque. Unfortunately, this criterion of identifying recently killed larvae did not work on the Kanawha River because of the larval species composition. Clupeids and Cyprinids predominated in the Kanawha River samples, whereas the Missouri River samples of King (1977) and Hergenrader et al. (1982) were predominately freshwater drum. Sacrificed cyprinids and clupeids from the Kanawha River appeared to turn opaque immediately upon death or while dying, making separation of recently killed larvae impossible. Therefore, for this study the larvae were separated into just two categories: live and dead. After sorting and counting, all samples were preserved in 5-10% buffered formalin.

Tows were made prior and immediately after barge passage, and comparisons of percent live larvae in "before" and "after" samples were used to examine impacts. Any decreases in percent live larvae were to be attributed to barge impacts.

Initial tows at midchannel indicated a high variability in percent living larvae, in both the "before" and "after" samples;

thereby requiring more samples than originally anticipated. This increase in samples required per site, combined with a time constraint, prompted a decision to concentrate efforts on the area where impacts are expected to be greatest: the midchannel or sailing line (Academy of Natural Sciences of Philadelphia 1980).

"Before" barge samples were collected by making tows at midchannel between river mile 34 and 37 (mortality sampling area) no less than one hour following the passage of a barge. The sample from each tow was rushed immediately to a sorting crew where live and dead larvae were sorted within 20 minutes of tow completion. When a barge entered the sampling area, a tow was made at midchannel, beginning approximately 100 m behind the barge, and proceeding in the same direction of travel. Upon completion of the tow, the samples was rushed to the sorting crew for immediate separation. The sampling boat then caught up to the barge and collected another sample. A maximum of two or three tows could be made before the barge left the sampling area. In June, 13 "before" and 10 "after" tows were made in two successive days, and in early July, 25 "before" and 16 "after" tows were completed in three successive days (Table IV.10.3).

Upon examination of the data, it was apparent that in several samples, the total number of larvae was < 15. For small numbers, the effect of just one larva on the "percent live" could be significant. Therefore, samples containing < 15 larvae were

Table IV.10.3. Summary of 1983 sampling activities for mortality.

Date	Sampling location	Number of samples Before	After
6/16	Lower Winfield Midchannel	6	4
6/17	Lower Winfield Midchannel	7	6
7/7	Lower Winfield Midchannel	4	2
7/8	Lower Winfield Midchannel	12	2
7/9	Lower Winfield Midchannel	9	12
Total samples		38	26

discarded from analysis. A two-sided Wilcoxon Rank Sum test was run on each month's samples (June and July) to determine if percent living larvae differed among "before" and "after" samples (decision level = 0.05).

Laboratory Experiments

A literature search failed to find any previously conducted research applicable to achieving the objective of this portion of the study. A study by Gregg and Bergersen (1980) used methods unsuitable for this investigation because of low levels of turbulence created by their apparatus. The work by Morgan et al. (1976) dealt with hull shear force alone; hull shear is only one of several possible causes of mortality associated with vessel passage. In addition, there may be mortality associated with acceleration, impact, abrasion, and abrupt changes in pressure. Therefore, a new procedure was developed to subject larvae to three levels of high velocity water flow and turbulence.

Two stress chambers, each consisting of a glass aquarium (interior dimensions of 30 cm width, 75 cm length, 45 cm depth) divided into two compartments, were constructed (Figure IV.10.5). Larval fish were confined to Side A of the chamber by a divider constructed of fine Nitex netting (130 micron mesh), resting at a 60 degree angle from the horizontal axis. During operation, water entered Side A via two parallel inflow pipes (spaced 8.0 cm

apart) that directed the high velocity water streams in such a manner as to 'sweep' down parallel to the inclined divider, minimizing impingement of larvae against the divider while producing turbulent flow within Side A of the chamber.
Water height was maintained at a fairly constant level by two 5.1 cm diameter PVC siphons connecting Side B with a constant level box equipped with two 7.8 cm diameter PVC spillways. During maximum flow conditions, the mean water level within Side A would rise 4.5 cm above a "resting" water height of 25.5 cm.

To eliminate the additional stress of changes in pH and temperature water for the stress experiments was drawn from the same supply that the larvae were held in. All water was filtered through 130 micron mesh Nitex netting to exclude "stray" larvae. A Homelite gasoline powered centrifugal pump (Model AP 320-1) supplied the required water pressure, and an adjustable gate valve controlled the flow of water to the chamber. All experiments were conducted in a shaded locality to avoid exposure of larvae to direct sunlight.

Three levels of turbulent high velocity flow were used to stress larval fish (Table IV.10.4). These flows were governed by the use of two sets of inflow pipes (0.9 and 1.3 cm inside diameter) and two inflow velocities (800 and 1000 cm/sec). An inflow velocity of 1000 cm/sec with the 1.3 cm inflow pipes was not possible due to limitations of the water pump.

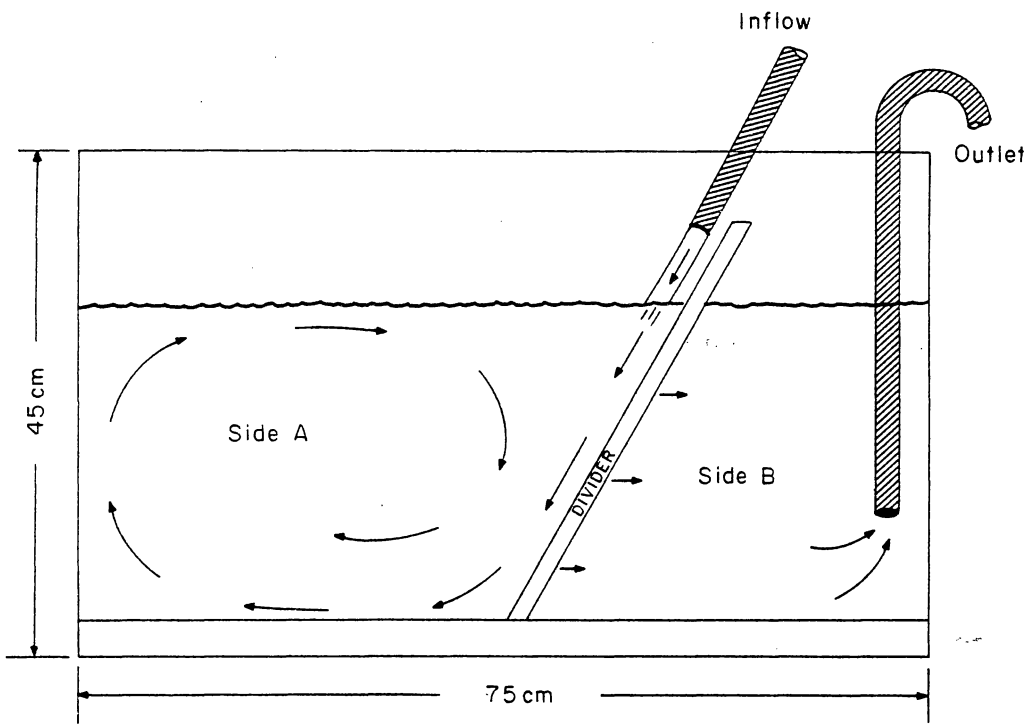

Figure IV.10.5. Profile of larval fish turbulent stress chamber.

Table IV.10.4. Flows used for stress chamber.

Inside Diameter of Inflow Pipes	Inflow Velocities in cm/sec	Volume of water passing through chamber in l/sec
0.9 cm	800	1.0179
0.9 cm	1000	1.2723
1.3 cm	800	2.1237

Four species of larval fish were stressed 0-2 days after swim-up: walleye, carp, bluegill, and channel catfish. Buffalo were to be used but the eggs died as a result of a malfunctioning dechlorinator. Gizzard shad were to be stressed, but sufficient numbers could not be collected undamaged from Smith Mountain Lake, Virginia. Walleye larvae were obtained from King and Queen State Fish Hatchery, Stephensville, Virginia. Experiments were conducted at the hatchery. Carp eggs were stripped from spawning adults collected from the Duck Pond, Blacksburg, Virginia. Recently hatched bluegill larvae were collected off nests in Bull Pond, Christiansburg, Virginia. Channel catfish prolarvae were obtained from Kurtz's Fish Hatchery, Elverson, Pennsylvania. Carp, bluegill, and channel catfish were reared in the laboratory until swim-up. Experiments involving these species were conducted at Cheatham Hall, Virginia Polytechnic Institute and State University, Blacksburg, VA.

A minimum of six replicates were run for each of the three flows, for each species, except catfish. The lower flow (0.9 cm pipes, 800 cm/sec inflow velocities) was not run on channel catfish because no mortalities occurred at the higher flows.

For each replicate, the pump was started, the controlling valve opened to a predetermined setting, and a minimum of 50 larvae were placed in Side A of the chamber (water level 25.5 cm) and allowed to disperse. Two ball valves (one per inflow pipe)

were thrown open simultaneously, allowing water to rush into the chamber. The ball valves were closed simultaneously after 60 seconds. Approximately 30 minutes later, the water level within the chamber was gently lowered to 3.5 cm, and the chamber removed to a counting platform where the larvae remained undisturbed until one hour had elapsed since exposure to the turbulence. The numbers of dead and live larvae were then counted. Larvae that did not respond to both the touch of forceps and immersion in 10% formalin were considered dead. All others were counted as living.

A control group was established for each replicate. A minimum of 50 larvae were placed in a holding tank for at least one hour, and subsequently examined for mortality. No mortalities were observed in the control groups.

All of the above experiments were run with filtered water lacking sediment particles. However, levels of potentially abrasive particles may become elevated for several minutes following barge passage (U.S. Army Corps of Engineers, unpublished data). Larval fish subjected to high velocities in the presence of abrasive particles may suffer higher mortality rates than in water devoid of similar sediments. To examine for an effect of sediment, paired runs (with and without sediment) were made with bluegill and channel catfish larvae.

Two pairs of runs were made with bluegills. For each pair, one tank had 7960 mg of sediment added prior the run (220 mg/l) while the other was run without sediment. Four pairs of runs were made with channel catfish; two pairs with 220 mg/l of sediment, and two pairs with 880 mg/l of sediment. All runs were made with 1.3 cm inflow pipes and 800 cm/sec inflow velocities. Sediments were obtained from the New River, Montgomery County, Virginia, and sifted through screens. Particles of 125-150 microns were used. Weights of sediments were determined with an electronic balance after drying at 60° C for 25 hours.

Channel catfish were excluded from statistical analysis because they exhibited no mortalities during the experiments. A Kruskal-Wallis test was used to determine if the percent mortality varied among the remaining three species for the 1000 cm/sec velocity runs, and a rank analogue of Fisher's Protected LSD was used to determine which pairs of species differed significantly. A Kruskal-Wallis test and rank analogue of Fisher's Protected LSD were also run on the bluegill results (all three levels of turbulence) to determine if mortality varied with level of turbulent flow. A two sample test of proportions (Ott 1984) was used to determine if sediment within the water influenced mortality of bluegill larvae under turbulent conditions. Decision level for all tests was 0.05.

IV.11. ENERGY FLOW MODEL

Model Description

K.R.E.A.M. (Kanawha River Ecosystem Analysis Model) is a linear donor-controlled model of total energy flow/trophic relationships throughout the Winfield pool ecosystem. It can be viewed from three perspectives. The model serves first as a synthesis of biological and physical data collected in 2 dominant habitats found in the Winfield pool of the Kanawha River. Secondly, it represents an extended, quantitative hypothesis of ecosystem function for this river reach. Practically, K.R.E.A.M. is a tool for assessing potential environmental impacts, specifically those associated with an increment to existing levels of tow traffic using the Winfield pool.

The model performs as an abstract and simplified substitute for the real-world ecosystem of the river. Model predictions are dependent on the interaction of its three conceptual parts: biological components, biological processes, and physical perturbation. Organisms and organic matter in the ecosytem are identified as the biological components of the model. The various actions and interactions of these organisms in nature are represented by biological processes in the model. Finally, the physical effects of individual tow passages are considered the perturbation in the model. The components, processes, and perturbation of the model are all represented in mathematical

terms. Each aspect of the model is discussed in more detail below.

Model Components

In all three conceptual portions of the model, there is a balance to be found between realism and model complexity. A considerable amount of simplification is needed to define a collection of organisms or organic matter in the river as a model component. The essential role of each organism in nature is effectively preserved, however, by grouping similar organisms into a common functional identity. For example, while sauger and walleye are distinct species, they share quite similar roles in nature as piscivorous fish, and so are aggregated into a single functional grouping. The characteristics of the piscivorous fish component, like all model components, are weighted based on the relative abundance of those individual species included in the group. Abundance, or standing stock of each trophic group is expressed as the biomass (in kilocalories) present throughout the water column or on the bottom under an average square meter of river surface. Averaging simplifies the model by homogenizing the diverse habitats of shoreline, near-shore, and mid-channel areas.

Biological Processes Simulated by the Model

All interactions and activities of organisms found in the river ecosystem have a common denominator when they are expressed in terms of energy flow. Such diverse processes as the harvest of fish, the fall of leaves into the river, the photosythetic activity of algae, and decomposition of organic matter on the bottom can be related quantitatively through energy flow. In this form, such diverse activities as consumption, assimilation, respiration, egestion, predation, and non-predatory mortality by individual model components are integrated as sources or sinks of energy flow. Conceptually, energy flow might be perceived as a mathematical expression of the "life pulse" of a community. This expression takes the visible form of a web of energy sources and sinks between, to, and from model components (Fig. IV.11.1). In the figure, numbered compartments signify the 19 trophic groups of the model. Energy flows are given by solid and dashed arrows connecting the compartments. Solid lines represent major flows (defined here as single energy flows greater than 10% of the total energy from all sources leading to a recipient component). Dashed lines represent minor flows, not all of which are shown for the sake of simplicity. Also not explicitly depicted in the diagram are the respiration, mortality, waste, or emergence flows. Inputs and outputs of energy to/from the system are denoted by the letters "A" through "H" (in triangles). Inputs include the capture of light energy by phytoplankton and periphyton (A,B respectively), the capture of suspended

particulate organic matter out of transport (C), leaf-fall (D), and utilization of terrestrial insect-fall (E). Harvest of three fish groups is represented by (F), (G), and (H).

Mathematical Development

Each trophic group in the ecosystem is assigned a mathematical counterpart in the model termed a component. Components are referred to by number using the designation X(i), where i ranges from one to nineteen. The complete list of model components with the dominant organisms in each is given in Table IV.11.1. The value of X(i) represents the abundance ($kcal/m^2$) of trophic group i per average square meter. Changes in the standing stock of the trophic groups are simulated through the use of a set of first order differential rate equations (Tables IV.11.2 and IV.11.3). The rate terms in the equation represent the various transfers of energy to and from model component X(i). Rates of input include consumption or other means by which trophic group (i) obtains energy, while rates of output include predation, mortality, transport out of the system, waste products, and respiration costs. Rates are expressed mathematically as simple linear functions of the current standing stock of the donor trophic group using a transfer coefficient. Transfer coefficients for linear donor-controlled models are calculated using the following relation:

$$\text{Flow}(i,j) = a_{ij}\, X(i)$$

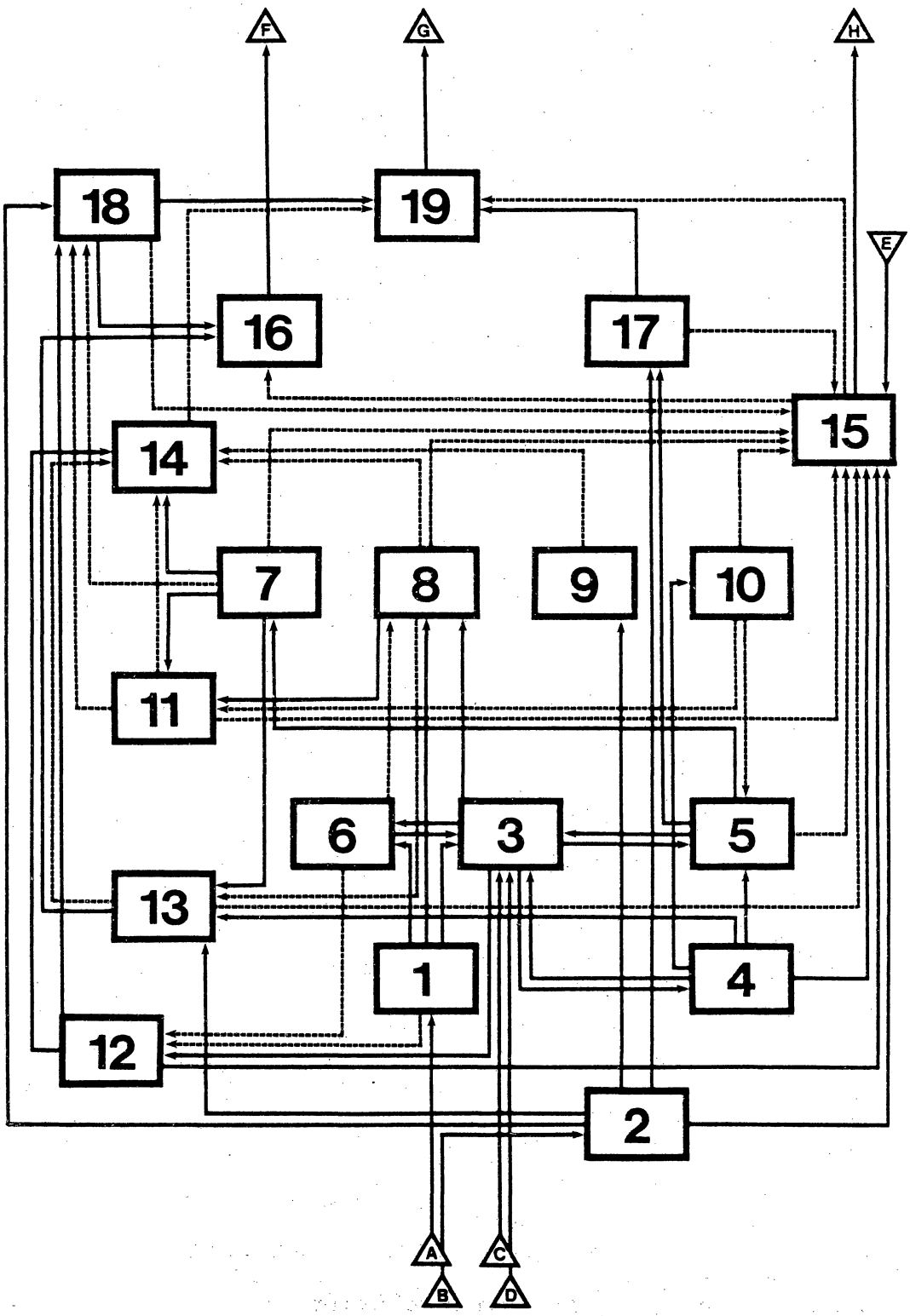

Fig. IV.11.1. Simplified energy flow diagram for K.R.E.A.M. (See text for definition of symbols).

$$a_{ij} = \text{Flow}(i,j) \; / \; X(i)$$

Where:

a_{ij} = transfer coefficient (yr^{-1})

Flow(i,j) = annual rate of flow from component X(i) to X(j) ($kcal/m^2/yr$)

X(i) = average standing stock of component i ($kcal/m^2$)

Thus each transfer coefficient functions as a constant turnover ratio which, when multiplied by the present value of the donor component, yields the rate of transfer between donor and recipient groups.

The differential equation is of the general form:

$$dX(i)/dt = \sum_{j=1}^{n} a_{ji}X(j) - \sum_{j=1}^{n} a_{ij}X(i) + U_i - O_i$$

$$\text{net rate of change} = (\text{rates of input}) - (\text{rates of output})$$

Where:

a_{ij} = transfer coefficient from compartment X(i) to X(j)

U_i = inputs to compartment X(i) from outside the system

O_i = outputs from compartment X(i) to outside the system

n = the number of compartments in the model

Table IV.11.1. Representative organisms for model components.

Model Component	Designation	Representative Organisms
	(Primary Producers)	
Phytoplankton	X(1)	phytoflagellates, green algae, diatoms, etc.
Periphyton	X(2)	diatoms, blue green algae
	(Organo-bacterial Complexes)	
SPOM	X(3)	suspended particulate organic
Benthic CPOM	X(4)	coarse particulate organic matter (> 1μm)
Benthic FPOM	X(5)	fine particulate organic matter (< 1μm)
Zooplankton	X(6)	rotifers, protozoans, etc.
	(Invertebrates)	
Collector Gatherers	X(7)	Orthocladinae, Chironomidae
Collector Filterers	X(8)	Hydropsyche, Chaematopsyche
Scaper Grazers	X(9)	some Tricoptera (rare), snails
Shredders	X(10)	some Plecoptera (rare)
Predators	X(11)	Odonata
Molluscs	X(12)	Corbicula
Crayfish	X(13)	Orconectes
	(Fishes)	
Benthic Invertivore	X(14)	redhorse, drum, etc.
Omnivore	X(15)	carp, catfish, etc.
Crayfish/Piscivore	X(16)	centrarchid basses
Herbi/Detritivore	X(17)	gizzard shad
Midwater Invertivore	X(18)	emerald shiner
Piscivore	X(19)	walleye, sauger, white bass

Table IV.11.2. Differential equations used in the model.

For definitions of terms used in equations, see Table IV.11.3.

```
DX(1)=( CFLOW(20,1)*X(20)*X(1) ) -
     (  (CFLOW(1,3)+CFLOW(1,6)+CFLOW(1,7)+CFLOW(1,8)+
     CFLOW(1,12)+CRESP(1)+CSURPL(1))*X(1)  )

DX(2)=( CFLOW(21,2)*X(21)*X(2) ) -
     (  (CFLOW(2,4)+CFLOW(2,5)+CFLOW(2,9)+CFLOW(2,13)+
     CFLOW(2,15)+CFLOW(2,17)+CFLOW(2,18)+CRESP(2) )*
     X(2)  )

DX(3)=(LEAVES(3)+CFLOW(1,3)*X(1)+
     CFLOW(4,3)*X(4)+CFLOW(5,3)*X(5)+
     CFLOW(6,3)*X(6)+DEFICT(3))-
     (  (CFLOW(3,4)+CFLOW(3,5)+CFLOW(3,6)+CFLOW(3,8)+
     CFLOW(3,12)+CRESP(3) )*X(3)  )

DX(4)=(CFLOW(2,4)*X(2)+
     CFLOW(3,4)*X(3)+CFLOW(7,4)*X(7)+
     CFLOW(8,4)*X(8)+CFLOW(9,4)*X(9)+
     CFLOW(10,4)*X(10)+CFLOW(11,4)*X(11)+
     CFLOW(12,4)*X(12)+CFLOW(13,4)*X(13)+
     CFLOW(14,4)*X(14)+CFLOW(15,4)*X(15)+
     CFLOW(16,4)*X(16)+CFLOW(17,4)*X(17)+
     CFLOW(18,4)*X(18)+CFLOW(19,4)*X(19))-
     (  (CFLOW(4,3)+CFLOW(4,5)+CFLOW(4,7)+CFLOW(4,10)+
     CFLOW(4,13)+CFLOW(4,14)+CFLOW(4,15)+CFLOW(4,17)+
     CRESP(4) )*X(4)  )

DX(5)=(CFLOW(2,5)*X(2)+
     CFLOW(3,5)*X(3)+CFLOW(4,5)*X(4)+
     CFLOW(7,5)*X(7)+CFLOW(8,5)*X(8)+
     CFLOW(9,5)*X(9)+CFLOW(10,5)*X(10)+
     CFLOW(11,5)*X(11)+CFLOW(12,5)*X(12)+
     CFLOW(13,5)*X(13)+CFLOW(14,5)*X(14)+
     CFLOW(15,5)*X(15)+CFLOW(16,5)*X(16)+
     CFLOW(17,5)*X(17)+CFLOW(18,5)*X(18)+
     CFLOW(19,5)*X(19))-(  (CFLOW(5,3)+CFLOW(5,7)+
     CFLOW(5,13)+CFLOW(5,14)+CFLOW(5,15)+CFLOW(5,17)+
     CRESP(5) )*X(5)  )

DX(6)=(CFLOW(1,6)*X(1)+CFLOW(3,6)*
     X(3))-(  (CFLOW(6,3)+CFLOW(6,7)+CFLOW(6,8)+
     CFLOW(6,12)+CFLOW(6,17)+CRESP(6)+CSURPL(6) )*X(6)  )

DX(7)=(CFLOW(1,7)*X(1)+CFLOW(4,7)*
     X(4)+CFLOW(5,7)*X(5)+CFLOW(6,7)*
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     X(6))-(  (CFLOW(7,4)+
     CFLOW(7,5)+CFLOW(7,11)+CFLOW(7,13)+CFLOW(7,14)+
     CFLOW(7,15)+CFLOW(7,16)+CFLOW(7,17)+CFLOW(7,18)+
     CFLOW(7,19)+CFLOW(7,25)+CRESP(7) )*X(7)  )

DX(8)=(CFLOW(1,8)*X(1)+CFLOW(3,8)*
     X(3)+CFLOW(6,8)*X(6))-(  (CFLOW(8,4)+
     CFLOW(8,5)+CFLOW(8,11)+CFLOW(8,13)+CFLOW(8,14)+
     CFLOW(8,15)+CFLOW(8,16)+CFLOW(8,17)+CFLOW(8,18)+
     CFLOW(8,19)+CFLOW(8,25)+CRESP(8) )*X(8)  )

DX(9)=(CFLOW(2,9)*X(2))-(  (CFLOW(9,4)+
     CFLOW(9,5)+CFLOW(9,14)+CRESP(9) )*X(9)  )

DX(10)=(CFLOW(4,10)*X(4))-(  (CFLOW(10,4)+
     CFLOW(10,5)+CFLOW(10,11)+CFLOW(10,13)+
     CFLOW(10,15)+CFLOW(10,16)+CFLOW(10,25)+
     CRESP(10) )*X(10)  )

DX(11)=(CFLOW(7,11)*X(7)+CFLOW(8,11)*
     X(8)+CFLOW(10,11)*X(10)+CFLOW(11,11)*
     X(11)+DEFICT(11))-
     (  (CFLOW(11,4)+CFLOW(11,5)+CFLOW(11,11)+
     CFLOW(11,13)+CFLOW(11,14)+CFLOW(11,15)+
     CFLOW(11,16)+CFLOW(11,17)+CFLOW(11,18)+
     CFLOW(11,25)+CRESP(11) )*X(11)  )

DX(12)=(CFLOW(1,12)*X(1)+
     CFLOW(3,12)*X(3)+CFLOW(6,12)*X(6))-
     (  (CFLOW(12,4)+CFLOW(12,5)+CFLOW(12,14)+
     CFLOW(12,15)+CFLOW(12,17)+CFLOW(12,18)+
     CRESP(12) )*X(12)  )

DX(13)=(CFLOW(2,13)*X(2)+CFLOW(4,13)*
     X(4)+CFLOW(5,13)*X(5)+CFLOW(7,13)*
     X(7)+CFLOW(8,13)*X(8)+CFLOW(10,13)*
     X(10)+CFLOW(11,13)*X(11))-
     (  (CFLOW(13,4)+CFLOW(13,5)+CFLOW(13,14)+
     CFLOW(13,15)+CFLOW(13,16)+CFLOW(13,19)+
     CRESP(13) )*X(13)  )

DX(14)=(TERR(14)+CFLOW(4,14)*X(4)+
     CFLOW(5,14)*X(5)+CFLOW(7,14)*X(7)+
     CFLOW(8,14)*X(8)+CFLOW(9,14)*X(9)+
     CFLOW(11,14)*X(11)+CFLOW(12,14)*X(12)+
     CFLOW(13,14)*X(13)+CFLOW(15,14)*X(15)+
     CFLOW(18,14)*X(18)+NSCTON(14))-
     (  (CFLOW(14,4)+CFLOW(14,5)+CFLOW(14,19)+
     CRESP(14) )*X(14)  )

DX(15)=(TERR(15)+CFLOW(2,15)*X(2)+
     CFLOW(4,15)*X(4)+CFLOW(5,15)*X(5)+
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     CFLOW(7,15)*X(7)+CFLOW(8,15)*X(8)+
     CFLOW(10,15)*X(10)+CFLOW(11,15)*X(11)+
     CFLOW(12,15)*X(12)+CFLOW(13,15)*X(13)+
     CFLOW(17,15)*X(17)+CFLOW(18,15)*X(18)+
     NSCTON(15))-
     ( (CFLOW(15,4)+CFLOW(15,5)+CFLOW(15,14)+
     CFLOW(15,16)+CFLOW(15,19)+CFLOW(15,26)+
     CRESP(15) )*X(15)  )

DX(16)=(CFLOW(7,16)*X(7)+
     CFLOW(8,16)*X(8)+CFLOW(10,16)*X(10)+
     CFLOW(11,16)*X(11)+CFLOW(13,16)*X(13)+
     CFLOW(15,16)*X(15)+CFLOW(17,16)*X(17)+
     CFLOW(18,16)*X(18)+NSCTON(16))-
     (  (CFLOW(16,4)+CFLOW(16,5)+CFLOW(16,26)+
     CRESP(16) )*X(16)  )

DX(17)=(CFLOW(2,17)*X(2)+
     CFLOW(4,17)*X(4)+CFLOW(5,17)*X(5)+
     CFLOW(6,17)*X(6)+CFLOW(7,17)*X(7)+
     CFLOW(8,17)*X(8)+CFLOW(11,17)*X(11)+
     CFLOW(12,17)*X(12)+NSCTON(17))-
     (  (CFLOW(17,4)+CFLOW(17,5)+CFLOW(17,15)+
     CFLOW(17,16)+CFLOW(17,19)+CRESP(17) )*X(17)  )

DX(18)=(NSCTON(18)+CFLOW(2,18)*X(2)+
     CFLOW(7,18)*X(7)+CFLOW(8,18)*X(8)+
     CFLOW(11,18)*X(11)+CFLOW(12,18)*X(12))-
     ( (CFLOW(18,4)+CFLOW(18,5)+CFLOW(18,14)+
     CFLOW(18,15)+CFLOW(18,16)+CFLOW(18,19)+
     CRESP(18) )*X(18)  )

DX(19)=(NSCTON(19)+CFLOW(7,19)*X(7)+
     CFLOW(8,19)*X(8)+CFLOW(13,19)*X(13)+
     CFLOW(14,19)*X(14)+CFLOW(15,19)*X(15)+
     CFLOW(17,19)*X(17)+CFLOW(18,19)*X(18))-
     ( (CFLOW(19,4)+CFLOW(19,5)+CFLOW(19,26)+CRESP(19) )*
     X(19)  )

DX(20)= ( ALIGHT(20) ) -
     ( LLOST(1) + (CFLOW(20,1)*X(20)*X(1)) )

DX(21)= ( ALIGHT(21) ) -
     ( LLOST(2) + (CFLOW(21,2)*X(21)*X(2)) )
```

Table IV.11.3. Definitions of terms used in model equations.

X(I,J)	term for the biomass (in kcal/m^2) of component (I) in river segment (J).
DX(I,J)	term for the rate of change of component (I) in river segment (J).
CFLOW(I,J)	transfer coefficient from X(I) to X(J) in river segment (K).
CRESP(I,J)	transfer coefficient for respiration by component (I) in river segment (J).
CSURPL(I,J)	transfer coefficient for extrasystem energy flow from component (I) in river segment (J).
DEFICT(I,J)	driving function representing extrasystem input of energy to component (I) in river segment (J).
NSCTON(I,J)	term for utilization of drifting aquatic insects by component (I) in river segment (J).
TERR(I,J)	term for utilization of terrestrial insects by component (I) in river segment (J).
ALIGHT(I,J)	ambient rate at which light energy is captured by primary producer (I) in river segment (J).
LLOST(I,J)	rate at which light energy potentially captured by primary producer (I) in river segment (J) is lost as a result of increased turbidity.

The equations calculate the component's rate of change (per year in this case), as the difference between rates of energy input and rates of energy output. A four step Runge-Kutta method of numerical integratation was used to solve the differential equations. Both the equations and the solution technique are imbedded within a time loop in the model program. The simulation is analogous to a "motion picture" with small incremental changes in 1) biomass, and 2) rates of energy transfer calculated each time the program executes the loop. In this case the loop represents a time step of 2.128 minutes (0.000005 years). With each iteration, the model computes the solution to a differential equation and multiplies this change-per-year term by the length of the time step (in years) to calculate the change in biomass of a component over one time step. The change (+/-) is added to component's prior biomass value and a new value is assigned to that component. In the following time step the new biomass value (through multiplication by various transfer coefficients) will cause increases/decreases in the rates of transfer for which that component is a donor. This in turn effects the rates of change calculated for both donor and recipient components when the differential equations are next solved.

Ecosystem Perturbation

Tow passage is known to affect the physical environment of the river in a predictable fashion given the physical layout of the waterway, the characteristics of the tow itself, and other factors. The primary avenue of impact was identified by the Academy of Natural Sciences of Philadelphia (1980) as the (temporary and local) resuspension of particulate matter associated with increased water turbulence in the vicinity of the passing towboat. It is important to recognize that the physical effects of tow passage on the river environment are understood on an individual-tow basis. Tow induced forces exist for only a certain length of time at any one river location. Consequently, the model is designed to simulate individual passages both in time and in space. This requires that tow location be included as a dimension of model components and processes.

Modeling of the Winfield Pool

To accomodate the dimensions in which the disturbance occurs, the model uses a river segmentation approach in which the Winfield pool is subdivided into two segments. Each river segment is simulated independently with uniquely specified model parameters. The more downstream model segment represents Kanawha River miles 31 through 55 (LW) while the remaining portion of the Winfield pool (river miles 56 through 67), are represented by the upstream model segment. The linkage between model segments

exists only conceptually, in that during the simulation of a tow passage the perturbation caused by the towboat is first represented in the one river segment, and progressively moves on to the adjacent river segment. Biological processes in one segment, however are independent of those simulated in the other segment.

Modeling of Tow Traffic

Tow passage frequency data at intervals along the pool were projected by the U.S. Army Corps of Engineers (Huntington District Corps of Engineers, unpublished data) for each traffic scenario and are listed in Table IV.11.4. Data pertaining to all intervals upstream or downstream of river mile 58 were averaged to obtain the tow frequencies used by the model in the UW and LW segments. These frequencies are independent of the direction traveled by the tow. It was assumed that over the course of a full year that upbound and downbound towboats were in equilibrium, and with this assumption the tow passage frequencies were halved and the halves assigned a direction of travel either upbound or downbound. The model was run for five different scenarios; a baseline simulation, three projected traffic levels, and an additional experiment.

Scenario #1: Baseline Conditions

Table IV.11.4. Traffic frequencies (in tows per year) for each scenario.

River mile interval	S1 1) Present tow traffic conditions	S2 2) Future, w/o Project conditions	S3 3) Future, Replace of Winfield	S4 4) Future, Replace W. and Marmet
30.6-39.6	4,167	6,678	7,508	9,250
39.6-49.0	4,167	6,010	6,307	9,250
49.0-55.0	3,959	5,676	5,931	8,788
55.0-58.0	3,584	5,075	5,256	7,955
58.0-64.0	3,167	4,407	4,505	7,030
64.0-67.7	3,334	4,674	4,805	7,400
	Traffic frequencies (tows per day) used by the model			
"UW"	9	12.5	13	20
"LW"	11	16	18	24
	Traffic increment (tows per day)			
"UW"	0	3.5	4	11
"LW"	0	5	7	13

The first scenario represented baseline conditions of tow traffic throughout the Winfield Pool, i.e. the levels of traffic present during the 1982-83 period when biological data were collected. Tow frequencies (bidirectional) ranged from 3,167 tows per year in the upper Winfield pool to 4,167 tows per year near the Winfield Locks. These frequencies, expressed on a per day basis, ranged from slightly under 9 to almost 11.5 tow passages per day. The maximum number of passages per day at Winfield Locks and Dam was 20.8.

As previously discussed, under baseline conditions the increment of tow traffic above and beyond 1982-83 conditions is null and the model does not recognize these tow passages explicitly in the simulation. It is assumed that the biological community of the river is representative of, and in equilibrium with, this level of traffic. A simulation was performed which demonstrates the model's ability to remain at steady state, i.e. no change in the standing stocks of any trophic group over the course of the simulation. The null simulation was needed only to provide a reference point for the output of other simulations. In the scenarios to follow (#2 - #4), simulations were conducted in which net traffic (in excess of baseline) was considered explicitly by the model.

Scenario #2: Without Project Conditions

The volume of tow traffic using the Winfield Pool is projected to increase beyond 1892-83 levels regardless of whether Winfield or Marmet are replaced (Table IV.11.4). Scenario #2 evaluates the potential impact of this increase in traffic.

Bidirectional frequencies were projected to be approximately 4,407 tows per year in the upper pool (between river miles 58. and 64.) and about 6,678 tows per year near the Winfield Locks. These frequencies ranged from slightly over 12 to over 18 tow passages per day, and represent an average traffic increment (modeled over baseline) of 3.5 tows per day in the upper river segment and 5 tows per day in the lower river segment. The maximum number of passages per day at Winfield Locks and Dam was 33.3, an increment of 12.5 tows per day over baseline conditions.

Scenario #3: Replacement of Winfield

Given the replacement or improvements in Winfield Locks and Dam, there is a projected increase in the volume of tow traffic which can be efficiently passed through the Winfield pool. The potential impact of this increase in traffic is evaluated in Scenario #3.

Tow frequencies projected for this scenario ranged from approximately 4,505 tows per year in the upper pool (between river miles 58. and 64.) to about 7,508 tows per year near the

Winfield Locks. There is very little projected increase in tow passage frequencies in the upper pool without improvements to Marmet navigation structures. On a per day basis, tow passage frequencies range from over 12 to over 20.5. The model explicitly recognizes an average traffic increment (over baseline) of 4 tows per day in the upper river reach and 7 tows per day in the lower river. The maximum number of passages per day at Winfield Locks and Dam is projected to be 37.4, an increment of 16.6 tows per day over baseline conditions.

Scenario #4: Replacement of Winfield and Marmet

This scenario represents the highest levels of navigation use in which both Winfield and Marmet lock and dam structures are replaced/improved to more efficiently pass traffic through the Winfield pool. Under this scenario, bidirectional tow passage frequencies range from approximately 7,030 in an upper reach of the pool to 9,250 in the lower reaches of the pool (20 to 24 tows per day). All areas of the Winfield pool would experience a significant increment over baseline traffic conditions (average increments of 11 and 13 tow passages per day modeled for UW and LW, respectively.)

Scenario #5: Theoretical Investigation of Ecosystem Dependence on Primary Production

The following scenario is included not as an investigation of potential impacts caused by tow traffic but rather as a theoretical evaluation of the role played by primary producers (phytoplankton and periphyton) in the Winfield pool. In this experiment the primary production rate was arbitrarily reduced to one-tenth of its original value for each primary producer.

The model does not recognize any level of tow traffic explicitly in this scenario. However, for the sake of comparison, a 90% reduction in gross primary production corresponds to a traffic level of 64.8 additional tows per day. This scenario represents about a five fold increase over the traffic levels projected for Scenario #4, and may well exceed the maximum number of tows which would ever be able to use the Kanawha River on account of physical limitations.

Modeling of Tow Passage

Tow passage simulation is performed by a program subroutine which monitors the entry, exit, and passage of individual tows within the Winfield pool. The subroutine keeps a running record of each tow's status, i.e. time of entry into pool, rate and direction of travel, origin, present location, and destination. This record is then used to compute, for each modeled river segment, the elapsed time which has passed since the last towboat occupied the segment. This information is then available to

compute the effective turbidity during the return to ambient or pre-passage conditions.

Tows are recruited into the pool from both Winfield and Marmet Locks at a constant rate specified by the traffic scenario. As discussed earlier, each simulated tow represents a tow in excess of the present number already operating on the river. The frequencies of departure were calculated on the basis of this future increment only.

All simulated tows which depart from Marmet into the Winfield pool are assigned a destination of Winfield Locks, while a random number generator is used to assign the destination of upbound tows departing from Winfield Locks. This assignment is based on a probability distribution function specific to the traffic scenario under consideration. In scenarios #2, #3, and #4 the probabilities of an upbound tow returning to Winfield after reaching Charleston is 30, 43, and 17%, respectively. The function results in a tow passage frequencies for each segment equal to the average frequency of passage in the UW and LW river miles. In all cases a higher frequency of passage results for the lower Winfield-like segment. In reality the distribution of destinations is more complex, but it is the tow frequency in each segment, and not the destinations, which are important. All simulated towboats are functionally equivalent, although the program subroutine includes a mechanism for specifying individual

tows with unique characteristics of horsepower, number and type of screws, number of barges, empty or full, etc.

Ecosystem processes which are sensitive to the level of suspended particulate matter are affected by the presence of individual towboats on the river. In an analogous manner, the model can modify the flows representing sensitive processes whenever and wherever the simulated physical environment is altered beyond ambient conditions by a recent tow passage. Numerous biological activities are potentially affected by tow passage, including primary production, respiration, food gathering, spatial orientation, sight feeding success, spawning behavior, and others. These may be roughly divided under two sources of impact; activities sensitive to water turbulence, and those sensitive to increased suspension of particulate matter. There is very little evidence for significant disturbance of biological processes (which naturally occur in a lotic environment) by turbulence alone. A worst case tow passage (upbound, fully loaded, high horsepower) can more than triple ambient velocities, yet the duration (maximum 11 minutes) and extent (less than one-third channel width) of elevated flows are relatively limited (Hochstein and Adams, 1985). The majority of the river experiences only mild, if not insignificant increases in current velocities. On the contrary increased suspension of particulates, if of sufficient magnitude and duration, would be expected to significantly affect certain processes. Due to

mixing the entire river, and not just a limited area, is subject to tow-induced increases in turbidity. The sensitivity of several biological processes to increased turbidity were considered, but evidence for significant impacts could be justified only for primary production. This is because tow-induced turbidity levels, which may temporarily exceed ambient turbidity conditions, still do not exceed natural levels to which heterotrophic river organisms are presumably already adapted.

Modeling of Primary Production

Primary production is modeled continuously during the simulation; no diurnal fluctuation is assumed. Primary producers are represented by two trophic groups in the model, phytoplankton and periphyton (X(1) and X(2) respectively). For each of these groups, the potential rate of primary production remains constant throughout the simulated year, although mechanisms are in place to make this a seasonal driving function of the model. The gross primary production rate is considered by the model to be a potential rate which applies under ambient (1982-83) turbidity conditions. When a simulated tow first enters a river segment, this elevates the effective turbidity level 50% over ambient. Effective turbidity then declines linearly, returning to ambient after a 20 minute recovery period. The realized rate of primary production is sensitive to the effective turbidity level in each

model segment at all times. Whenever the effective turbidity exceeds the ambient level, all potential primary production in the effected segment is suspended, i.e. the realized rate of primary production equals to zero. Realized primary production returns to the pre-passage rate at the same time the effective turbidity returns to ambient levels.

In reality, primary production may not be entirely suspended, even in the sailing line, and further it is not reduced to the same extent in all areas of the river occupied by a passing towboat. There is also a gradual return to the pre-passage rate of photosynthesis from the depressed rate. Lacking field measurements of these phenomena, and in the interest of conservatism, the step function was selected.

Derivation of Biological Parameters

This section documents the sources of biological parameters used in the model. In order to minimize repetition of similar information, the nineteen trophic groups of the model are simplified into six classes; primary producers, organic matter, zooplankton, insect invertebrates, other invertebrates, and fish. For each class of trophic groups, general derivations of consumption, waste, respiration, production, extrasystemic inputs/outputs, and standing stock are presented. Specific exceptions are noted where the treatment of a particular trophic

group differs from the norm in its class. Parameter derivations are presented sequentially, that is, the most direct or independent estimates preceed other estimates which depend on the former in their own derivation. Average annual standing stock estimates are expressed in $kcal/m^2$, while the remaining parameters are expressed as annual average energy flows in $kcal/m^2/yr$.

Before commencing with the methods used to derive parameters, it is important to note that in several instances all flows pertaining to a trophic group have been adjusted by a constant factor. This was done in order to reconcile different trophic levels where apparent production at the lower level was insufficient to meet the estimate of total consumptive demand by trophic groups at higher levels. In such cases greater confidence was placed on the estimate of consumptive demand, and the production of the trophic group at the lower trophic level was increased along with the biomass, consumption, waste, and respiration flows. Adjustment factors are listed in Table IV.11.5. In the sections to follow the application of an adjustment factor will be noted by an asterisk following the names of the affected trophic groups.

Class: Primary producers - Phytoplankton X(1),
Periphyton* X(2)

Table IV.11.5. Adjustment factors used to reconcile different trophic levels in the model.

Trophic Group	Name	River segment UW	River segment LW
x(1)	Phytoplankton	-	-
x(2)	Periphyton	2	2
x(3)	SPOM	-	-
x(4)	CPOM	-	-
x(5)	FPOM	-	-
x(6)	Zooplankton	-	-
x(7)	Coll/Gatherers	-	2
x(8)	Coll/Filterers	-	70
x(9)	Scraper Grazers	-	-
x(10)	Invt Shredders	-	25
x(11)	Invt Predators	-	2
x(12)	Molluscs	-	-
x(13)	Crayfish	-	-
x(14)	B. Invertivores	-	-
x(15)	Omnivores	-	-
x(16)	Cray/Piscivores	-	-
x(17)	Herb/Detritivores	-	-
x(18)	M. Invertivores	10	10
x(19)	Piscivores	-	-

The data on which primary producer energy flows are based comes from Sections V.5 and V.6. The sole input of energy to phytoplankton and periphyton is modeled specifically as average annual gross primary production. Field measurements of gross primary production and respiration are given in Table V.5.6 for phytoplankton and V.6.1 for periphyton. Note that respiration by the light/dark bottle methodology does not represent primary producers exclusively, but rather the autotrophs in concert with whatever micro and macrofauna were present in the samples. In order to model energy flow through the primary producer trophic groups, production and respiration energy flows must be comparable (pertain only to the autotrophs). As such, the respiration data collected in the field are not appropriate for use in the model. Respiration was assumed to be 25% of annual average gross primary production. Extracellular release of photosynthetic products was assumed to be 10% of gross primary production for both trophic groups. After satisfying consumptive demands, the remaining phytoplankton production is partitioned equally between export through downstream transport and non-predatory production loss to S.P.O.M.. In the case of periphyton all net production in excess of consumptive demand is fated to C.P.O.M.

The methods for determining standing stocks are given in Sections IV.5 and IV.6. The original calculation assumed ten square meters of substrate suitable for periphyton per meter of

river length. This estimate was doubled to provide sufficient production to support other trophic groups which graze on periphyton.

Class: Particulate organic matter - S.P.O.M. X(3), C.P.O.M. X(4), F.P.O.M. X(5)

Particulate organic matter groups were defined to include dead organic matter as well as associated microbial decomposers. Energy flows among the particulate organic matter trophic groups are more speculative than those associated with the remaining sixteen groups of the model. This is due to the difficulty in making direct field measurements of detrital processes in a large river. The following section outlines the hypotheses made in order to obtain energy flow estimates for the organic matter groups.

Suspended, benthic-coarse, and benthic-fine particulate matter (hereafter S.P.O.M., C.P.O.M., and F.P.O.M.) are proposed as existing in a dynamic equilibrium within the river environment. Under this hypothesis, assumptions made regarding energy flows for any one of these trophic groups necessarily constrain flows involving the other two groups. Consequently energy flows were estimated for F.P.O.M. first, C.P.O.M. second, and S.P.O.M. last, as then any constraints which might arise from initial hypotheses would be extended "backwards" through the

particulate organic matter groups and away from conflicts with more solidly established energy flows for living groups in the rest of the model.

Benthic F.P.O.M. - It is assumed that from year to year F.P.O.M. neither accumulates nor declines; that the combined sources of F.P.O.M. are in equilibrium with all sinks. Sources of F.P.O.M. in the model include the waste products of all non-planktonic biota (W), the breakdown of materials originally present as C.P.O.M. (B), and the settling of materials once present as S.P.O.M. (S1). Sinks of F.P.O.M. include that consumed by other groups (P), that which is resuspended to S.P.O.M. (S2), and that which is decomposed by bacteria with their associated respirtory losses (R). Thus:

$$W + B + S1 = P + S2 + R$$

We further assume that settling (S1) and resuspension (S2) are approximately equal over the year (S1 = S2 = S), and that about half of the total combined sources of F.P.O.M. are respired by decomposers per year. Then:

$$(W + B + S)\ (0.5) = R$$

$$(W + B + S)\ (0.5) = P + S$$

The above pair of equations were then solved for B and R after hypothesizing a value for S (the amount of settling/resuspension) of ten times the amount of F.P.O.M. consumed by all biota per year.

C.P.O.M. - As with the previous example, we assume that there is no net yearly change in the amount of C.P.O.M., and that the combined sources are in equilibrium with the combined sinks. Sources of C.P.O.M. include that supplied from mortality of nonplanktonic biota (M) and the settling out of leaves onto the bottom (L). C.P.O.M. consumed by organisms (P), that resuspended off the bottom (S), that which is broken down into F.P.O.M. (B), and that used in respiration by decomposers (R) are considered as sinks. Thus:

$$M + L = P + S + B + R$$

It is assumed that half of all sources of C.P.O.M. are respired by bacterial decomposers each year. It is also hypothesized that the amount of C.P.O.M. resuspended over the year equals the amount processed to F.P.O.M. (S = B). Thus:

$$(M + L)\ (0.5) = R$$

$$(M + L)\ (0.5) = P + S + B = P + (2)B$$

The pair of equations are then solved to give estimates of L and R.

S.P.O.M. - The balance of sources and sinks of S.P.O.M. is nearly achieved by the above constraints on energy flows involving S.P.O.M., given measured estimates of average annual respiration in the water column. Sources include the amount of S.P.O.M. removed or utilized from that being transported downstream yearly (I), that contributed through waste and mortality of planktonic biota (WM), leaffall onto the river surface (LF), and that resuspended from C.P.O.M. and F.P.O.M. (R1 and R2, respectively). Sinks of S.P.O.M. include that which settles out to C.P.O.M. and F.P.O.M. (S1, S2), that consumed by other trophic groups (P), and that lost to respiration during decomposition (R). Thus:

$$I + WM + LF + S1 + S2 = R1 + R2 + R + P$$

All but the first term above are measured values or are otherwise constrained by prior assumptions. This equation was then solved to estimate the amount of S.P.O.M. removed from downstream transport.

The average concentration of S.P.O.M. throughout the year was estimated by subtracting from the average concentration of seston (Section V.4, includes living and dead organic matter) the

average concentrations of phytoplankton and zooplankton, yielding 0.0031 g AFDW per liter. Standing stock of S.P.O.M. was estimated by assuming 4.8 kcal per gram AFDW and multiplying by the volume of water over the average square meter of river bottom. Average depths for UW and LW were taken as 5 and 7 meters, respectively.

Direct field measurements of benthic coarse and fine particulate organic matter abundance were not made. Standing stocks are estimated by using the total annual energy flux through each component and assuming turnover ratios of 100 and 50 times per year for C.P.O.M. and F.P.O.M., respectively. Estimates were then rounded to indicate the many approximations used in their derivations.

Class: Zooplankton - Zooplankton X(6)

Our estimates for energy flows involving zooplankton come directly from methods listed in Section IV.7. Food types consumed by zooplankton included phytoplankton and suspended particulate organic matter. Gross consumption was estimated as that required to support measured production. The production estimate was first divided by an assumed production efficiency of 30%, and the result divided by a composite assimilation efficiency of 61%. Net production in excess of consumptive demand was split evenly between S.P.O.M. and downstream transport out of the system.

Class: Insect invertebrates - Collector Gatherers (* LW only), Collector Filterers (* LW only), Scraper Grazers, Shredders (* LW only), Predators (* LW only)

Four of the five aquatic insect groups were sufficiently abundant to make biomass and production estimates from the field data collected; collector/gatherers, collector filterers, shredders, and predators. The production estimates given in Section V.8 are made for suitable substrates only, and do not represent the average habitat simulated by the model. These data are first converted to kilocalories and then adjusted by the proportions of suitable substrate found at each site (see Table IV.11.6 and IV.11.7). Production estimates for these four benthic invertebrate groups, after the adjustment for the average square meter of substrate at LW, fell short of the combined predatory demands of the fish trophic groups at that site. As greater confidence was placed on fish abundance, production, and consumption estimates, we elected to arbitrarily multiply the biomass, production, consumption, and respiration, and waste flows of the LW insect groups by the following factors: collector/gatherers (x 2), collector filterers (x 70), shredders (x 25), and predators (x 2). The most probable explanation for this deficit is that snags or other debris provide suitable substrates and that this factor was underestimated in our original calculation.

Table IV.11.6. Conversion factors used to derive model biomass and production estimates of UW benthic invertebrate groups.

Trophic Group	Name	kilocalorie conversion [1]	% suitable substrate [2]
X(7)	Collector Gatherers		
	Oligochaetes	5.0	0.72
	Baetidae	5.3	0.28
	Stenacron	5.3	0.28
	Stenonema	5.0	0.28
	Ephemerella	5.0	0.28
	Tricorythodes	5.0	0.28
	Caenis	5.0	0.28
	Cyrnellus	5.0	0.28
	Orthocladiinae/ Chironominae CP	5.5	0.28
	Orthocladiinae/ Chironominae SS	5.5	0.72
X(8)	Collector Filterers		
	Cheumatopsyche	5.6	0.28
	Isonychia	5.0	0.28
	Hydropsyche	5.6	0.28
	Simulium	5.0	0.28
X(10)	Shredders		
	Taeniopteryx	5.0	0.28
	Strophopteryx	5.0	0.28
X(11)	Predators		
	Tanypodinae CP	5.5	0.28
	Tanypodinae SS	5.5	0.72

[1] taken from Cummins and Wuycheck (1971) for same or similar taxa
[2] Corps of Engineers unpublished sediment survey data
CP = cobble/pebble substrates
SS = silt/sand substrates

Table IV.11.7. Conversion factors used to derive model biomass and production estimates of LW benthic invertebrate groups.

Trophic Group	Name	kilocalorie conversion [1]	% suitable substrate [2]
X(7)	Collector Gatherers		
	Oligochaetes	5.0	0.99
	Baetidae	5.3	0.01
	Stenacron	5.3	0.01
	Stenonema	5.0	0.01
	Ephemerella	5.0	0.01
	Tricorythodes	5.0	0.01
	Caenis	5.0	0.01
	Cyrnellus	5.0	0.01
	Orthocladiinae/ Chironominae CP	5.5	0.01
	Orthocladiinae/ Chironominae SS	5.5	0.99
X(8)	Collector Filterers		
	Cheumatopsyche	5.6	0.01
	Hydropsyche	5.6	0.01
X(10)	Shredders		
	Taeniopteryx	5.0	0.01
	Strophopteryx	5.0	0.01
X(11)	Predators		
	Argia	5.1	0.01
	Tanypodinae CP	5.5	0.01
	Tanypodinae SS	5.5	0.99

[1] taken from Cummins and Wuycheck (1971) for same or similar taxa
[2] Corps of Engineers unpublished data (sediment survey data)
CP = cobble/pebble substrates
SS = silt/sand substrates

Section V.8 details the trophic basis for production for the major insect invertebrate taxa. Food types included the following categories: animal, diatom, other algae, detritus, and vascular plant. Total consumption for the four trophic groups is calculated as that required to support measured production, assuming 1) net production efficiency of 50% and 2) a 10, 30, and 70% assimilation efficiencies for detritus/vascular plants, diatoms/other algaes, and animal matter in the diets, respectively. Gross consumption was then partitioned into individual feeding flows between trophic groups. Partitioning was based on individual taxa diet percentages (see Table V.8.6) weighted according to the relative proportions of the taxon in each trophic group. Net production in excess of consumptive demand was divided between emergence by adults (20%) and natural mortality losses to C.P.O.M. (80%). Estimates of standing stock come from data in Tables V.8.4 and V.8.5. Again, these data pertain specifically to suitable substrates, and do not represent the average square meter of river habitat simulated by the model. Abundance estimates are first converted from dry weights to kilocalories, then weighted to represent the average square meter of available substrate. Conversion and weighting factors are given in Tables IV.11.6 and IV.11.7.

One group, the scraper grazers X(9), was not well represented in the field collections and was given only token representation in the model. Energy flow estimates for this

group were generated on the basis of the only data available, that being the total consumption of scraper grazer organisms (primarily snails) by other trophic groups. Scraper/Grazer biomass was estimated indirectly by establishing a total production value which would satisfy consumptive demand for these organisms, and then assuming a production to biomass ratio of 0.5. Consumption and respiration were back calculated from the assumed production value using a net production efficiency of 34% and an assimilation efficiency of 30%. Evidently scraper grazers play an insignificant role in the functioning of the Kanawha River ecosystem, thus the lack of field derived measurements is not expected to influence the results of the modeling effort.

Class: Noninsect invertebrates - Molluscs X(12), Crayfish X(13)

No direct measurements of production or abundance were made for molluscs or crayfish. These were estimated using methods similar to Scraper/Grazers above. A production value sufficient to satisfy consumptive demands was adopted, after which a P/B ratio of 4.1 was used to approximate mollusc biomass (Aldridge and McMahon 1978), while crayfish biomass was estimated using a P/B ratio of 1.25 (Momot and Gowing 1977). Net production efficiencies of 60 and 50%, and assimilation efficiencies of 13 and 15% were used in the back calculation of total consumption by molluscs and crayfish, respectively.

Class: Fish - Benthic Invertivores X(14), Omnivores X(15), Crayfish/Piscivores X(16), Herbivore/Detritivores X(17), Midwater Invertivores* X(18), Piscivores X(19)

The original estimates of biomass, production, and consumption in Section V.9 pertain to the near-shore habitats that were sampled, and require adjustment to represent the average habitat of the river. Since the model represents the river on an average square meter basis, and near-shore and off-shore habitats are not utilized equally by all species of fish, the nearshore data are not appropriate for direct use in the model. Estimates of the biomass supported by average habitat were made by averaging the near-shore biomass data with off-shore biomass approximations. Approximations of off-shore biomass for each trophic group were obtained by multiplying the near-shore biomass estimates by a species-weighted open-water conversion factor modified from Davies and Shelton (1983). Weighting is based on the relative contribution to total trophic group biomass made by each species for which conversion factors were available. The biomass estimates used by the model are the weighted average of the near-shore estimates and off-shore approximations. Weighting here is based on the proportion of each habitat type at each site determined by bathymetric charts of the river channel. These adjustments had the effect of reducing the standing stocks

of three trophic groups: benthic invertivores, crayfish/piscivores, and piscivores.

Our estimate of midwater invertivores production, even after the above reduction in average standing stocks of predatory groups, was still grossly less than that which could reasonably be expected to support consumptive demand. Small fish (<100mm) could not be collected with the same efficiency as larger individuals, and it was felt that the various shiners and young-of-year fishes which dominate this trophic group were significantly underrepresented in our collections. Attempts at seining for small minnows were made but discontinued due to the lack of shallow water free from obstructions. We arbitrarily increased both the standing stock and energy flows of midwater invertivores by ten fold in order to supply the consumptive demands of other fish.

Production was adjusted by weighting the observed turnover ratios of the major fish species in each trophic group according to relative contribution to total group biomass, and multiplying this weighted average P/B ratio with the revised (habitat-weighted) biomass estimates.

Consumption estimates were obtained indirectly from production, and consequently these also required re-calculation. First a composite assimilation efficiency was established based on the proportions of various food types in the diets, assuming

that animal foods, vegetable foods, and detrital foods are 70%, 30%, and 10% assimilated (Table IV.11.8). A net production efficiency of 30% was assumed (Kozlofsky 1968) to complete the calculation.

Production in excess of consumptive demand (surplus) was partitioned between natural mortality and harvest losses. Harvest was assumed to be 25, 50, and 50% of the production surplus for omnivores, crayfish/piscivores, and piscivores, respectively.

Types of Model Output

Model output consists of two types. The first type of output consists of a time series of the standing stocks for all trophic groups. Each time series includes instantaneous standing stock values (at some interval requested of the user prior to model operation), along with the percentage change between initial and final values for the abundance of each trophic group.

The second form of simulation results is a set of energy budgets for each trophic group in each river segment. Energy budgets may be obtained for either pre- or post-simulation as a user option. An energy budget details all sources and sinks of energy flow through a trophic group, and can be used to evaluate the various pathways with which energy flows through the

Table IV.11.8. Weighted assimilation efficiencies by fish trophic groups.

Trophic Group	Name	Composite efficiency UW	Composite efficiency LW
X(14)	Benthic Invertivores	0.6778	0.6514
X(15)	Omnivores	0.4885	0.5056
X(16)	Crayfish/Piscivores	0.7000	0.7000
X(17)	Herbivore/Detritivores	0.1480	0.1576
X(18)	Midwater Invertivores	0.6320	0.6320
X(19)	Piscivores	0.7000	0.7000

ecosystem. This information is further summarized in a flow analysis to be described below.

Ecosystem Flow Analysis

The flow analysis used (Finn 1976,1980) integrates all energy flows in the model to account for the ultimate sources and sinks of energy from the perspective of any trophic group. The "ultimate" nature of the analysis lies its ability to partition the flow through any model component beyond the immediate components to which it is directly linked. In other words, energy flows are traced throughout the entire ecosystem model, and information about indirect linkages is revealed. Two matrices are produced in the analysis which yield concise and useful summaries of the various energy pathways in the ecosystem.

The first matrix, hereafter refered to as the "N^{**}" matrix, contains information which addresses the eventual fate of energy passing through a particular trophic group. Each element of the N^{**} matrix, $N^{**}(I,J)$, gives the relative proportion of energy flow through trophic group (J) which ultimately will flow through trophic group (I) as well. Thus the N^{**} matrix provides a quantitative expression of how any chosen trophic group eventually (directly or indirectly) supports any other. In a simple cascading model in which all energy flow proceeds directly out of the system, these fractions would all be less than or

equal to unity. In this model, however, there are instances in which flows (directly and indirectly) pass through a trophic group more than once. In these cases fractions in excess of unity result to indicate the internal cycling which is involved. In all cases the percentage is based on the annual energy flux passing through the donor component.

The N^{**} matrix is derived by first defining an energy flow matrix in which any element F(I,J) of the matrix represents the annual flow of energy from trophic group X(J) to trophic group X(I), and a vector T, where element T(J) represents the total flow of energy passing through the donor trophic group X(J) per year. The F matrix is then divided (in elementwise fashion) by the T vector to produce a Q'' matrix:

$$Q''(I,J) = F(I,J) / T(J) \qquad (1)$$

The N^{**} matrix is then calculated first by subtracting the Q'' matrix from an appropriately dimensioned identity matrix I, and inverting the difference:

$$N^{**} = (I - Q''(I,J))^{-1} \qquad (2)$$

In any ecosystem energy dissipates in a predictably rapid fashion as it is successively passed between trophic levels. Only a small percentage of the energy entering the system at a lower trophic level ever reaches organisms at the highest trophic

levels. In theory the structure of the ecosystem reflects, and conforms to, those sources of energy which are most readily available. Assuming this to be true, then we would expect to find certain sources of energy more effectively retained through higher trophic levels than other sources on which the system does not critically depend. This first half of the flow analysis will reveal the degree to which the Kanawha River ecosystem reflects one or another primary energy source.

The second matrix, hereafter referred to as the "N^*" matrix, addresses the relative importance to a given trophic group of the sources of energy presently embodied in other trophic groups. These sources are not limited to the groups on which a particular group feeds. As with the N^{**} matrix above, the N^* matrix reflects indirect sources of energy throughout the ecosystem. Specifically, any element of the N^* matrix $N^*(I,J)$, gives the relative proportion of energy flow through trophic group (I) which once flowed through trophic group (J) as well. Thus the N^* matrix provides a quantitative expression of a particular trophic group's dependence on others for support.

The N^* matrix is obtained using a derivation similar to the one above. The first difference is in the original flow matrix used. In the second derivation, the F matrix defined above is expanded to a square matrix by adding an extra row and column. The "new" last column can be considered as a column vector D

whose elements D(J) are the total flow of energy entering the ecosystem through trophic group X(J). Elements of the "new" last row are null. Again we define an energy flow matrix in which any element F(I,J) of the matrix represents the annual flow of energy from trophic group X(J) to trophic group X(I), but now the vector T is defined with elements T(I) represents the total flow of energy passing through the recipient trophic group X(I) per year. The expanded flow matrix is then divided (in elementwise fashion) by the T vector to produce a Q' matrix:

$$Q'(I,J) = F(I,J) / T(I) \qquad (3)$$

The N^* matrix is calculated by subtracting the Q' matrix from an appropriately dimensioned identity matrix I, and inverting the difference:

$$N^* = (I - Q'(I,J))^{-1}$$

The information contained in the N^* matrix can be used both to interpret, and to some extent predict, the simulation results obtained by using the model. In theory, the most successful trophic groups are those in a position to take advantage of the most prevalent energy sources. The N^* matrix will elucidate the sources of energy supporting the dominant organisms of the Kanawha River.

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V. RESULTS

V.0. WATER QUALITY

Vertical profiles of temperature, conductivity, light extinction, and dissolved oxygen for each sampling date are presented in Appendix A, Tables A0.1.1-13. Each of the parameters reported below are summarized in Tables V.0.1-6.

Temperature and Specific Conductance

Water temperature (Table V.0.1) followed expected seasonal trends with a minimum in January (2.5°C) and a maximum in July-August (28.5°C). Differences between Top and Bottom never exceeded 1°C. LW generally had temperatures equal to or higher than those of UW. The difference ranged from -0.5 to +2.7°C. Ice formation was not observed beyond a meter from the shore at either site during the winter season.

Specific conductance (Table V.0.2) ranged from 60 to 360 µmhos/cm (UW from 60 to 275 µmhos/cm and LW from 110 to 360 µmhos/cm). Bottom samples were equal to or higher in conductance than Top samples, with differences as high as 15 µmhos/cm at UW and as high as 40 µmhos/cm at LW. LW had equal or greater values for every month except January, and the difference ranged from -40 to +130 µmhos/cm.

pH and Alkalinity

Table V.0.1. Summary of temperature (° C) measured on each sampling date. UW = Upper Winfield; LW = Lower Winfield.

Month	Depth	site	
		UW	LW
Oct	Top	15.8	18.5
	Bot	15.8	17.8
Nov	Top	9.5	9.8
	Bot	9.0	9.5
Dec	Top	4.3	6.8
	Bot	4.3	6.8
Jan	Top	2.5	2.7
	Bot	2.5	2.9
Feb	Top	4.0	3.5
	Bot	4.0	3.7
Mar	Top	6.5	7.0
	Bot	6.5	7.0
Apr	Top	8.0	9.0
	Bot	8.0	8.5
May	Top	14.0	15.0
	Bot	14.0	14.5
Jun	Top	22.0	22.5
	Bot	22.0	22.0
Jul	Top	28.0	28.5
	Bot	28.0	27.5
Aug	Top	28.0	28.7
	Bot	28.0	28.5
Sep	Top	25.5	25.5
	Bot	25.5	25.5

Table V.0.2. Summary of conductivity (μmhos/cm) on each sampling date. UW = Upper Winfield; LW = Lower Winfield.

Month	Depth	site	
		UW	LW
Oct	Top	185	235
	Bot		235
Nov	Top	140	240
	Bot	140	250
Dec	Top	165	210
	Bot	165	220
Jan	Top	270	230
	Bot	275	240
Feb	Top	170	185
	Bot		190
Mar	Top	130	125
	Bot		135
Apr	Top	155	185
	Bot	155	190
May	Top	60	110
	Bot	70	110
Jun	Top	100	120
	Bot	100	120
Jul	Top	225	230
	Bot	240	240
Aug	Top	230	360
	Bot	230	360
Sep	Top	200	260
	Bot	210	300

Values for pH (Table V.0.3) ranged from 6.0 to 7.3 during the six months of the spring and summer periods. Consistent trends between sites or with depth were not apparent.

Alkalinity measurements (Table V.0.4) ranged from 22 to 77 mg/l. Alkalinity was usually equal or higher at UW with the lxception of November when a difference of 9 mg/l existed between UW and LW. Differences between Top and Bottom were not consistent.

Dissolved Oxygen

Dissolved oxygen concentrations (Table V.0.5) ranged from 5.0 to 13.5 mgO_2/l and followed the seasonal trends expected from temperature changes in a system near oxygen saturation. Percent saturation (American Public Health Association et al. 1981) showed UW near or above 100% saturation for the entire year. LW had lower dissolved oxygen concentrations and percent saturation with the exception of the Top for July. This supersaturation was due to the high production rate of an algae bloom. The large difference between Top and Bottom for this date was probably also due to the high rate of respiration in the water column. The lowest percent saturation was at LW in August (66%).

Transparency and Light Penetration

Transparency (Table V.0.6) is given as the Secchi disk depth (the depth at which a 20 cm disk with alternating black and white quadrants disappears from view), as the depth of 1% incident

Table V.0.3. Summary of pH measured on each sampling date. UW = Upper Winfield; LW= Lower Winfield.

Month	Depth	site	
		UW	LW
Apr	Top	6.3	6.3
	Bot	6.3	6.4
May	Top	6.1	6.0
	Bot	6.2	6.0
Jun	Top	6.6	
	Bot	6.0	6.2
Jul	Top	7.2	6.8
	Bot	6.9	7.1
Aug	Top	7.3	6.9
	Bot	7.3	6.9
Sep	Top	6.5	6.3
	Bot	6.5	6.3

Table V.0.4. Summary of alkalinity (mg/l $CaCO_3$) measured on each sampling date. UW = Upper Winfield, LW = Lower Winfield.

Month	Depth	site	
		UW	LW
Oct	Top	29	25
	Bot	29	25
Nov	Top	37	48
	Bot	37	46
Dec	Top	34	27
	Bot		
Jan	Top	47	45
	Bot	46	35
Feb	Top	30	29
	Bot		29
Mar	Top	29	26
	Bot		24
Apr	Top	30	28
	Bot	31	28
May	Top	30	28
	Bot	22	25
Jun	Top	37	
	Bot	38	37
Jul	Top	76	68
	Bot	77	
Aug	Top	45	40
	Bot	45	40
Sep	Top	37	40
	Bot	42	42

Table V.0.5. Summary of dissolved oxygen (mg/l) measured on each sampling date. UW = Upper Winfield; LW = Lower Winfield.

Month	Depth	site	
		UW	LW
Oct	Top	8.9 (90%)	7.3 (77%)
	Bot	8.9 "	7.3 "
Nov	Top	11.5 (100%)	10.1 (89%)
	Bot	11.5 "	10.1 "
Dec	Top	13.1 (100%)	11.3 (93%)
	Bot		11.4 "
Jan	Top	13.3 (98%)	12.0 (90%)
	Bot	13.3 "	12.3
Feb	Top	13.5 (103%)	13.1 (102%)
	Bot	13.5 "	13.1 "
Mar	Top	13.0 (106%)	12.5 (103%)
	Bot		12.4 "
Apr	Top	12.8 (108%)	12.2 (105%)
	Bot	12.8 "	12.2 "
May	Top	10.3 (100%)	9.8 (97%)
	Bot	10.2 "	9.8 "
June	Top	8.6 (99%)	7.9 (90%)
	Bot	8.6 "	7.8 "
Jul	Top	7.7 (99%)	9.2 (115%)
	Bot	7.7 "	6.9 (87%)
Aug	Top	7.7 (99%)	5.2 (66%)
	Bot	7.7 "	5.0 "
Sep	Top	8.5 (104%)	6.1 (75%)
	Bot	8.5 "	6.0

light, and as a vertical extinction coefficient (both derived from vertical light readings). Extinction as measured by either method had similar monthly changes. During June the extinction was high from muddy water.

V.1. TERRESTRIAL INPUTS

Leaf Fall

The measurements of leaf fall are summarized in Fig. V.1.1. The total annual leaf fall amounted to 371.2 gDW/m^2. There was very little leaf fall except for the normal period of abscission. Almost all leaf fall (96%) occurred from August - December, with maximum values of 118.8 and 142.7 gDW/m^2 in October and November, respectively. In the spring and early summer leaf fall was negligible, ranging from 0.6 - 6.8 gDW/m^2. All leaf fall data are recorded in Appendix A, Tables A1.1.1 - 11.

Leaf Blow-In

Annual leaf blow-in was considerably less than leaf fall, with a total of 68.4 gDW/m. Maximum blow-in (31.1 gDW/m) occurred in October. The measurements of leaf blow-in are summarized in Fig. V.1.2. After the peak of blow-in during October, the values dropped sharply in November and December (25.3 and 6.4 gDW/m, respectively) and became negligible during winter and spring (0.7 - 1.1 gDW/m). Apparently the fallen leaves become packed on the ground shortly after abscission and

Table V.0.6. Summary of light penetration measured on each sampling date. Z_{SD} = secchi disc; $Z_{1\%}$ = photometer; n" = light extinction coefficient; UW = Upper Winfield; LW = Lower Winfield.

Month	UW			LW		
	Z_{SD}	$Z_{1\%}$	n"	Z_{SD}	$Z_{1\%}$	n'
Oct	9.2	3.5	1.18	0.9	3.4	1.22
Nov	0.6	2.1	1.93	1.1	3.5	1.30
Dec	0.4	1.0	4.17	1.0	3.9	1.15
Jan	1.8	6.5	0.85	1.5	7.0	0.62
Feb	0.5	1.4	3.30	0.5	2.6	1.67
Mar	0.4	--	--	0.2	1.0	4.95
Apr	0.7	2.2	2.20	0.8	2.8	1.42
May	0.4	2.1	2.21	0.5	2.1	2.16
Jun	0.3	1.8	2.85	0.2	0.9	5.07
Jul	1.0	--	--	1.0	--	--
Aug	1.0	3.3	1.34	0.9	2.8	1.52
Sep	0.8	2.0	2.04	1.0	3.1	1.39

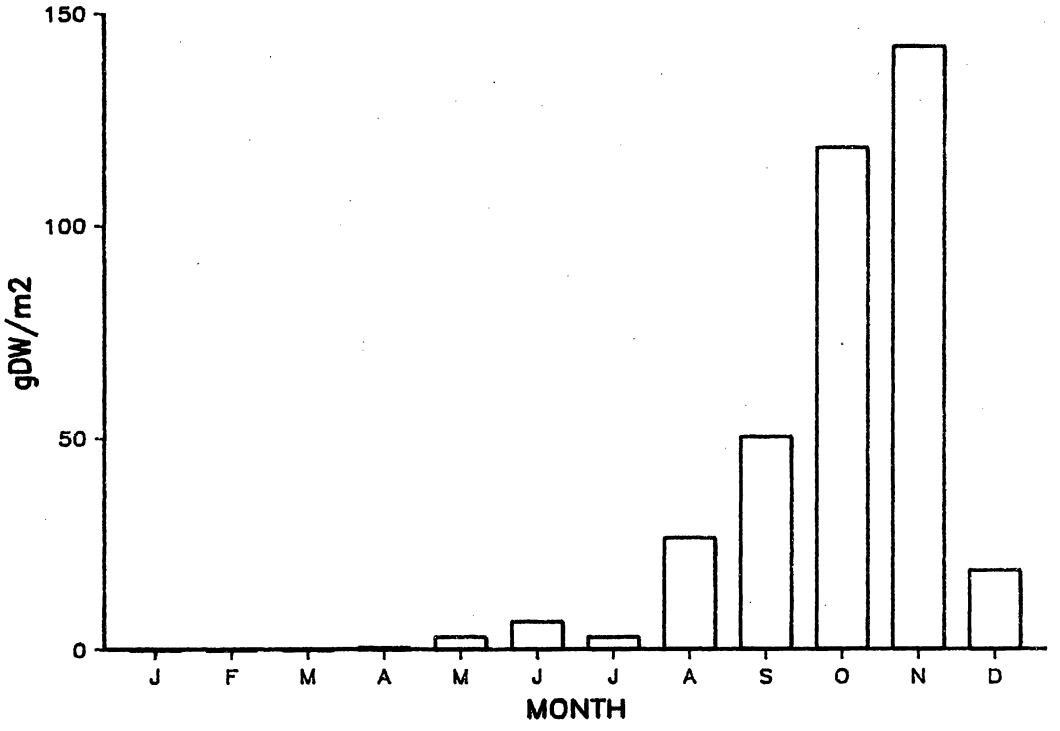

Fig. V.1.1. Average leaf fall (gDW/m^2) in riparian areas of the Winfield Pool.

are no longer subject to movement by the wind. In the summer, the dense brush along the river banks makes leaf blow-in negligible. All leaf blow-in data are recorded in Appendix A, Tables A1.1.1 - 11.

Insect Fall

The biomass of insects that fell from the tree canopy totalled 28.33 gDW/m^2 for the year. The estimates of insect fall are summarized in Fig. V.1.3. During the spring and summer, insect fall was consistent. Insect fall peaked in October, possibly because insects descended with the leaves as they were shed by trees. The sharp drop in insect fall during November probably resulted from the cessation of insect activity with the onset of consistent cold weather. Insect fall data are recorded in Appendix A, Tables A1.1.1 - 11).

V.2. BENTHIC DETRITUS

Information on the composition of the bottom sediment (% organic content) is summarized in Figs. V.2.1 and V.2.2. The sediments at LW were generally higher in organic content (average 12%, range 4 - 20%) then the sediments at UW (average 8%, range 3 - 20%). All data on bottom composition are recorded in Appendix A, Table A2.1.1.

V.3. SESTON

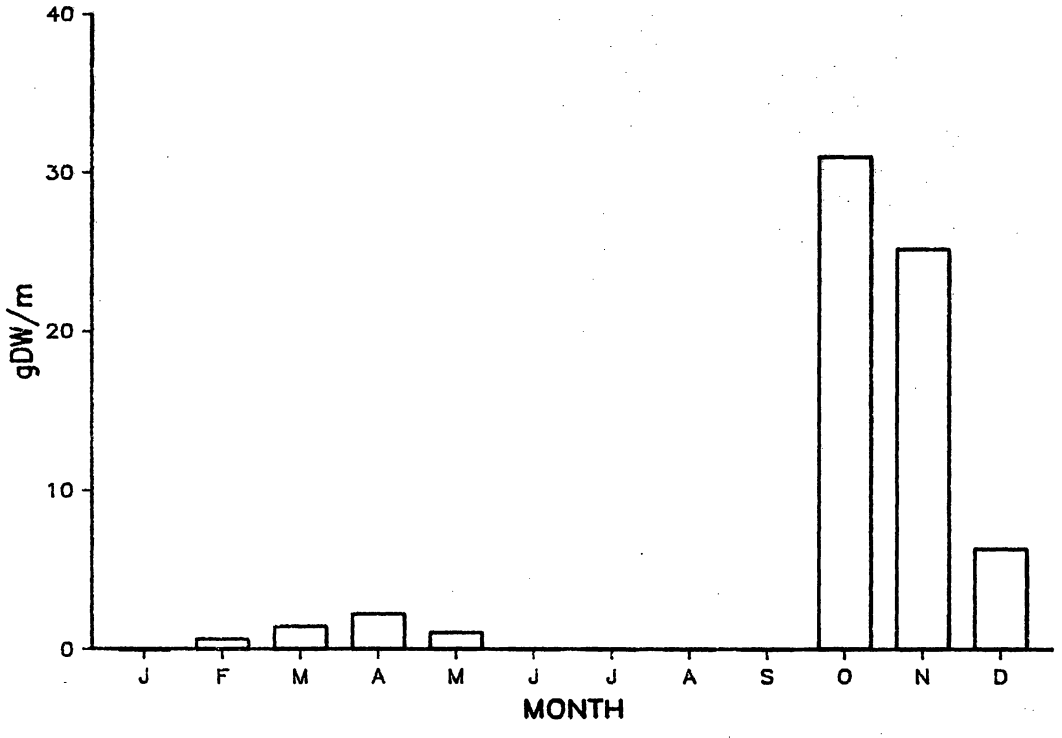

Fig. V.1.2. Average leaf blow-in (gDW/m) from riparian areas of the Winfield Pool.

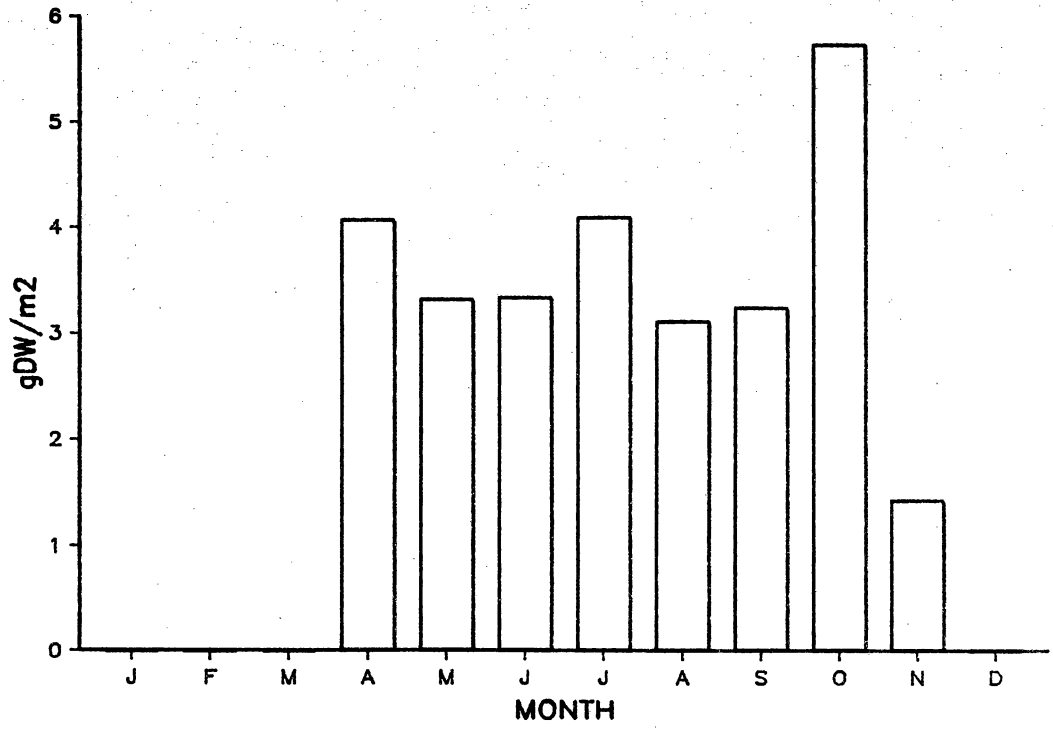

Fig. V.1.3. Average insect fall (gDW/m^2) in riparian areas of the Winfield Pool.

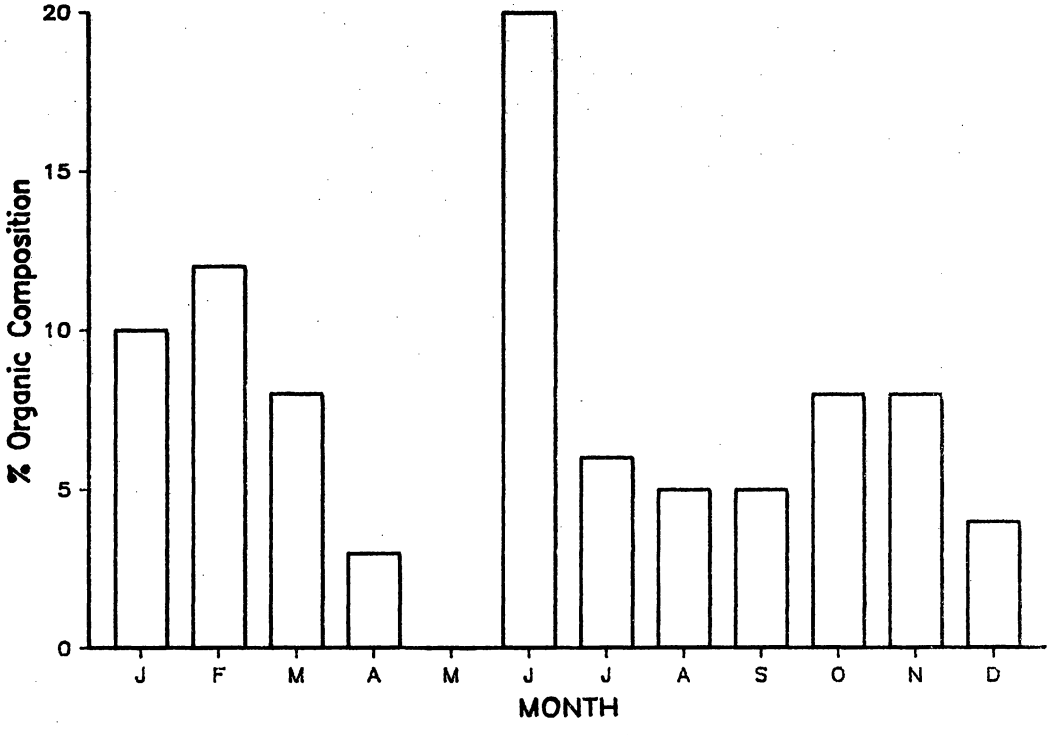

Fig. V.2.1. Composition of bottom sediments (% organic) at UW.

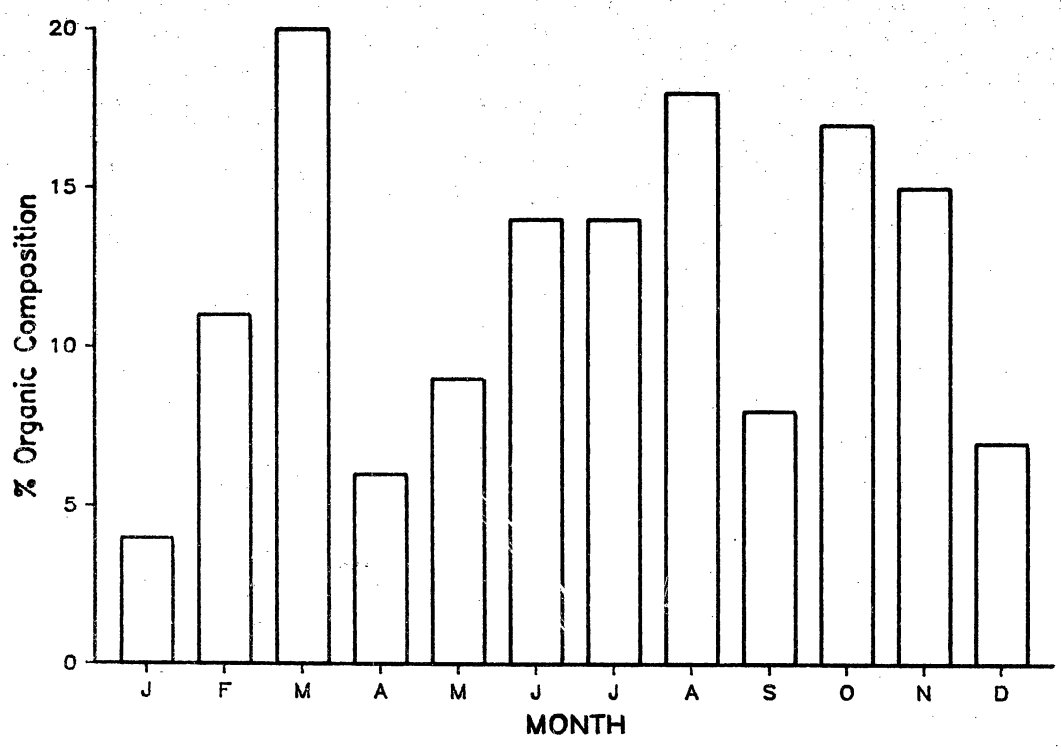

Fig. V.2.2. Composition of bottom sediments (% organic) at LW.

The concentrations of organic seston are recorded in Appendix A, Tables A3.1.1 - 24 and summarized in Table V.3.1. The annual averages for total organic seston (gAFDW/l) were: 0.00322 UW-Top, 0.00221 UW-Bottom, 0.00226 LW-Top, and 0.00280 LW-Bottom. The monthly concentrations of organic seston (gAFDW/l) ranged as follows: 0.00099 - .01143 UW-Top, 0.00098 - 0.00308 UW-Bottom, 0.00093 - 0.00755 LW-Top, 0.00094 - 0.00895 LW-Bottom. The lowest concentrations at both sites and depths occurred in January (0.00093 - 0.00099). The highest concentration at UW occurred in December at the Top depth (0.01143). On that date the current was too swift to sample the Bottom depth; the highest concentration at UW-Bottom was recorded in May (0.00314). The highest concentration at LW occurred in March at the Top depth, (0.00755), but samples could not be collected at that time from the Bottom depth. The concentration of organic seston was highest at LW-Bottom (0.00895) in May.

The proportion of the total seston that was composed of organic matter (% by weight) is recorded in Appendix A, Tables A3.2.1 - 24 and summarized in Table V.3.2. The annual averages indicated that the percentage of organic seston was slightly higher at LW than UW and slightly higher at the Top depth than the Bottom depth (29.4 UW-Top, 27.2 UW-Bottom, 36.0 LW-Top, 32.5 LW-Bottom). The ranges of the monthly values for percentage of organic seston were similar at UW-Top (22.0 - 43.0), UW-Bottom (19.7 - 37.8), and LW-Bottom (20.7 - 42.5). The minimum

Table V.3.1. Summary of organic seston concentrations (gAFDW/l). UW = Upper Winfield; LW = Lower Winfield; * = same value in 2 or more months.

Site	Depth	Size Class	Mean	Max.	(Mo.)	Min.	(Mo.)
UW	Top	L	.00032	.00345	(Jul)	.00001	(*)
		ML	.00013	.00119	(Dec)	.00001	(*)
		S	.00031	.00298	(Dec)	.00001	(*)
		F	.00027	.00179	(Dec)	.00002	(Nov)
		VF	.00006	.00022	(Dec)	.00001	(*)
		UF	.00214	.00503	(Dec)	.00092	(Jan)
		Total	.00322	.00143	(Dec)	.00099	(Jan)
	Bot[a]	L	.00002	.00006	(Mar)	.00001	(*)
		ML	.00005	.00016	(*)	.00001	(*)
		S	.00007	.00026	(May)	.00001	(Aug)
		F	.00012	.00056	(May)	.00004	(Jan)
		VF	.00003	.00013	(May)	.00001	(*)
		UF	.00191	.00292	(Sep)	.00089	(Jan)
		Total	.00221	.00314	(May)	.00098	(Jan)
LW	Top	L	.00001	.00003	(Mar)	.00001	(*)
		ML	.00004	.00033	(Mar)	.00001	(*)
		S	.00010	.00082	(Mar)	.00001	(*)
		F	.00014	.00118	(Mar)	.00001	(*)
		VF	.00004	.00021	(Mar)	.00001	(*)
		UF	.00193	.00498	(Mar)	.00088	(Jan)
		Total	.00226	.00755	(Mar)	.00093	(Jan)
	Bot[b]	L	.00002	.00015	(May)	.00001	(*)
		ML	.00006	.00056	(May)	.00001	(*)
		S	.00009	.00063	(May)	.00001	(*)
		F	.00016	.00139	(May)	.00001	(*)
		VF	.00036	.00250	(Aug)	.00001	(*)
		UF	.00210	.00491	(May)	.00089	(*)
		Total	.00280	.00895	(May)	.00094	(Jan)

[a] December Bottom samples could not be collected because of high discharge.

[b] March Bottom samples could not be collected because of high discharge.

percentage recorded at LW-Top (17.9) was similar to the other sites and depths, but an unusually high percentage of organic seston (72.0) was recorded at LW-Top in July.

The concentrations of organic seston and the percentages of organic matter in the seston according to size classes are also recorded in Appendix A, Tables A3.1.1 - 24 and A3.2.1 - 24, and summarized in Tables V.3.1 and V.3.2., respectively. In general, there was a trend for higher concentrations of organic seston in the smaller size classes. The majority of the organic seston occurred in the UF size class. The mean percent of the total weight of organic seston that occurred in the UF size class at each site and depth was as follows: UW - Top 66.5%, UW - Bottom 86.4%, LW - Top 85.4%, LW - Bottom 75.0%. There was also a trend for higher percentages of organic matter in the larger size classes. The UF size class was composed of about 17% organic matter, while the VF-L size classes averaged from about 20 - 50 % organic matter.

The concentrations of total seston and inorganic seston are recorded in Appendix A, Tables A3.3.1 - 24 and Tables A3.4.1 - 24, respectively.

V.4. BACTERIA

Analysis of seston by weight showed the major portion occurred in the UF size class (Section V.3.). This was true regardless of site, depth, or time of year. A total of 237

Table V.3.2. Summary of percent organic content of seston (% by weight). UW = Upper Winfield; LW = Lower Winfield.

Site	Size Depth	Class	Mean	Max. (Mo.)	Min. (Mo.)
UW	Top	L	48.9	98.9 (Oct)	25.1 (Jun)
		ML	39.9	67.3 (Oct)	22.4 (Dec)
		S	28.7	46.2 (Jul)	19.9 (Dec)
		F	20.8	37.7 (Feb)	13.0 (Dec)
		VF	21.0	31.2 (Jul)	16.0 (Feb)
		UF	17.3	27.6 (Jul)	11.0 (Feb)
		Total	29.4	43.0 (Oct)	22.0 (Dec)
	Bot[a]	L	41.6	77.1 (Oct)	20.4 (Sep)
		ML	38.4	58.4 (Oct)	13.2 (Mar)
		S	28.0	53.0 (Jan)	14.6 (Mar)
		F	18.3	22.2 (Jul)	15.1 (Mar)
		VF	18.8	24.9 (Jul)	12.4 (Nov)
		UF	17.8	31.2 (Aug)	8.9 (Nov)
		Total	27.2	37.8 (Oct)	19.7 (Mar)
LW	Top	L	47.2	80.5 (Jul)	13.5 (Aug)
		ML	50.2	87.7 (Jul)	21.1 (Mar)
		S	43.5	78.8 (Jul)	17.1 (Mar)
		F	30.9	75.3 (Jul)	11.3 (Mar)
		VF	27.1	73.9 (Jul)	16.0 (Feb)
		UF	17.2	35.7 (Jul)	9.1 (Jun)
		Total	36.0	72.0 (Jul)	17.9 (Mar)
	Bot[b]	L	45.6	96.6 (Oct)	14.0 (Sep)
		ML	46.1	58.7 (May)	22.0 (Jun)
		S	39.4	51.1 (May)	26.7 (Jun)
		F	23.7	36.0 (Sep)	17.0 (Feb)
		VF	22.4	52.9 (Aug)	13.4 (May)
		UF	17.6	33.2 (Oct)	8.4 (Aug)
		Total	32.5	42.5 (Oct)	20.7 (Jun)

[a] December Bottom samples could not be collected because of high discharge.

[b] March Bottom samples could not be collected because of high discharge.

slides were examined for total numbers for bacterioplankton, phytoplankton, and zooplankton, and the surface area of these plankton and tripton were measured as well.

Analysis of bacteria associated with the seston size fractions showed the greatest number of bacteria were also associated with the UF size class (Appendix A, Tables A4.2.1-4; Figs. V.4.1-4). This was the case regardless of site, depth, or time of year. The size class with the next greatest number of bacteria was the F. The VF and S size classes had about equal densities, but were 2-3 orders of magnitude below the density in the UF class. The ML size class had less than the VF and S classes, but more than the L class. The L class had the least bacteria of any of the size classes.

In Appendix A, Tables A4.1.1-24 show the composition of the seston with respect to the size classes. Quantitative estimates of seston composition were based upon area in square microns of tripton, bacteria, and phytoplankton and zooplankton. There was, in general, a greater percentage of bacteria associated with the larger particle sizes than the smaller sizes, even though the greatest amount of bacterial biomass was in the UF class. The largest percent composition of the seston was tripton, followed by the large plankton (phytoplankton and zooplankton), and the bacteria constituted the smallest percentage. The annual means for each station and depth are summarized in Table V.4.1.

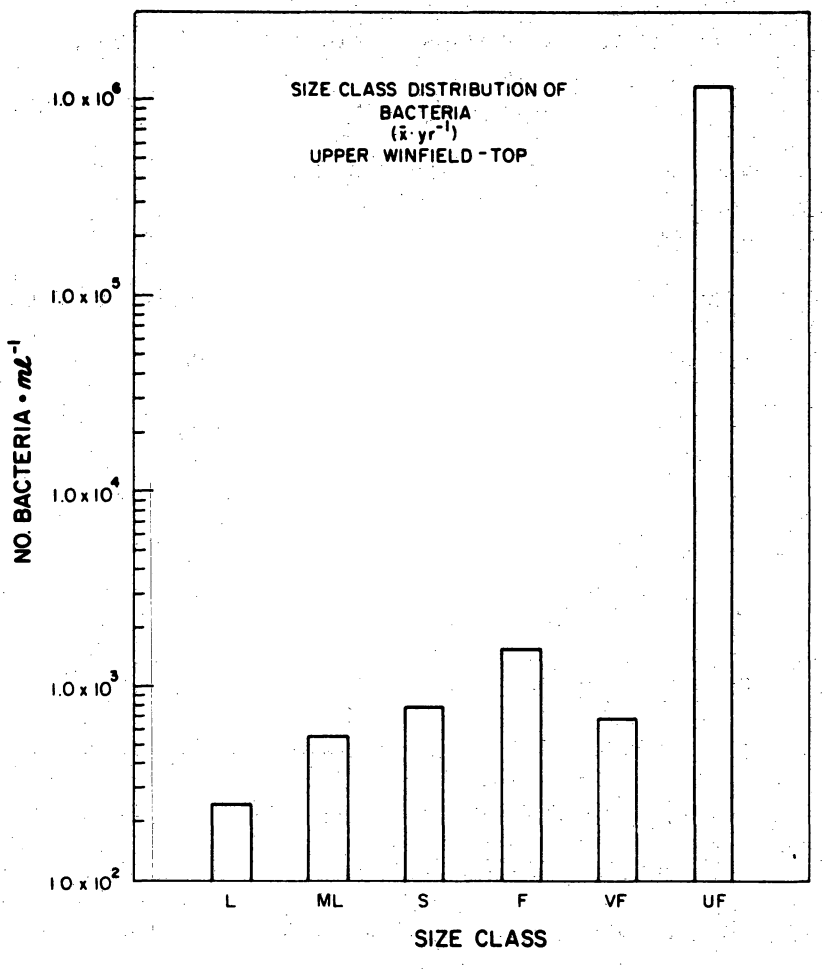

Fig. V.4.1. Mean bacterial densities for each seston size class from Top samples at UW.

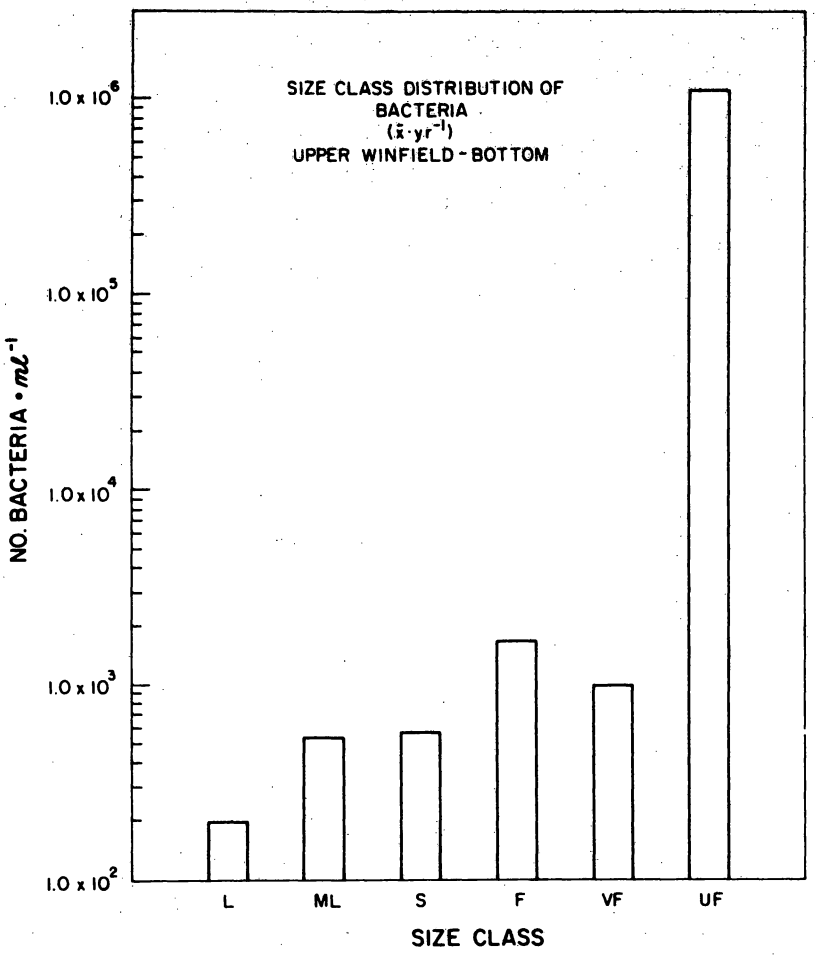

Fig. V.4.2 Mean bacterial densities for each seston size class from Bottom samples at UW.

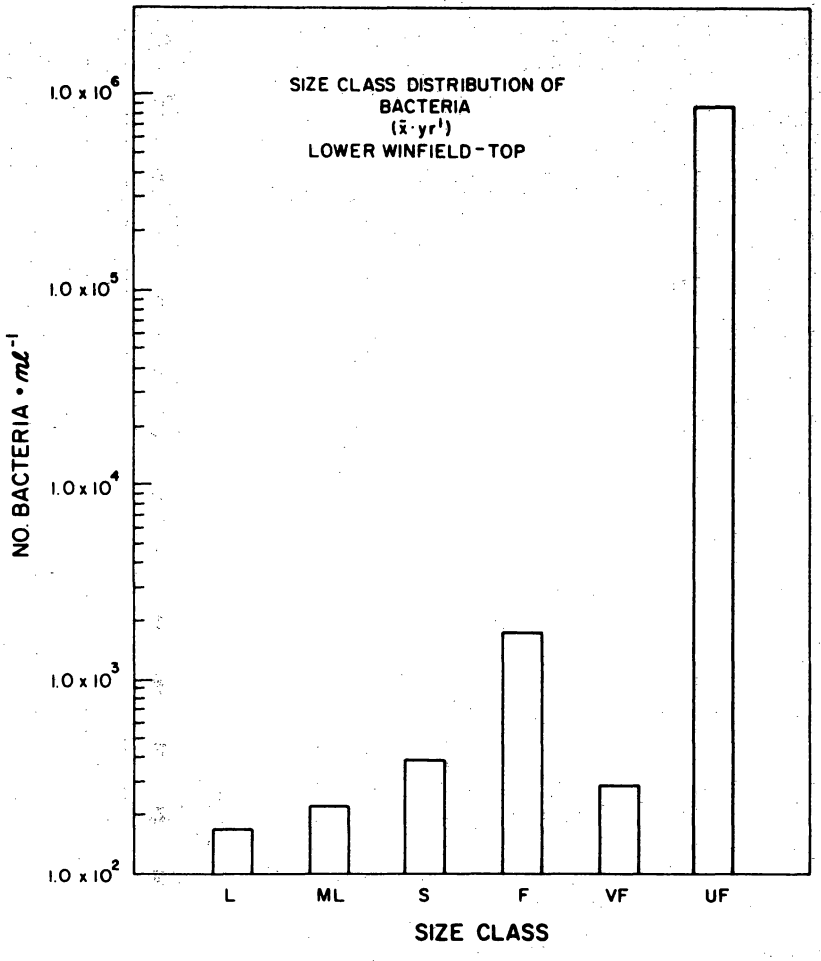

Fig. V.4.3. Mean bacterial densities for each seston size class from Top samples at LW.

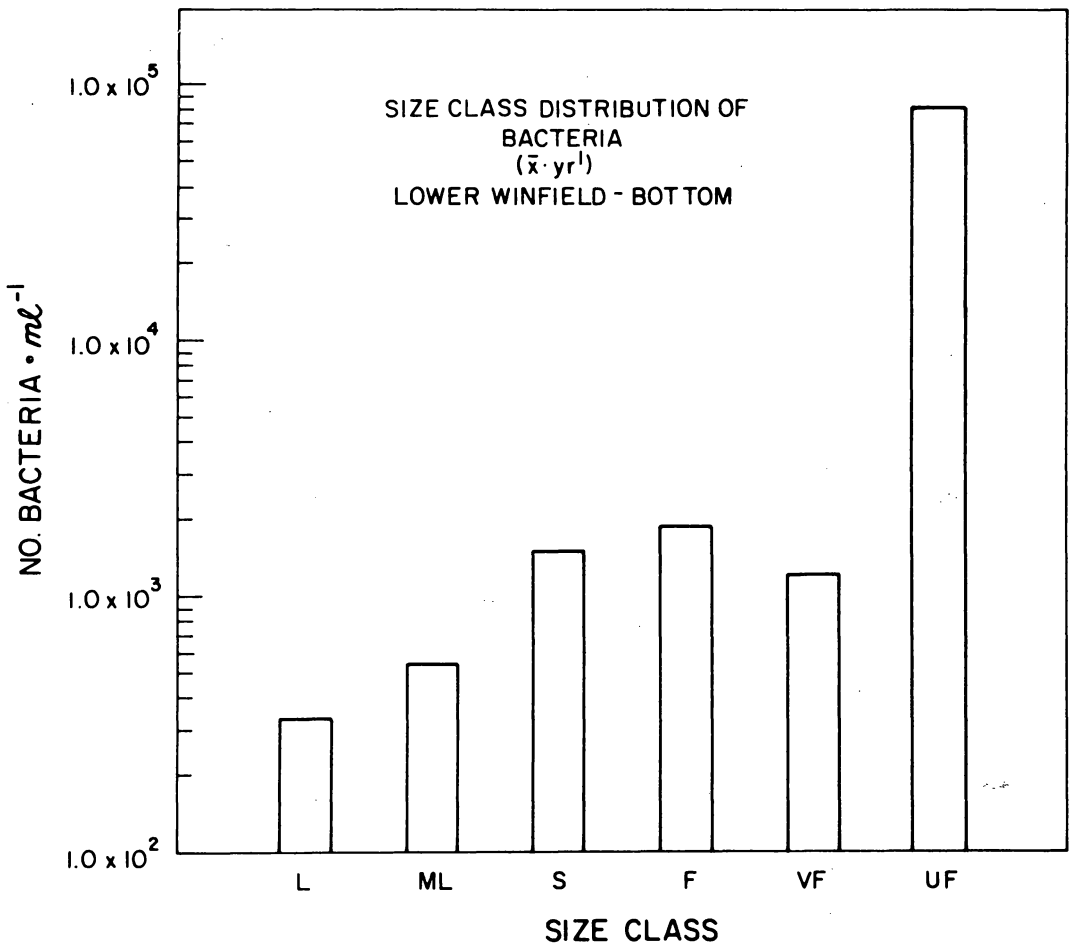

Fig. V.4.4. Mean bacterial densities for each seston size class from Bottom samples at LW.

Table V.4.1. Summary of seston composition (%) according to size class.

Site	Depth	Size Class	Bacterio-plankton	Other Plankton	Tripton
UW	Top	L	10.4	12.7	76.9
		ML	5.7	10.8	83.5
		S	4.0	10.9	85.1
		F	2.2	9.9	87.9
		VF	4.1	12.5	83.4
		UF	5.2	0.1	94.7
UW	Bot	L	20.8	3.5	75.7
		ML	8.0	4.9	87.0
		S	3.8	8.0	88.1
		F	2.5	5.2	92.2
		VF	4.6	5.9	89.5
		UF	4.2	5.5	90.3
LW	Top	L	13.2	20.5	66.3
		ML	4.7	6.1	89.2
		S	8.5	12.2	79.3
		F	4.0	8.7	87.3
		VF	4.7	8.2	87.1
		UF	3.5	0.0	96.4
LW	Bot	L	10.5	16.3	73.2
		ML	8.9	10.8	80.3
		S	3.2	15.0	81.8
		F	1.9	9.8	88.3
		VF	3.0	7.8	89.2
		UF	3.0	2.9	94.1

Figs. V.4.5 - V.4.8 are three dimensional graphs which show the distribution of bacterial density for each monthly sampling period for all size fractions larger than the UF size fraction. The UW stations showed a greater representation of bacteria in all the larger size fractions for nearly all months. At the LW stations, there were some months when there were few or no bacteria that were counted in the larger size classes. Bacterial density peaks in these larger size classes appeared to be independent of station and sampling period.

Because the greatest number of bacterial densities occurred in the UF size class, we will confine the remainder of this section to the densities in the UF class.

Figs. V.4.9 and V.4.10 show the trends in density changes through time in the Winfield Pool. While there was some variation in density, most were near 10^6 bacteria/ml. Using the Wilcoxon Rank Sum Test (Wilcoxon and Wilcox 1964, Woolf 1968), there were no significant differences in density between sites or depths ($P = .05$). The data suggested a fall minimum with a return to yearly maximum levels by mid-winter. Variation in density was generally less at LW than UW. The greater variability in density at UW in comparison to LW probably reflected greater water turbulence at UW.

Figs. V.4.11 and V.4.12 show the relationship between bacterial densities and phytoplankton biomass as measured by chl a concentration. One should note that the time scale is between

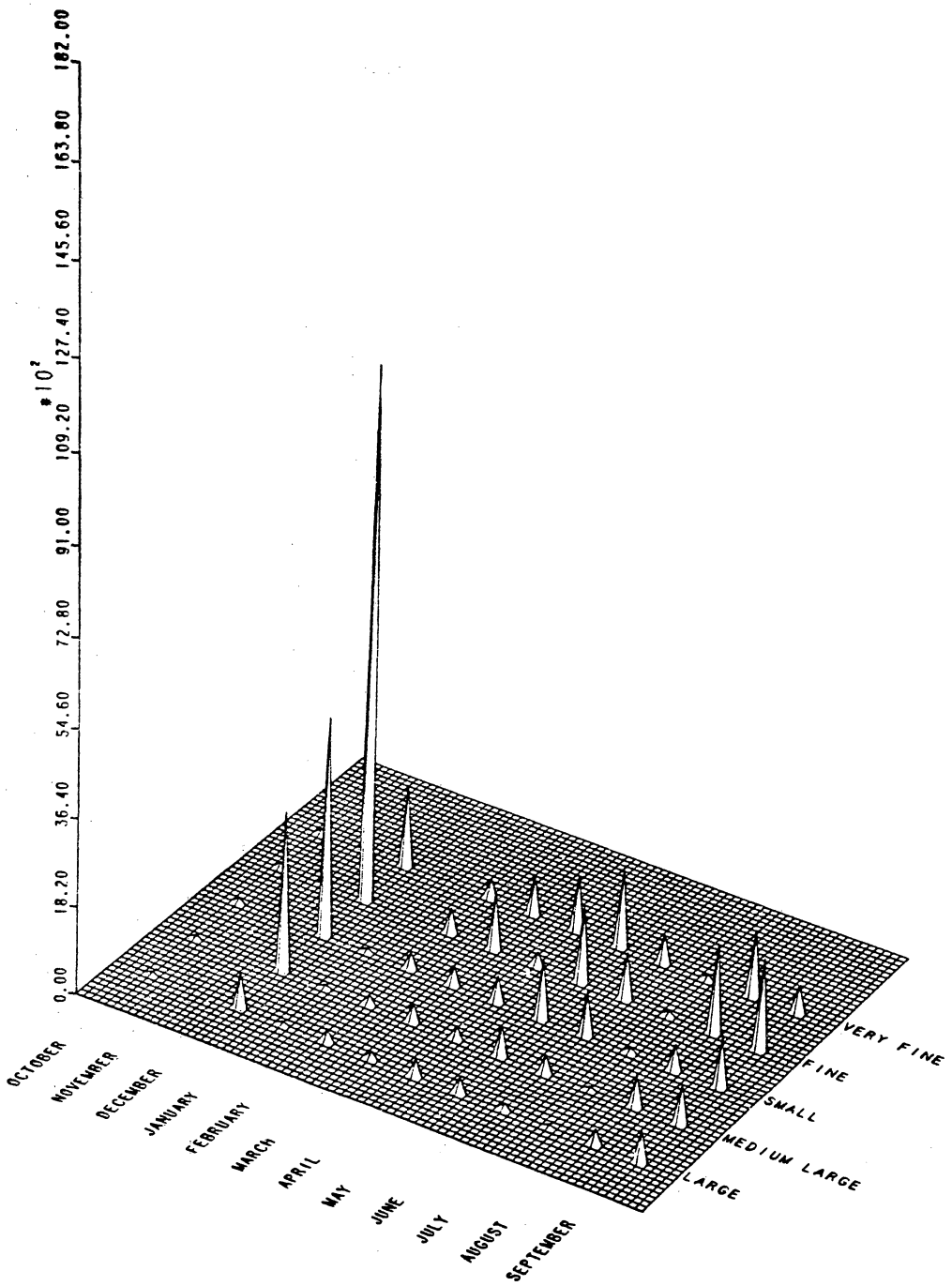

Fig. V.4.5. Bacterial density distribution in seston size classes > 25 µm from Top samples at UW.

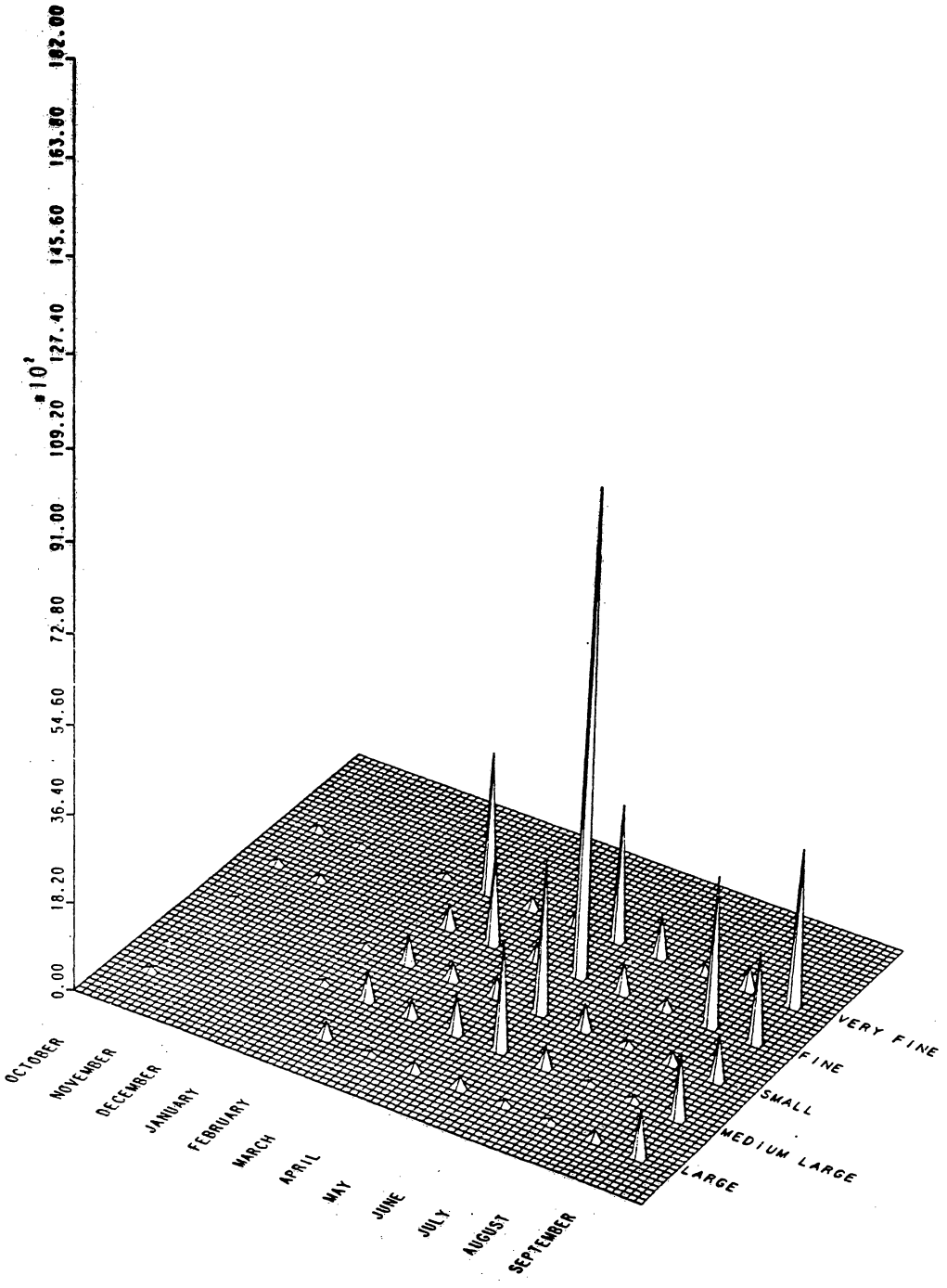

Fig. V.4.6. Bacterial density distribution in seston size classes > 25 µm from Bottom samples at UW.

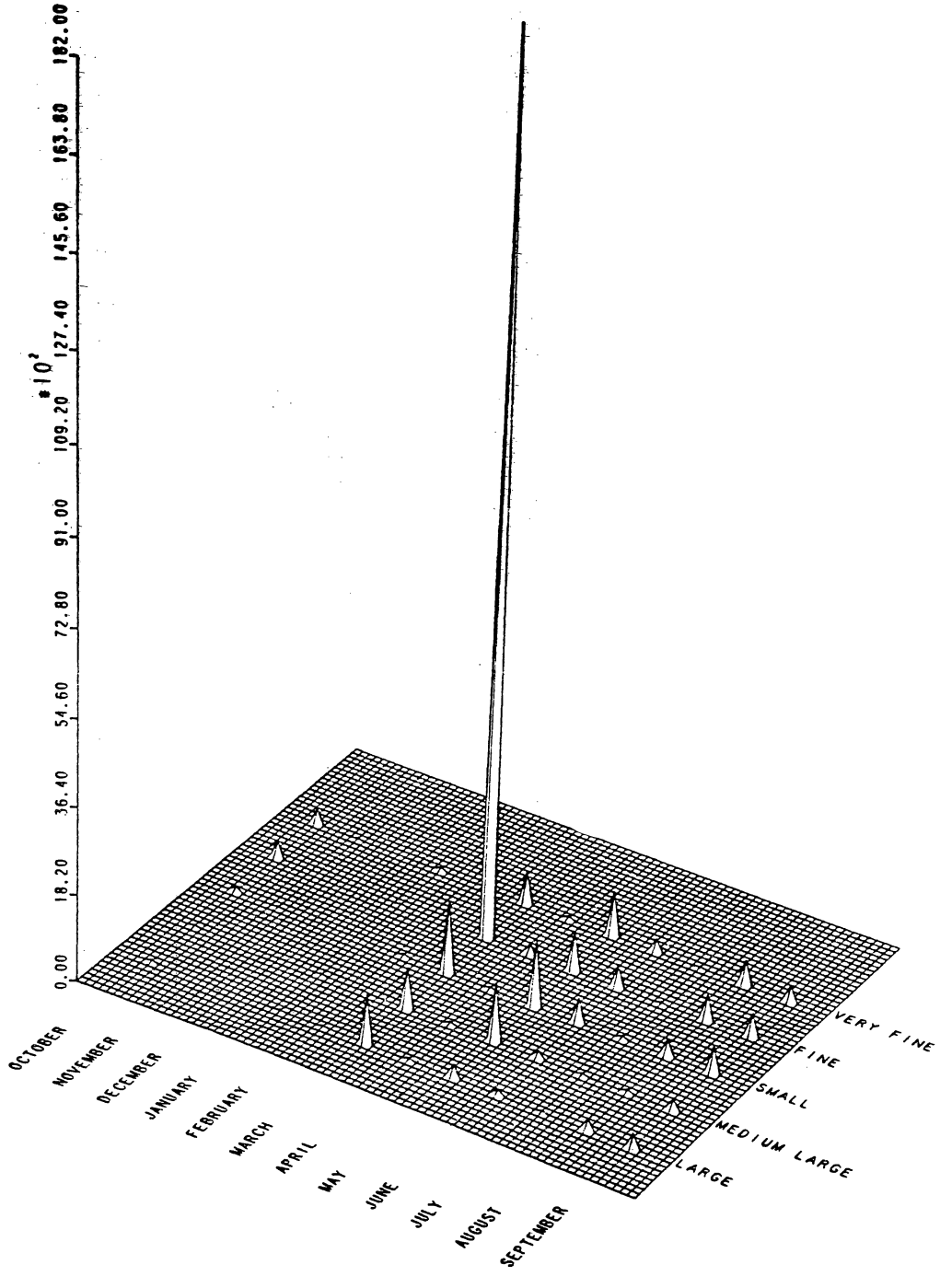

Fig. V.4.7. Bacterial density distribution in seston size classes > 25 µm from Top samples at LW.

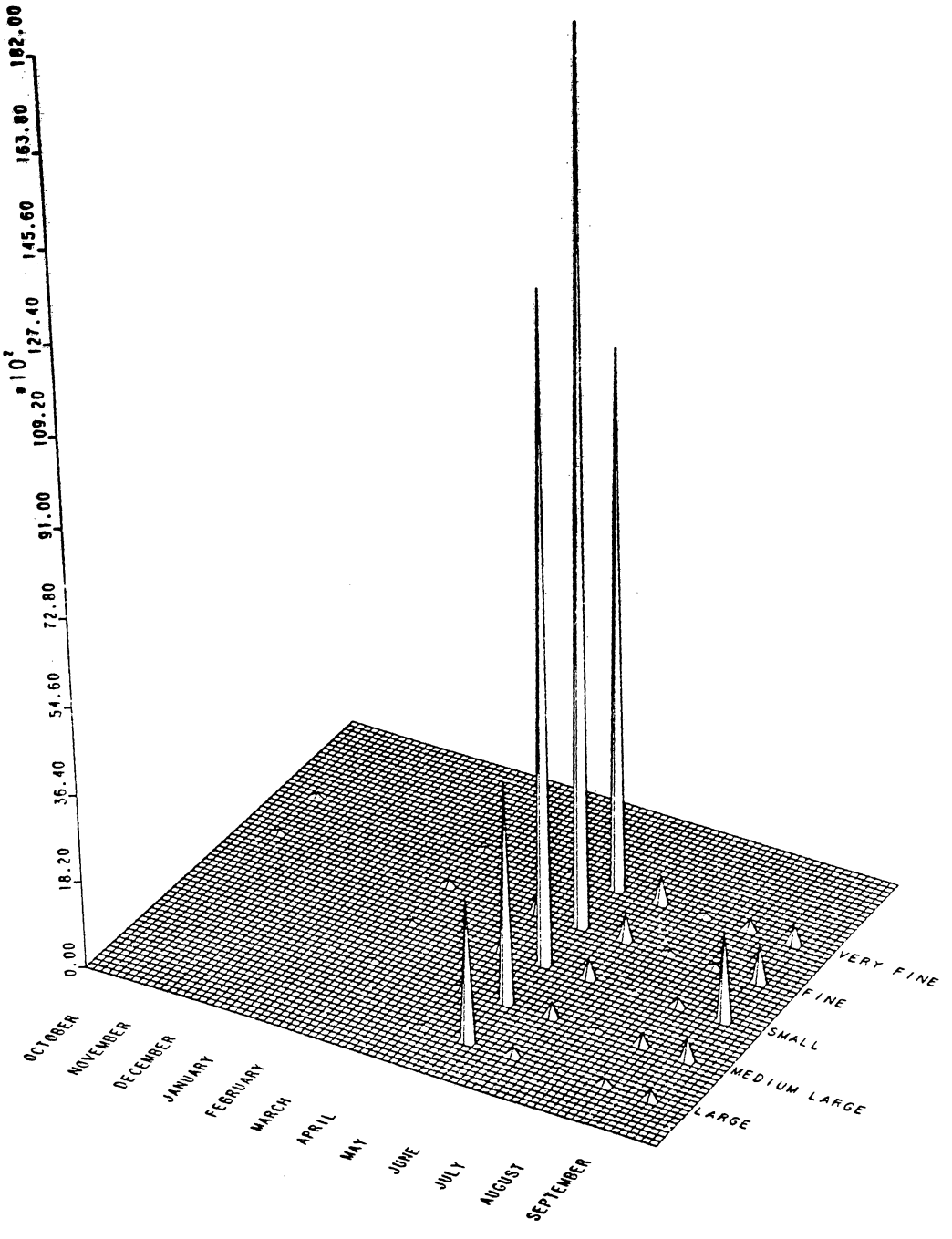

Fig. V.4.8. Bacterial density distribution in seston size classes > 25 µm from Bottom samples at LW.

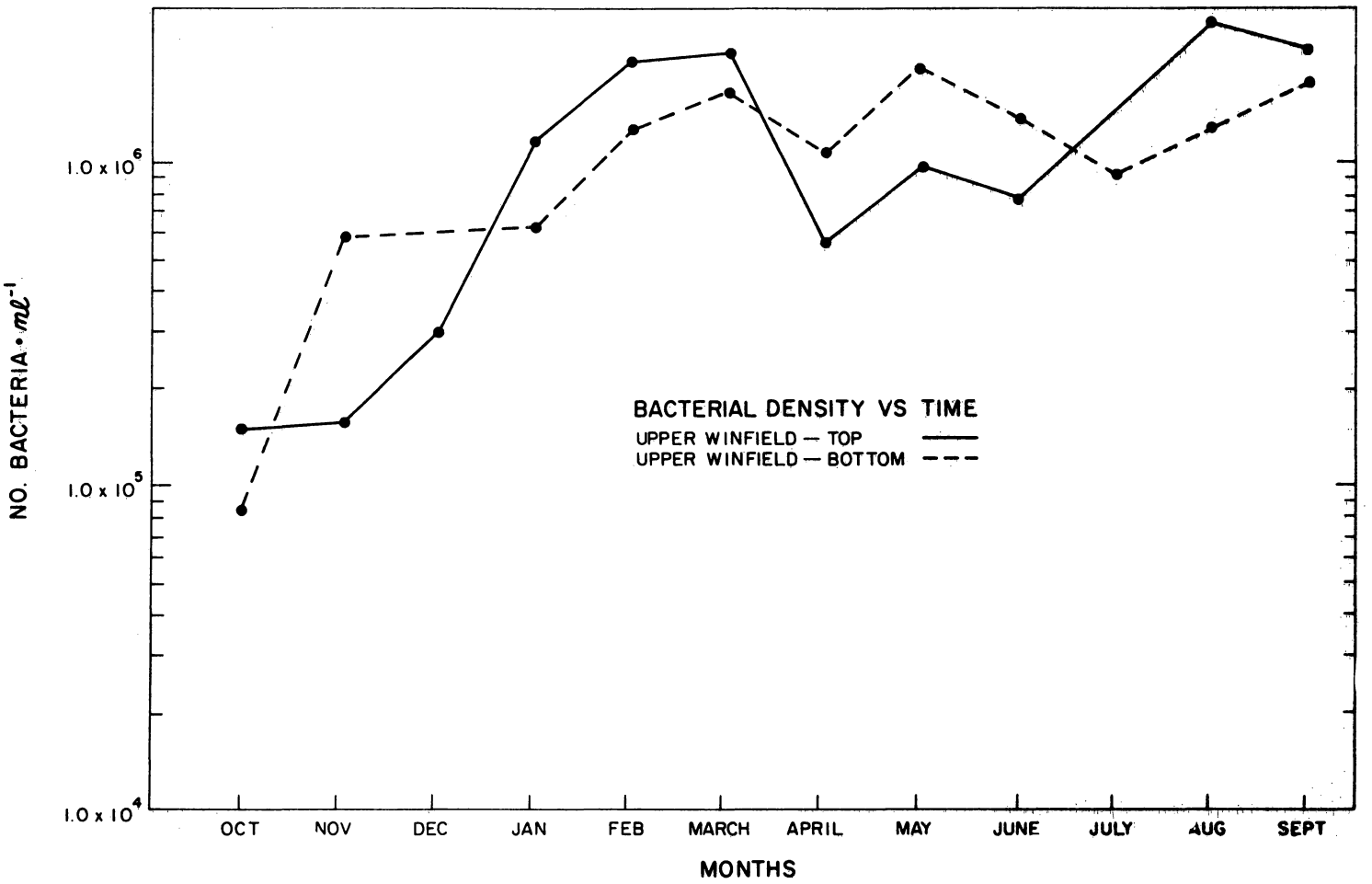

Fig. V.4.9. Total bacterial density in waters at UW.

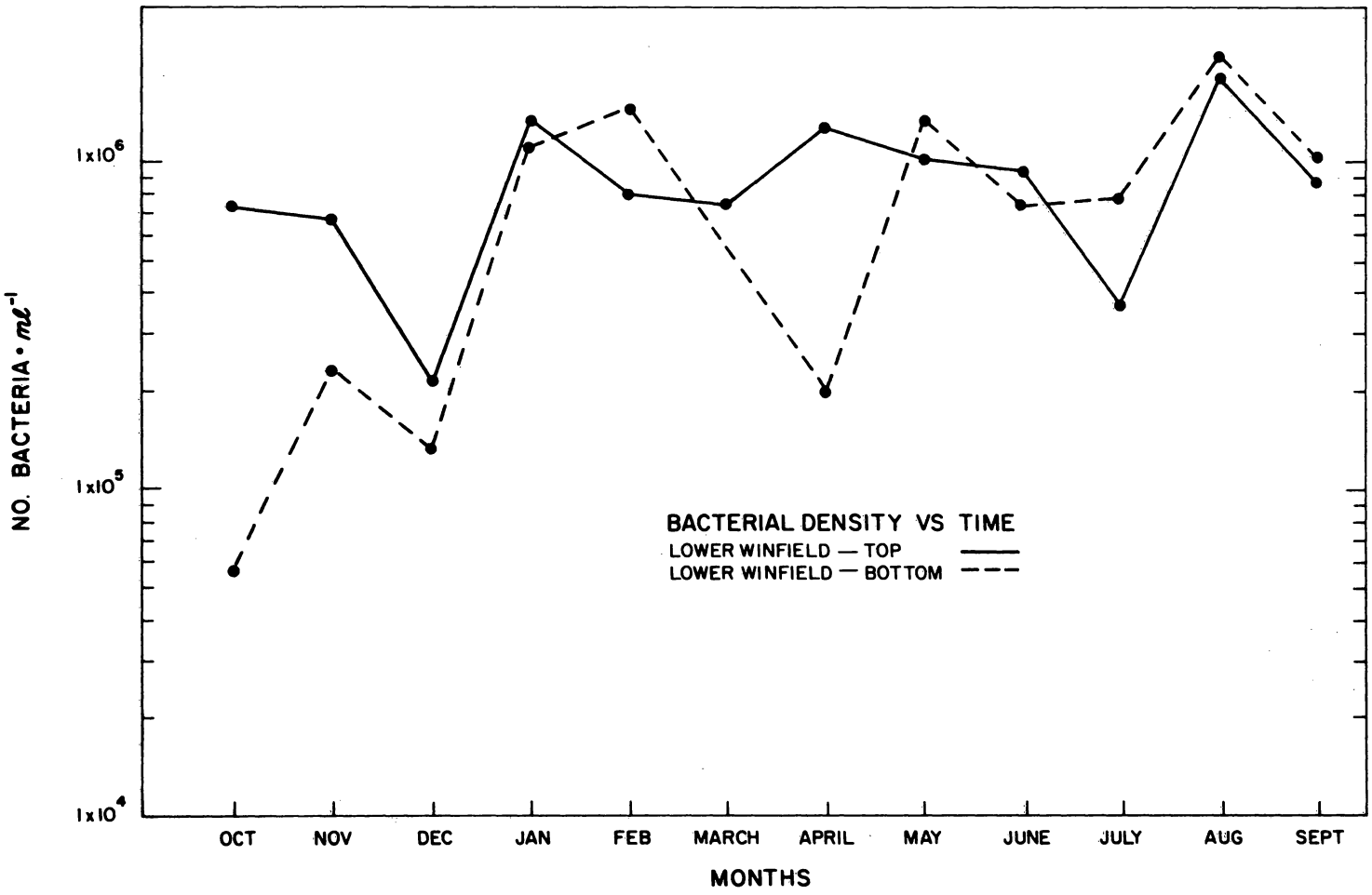

Fig. V.4.10. Total bacterial density in waters at LW.

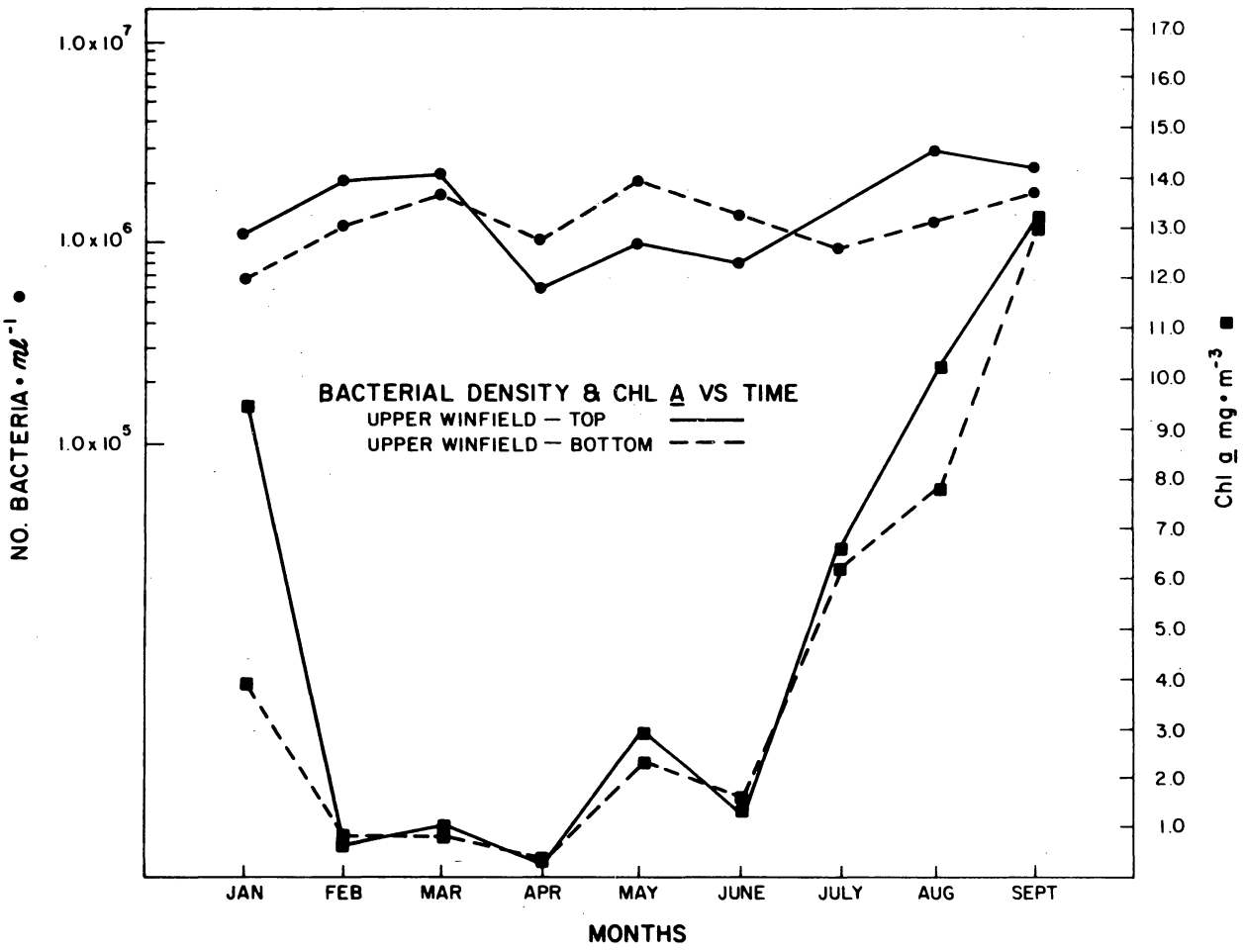

Fig. V.4.11. Relationship between bacterial density and phytoplankton biomass (Chl a) at UW from January through September 1983.

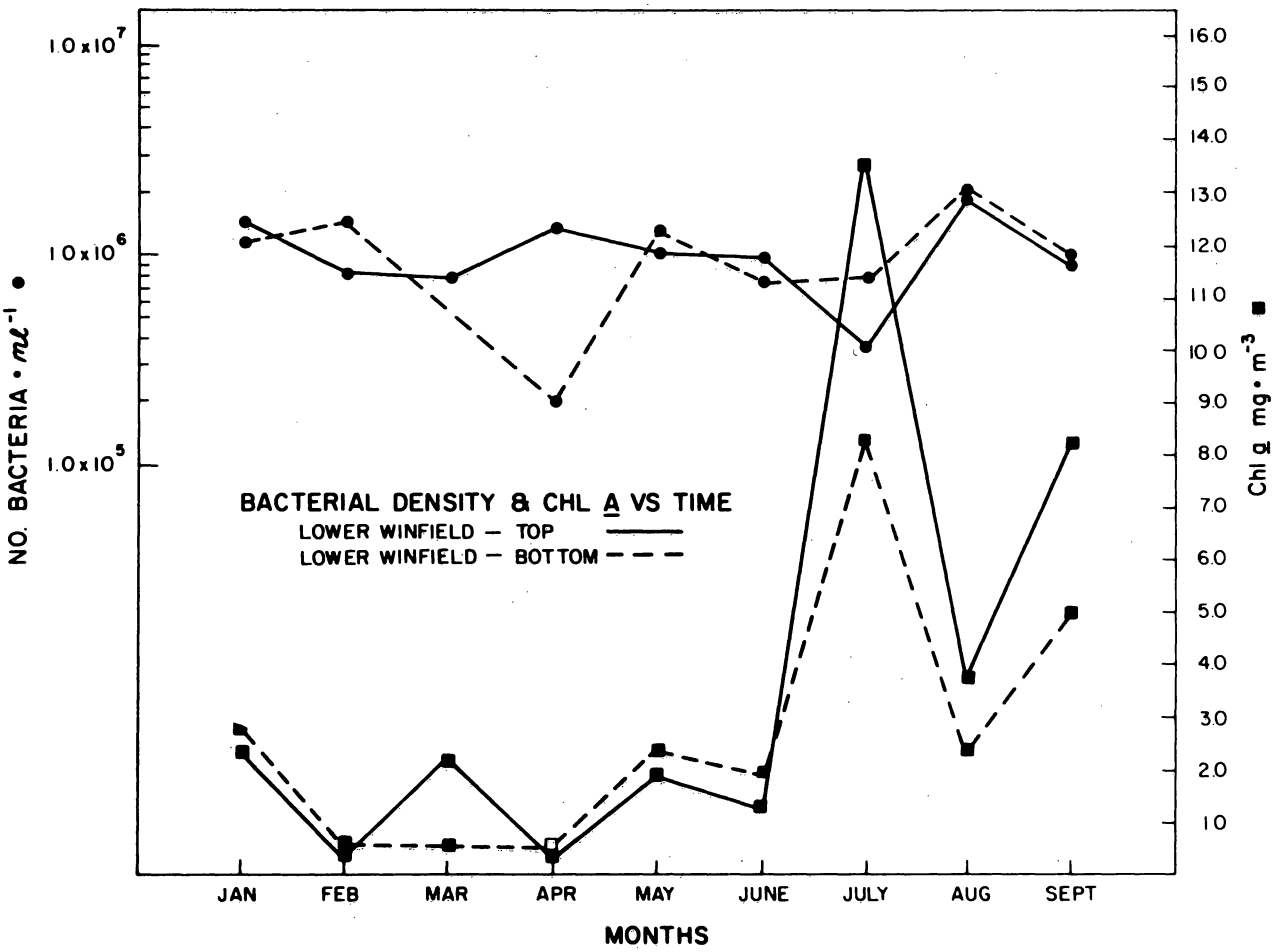

Fig. V.4.12. Relationship between bacterial density and phytoplankton biomass (Chl a) at LW from January through September 1983.

January and September and the bacteria already have reached density levels that will continue through the remaining months. Phytoplankton biomass, as measured by chl a, did not begin to show an increase until after June and there was little difference between Top and Bottom chl a levels, as noted by Lizotte (1984).

V.5 PHYTOPLANKTON

Taxonomic Composition and Cell Density

Phytoplankton are combined with the periphyton in the taxonomic list presented in Appendix A, Table A5.1.1. The total number of taxa identified from the phytoplankton was 126, representing 87 genera and 7 classes. This includes two large, diverse groups as individual taxa, the phytoflagellates and the naviculoid diatoms. Phytoflagellates may be members of the classes Chlorophyceae, Euglenophyceae, Chrysophyceae, Dinophyceae, or Cryptophyceae which were not sufficiently distinguishable to be identified further. Naviculoid diatoms may be members of the genera Navicula, Pinnularia, Diploneis, Carpatogramma, Stauroneis, Neiduim, Gomphonema, Anomoeneis, Frustulia, Rhoicosphenia, Achnanthes, Cymbella, or Nitzschia which had a naviculoid shaped frustule but could not be identified further for cells containing cytoplasm. Two taxa which were previously reported in the Reconnaissance Report (Voshell et al. 1982) have been listed under their more proper

names. Schroederia judayi was changed to Ankyra judayi, and Merismopedia tenuissima was changed to Agmenellum tenissima.

Phytoplankton cell densities are recorded in Appendix A, Tables A5.2.1-29 as cell per milliliter (cells/ml) which equals millions of cells per cubic meter (cells x $10^6/m^3$). Totals ranged from 16 to 11,620 cells/ml; were lower in winter and higher in summer. The percent composition of the phytoplankton by taxonomic class is listed in Tables A5.2.30-41. The algae are grouped by class (Smith 1950) into Chlorophyceae (green algae), Euglenophyceae, Chrysophyceae, Bacillariophyceae (diatoms), Dinophyceae, Cryptophyceae, or Cyanophyceae (blue-green algae). Four of these groups, the Chlorophyceae, Bacillariophyceae, Cryptophyceae and Cyanophyceae represented from 72 - 100% of the cells counted. The Euglenophyceae made up 12% of the cells at LW in January, the Chrysophyceae made up 14% at LW in May, and 10% at UW in October and May. In Figs.V.5.1 and V.5.2 the four major classes are arranged on a cumulative scale, and the percent represented by a single group is equivalent to the space between lines. A pattern of succession was clearest for the LW plankton (Fig.V.5.1): a blue-green algae dominance from December to April; a diatom dominance from May to June; a green algae dominance from July to August; and no single group dominating in the fall. For the UW plankton (Fig.V.5.2), the diatoms were the dominant group for most of the year, particularly from May to September. The cryptophytes were equivalent to the diatoms in

November and January, the green algae were dominant in October and April, and the blue-green algae were dominant in December. The successional trends common to both stations were the dominance of cryptophytes in November, the dominance of blue-green algae in December, the diatom dominance in the late spring, and the increasing blue-green algae contribution in the late summer.

Phytoplankton taxa which comprised more than 10% of the total population for each monthly sample, at either station, are summarized in Table V.5.1 as cell densities and percent of total. From October 1982 to June 1983, cell densities were on the order of 10^1 to 10^2 cells/ml (range recorded = 16 to 359 cells/ml). From July 1983 to September 1983, cell densities were on the order of 10^3 to 10^4 cells/ml (range recorded = 1,160 to 11,620 cells/ml). Differences in cell density between the stations were not significant until the summer season. The LW station had a greater population density in July due to a bloom of green algae, primarily the volvocids Pandorina morum and Eudorina elegans. The UW site had greater cell densities for August and September because of high numbers of very small cells, primarily the green algae Actinastrum Hantzschii, the blue-green Agmenellum tenuissima, and the centric diatoms Skeletonema potamos, Cyclotella meneghiniana, and a Cyclotella sp. Taxa which were present year round and were found at relatively high densities for particular samples included: the crytophytes Chroomonas

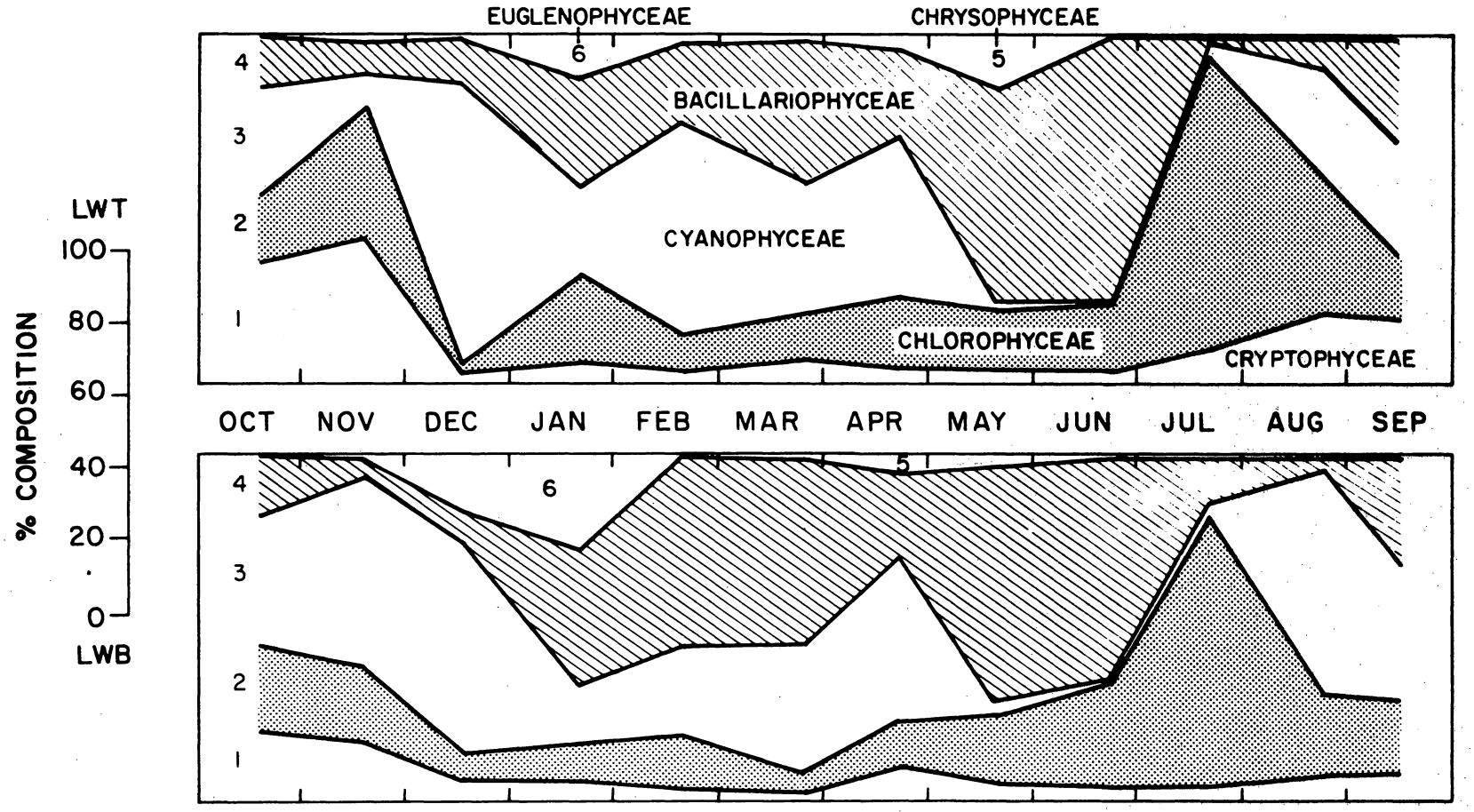

Fig. V.5.1. Composition of phytoplankton at LW.

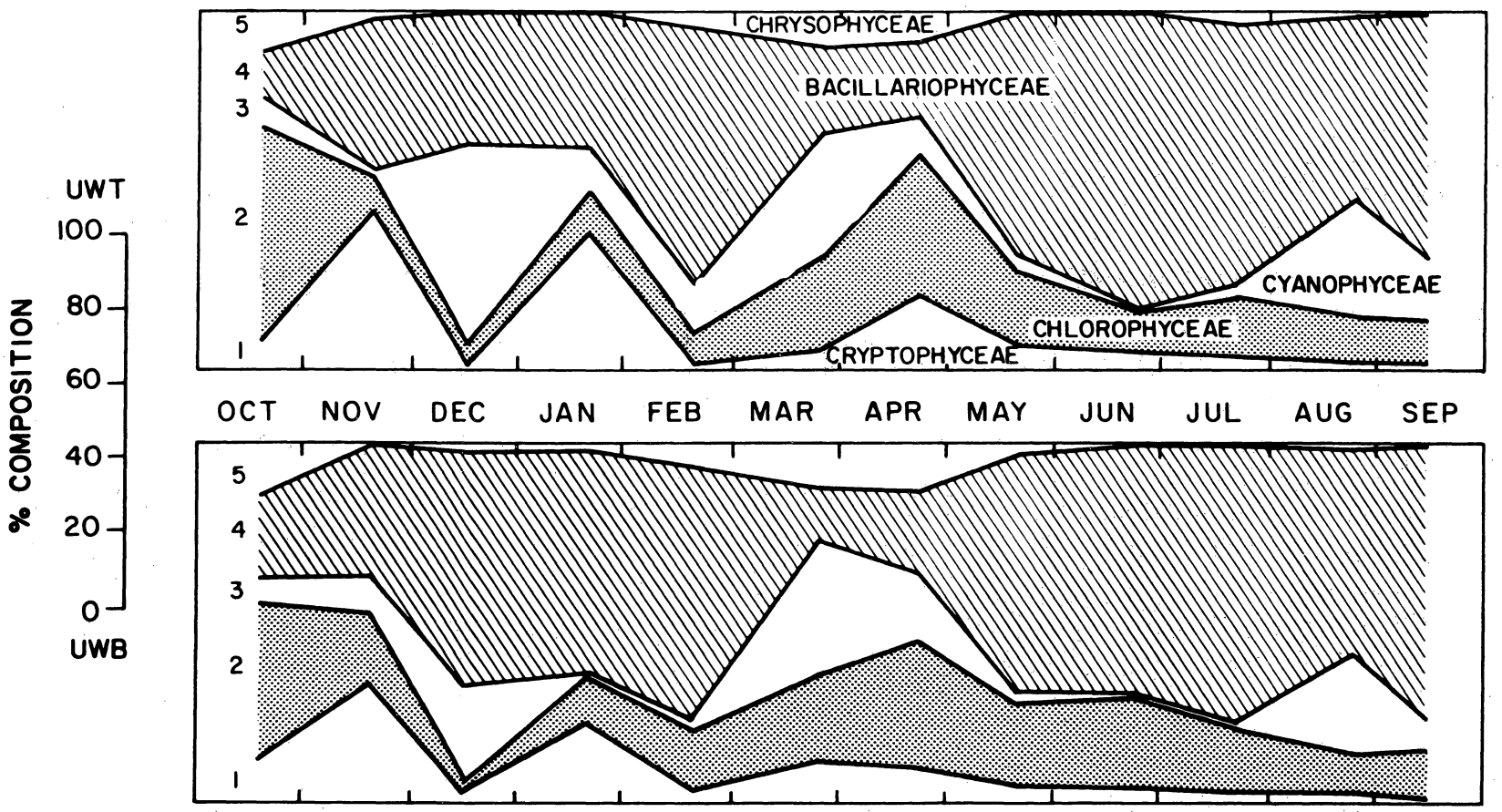

Fig. V.5.2. Composition of phytoplankton at UW.

minuta and Cryptomonas sp. the green algae species of Chlamydomonas the blue-green alga Phormidium angustissimum and the diatoms, Melosira varians and various species collectively called Naviculoid.

Autecology of Dominant Species

Stichococcus bacillaris

Stichococcus bacillaris is a green alga which is composed of simple, unbranched filaments of short, cylindrical cells, which are often loosely connected, so that interrupted series are formed. The tendency of this alga to fragment results in filaments that are rarely more than several cells in length.

Melosira varians

Melosira varians is a centric diatom with cylindrical cells which are closely united into long filaments. It is periphytic (occurring on the substrate and submerged objects). M. varians is a cosmopolitan species with an extraordinarily large ecological span (Lowe 1974). It has a pH range of 6.4-9.0 (optimum of about 8.5, Cholnoky 1968) and occurs over a broad range of temperatures. It may produce massive growths in eutrophic waters in the summer, as well as large growths in katharobic water (waters in which dissolved organic nutrients are very low or absent) in January and February (Hornung 1959). This species tolerates small amounts of salt and is indifferent to

Table V.5.1. Phytoplankton taxa and densities (cells/ml) for each taxon comprising more than 10% of total at either Lower Winfield (LW) or Upper Winfield (UW) during one sampling period.

Date	Taxa	Density (% Total) UW	LW
Oct	Stichococcus bacillaris	34 (23%)	0
	Chroomonas minuta	10 (7%)	33 (26%)
	Agmenellum tenuissima	0	20 (16%)
Nov	Chlamydomonas spp.	3 (4%)	7 (13%)
	Crucigena fenestrata	0	6 (11%)
	Melosira varians	16 (22%)	0
	Chroomonas minuta	28 (39%)	10 (19%)
	Cryptomonas sp.	3 (4%)	9 (17%)
Dec	Naviculoid	20 (22%)	2 (3%)
	Anabaena spp.	19 (21%)	0
	Phormidium angustissimum	30 (33%)	40 (52%)
	Phormidium fragile	0	20 (26%)
Jan	Chlamydomonas spp.	7 (12%)	4 (25%)
	Cyclotella meneghiniana	0	3 (19%)
	Diatoma vulgare	14 (23%)	0
	Chroomonas minuta	11 (18%)	0
	Cryptomonas sp.	9 (15%)	0
	Dactylococcopsis raphidioides	7 (12%)	4 (25%)
Feb	Chlamydomonas spp.	4 (7%)	8 (11%)
	Melosira varians	12 (21%)	1 (1%)
	Naviculoid	7 (13%)	2 (2%)
	Oscillatoria geminata	0	11 (15%)
	Phormidium angustissimum	6 (11%)	30 (42%)
Mar	Chlamydomonas spp.	26 (12%)	10 (6%)
	Hyalobryon mucicola	22 (10%)	0
	Naviculoid	0 (5%)	29 (19%)
	Oscillatoria spp.	0	48 (31%)
	Phormidium angustissimum	60 (28%)	0
Apr	Actinastrum hantzschii v. fluviatile	10 (12%)	0
	Chlamydomonas spp.	18 (22%)	4 (4%)
	Chroomonas minuta	9 (11%)	3 (3%)
	Phormidium angustissimum	0	40 (36%)

May	Mastigosphaera sp.	40 (14%)	0
	Melosira varians	160 (54%)	120 (47%)
Jun	Dictyosphaerium ehrenbergianum	0	50 (14%)
	Fragilaria crotenensis	0	50 (14%)
	Melosira varians	200 (70%)	200 (56%)
Jul	Pandorina morum	190 (6%)	3600 (62%)
	Skeletonema potamos	1990 (65%)	52 (1%)
Aug	Actinastrum hantzschii	1000 (11%)	0
	Skeletonema potamos	3170 (33%)	0
	Agmenellum tenuissima	3100 (33%)	920 (28%)
Sep	Cyclotella sp.	2360 (20%)	6 (1%)
	Melosira granulata	0	286 (25%)
	Skeletonema potamos	5550 (48%)	48 (4%)
	Cryptomonas sp.	170 (1%)	140 (12%)
	Agmenellum tenuissima	2000 (17%)	360 (31%)

iron concentration (Niessen 1956). It is beta-mesosaprobic (characteristic of zones where oxidation of the organic load is proceeding and nitrogen is in the form of ammonia compounds) (Lowe 1974) and is probably an obligate nitrogen heterotroph (Cholnoky 1968).

Chroomonas minuta

Chroomonas is a unicellular, free-swimming flagellate. It has compressed cells with the anterior end truncate and the posterior end rounded (Smith 1950).

Phormidium angustissimim

Phormidium angustissima is a filamentous, periphytic blue-green alga. It occurs primarily in rivers and streams other than the organically polluted ones but also occurs in the presence of pollution (Van Landingham 1982).

Phormidium fragile

Phormidium fragile is a blue-green alga with a slimy, yellowish plant mass. It occurs in fresh water, brackish, and salt water (Cocke 1967).

Diatoma vulgare

Diatoma vulgare is a cosmopolitan, periphytic and epiphytic diatom. In summer it is dominant on Cladophora in large

eutrophic lakes, and in winter it is found on other substrata in the surf zone (Jorgensen 1948). It has been collected over a broad range of temperatures and a pH range of 6.4-8.3 (optimum about 8.2, Cholnoky 1968). It is present over a range of temperatures but is characteristically a winter dominant form. It is found in eutrophic waters and is beta-mesaprobic to oligosaprobic (characteristic of the zone where oxidation of biodegradable compounds is complete and the concentration of inorganic nutrients is usually high) (Lowe 1974).

Datylococcopsis raphidioides

Dactylococcopsis raphidioides is a blue-green alga with sigmoid cells arranged in colonies of 4-8 within a hyaline, gelatinous envelope. It is epiphytic on species such as Utricularia. (Prescott 1962).

Hyalobryon mucicola

The cells of the yellow-green alga Hyalobryon mucicola, may be solitary or united in branching colonies consisting of a series of nesting cups representing growth stages. It is epiphytic on various algae. Smith (1950) reported that this species had been found only in Wisconsin where it is rare to common in lakes and swamps.

Actinastrum hantzschii v. fluviatile

Actinastrum hantzschii v. fluviatile is a green alga with cells which are spindle-shaped, narrowed toward the apices, and arranged in simple or compound colonies of 4 or 8, with the long axes of the cells radiating from a common center (Prescott 1962). Actinastrum is of widespread occurrence in the plankton of lakes and ponds (Smith 1950).

Dictyosphaerium ehrenbergianum

Dictyosphaerium ehrenbergianum is a green alga which forms ovoid colonies composed of 8-30 ellipsoidal cells. The cells are attached in groups of 2 or 4 at the ends of fine, branched strands. It is common in the plankton of many soft-water lakes (Prescott 1962).

Pandorina morum

Colonies of the green alga, Pandorina morum are composed of 8-16 flagellated cells which are compactly arranged and enclosed by a common gelatinous envelope. P. morum is common in the plankton of both hard and soft water lakes but is more frequent among dense growths of algae in the shallows, especially in water rich in nitrogenous matter (Prescott 1962).

Skeletonema potamos

Skeletonema potamos is a species of diatom which grows in freshwater lakes and rivers as well as in brackish water. When

grown at a salinity of 0%, the processes are extremely short; at salinities of 2% or more, the processes are much longer (Hasle and Evensen 1976). It has been collected from localities regarded as entrophic.

Agmenellum (Merismopedia) tenuissima

Agmenellum tenuissima is a blue-green alga with a flat plant mass composed of small spherical cells arranged in definite rows at right angles to each other. It is common in the plankton of brackish and fresh-water and is also found on wet sand and mud (Cocke 1967). It occurs under a variable range of organic nutrient content and salt content and is without value as an indicator of pollution (Van Landingham 1982). It is tychoplanktonic (normally associated with periphytic habitats but often suspended in the water). It occurs over a broad temperature spectrum with optimum growth during the summer.

Melosira granulata

Melosira granulata is a cosmopolitan, euplanktonic (suspended in the water) diatom which occurs in eutrophic lakes, ponds, and rivers. It has been collected within a pH range of 6.3-9.0 with an optimum of 7.9-8.2 (Cholnoky 1968). It tolerates small amounts of salts and is characteristic of the zone where oxidation of the organic load is proceeding (Lowe 1974). Optimal growth is obtained during the summer.

Biomass

Chlorophyll a (chl a) concentrations by size class are recorded in Appendix A, Tables A5.3.1-18 and are summarized in Table V.5.2. The majority of the biomass was in the 0 - 25µm size range (43 to 99%), and the average percentage was 80%. The large size fractions had average percentages of approximately 4% for 25 - 43µm, 6% for 105 - 243µm, 2% for 243 - 1000µm and less than 243µm occasionally had relative contributions above 10%, and as high as 34%.

Total chl a concentrations (Table V.5.3) ranged from 0.278 to 13.507 mg/m^3 with an average of approximately 4 mg/m^3. UW generally had a higher chl a concentration than LW, but that was not predictable from month to month. From February through June, the two sites had comparable values; from July through September, great differences were seen between sites; and for January, much higher values were recorded for UW than the LW. Bottom samples had consistently lower concentrations than Top samples for LW during the full 24 hr. Estimates are presented in terms of carbon based on summer months of July, August, and September. The relationship between phytoplankton cell density and biomass (chl a) for the study period is represented in Figures V.5.3 - V.5.6.

Gross Primary Production and Respiration

Gross primary production and respiration rates determined by the light-dark bottle method for a period of one year are

Table V.5.2. Chl a concentrations for seston size classes.

Size Class	Site	Depth	Percent range	Percent mean
UF	UW	Top	43.3 - 99.0	80.8
		Bot	49.2 - 97.9	80.3
	LW	Top	59.4 - 99.2	80.9
		Bot	52.9 - 97.2	76.7
VF	UW	Top	0.3 - 8.3	2.9
		Bot	0.3 - 9.0	3.0
	LW	Top	0.1 - 12.1	3.7
		Bot	0.5 - 34.2	6.8
F	UW	Top	0.5 - 25.2	7.4
		Bot	1.0 - 23.9	8.2
	LW	Top	0.4 - 20.2	7.4
		Bot	0.2 - 26.8	9.0
S	UW	Top	0.2 - 17.2	6.8
		Bot	0.1 - 17.3	5.7
	LW	Top	0.2 - 20.5	6.4
		Bot	0.1 - 23.9	6.0
ML	UW	Top	0 - 5.9	1.8
		Bot	0 - 6.8	2.1
	LW	Top	0 - 4.7	1.3
		Bot	0 - 6.5	1.2
L	UW	Top	0 - 6.9	1.3
		Bot	0 - 2.1	0.6
	LW	Top	0 - 1.8	0.4
		Bot	0 - 0.8	0.3

Table V.5.3. Summary of total phytoplankton chl *a* concentrations (mg Chl *a*/m^3) and biomass (mg C/m^3).

Month	Depth	Concentration		Biomass	
		UW	LW	UW	LW
Jan	Top	9.464	2.310	161.4	39.4
	Bot	3.904	2.793	66.6	47.6
Feb	Top	0.606	0.318	10.3	5.4
	Bot	0.796	0.505	13.6	8.6
Mar	Top	1.012	2.151	17.3	36.7
	Bot	0.852		14.5	
Apr	Top	0.282	0.339	4.8	5.8
	Bot	0.278	0.404	4.7	6.9
May	Top	2.939	1.865	50.1	31.8
	Bot	2.312	2.309	39.4	39.4
Jun	Top	1.324	1.263	22.6	21.5
	Bot	1.539	1.948	26.2	33.2
Jul	Top	6.592	13.507	112.4	230.3
	Bot	6.216	8.291	106.0	141.4
Aug	Top	10.212	3.783	174.1	64.5
	Bot	7.894	2.385	134.6	40.7
Sep	Top	13.236	8.242	225.7	140.5
	Bot	13.097	4.953	223.3	84.4

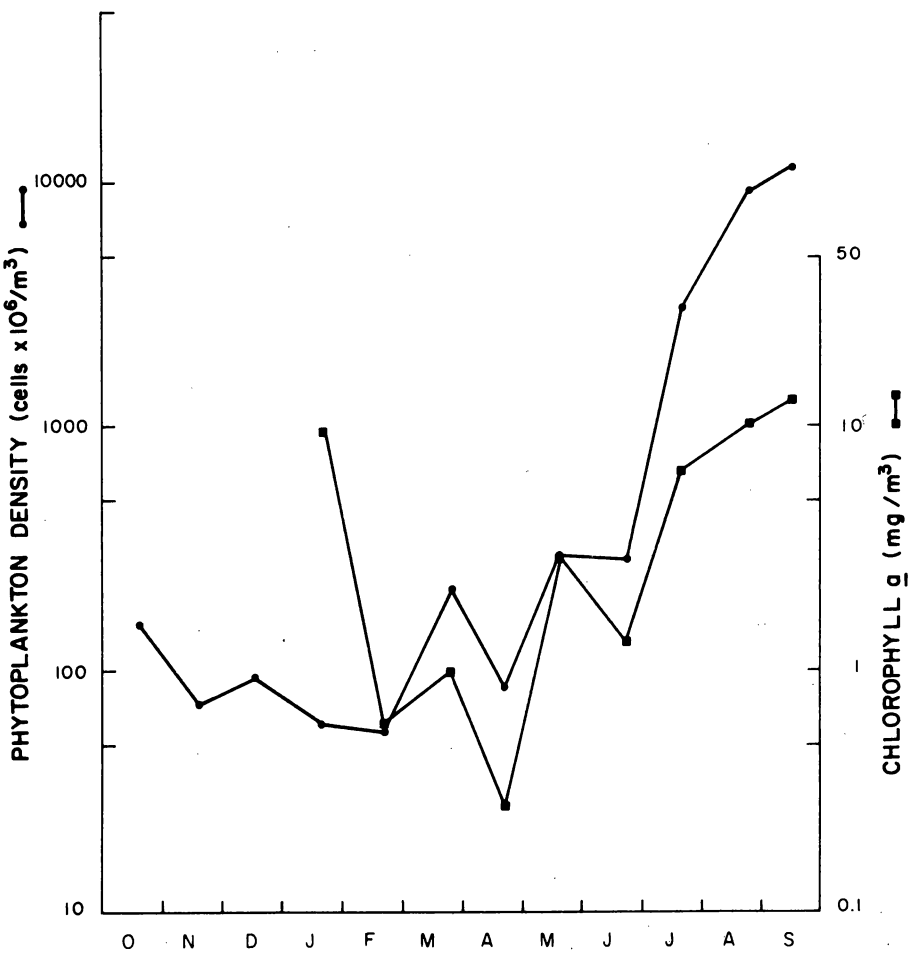

Fig. V.5.3. Relationship between phytoplankton cell density (cells/m^3) and phytoplankton biomass (mg Chl *a*/m^3) in Top samples from UW.

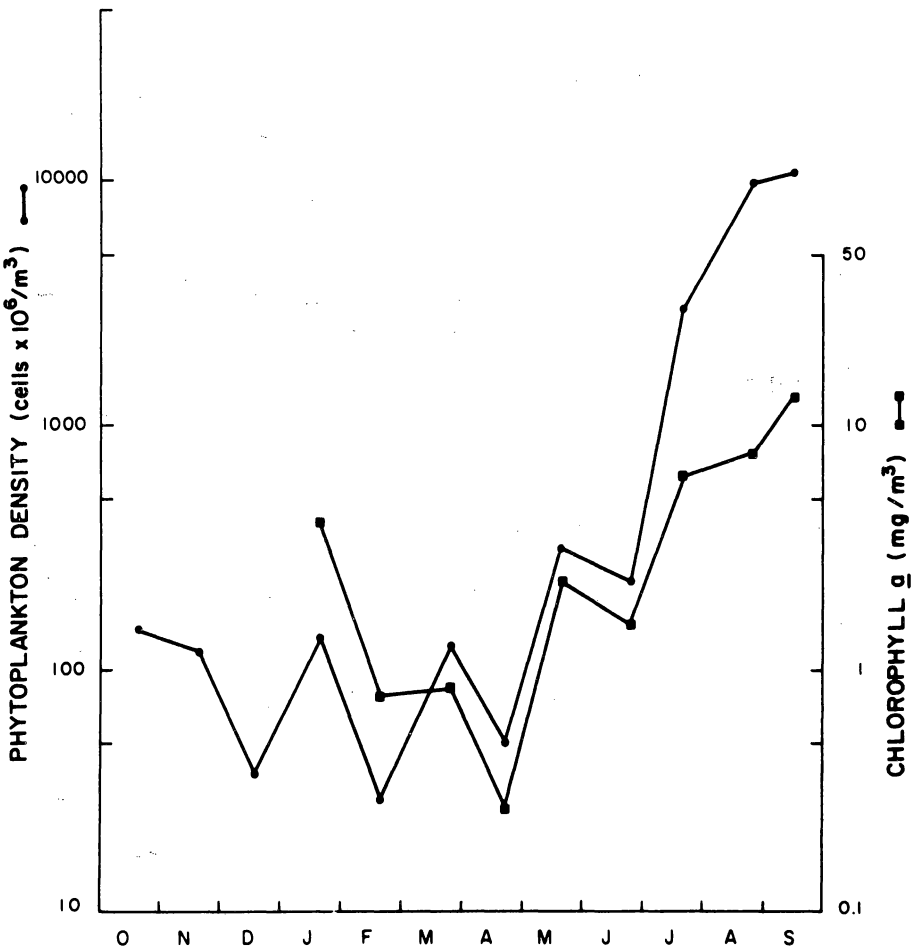

Fig. V.5.4. Relationship between phytoplankton cell density (cells/m^3) and phytoplankton biomass (mg Chl *a*/m^3) in Bottom samples from UW.

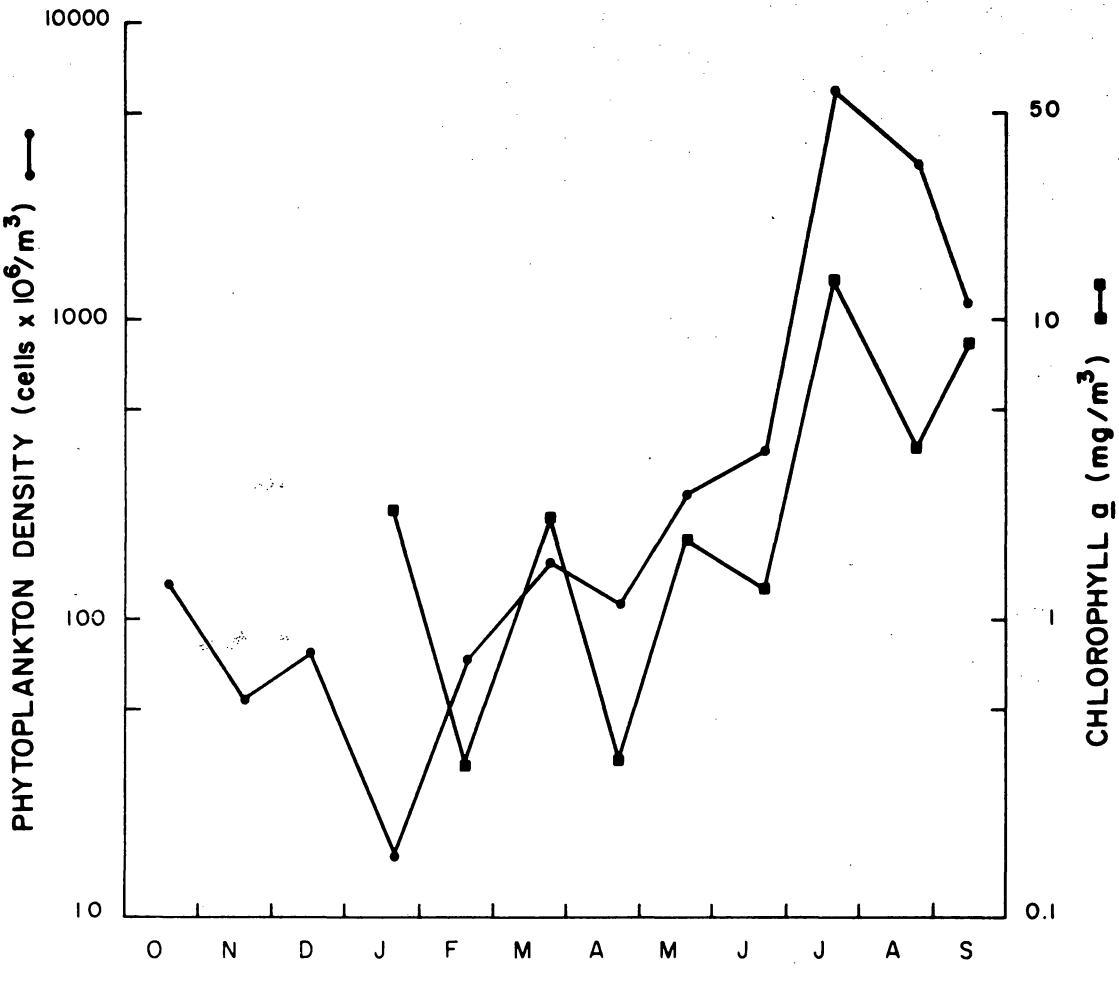

Fig. V.5.5. Relationship between phytoplankton cell density (cells/m^3) and phytoplankton biomass (mg Chl *a*/m^3) in Top samples from LW.

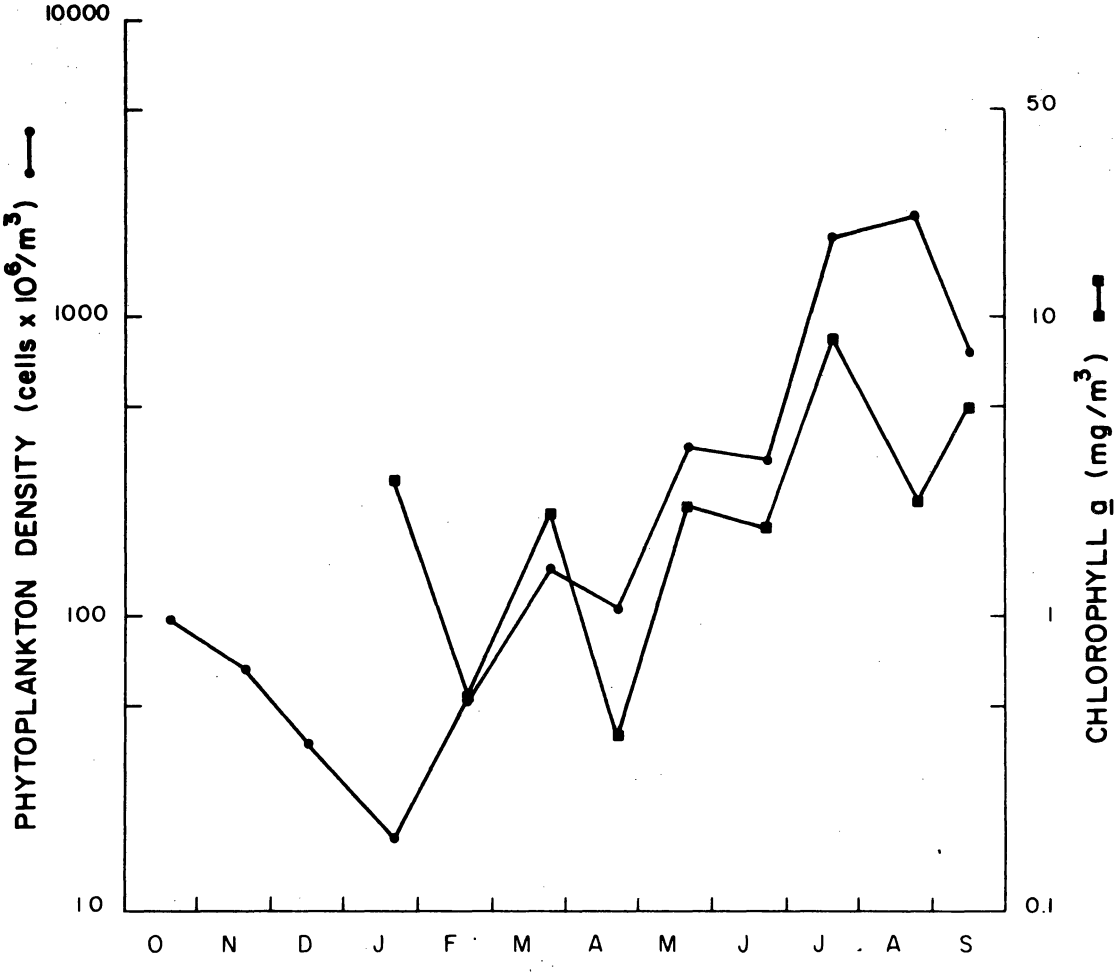

Fig. V.5.6. Relationship between phytoplankton cell density (cells/m^3) and phytoplankton biomass (mg Chl *a*/m^3) in Bottom samples from LW.

recorded in Appendix A, Tables A5.4.1-24 and are summarized in Table V.5.4 and Fig V.5.7. When gross and net primary production are compared (Fig.V.5.7.) the heterotrophic nature of the river is apparent, and only the months of July and August showed a net production rate. When the rates were integrated for the 10 m deep water column, respiration was always greater than primary production. Periphyton communities generally had production: respiration ratios of one and surveys of the length of the Winfield pool did not reveal any beds of submerged macrophytes.

Size fractionation studies were begun in April 1983, but significant rates were not measured until June for respiration and July for gross primary production. Activity may have been limited by low temperatures, low light levels (Table V.5.5), or low phytoplankton populations.

The size classes producing a significant amount of oxygen are presented in Table V.5.6. For each month from July through September there were two size classes with significant activity. The 3 - 25μm size class was always a major contributor, and always included the majority of the phytoplankton biomass (Table V.5.7). If the rate of oxygen production is placed on a biomass basis (Table V.5.8), the productivity of the each of the other size classes is proportionately greater.

The total number of phytoplankton species identified for each of these three months ranged from 26 to 29, and the majority occurred as cells or colonies 3 - 25μm in diameter. From algae

Table V.5.4. Summary of daily gross primary production (GPP) and daily respiration (Resp) rates of the plankton. (mg $O_2/m^2/day$)

Month	UW		LW	
	GPP	Resp.	GPP	Resp.
Oct	850	3,100	1,700	18,000
Nov	0	600	100	1,900
Dec	60	170	1,000	4,100
Jan	0	0	0	0
Feb	20	1,300	60	1,400
Mar	650	2,800	100	4,000
Apr	0	0	0	0
May	300	740	230	2,000
Jun	880	750	10	690
Jul	4,600	2,600	7,700	23,000
Aug	7,100	2,900	1,600	3,500
Sep	3,100	4,500	2,900	18,000

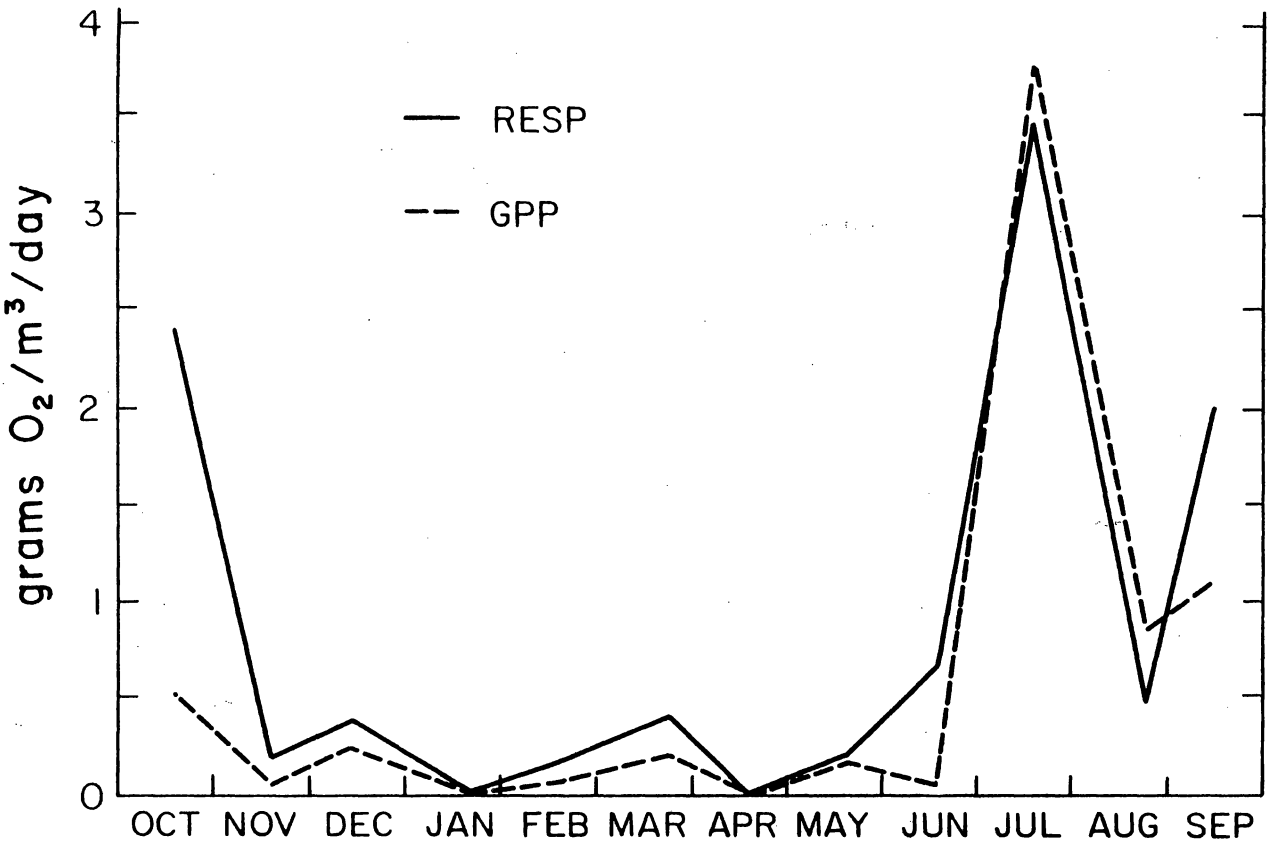

Fig. V.5.7. Plankton respiration (RESP) and gross primary production (GPP) in Top samples at LW.

Table V.5.5. Temperature and light intensity at LW. Measurements taken in mid-channel, for a depth of 1 m, at approximately noon of the dates listed.

Date	Temperature	Light	
(1983)	(°C)	($E/cm_2/s$)	(% of Incident)
4-22	9.0	3.5×10^{-8}	18
5-20	15.0	7.0×10^{-9}	8
6-23	22.5	1.2×10^{-9}	1
7-21	28.5	1.2×10^{-9}	20[a]
8-25	28.7	3.0×10^{-8}	20
9-15	25.5	2.8×10^{-8}	19

[a]Calculated from Secchi disk depth relationship determined for the Kanawha River: S.D. depth : 20% incident light intensity.

Table V.5.6. Oxygen production rates for each size class making a significant ($P \geq 0.95$) contribution to total gross primary production at LW.

1983	µg O_2/l/hr			
	<3 µm	3-25 µm	25-43 µm	43-105 µm
7-21	22	210	0	0
8-25	0	20	27	0
9-15	0	39	0	31

Table V.5.7. Biomass of phytoplankton in seston size classes at LW. Any difference from 100% is due to phytoplankton retained by the 105 µm screen.

Date (1983)	µg Chl a/l (% Total) 43-105µ	25-43µ	<25µ	<3µ
7-21	0.103 (0.8%)	0.267 (2.0%)	13.008 (97.0%)	0.100 (0.7%)
8-25	0.066 (1.7%)	0.141 (3.7%)	3.479 (92.0%)	Not Sampled
9-15	1.227 (14.9%)	0.157 (1.9%)	5.090 (61.8%)	Not Sampled

Table V.5.8. Oxygen production on a biomass basis for each size class making a significant (P≥ 0.95) contribution to total gross primary production at LW.

Date (1983)	$\mu gO_2/\mu g$ Chlorophyll a/hr				
	<3μ	3-25μ	25-43μ	43-105μ	Total
7-21	220	16	0	0	18
8-25	0	6	193	0	12
9-15	0	8	0	25	9

cell counts of the samples with size fractions removed, particular species may be implicated with the primary production attributed to other size fractions. The taxa selectively retained or selectively passed and their population densities are listed in Table V.5.9.

The size fractions consuming a significant amount of oxygen for day and night experiments are presented in Table V.5.10. The <3µm size class was the only significant contribution for 3 of the dates, and a significant contribution by the 3 - 25µm group was measured in July. This contribution by the 3 - 25µm fraction occurred in conjunction with the highest gross primary production by the phytoplankton in that size fraction. Significant differences between day and night rates were observed in July for the 0 - 3µm and 3 - 25µm fractions, and in September for the 0 - 3µm fraction. For the 0 - 3µm size class, night respiration was greater than day respiration.

The direct counts of bacteria in the fraction passing a 3µm Nuclepore filter were compared to unfiltered ("total") samples (Table V.5.11). The bacterial population represented in the <3µm size fraction ranged from 83 - 96%, and averaged 89% of the total.

Table V.5.9. Algal species which may be implicated with the productivity attributed to size classes other than 3-25 µm at LW.

Date 1983	Species	Cells/ml <3µm	<25µm	<43µm	<105µm
7-21	Dactylococcopsis raphidioides	16	64	--	--
8-25	Eudorina elegans	0	10	50	--
	Pandorina morum	0	10	50	--
	Platydorina caudata	0	10	100	--
	Melosira granulata	0	1	6	--
9-15	Errerella bornhemiensis	0	0	0	10
	Eudorina elegans	0	0	0	10
	Pandorina morum	0	0	20	20
	Platydorina caudata	0	0	6	6
	Melosira granulata	0	0	36	160

Table V.5.10. Oxygen uptake rates for each size class making a significant ($P \geq 0.95$) contribution to total community respiration at LW. Dates when there were significant ($P \geq 0.95$) differences between night and day rates are marked by an asterisk(*).

Date		$\mu gO_2/l/hr$	
1983	Period	<3 µm	3-25 µm
6-23	Night	25	0
	Day	20	0
7-21	Night	14	32
*	Day	65	9
8-25	Night	21	0
	Day	29	0
9-15	Night	4	0
*	Day	24	0

Table V.5.11. Concentrations of bacteria at LW. Direct counts were made on unfiltered water and water which had passed a 3 µm Nuclepore filter.

Date 1983	Period	10^6 cells/ml < 3 µm	Total	% Passing 3 µm filter
6-22	Night	2.2	2.3	96
	Day	2.1	2.3	91
7-21	Night	2.0	2.2	91
	Day	2.1	2.5	84
8-25	Night	2.0	2.4	83
	Day	1.5	1.6	94
9-15	Night	2.0	2.3	87
	Day	2.0	2.2	91

V.6. PERIPHYTON

Taxonomic Composition and Cell Density

Periphyton are combined with the phytoplankton in the algae taxa list in Appendix A, Table A5.1.1. The total number of periphyton taxa identified was 60, representing 42 genera and 4 classes.

Periphyton cell densities are recorded in Appendix A, Tables A5.2.1-29. Total cell density ranged from 1.99×10^7 to 4.96×10^{11} cells/m^2 and were lowest in winter in comparison to fall, spring, or summer periods. Changes in the respective composition of the periphyton community by algae class are recorded in Appendix A, Tables A6.2.1-12 and are illustrated in Fig. V.6.1. As these figures show, the dominant class was Bacillariophyceae through most of the year, except for the late summer months when the Cyanophyceae were more dominant. Rhizoclonium hieroglyphicum C. A. Agardh, a filamentous Chlorophycean, was found in large masses around our buoy lines in July 1983 and may be an important taxa though it was not found on the other substrates sampled.

Biomass

Figs. V.6.2 and V.6.3 show the relationship between periphyton cell density and chl a concentration for the study period. The two measurements of community change followed each other very closely.

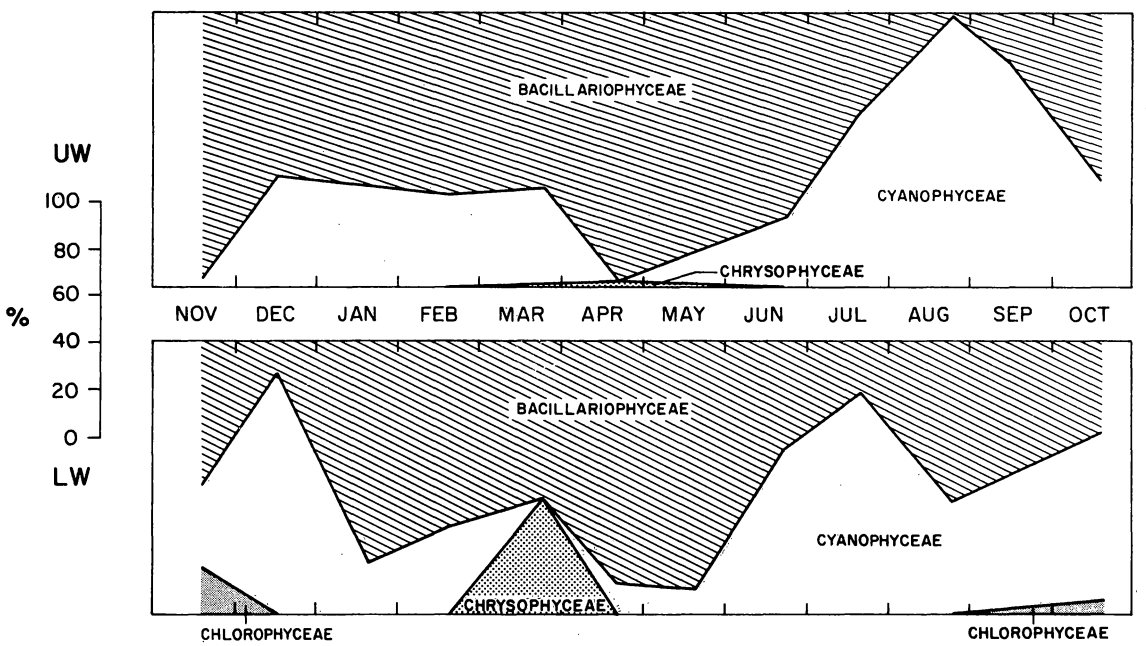

Fig. V.6.1. Composition of periphyton at UW and LW.

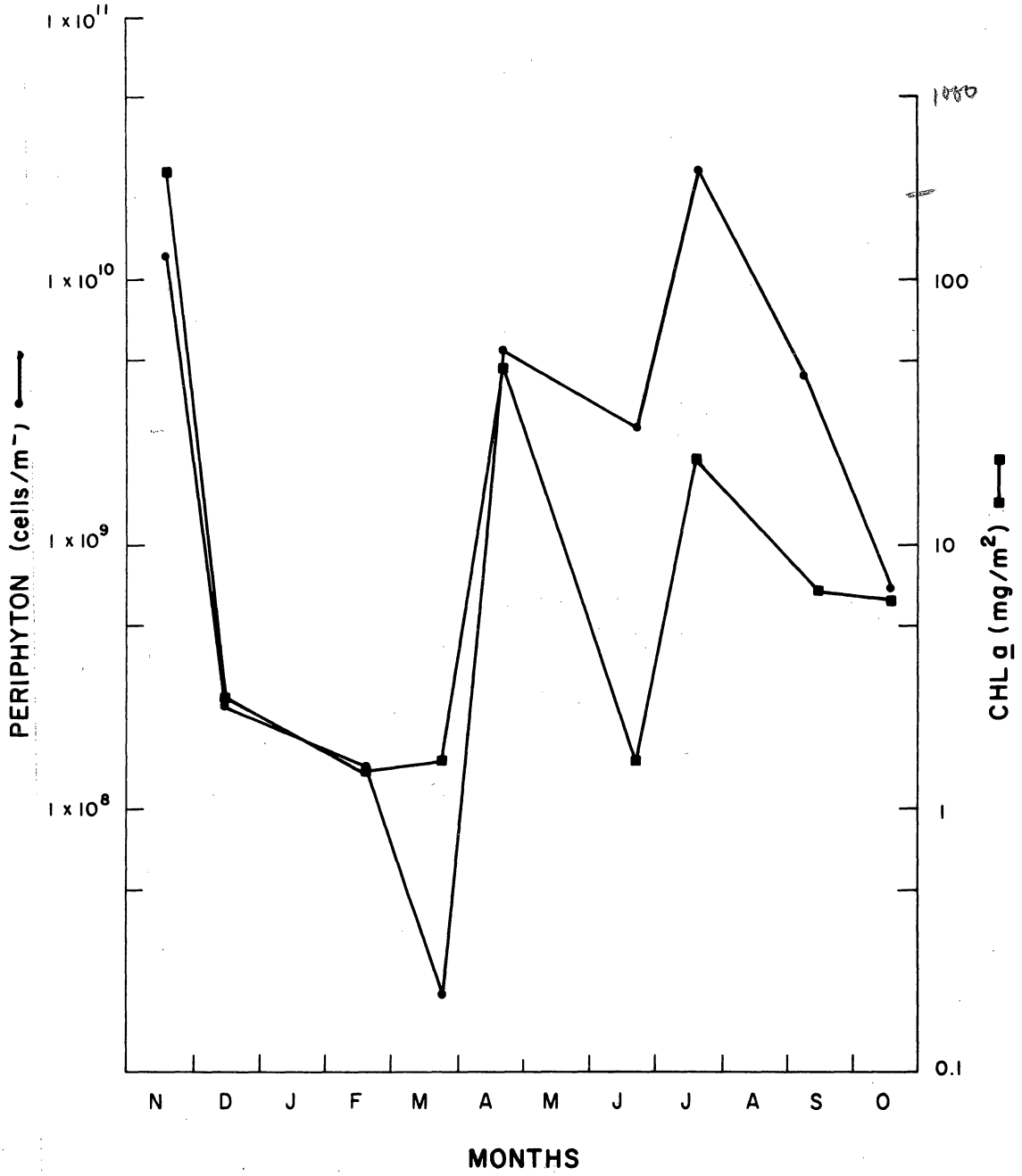

Fig. V.6.2. Relationship between periphyton cell density (cells/m^2) and periphyton biomass (mg chl a/m^2) at UW.

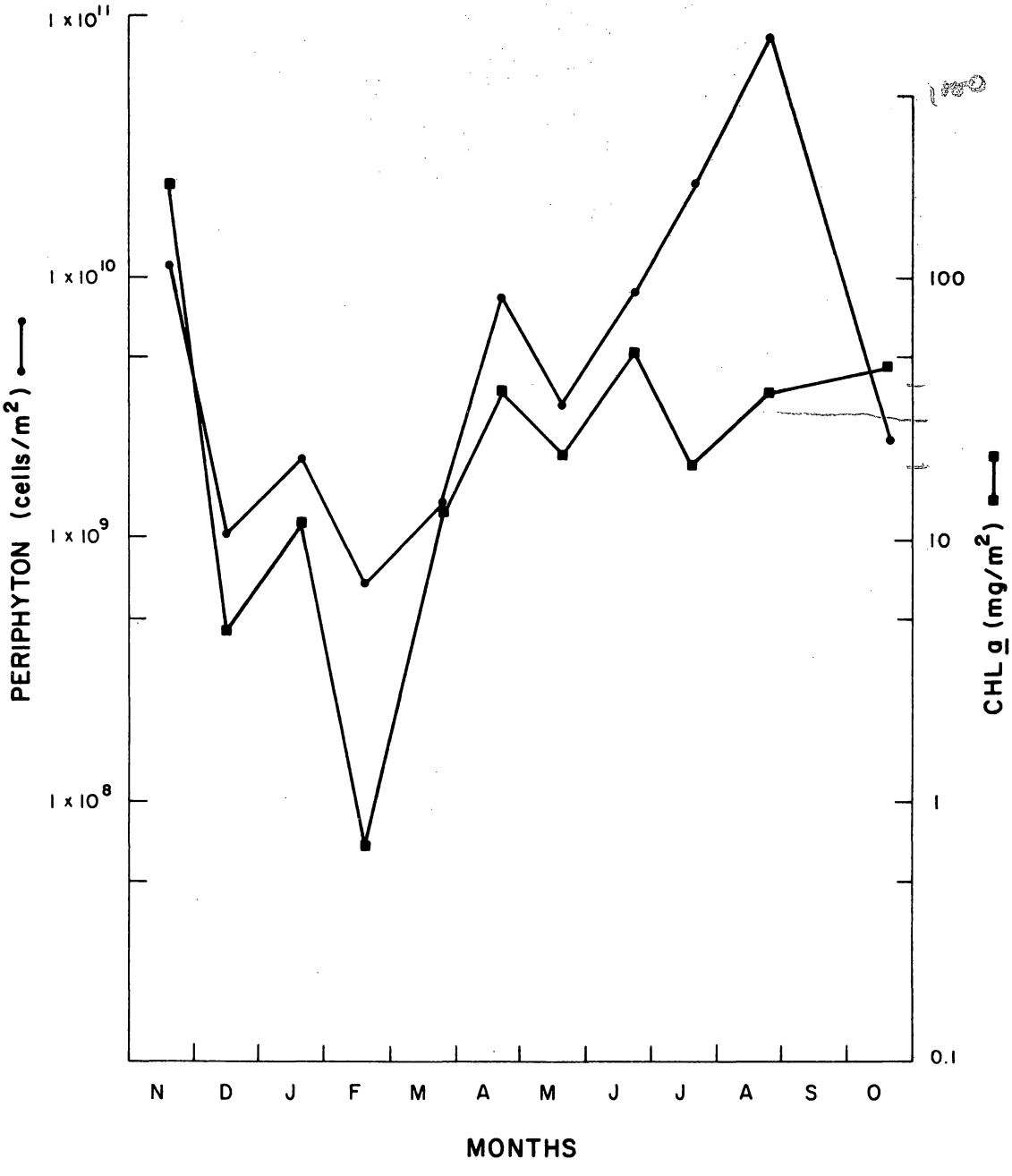

Fig.V.6.3. Relationship between periphyton cell density (cells/m^2) and periphyton biomass (mg chl a/m^2) at LW.

Gross Primary Production and Respiration

The daily gross primary production and respiration rates of the periphyton for the study period are recorded in Appendix A, Tables A6.3.1-17 and are summarized in Table V.6.1. Even though several sampling periods were missed in UW due to loss of slides, the data show that the highest levels of GPP at both sites were between the months of April and August. Respiration rates were more variable and were probably influenced by the assemblage of micro and macroinvertebrates on the glass slides during certain times of the year. The highest GPP rate was 2,080 mg O_2/m^2 substrate/day in July at LW. Values >1,000 mg O_2/m^2 substrate/day during the summer months were common. January showed a GPP of 0 mg O_2/m^2 substrate/day which was the lowest value recorded.

The monthly conversions of biomass (mg chl a/m^2) to caloric equivalents (kcal/m^2) per slide, and to caloric equivalents for the pool (kcal/m^2) are presented in Table V.6.2.

V.7. ZOOPLANKTON

Seasonal

Total rotifer concentrations were characterized by a single summer population peak (July-August), with total rotifer concentration at UW and LW approaching or exceeding 1 x 10^6 individuals m^{-3} (Fig. V.7.1). Spring concentrations (April-June)

Table V.6.1. Summary of daily gross primary production (GPP) and daily respiration (Resp) of the periphyton (mg O_2/m^2 substrate/day).

Date	UW		LW	
	GPP	Resp	GPP	Resp
Nov	750	500	1,140	1,410
Dec	40	1,540	90	30
Jan			0	70
Feb			170	60
Mar	50	40	100	70
Apr	1,990	1,260	1,530	1,210
May			850	260
Jun	1,420	500	1,850	720
Jul	1,020	810	2,080	640
Aug			1,510	1,120
Sep	380	550		

Table V.6.2. Conversion of periphyton biomass (mg chl a/m^2) to caloric equivalents.

Date	Site	Firm substrate in photic zone: mg chl a/m^2	Firm substrate in photic zone: kcal/m^2	Average for entire study site: kcal/m^2
Nov	UW	2.6012×10^{2}	3.6313×10^{1}	3.4313×10^{-1}
Dec		2.5924×10^{0}	3.6190×10^{-1}	3.4019×10^{-3}
Jan				
Feb		1.4644×10^{0}	2.0442×10^{-1}	1.9215×10^{-3}
Mar		1.5690×10^{0}	2.1903×10^{-1}	2.0589×10^{-3}
Apr		4.8120×10^{1}	6.7169×10^{0}	6.3138×10^{-2}
May				
Jun		1.5690×10^{0}	2.1903×10^{-1}	2.0589×10^{-3}
Jul		2.1181×10^{1}	2.9569×10^{0}	2.7795×10^{-2}
Aug				
Sep		6.7992×10^{0}	9.4912×10^{-1}	8.9217×10^{-3}
Oct		6.223×10^{0}	8.687×10^{-1}	5.8496×10^{-2}
Mean		3.8849×10^{1}	5.4231×10^{0}	5.4231×10^{-1}
Nov	LW	2.262×10^{2}	3.1578×10^{1}	2.9683×10^{-1}
Dec		4.4819×10^{0}	6.2567×10^{-1}	5.8813×10^{-3}
Jan		1.1506×10^{1}	1.6062×10^{0}	1.510×10^{-2}
Feb		6.7990×10^{-1}	9.4910×10^{-2}	8.9215×10^{-4}
Mar		1.2760×10^{1}	1.7814×10^{0}	1.6745×10^{-2}
Apr		3.660×10^{1}	5.1091×10^{0}	4.8026×10^{-2}
May		2.1180×10^{1}	2.9569×10^{0}	2.7795×10^{-2}
Jun		5.1254×10^{1}	7.1555×10^{0}	6.7262×10^{-2}
Jul		1.9525×10^{1}	2.7257×10^{0}	2.5621×10^{-2}
Aug		3.6611×10^{1}	5.1106×10^{0}	4.8040×10^{-2}
Sep				
Oct		4.062×10^{1}	5.6704×10^{0}	3.8183×10^{-1}
Mean		4.1304×10^{1}	5.7659×10^{0}	5.7659×10^{-1}

were significantly lower than summer populations (Duncans Multiple Range Test, α = .05) and significantly higher than winter and fall populations. Winter and fall concentrations were not significantly different from each other. Total rotifer concentrations were significantly higher in July than in August, and were significantly higher in August than in all remaining months. All remaining months (October 1982-June 1983, and September 1983) were not significantly different from each other.

Total rotifer concentrations showed significant positive correlations with chl *a* and temperature at UW and LW, and negative correlations with dissolved oxygen and discharge. Specific conductance correlated positively at UW only. Other parameters showed no significant correlation (Table V.7.1).

Analysis of the seasonal data by Multivariate Analysis of Variance statistics showed that there was a month effect on species composition at both UW and LW. Correlations of species with physical, chemical, and biological parameters (results of which are grouped by family) are given in Table V.7.2. Species composition clustered by season (Pinkham & Pearson similarity index cluster analysis, [Pinkham and Pearson, 1976]) showed that winter and fall were most similar to each other. Spring and summer were dissimilar to each other and both were dissimilar to fall and winter, summer more so than spring.

Longitudinal

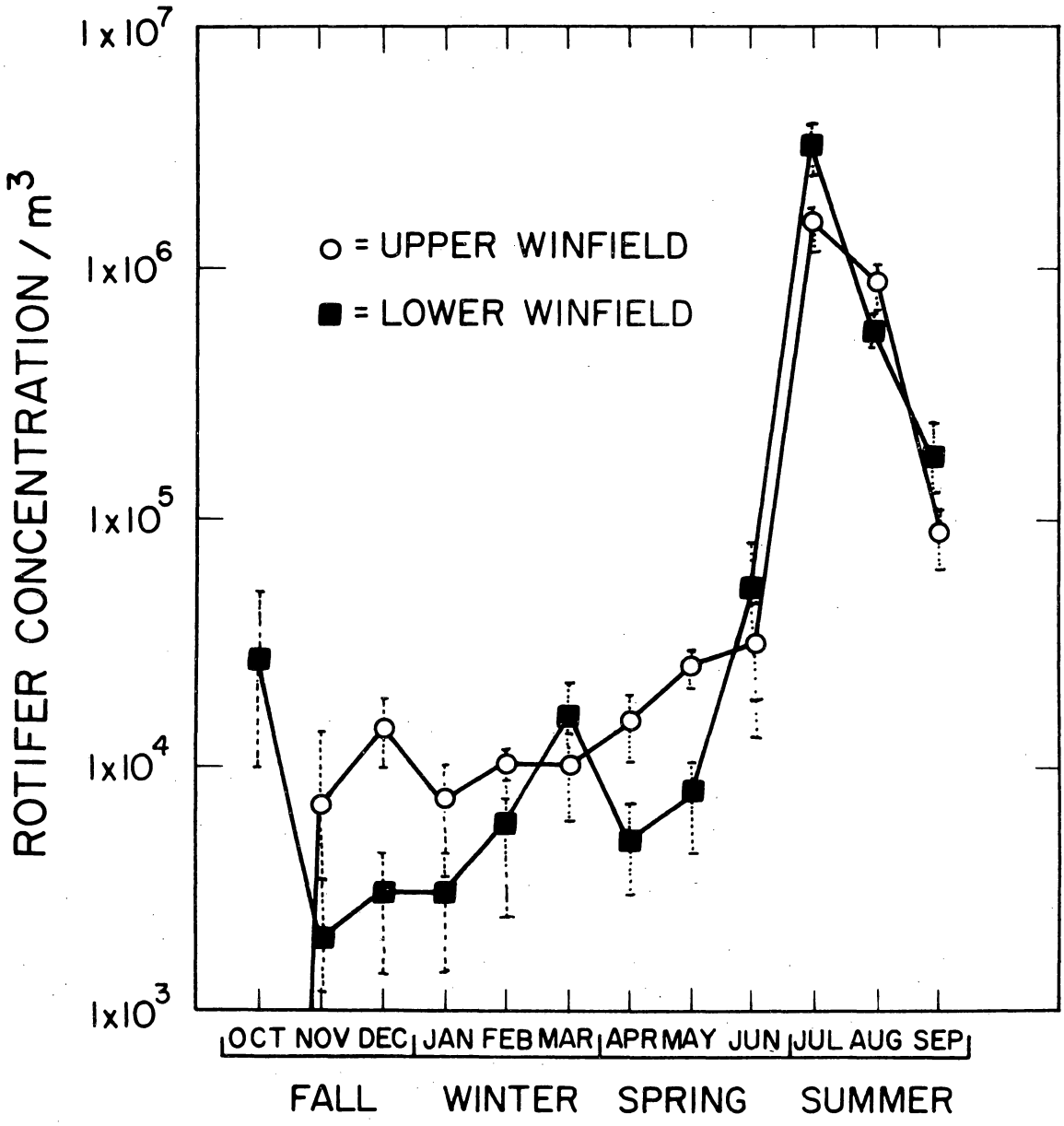

Fig. V.7.1. Mean total rotifer population densities (number/m^3) in Top samples at UW and LW.

Table V.7.1. Total rotifer concentrations from Upper Winfield (UW) and lower Winfield (LW) sampled at a depth of 1 m correlated with physical, chemical, and biological parameters.

Parameter	Station	r^2
Chlorophyll *a*	UW	14.4
Chlorophyll *a*	LW	73.6
Temperature	UW	44.8
Temperature	LW	25.2
Discharge	UW	20.8
Discharge	LW	12.6
Dissolved O_2	UW	38.1
Dissolved O_2	LW	8.4
Conductivity	UW	35.6
Conductivity	LW	4.1*
pH	UW	*
pH	LW	*
Light	UW	*
Light	LW	*

* r^2 not significant at α .05

Table V.7.2. Concentrations of species (grouped by family) correlated with chemical, physical, and biological parameters. The range of r values is given. NS = not significant at α .05.

Parameter	Family (& Station)		
	Synchaetidae Trichocercidae (UW & LW)	Asplanchnidae Brachionidae Hexarthridae (UW & LW)	Dicraniphoridae (UW)
Temperature	.47 to .68	.38 to .49	-.36
Conductivity	.45 to .69[a]	-.37 to .65[b] -.53[c]	NS
Dissolved O^2	-.41 to -.70	.44 to .51	.38
Alkalinity	.78 to .87	.44 to .81	NS
Discharge	-.38 to -.50	NS[d]	NS
Chlorophyll *a*	.50 to .57[e]	NS[f]	NS

[a] Not significant at LW
[b] Values for LW
[c] Values for UW
[d] Exception: *Euchlanis alata*, .53
[e] Not significant at UW
[f] Exception: *Brachionus calyciflorus*, .56

Total rotifer concentrations showed significant differences between stations in longitudinal transect (Table V.7.3). Longitudinal total rotifer concentration at 1 m showed significant positive correlation with pH (mean r = .53 ; α < .05) on all sampling dates. It should be noted that data analyzed for the longitudinal study included the last three sampling dates only; the July 28 samples were collected after a spate which temporarily eliminated the rotifer population. No significant correlation was found between longitudinal total rotifer concentration and chl a sampled at 1 m when all eight stations were analyzed (r = .28). Stations 1-6 were significantly correlated (r = .82) with chlorophyll a.

Total rotifer concentrations at 1 m were significantly higher than at 4 m, with the exception of station 8 where there was no significant difference. Stations 1, 2, and 7 had 8 m samples which were not significantly different from the respective 4 m samples, with the exception of station 2 where the 8 m sample concentration was significantly lower than that of the 4 m sample.

Analysis of the longitudinal data by Multivariate Analysis of Variance statistics showed that there was a station effect on species composition. Correlation of species composition with physical, chemical and biological parameters showed that Branchionidae, Conochilidae and Hexarthridae showed no significant correlation with specific conductance (except

Table V.7.3. Mean total rotifer concentration per liter from 1 m at each station in the longitudinal transect. Concentrations at stations preceded by the same group letter are not significantly different (Duncan's Multiple Range analysis).

Group	Station	no/l
A	5	1634.0
B	4	1244.0
B	3	1242.9
B	2	1203.1
C	1	878.2
C	6	809.3
D	7	437.8
E	8	49.6

Hexarthridae) and significant negative correlations with chl a and dissolved oxygen. Some Branchionidae showed positive correlations with pH. The Synchaetidae showed significant negative correlations with alkalinity and specific conductance, and either no trend or no significant correlation with temperature, vertical light intensity, and chl a. The Trichocerchidae showed no trend or correlation with any parameter measured.

Analysis of the vertical profile data showed that species composition did not change with depth in 96% of the profiles taken.

Effects of Tow Passage

Samples collected behind the towboats Interstate and James K. Ellis had no more damaged rotifers than did samples collected away from the vicinity of towboats.

Autecology of Dominant and Important Species

Fifty-four species of rotifers were found in plankton samples from the Kanawha River. Notes on species comprising > 40% of the rotifer population or > 200 individuals per liter are given below.

Branchionus calyciflorus

Branchionus calyciflorus is a commonly encountered planktonic species of ponds and lakes throughout the world

(Ahlstrom 1940, Gilbert and Starkweather 1977). It is also a common constituent of the plankton in rivers: Sokoto River, Nigeria (Green 1960); Atchafalya River, Louisianna, USA (Holland et al. 1983); Grabia River, Poland (Pawlowski 1973); Ohio River, USA (Ohio River Valley Sanitation Commission 1962); and the Yamuna River, India (Rai 1974). B. calyciflorus was found in the Kanawha River at station 1 (LW) on July 21, 1983 and October 8, 1983 in quantities > 200 individuals per liter, and also at station 2 on August 16, 1983. It is a summer form.

B. calyciflorus is commonly found in alkaline and eutrophic habitats. This species is generally recognized as an indicator of eutrophy (Pawlowski 1973, Sladacek 1983, Stemberger 1979). B. calycilflorus is primarily an herbivore, although it can ingest detritus and bacteria (Gilbert and Starkweather 1977, Pourriot 1977).

Brachionus quadridentatus

Brachionus quadridentatus is a cosmopolitan species, common to ponds and rivers (Ahlstrom 1940). It has been found in the Sokoto River, Nigeria (Green 1960); the Canard River, Canada (Hodgkinson 1970); the Illinois River, USA (Kofoid 1908); the Ohio River, USA (Ohio River Valley Sanitation Commission 1962); and the Yamuna River, India (Rai 1974). B. quadridentatus was found in the Kanawha River at stations 1 and 2 in quantities > 200 individuals per liter on October 8, 1983. It is a summer form (Kofoid 1908). B. quadridentatus is commonly found in

alkaline habitats (Ahlstrom 1940). This species is generally recognized as an indicator of eutrophy (Pawlowski 1973, Sladacek 1983, Stemberger 1979).

Cephalodella gibba

Cephalodella gibba is common to littoral areas and is normally epiphytic or benthic although it occasionally occurs in the plankton (Edmondson 1959, Stemberger 1979). Generally the genus Cephalodella is considered an acid water species (Edmondson 1959).

It has been found in activated sludge plants in Europe (Sladacek 1983), as well as in the Grabia River, Poland (Pawlowski 1973), and in the Yamuna River, India (Rai 1974). C. gibba was found in the Kanawha River at station 1 on April 21, 1983, comprising > 40% of the rotifer population. C. gibba is considered an indicator of eutrophy (Pawlowski 1973, Sladacek 1983). It is an omnivore which feeds on other rotifers, diatoms, and green algae (Stemberger 1979).

Colurella gastrocantha

Colurella gastrocantha is a littoral species, although it occasionally occurs in the plankton (Edmondson 1959). Colurella species have been found in the plankton of the Ocqueoc River, Michigan USA (Beach 1960); the Atchafalaya River, Louisianna (Holland 1977); the Illinois River, USA (Kofoid 1908); and the Grabia River, Poland (Pawlowski 1973). C. gastrocantha was found

in the Kanawha River plankton at UW on March 25, 1983 and at LW on May 20, 1983. Colurella species, although considered littoral often become abundant in the limnetic zone of eutrophic waters (Gannon and Stemberger 1978). Colurella species are often found in eutrophic waters, although they have been recorded in oligotrophic waters (Sladacek 1983). Colurella species feed by scraping up small organisms with their head shield (Edmondson 1959).

Euchlanis sp.

Euchlanis species are littoral rotifers (Stemberger 1979). They have been noted in the Sokota River, Nigeria (Green 1960); the Atchafalaya River, Louisianna, USA (Holland 1977); the Illinois River, USA (Kofoid 1908); and the Yamuna River, India (Rai 1974). Euchlanis species are eutrophic to mesotrophic species (Pawlowski 1973; Sladacek 1983). Euchlanis species may eat green algae, diatoms, or bacteria (Pourriot 1977, Stemberger 1979).

Keratella cochlearis var. cochlearis.

K. cochlearis is cosmopolitan, and is probably the world's most common rotifer (Edmondson 1959). It is perennial and lives in a wide range of conditions: cold Bare Lake in Alaska to temperate South Africa (Hutchinson 1967). It has been found in rivers world-wide: e.g. the Ocqueoc River, Michigan, USA (Beach 1960) to the Motala River, Sweden (Carlin 1943). K. cochlearis

constituted 40% or more of the population in the Kanawha River at LW on October 18, 1982, November 18, 1982, and March 24, 1983, and 40% or more of the population at UW on November 19, 1982, February 18, 1983, and June 22, 1983. It is found in a wide range of trophic conditions (Sladacek 1983). K. cochlearis consumes particles of detritus (with associated bacteria) up to 12 microns in diameter (Edmondson 1964). It can eat larger items such as cryptomonads and chrysomonads (Pourriot 1977).

Lecane n. sp.

Lecane n. sp. had a very limited distribution, known only from the upper reaches of the Kanawha River, particularly UW station. It appeared in small quantities on occasions throughout the year in the Kanawha, however, it is a summer form. Large populations were noted on June 28, 1982 and September 13, 1983. Other Lecane species are known to be microphageous and feed on bacteria and detritus (Stemberger 1979). This Lecane species apparently feeds on bacteria and detritus because its populations did not correlate with chl a.

Lepadella sp.

Lepadella sp. are littoral rotifers, common in hard waters world-wide (Harring 1916). They have been found in river systems: the Ocaueoc River, Michigan, USA (Beach 1960); the Illinois River, USA (Kofoid 1908); the Ohio River, USA (Ohio River Valley Sanitation Commission 1962); the Grabia River, Poland (Pawlowski 1973); and the Yamuna River, India (Rai 1974).

Lepadella species were relatively abundant in the Kanawha River on March 25, 1983.

Polyarthra dolichoptera

Polyarthra dolichoptera is a widely distributed planktonic species. Records of the distribution of this species are confusing and incomplete due to the synonymy of P. platyptera and P. trigla with P. dolichoptera and with P. vulgaris (Bartos 1959). P. dolichoptera is generally considered to be a cold stenotherm (but not a winter form), although it can live in temperatures up to 19° C (Chengalath 1982. unpublished; Hutchinson 1967). It is often found near or in the hypolimnion at low dissolved oxygen levels, apparently excluded from the epilimnion by the competition of other Polyarthra. P. dolichoptera is found in large numbers in the Great Lakes in late spring (Hutchinson 1967). It is found in the Motala River, Sweden, in late spring (Carlin 1943). It was found in the Kanawha River on August 16, 1983.

Polyarthra dolichoptera is found in oligotrophic and eutrophic waters (Sladacek 1983). It is exclusively algae eating (Pourriot 1977).

Polyarthra remata

Polyarthra remata is an epilimnetic species (Hutchinson 1967) and is the smallest Polyarthra species (Stemberger 1979). Much synonomy exists for this Polyarthra species (Bartos 1959).

P. remata is a late summer species (Hutchinson 1967). It occurs only at high summer temperatures in Lake Osbysjon, Sweden (Pejler 1961). It also occurs in river systems, e.g. in the Motala River, Sweden (Carlin 1943), and the Yamuna River, India (Rai 1974). P. remata may prefer oligotrophic lakes (Maemets 1983), but it is found in oligotrophic to eutrophic lakes (Sladecek 1983). It was abundant in the Kanawha River August 25, 1983.

Polyarthra vulgaris

Polyarthra vulgaris is a widely distributed perennial form that inhabits the epilimnion (Carlin 1943, Hutchinson 1967). It usually shows a population pulse in late spring or early summer when the water temperature is 15-20° C (Carlin 1943), and may be a temperature dependent species (Edmondson 1964). Extensive synonymy exists for this species (Bartos 1959).

P. vulgaris has been found in other river systems, e.g. the Atchafalaya River, Louisianna, USA (Holland 1977). It was found in the Kanawha River August 24, 1983 at UW. This species may be a useful indicator of eutrophy (Sladacek 1983).

Polyarthra vulgaris eats algae almost exclusively and probably no algae smaller than 15 microns (Pourriot 1977, Edmondson 1964). P. vulgaris also eats flagelated protozoans (species of the genus Bodo in culture (Buikema et al. 1977). In nature P. vulgaris may feed primarily on cryptomonads (Edmondson 1964).

Synchaeta stylata

Synchaeta stylata occurs during late spring through the fall, usuall with maximum populations in late spring and summer, particularly in August (Stemberger 1979, Carlin 1943). It has been noted in river systems such as the Motala River, Sweden (Carlin 1943), and the Atchafalaya River, Louisianna (Holland 1977). It was found in the Kanawha River at UW on July 20, 1983 and August 24, 1983.

S. stylata is considered an indicator of oligotrophic conditions (Hutchinson 1967). Synchaeta species, however, have been noted in eutrophic systems (Sladecik 1983). S. stylata is a grasping species and is considered by Pourriot (1977) to be an obligate herbivore. However, it should be noted that some species may feed on bacteria in eutrophic systems (Gliwicz 1969, Johansen 1983).

V.8. BENTHIC MACROINVERTEBRATES

Taxonomic Composition

Most taxa could be indentified to the generic level and some could be reliably identified to species. Chironomidae were only identified as either the subfamily Tanypodinae or a combination of two subfamilies, Orthocladiinae and Chironominae. Sixty-one taxa were collected (Table V.8.1; see also Appendix A, Tables A8.1.1), representing 6 aquatic insect orders as well as several

noninsect orders. A greater number of taxa were collected at UW (60) than at LW (38). The organisms which were collected represented various trophic groups including: collector-filterers, collectorgatherers, shredders, and predators.

Total density and diversity of benthic macroinvertebrates are recorded in Appendix A, Tables A8.2.1-24 and A8.3.1-21 and are summarized in Table V.8.2. The artificial substrate samplers, which mimicked the cobble/pebble substrates found in the Winfield Pool, were colonized by a greater density of macroinvertebrates than the sand/silt substrates sampled with the Ponar grab (Table V.8.2). At UW the greatest number of individuals occurred on cobble/pebble substrates in July (79,640.0/m^2) while the lowest occurred in February (236.8/m^2). At LW the greatest number of individuals occurred on cobble/pebble substrates in November (24,261.9/m^2) with the Orthocladiinae/Chironominae group making up most of this total. Low numbers of individuals occurred on cobble/pebble substrates at LW during the winter months, with the lowest colonization occurring in January (195.9/m^2). Patterns of colonization of sand/silt substrates were distinctly different from those on cobble/pebble substrates. Low numbers of individuals were found on sand/silt substrates at both UW and LW during April (42.0/m^2 at both sites), with the number of organisms collected at UW being equally low in May (41.8/m^2). High numbers were found during October at UW (15,291.4/m^2) and during November at LW (2,694.0/m^2).

Table V.8.1. Aquatic macroinvertebrates collected from the Winfield Pool. AAD = average annual density (organisms/m2). * = taxa less than .01% of community.

Organism	UW AAD	UW %	LW AAD	LW %
COBBLE/PEBBLE SUBSTRATES				
Misc. Invertebrates				
Turbellaria	2.5	.01		
Nematoda	8.6	.05	5.6	.07
Oligochaeta	8.4	.04	12.4	.16
Gammarus	.2	*	9.3	.12
Hyalella	4.5	.02	.5	.01
Isopoda	.9	*		
Arachnida	16.5	.09	10.8	.14
Ferrissia	5.2	.03		
Orconectes	.2	*		
Ephemeroptera				
Isonychia bicolor	86.1	.46		
Baetidae	17.9	.09	10.3	.14
Stenacron	260.9	1.38	37.0	.49
Stenonema	358.0	1.90	40.5	.54
Ephemerella	57.0	.30	5.2	.07
Tricorythodes	30.1	.16	83.5	1.11
Caenis	5.6	.03	7.5	.10
Hexagenia	.4	*		
Odonata				
Argia	11.8	.06	11.4	.15
Macromiidae	.7	*	.4	.01
Plecoptera				
Taeniopteryx	8.6	.05	25.4	.34
Strophopteryx	15.1	.08	18.1	.24
Amphinemoura	2.7	.01	.4	.01
Phasganophora	.2	*		
Acroneuria	.5	*		
Paragnetina	.2	*		
Neoperla	1.1	.01		
Isogenus	.5	*		
Isoperla	.2	*		
Megaloptera				
Corydalus	.4	*		

Coleoptera				
Hydrophilidae	.4	*		
Psephenus	.9	*		
Stenelmis	1.6	.01		
Microcylloepus	9.0	.05	.7	.01
Macronychus	1.8	.01		
Dubiraphia	.5	*		
Promoresia	1.4	.01		
Trichoptera				
Chimarra	5.6	.03		
Cyrnellus fraternus	50.8	.27	447.3	5.94
Neureclipsis	4.5	.02	.4	.01
Polycentropus	.2	*		
Cheumatopsyche	886.1	4.69	71.5	.95
Hydropsyche aerata	.5	*	1.6	.02
Hydropsyche morosa	61.8	.33	1.9	.03
Hydropsyche phalerata	15.2	.08	.2	*
Hydropsyche orris	6.1	.03	.7	.01
Potamyia	.5	*		
Rhyacophila	.4	*		
Hydroptila	17.9	.09	6.8	.09
Diptera				
Tipulidae	.2	*		
Antocha	.5	*	.2	*
Ceratopogonidae	.2	*		
Orthoclad/Chironominae	16,675.5	88.34	6,546.8	87.00
Tanypodinae	123.1	.65	160.0	2.13
Prosimulium	.9	*	.2	*
Simulium jenningsi	88.7	.47	6.5	.09
Simulium vittatum	5.7	.03	.9	.01
Atherix	.2	*	.4	.01
Empididae	.5	*	.2	*
Hemerodromia	10.4	.06	.5	.01
Tabanidae	.2	*		
TOTAL	18,876.3		7,525.1	

SAND/SILT SUBSTRATES

Misc. Invertebrates				
Oligochaeta	1.9	.09	5.9	.85
Hyalella	.3	.01		
Arachnidae	1.9	.09	2.4	.35
Ferrissia	.3	.01		
Ephemeroptera				
Stenacron	.3	.01	2.8	.41
Stenonema	1.6	.08		
Tricorythodes	.3	.01		

Hexagenia	1.6	.08	9.0	1.30
Plecoptera				
Strophopteryx			.3	.04
Coleoptera				
Dubiraphia	.3	.01	.3	.04
Trichoptera				
Cyrnellus fraternus			1.0	.14
Cheumatopsyche	2.2	.11		
Diptera				
Chaoboridae			1.9	.27
Ceratopogonidae			.7	.10
Orthoclad/Chironominae	979.0	47.56	247.2	35.77
Tanypodinae	1,068.2	51.89	419.5	60.71
Prosimulium	.6	.03		
TOTAL	2,058.5		691.0	

According to the Dbar index (Lloyd et al. 1968), the greatest diversity was found at UW on cobble/pebble substrates in May (2.2) (Table V.8.2). Cobble/pebble substrates at UW were colonized by a more diverse community throughout most of the year than those at LW (1.2 UW, 0.7 LW). In June, and August, however, the cobble/pebble substrates at LW had more diverse communities. Diversity was lower at UW on the cobble/pebble substrates during the summer months because of the dominance of the Orthocladiinae/Chironominae group. The cobble/pebble substrates were colonized by a much greater diversity of benthic macroinvertebrates than the sand/silt communities. During much of the year diversity in sand/silt communities was very low, often with only one or two taxa being represented at either UW or LW. At UW, from January to June, only members of the Orthocladiinae/Chironominae group were found in the sand/silt community. Annual mean diversity of taxa on sand/silt substrates was slightly greater at LW (0.6) than at UW (0.4). It must be remembered however, that classification of Chironomidae past the subfamily level used in this study could affect the Dbar index value.

Autecology of Major Taxa

Of the 66 taxa collected in the Kanawha River as part of this study, 14 accounted for more than 0.25% of the community. A

Table V.8.2. Summary of community density (number/m2), number of taxa, and diversity index (Dbar) values. Sand/silt samples not taken March UW and LW, and May LW.

Month	Mean Density	Mean # of Taxa	Total # of Taxa	Mean Diversity
UW - COBBLE/PEBBLE				
JAN	266.9	4.6	9	1.7
FEB	236.8	4.8	11	1.0
MAR	1,384.2	8.8	19	.7
APR	2,368.1	15.2	27	2.1
MAY	4,695.2	18.4	33	2.2
JUN	40,937.3	16.2	28	.7
JUL	79,640.0	15.0	25	.8
AUG	46,842.4	14.2	20	.6
SEP	25,680.5	11.6	16	1.0
OCT	16,641.0	10.4	13	1.3
NOV	5,616.6	9.0	13	1.2
DEC	2,234.6	8.8	19	.7
MEAN	18,878.8	11.4	19.4	1.2
LW - COBBLE/PEBBLE				
JAN	195.9	2.2	6	.4
FEB	856.8	3.0	6	.4
MAR	1,696.4	3.2	8	.2
APR	1,509.1	5.4	13	.6
MAY	927.8	8.2	16	1.6
JUN	16,074.8	13.4	18	1.1
JUL	8,891.0	10.0	17	.6
AUG	6,075.2	7.8	11	1.4
SEP	7,278.6	5.6	9	.8
OCT	17,644.2	5.6	11	.8
NOV	24,261.9	6.6	10	.3
DEC	4,929.9	5.0	9	.5
MEAN	7,528.3	6.3	11.2	.7

Table V.8.2. (continued)

Month	Average Density	Mean # of Taxa	Total # of Taxa	Mean Diversity
UW - SAND/SILT				
JAN	2,189.0	1.0	1	.0
FEB	428.6	.8	1	.0
MAR				
APR	42.0	1.0	1	.0
MAY	41.8	.8	1	.0
JUN	141.4	1.0	1	.0
JUL	294.4	2.0	12	.6
AUG	478.4	3.4	7	1.1
SEP	408.0	2.4	3	.9
OCT	15,291.4	3.4	6	.8
NOV	1,908.4	3.4	7	.5
DEC	673.4	1.2	2	.0
MEAN	1,990.6	1.9	3.8	.4
LW - SAND/SILT				
JAN	313.8	1.8	2	.5
FEB	413.0	2.0	2	.8
MAR				
APR	42.0	1.0	1	.0
MAY				
JUN	61.8	.8	1	.0
JUL	784.6	2.0	2	.6
AUG	57.0	1.6	5	.6
SEP	238.4	3.4	6	1.4
OCT	665.8	3.4	6	1.1
NOV	2,694.0	3.0	5	.9
DEC	1,664.4	2.2	5	.3
MEAN	693.5	2.1	3.5	.6

short description of the autecology of these 14 taxa is given below. The autecology of two other taxa for which production estimates were made, Caenis sp. and Argia spp., are also discussed. During this study it was seldom possible to make species determinations of the immature stages that were collected. However, R.F. Kirchner (Huntington District Corps of Engineers, unpublished data) has compiled an extensive species list for the Kanawha River, based upon collections of adults and immatures at each of the locks and dams. The species listed by Kirchner (Huntington District Corps of Engineers, unpublished data) are considered to be the most likely ones to have been collected during this study and are emphasized in the following review.

Isonychia bicolor (Walker)

(Ephemeroptera: Oligoneuriidae)

Members of the mayfly genus Isonychia are found commonly in eastern North America and are found as far south as Honduras (Edmunds et al. 1976). Kondratieff and Voshell (1984) reviewed the genus Isonychia and provided a detailed discussion about the biology of this organism. Isonychia bicolor typically has two generations per year throughout most of its range. This is also the case in the Kanawha River (see Appendix A, Table A8.4.1). The winter generation emerges in late spring, a smaller, more rapidly growing summer generation then follows. Sweeney (1978) showed

that temperature had a great effect on determining the distribution, development and fecundity of this organism. Kondratieff and Voshell (1984), in a study of I. bicolor in a small stream, showed overwintering of various developmental stages, with little growth occurring from December to March. During the summer the increasing water temperatures apparently resulted in faster development and smaller-sized adults. Isonychia bicolor nymphs are strong swimmers and may be found in fast flowing streams and rivers on vegetation, debris, and rock surfaces. The forelegs have long setae which extend like a comb or rake. The nymphs face the current with the forelegs held in front of them, the setae filtering materials from the water. After a time the fore-leg setae are cleaned off with the mouthparts and the collected materials ingested. In the Kanawha River I. bicolor nymphs were found only at UW; LW was apparently too lacustrine to support this organism.

Stenacron sp.

(Ephemeroptera: Heptageniidae)

Stenacron interpunctatum (Say) has been collected by Kirchner (Huntington District Corps of Engineers, unpublished data) from the Winfield Pool and it is thought that the individuals collected as part of this study were of this same species. Found in almost any unpolluted stream east of the Rocky Mountains (Lewis 1974), Stenacron sp. individuals were collected

principally during the late summer and fall months in the Kanawha River. Six subspecies have been reported with Stenacron interpunctatum interpunctatum (Say) common in the Ohio River drainage (Lewis 1974). Members of the genus Stenacron have been reported as passing through as many as 45 instars before emergence and two generations per year may occur (Edmunds et al. 1976). Merritt and Cummins (1984) list them as primarily scrapers, and secondarily collector - gatherers. This organism is dorso-ventrally flattened and clings to hard substrates.

Stenonema spp.

(Ephemeroptera: Heptageniidae)

Two species of Stenonema were collected by Kirchner (Huntington District Corps of Engineers, unpublished data) in the Winfield Pool of the Kanawha River: Stenonema mediopunctatum (McDunnough) and S. terminatum (Walsh). Earlier reports place Stenonema mediopunctatum as being limited to areas near the Great Lakes (Lewis 1974), however, Edmunds et al. 1976) list this species as having North Eastern and Central United States distribution. Stenonema terminatum has been reported as having a Central and South Eastern (U.S.) distribution pattern (Edmunds et al. 1976). This species appears to be much more tolerant of polluted rivers than S. mediopunctatum (Lewis 1974) and is probably the species found in the Winfield Pool in greatest abundance. Stenonema terminatum is found in larger streams and

rivers among coarse gravels (Lewis 1974). Stenonema nymphs, like Stenacron nymphs, are dorso-ventrally flattened and cling tightly to hard substrates. Edmunds et al. (1976) report that members of this genus can be found in swift streams or in quiet areas, or even in lakes where there is some wave action. Although they are awkward swimmers, they can walk quickly in almost any direction. Merritt and Cummins (1984) describe members of the genus Stenonema as scrapers and collector - gatherers.

Ephemerella sp.

(Ephemeroptera: Ephemerellidae)

Kirchner (Huntington District Corps of Engineers, unpublished data) collected Ephemerella needhami (McDunnough) from the Winfield Pool. Edmunds et al. (1976) reported a Central and North Eastern (U.S.) distribution for this species. Ephemerella spp. are classified by Merritt and Cummins (1984) as collector - gatherers. Diet studies by Shapas and Hilsenhoff (1976) support this classification. The nymphs are stocky, and are apparently awkward swimmers.

Baetidae

(Ephemeroptera)

Three genera from the family Baetidae have been collected by Kirchner (Huntington District Corps of Engineers, unpublished data) from the Winfield Pool: Baetis pygmaeous (Hagen), Cloeon

spp., and Pseudocloeon sp.. Edmunds et al. (1976) reported that Baetis pygmaeous has a Central and North Eastern (U.S.) distribution. Life cycles of Baetis spp. range from univoltine in colder streams to multivoltine in warmer waters (Edmunds et al. 1976). Cloeon spp. are commonly found in slower streams or even in some large lakes, and Pseudocloeon have been reported as inhabiting shallow, flowing water in all sizes of streams and rivers. All three of these taxa have streamlined bodies and are good swimmers. They are all classified as collector - gatherers and scrapers (Merritt and Cummins 1984).

Tricorythodes spp.

(Ephemeroptera: Tricorythidae)

Members of the genus Tricorythodes exhibit considerable variation in life history patterns. In some locations a species may be univoltine, in others, bivoltine or even polyvoltine. Edmunds et al. (1976) report two species with North Eastern (U.S.) distribution, T. allectus Needham and T. minutus Traver. Members of this genus are classified as collector - gatherers (Merritt and Cummins 1984). Shapas and Hilsenhoff (1976) found T. minutus guts to contain only fine detritus. Tricorythodes spp. have a rather stocky appearance and are awkward swimmers. Nymphs may be found in a variety of habitats and even in areas with silted substrates (Edmunds et al. 1976).

Caenis spp.

(Ephemeroptera: Caenidae)

Members of the genus Caenis are very similar in general appearance to Tricorythodes spp. Caenis spp. are also classified as collector - gatherers (Merritt and Cummins 1984). Eight of the thirteen species of Caenis have a North Eastern distribution (Edmunds et al. 1976). Edmunds et al. (1976) report that some of the members of this genus have considerable tolerance for pollution and may be found in slow moving water or even at the edges of lakes.

Argia spp.

(Odonata: Coenagrionidae)

Two species of Argia have been collected from the Kanawha River, Argia translata Hagen and A. tibialis (Rambur) (Huntington District Corps of Engineers, unpublished data). Walker (1953) reported that A. tibialis can be found in a variety of habitats from small streams to river sloughs. A. translata is usually collected in larger, slower streams and rivers, although it may be collected in smaller streams (Walker 1953). Argia spp. are predators on other insect larvae (Corbet 1963, Merritt and Cummins 1984).

Taeniopteryx spp.

(Plecoptera: Taeniopterygidae)

Taeniopteryx stoneflies, unlike most of the other taxa found in the Kanawha River, emerge as adults during the winter. The eggs hatch shortly after oviposition then the early instar nymphs diapause in the substrates during the warm summer months. In the Kanawha River, nymphs develop to their maximum size in the early winter months and were only collected during November and December. Kirchner collected two species from the Winfield Pool: Taeniopteryx burksi Ricker and Ross and T. parvula Banks (Huntington District Corps of Engineers, unpublished data). Fullington and Stewart (1980) report that T. burksi is the most common species of Taeniopteryx in North America. The nymphs are generally classified as being shredders (Merritt and Cummins 1984).

Strophopteryx spp.

(Plecoptera: Taeniopterygidae)

Two species, Strophopteryx fasciata (Burmeister) and S. appalachia Ricker and Ross, have been collected by Kirchner (Huntington District Corps of Engineers, unpublished data) in the Winfield Pool. S. fasciata is commonly collected in a wide range of stream sizes (Hitchcock 1974). Nymphs are shredders, feeding on decaying leaves (Hitchcock 1974). Harper and Hynes (1970) reported that eggs hatch immediately after being laid, and that the nymphs, like those of Taeniopteryx spend the summer months in diapause. Small nymphs began to appear in samples

taken as part of this research in November, with larger nymphs appearing in January and being present through March.

Cyrnellus fraternus Banks
(Trichoptera: Polycentropodidae)

Cyrnellus fraternus has a wide distribution, from Minnesota to Texas and eastward with some individuals having been collected as far south as the Amazon River (Wiggins 1977). Larvae are common in large rivers and may also be found in lakes and reservoirs. Larvae are only lightly sclerotized and may grow up to 9mm in length. This organism builds a small retreat by covering a depression with a roof of silk and leaving a front and rear exit. The silk quickly accumulates a layer of fine organic particles. The retreat is large enough for the organism to be able to turn around. Guts of C. fraternus commonly contain fine organic particles and rarely some arthropod remains. Merritt and Cummins (1984) reported C. fraternus as being a collector - filterer. Wiggins (1977) reported arthropod remains in one out of the three individuals examined. In this study, arthropod remains were found in only one of 16 individuals examined while the others contained only detritus. C. fraternus is probably omnivorous in its feeding behavior and is classified in this study as a collector-gatherer.

Cheumatopsyche spp.
(Trichoptera: Hydropsychidae)

A large and common genus, approximately 39 species of Cheumatopsyche are found in North America. Unfortunately larval characters to aid in the separation of the various species have been almost impossible to find (Wiggins 1977). Most Cheumatopsyche spp. have a median notch on the anterior margin of the frontoclypeal apotome. Cheumatopsyche spp. without this notch were extremely rare in the Kanawha River. Ross (1959) reports that Cheumatopsyche spp. are more dominant in warmer and more polluted streams than Hydropsyche spp. Cheumatopsyche larvae construct silk retreats on solid surfaces in flowing waters. These retreats include nets which are used to filter suspended materials. Larvae feed opportunistically on whatever is on the net. Cheumatopsyche spp. generally have two generations per year, a long overwintering generation and a much shorter summer generation. Individuals from these generations may differ in size (Mackay 1985).

Hydropsyche morosa Hagen
(Trichoptera: Hydropsychidae)

Hydropsyche morosa (Symphitopsyche morosa (Hagen) in Schuster and Etnier 1978) and the closely related species H. cheilonis (Ross) and H. bifida (Banks) can be found in medium-sized rivers in the Appalachian region. H. morosa is a net-spinning collector - filterer like Cheumatopsyche spp. and is reported to be widespread from Maine to Wisconsin and as far

south as Tennessee (Schuster and Etnier 1978). Other Hydropsyche species such as H. orris have been shown to have a bivoltine life cycle (Beckett 1982) in large rivers, and this is apparently the case with H. morosa in the Kanawha River.

Chironomidae

(Diptera)

Chironomids may make up a significant portion of benthic macroinvertebrate communities in large rivers. Because of the difficulty in identifying these organisms and their high densities, frequently identifications are left at the family or subfamily level. A summary of the autecology of this family is given in Brigham et al. (1982). Eggs may be attached to vegetation or allowed to sink to the stream bottom. The eggs are frequently laid in a gelatinous matrix and hatch within a few days or weeks. Larvae pass through four instars before entering a free-living pupal stage. The adults emerge during early evening and although they do not bite, may create a nuisance by their abundance. Most subfamilies of Chironomidae live in small silk tubes or cases. Orthocladiinae are generally found in cooler lakes and streams, while Chironominae inhabit quiet waters in large rivers or lakes (Brigham et al. 1982). These subfamilies are classified as collector - gatherers, although some genera may also be scrapers (Merritt and Cummins 1984). The larvae of the subfamily Tanypodinae are predaceous, primarily consuming

oligochaetes and other chironomids as well as other small invertebrates (Roback 1969). Tanypodinae are generally free living, moving over the substrate in search of prey (Brigham et al. 1982).

Simulium jenningsi Malloch
(Diptera: Simuliidae)

The biology of black flies was reviewed by Voshell (1983). Larvae generally pass through seven instars before pupation. In the New River at Hinton, West Virginia, Voshell (1983) found that S. jenningsi had five generations each year. The larvae secrete a small pad of silken material on a hard substrate and then attach themselves to it using a posterior circle of hooks. The larvae have two cephalic fans which they place in the current to filter material from the water. At intervals the larvae clean these fans with their mouthparts and ingest the collected materials.

Community Structure and Function

The structure and function of the Winfield Pool benthic macroinvertebrate community are recorded in Appendix A, Tables A8.2.1-24 and A8.3.1-21 and are summarized in Tables V.8.3 and V.8.4. Diptera dominated community structure on cobble/pebble

substrates at both sites (Table V.8.3). At UW, on cobble/pebble substrates, Diptera made up 84.4% of the community, while at LW the percentage was slightly higher, 89.0%. On the cobble/pebble substrates the Orthocladiinae/Chironominae group made up most of the Diptera in the community (78.7% UW, 86.3% LW). Other important groups on cobble/pebble included Trichoptera (5.2% UW, 5.5% LW), Ephemeroptera (5.2% UW, 3.2% LW), and Plecoptera (3.5% UW, 0.9% LW). Trichoptera on cobble/pebble substrates were represented primarily by Cheumatopsyche spp. during the summer, and Cyrnellus fraternus during the fall. Ephemeroptera were represented on cobble/pebble substrates at UW by high numbers of Stenonema spp., Stenacron spp., and Isonychia bicolor. At LW, Stenonema spp. and Stenacron spp. were also found on cobble/pebble substrates, but the mayflies found in highest abundance were Tricorythodes spp. Isonychia bicolor was only found at UW. Two winter stonefly genera, Strophopteryx and Taeniopteryx, made up most of the Plecoptera component of the cobble/pebble communities, both at UW and LW. Diptera also dominated community structure on sand/silt substrates at both sites, however, the dominance of the Orthocladiinae/Chironominae group was not so great as on the cobble/pebble substrates. The Orthocladiinae/Chironominae and Tanypodinae subfamilies were present in nearly equal parts (47.6 and 51.9%, respectively) at UW.

Table V.8.3. Summary of community makeup (%) according to higher taxonomic groups. * = percentage is less than 0.1. Sand/silt samples not taken March UW and LW and May.

Month	Misc.	Ephem.	Odon.	Plecop.	Coleop.	Tricop.	Dipt.
UW - COBBLE/PEBBLE							
JAN	2.4	2.4	-	30.6	-	.8	63.7
FEB	.6	.9	-	2.7	-	2.7	90.9
MAR	2.6	.8	-	.8	.8	.8	94.2
APR	1.8	8.2	-	1.0	3.0	5.8	80.2
MAY	1.1	14.2	-	.6	1.7	6.6	75.8
JUN	.3	3.6	*	*	*	4.3	91.8
JUL	*	2.7	*	-	-	3.7	93.5
AUG	.2	2.0	*	-	*	6.2	91.5
SEP	.3	6.0	.1	-	-	8.3	85.3
OCT	.2	14.4	.5	-	-	10.1	74.7
NOV	.5	6.7	*	1.6	*	11.1	79.9
DEC	1.5	.8	.1	4.3	.1	1.9	91.2
MEAN	1.0	5.2	.1	3.5	.5	5.2	84.4
LW - COBBLE/PEBBLE							
JAN	.6	-	-	1.1	-	-	95.6
FEB	4.0	-	-	1.0	-	-	95.0
MAR	1.8	.4	-	-	-	.8	97.1
APR	1.7	1.0	-	.3	.3	.9	95.9
MAY	1.6	20.0	-	-	.5	5.1	72.9
JUN	1.3	8.7	.1	-	-	5.4	84.5
JUL	.2	2.1	.2	-	-	2.8	94.6
AUG	5.6	6.0	.2	-	-	20.9	71.9
SEP	.2	.3	.3	-	-	13.7	85.5
OCT	*	.1	.3	-	-	15.0	84.5
NOV	.1	.1	.1	.6	-	1.1	98.0
DEC	.1	*	-	7.3	-	.3	92.3
MEAN	1.4	3.2	.1	.9	.1	5.5	89.0

Table V.8.3. (continued)

Month	Misc.	Ephem.	Odon.	Plecop.	Coleop.	Tricop.	Dipt.
UW - SAND/SILT							
JAN	-	-	-	-	-	-	100.0
FEB	-	-	-	-	-	-	100.0
MAR							
APR	-	-	-	-	-	-	100.0
MAY	-	-	-	-	-	-	100.0
JUN	-	-	-	-	-	-	100.0
JUL	-	-	-	-	-	-	100.0
AUG	.8	.4	-	-	-	1.6	93.6
SEP	4.7	-	-	-	-	-	95.3
OCT	.1	*	-	-	*	*	99.8
NOV	.5	.4	-	-	-	.8	97.3
DEC	-	-	-	-	-	-	100.0
MEAN	.6	.1	.0	.0	.0	.2	98.7
LW - SAND/SILT							
JAN	-	-	-	-	-	-	100.0
FEB	-	-	-	-	-	-	100.0
MAR							
APR	-	-	-	-	-	-	100.0
MAY							
JUN	-	-	-	-	-	-	100.0
JUL	-	-	-	-	-	-	100.0
AUG	26.7	-	-	-	-	-	73.3
SEP	20.8	1.6	-	-	-	1.6	76.0
OCT	2.3	9.2	-	-	.6	-	88.0
NOV	-	2.4	-	-	-	.3	97.3
DEC	.7	-	-	.2	-	-	99.1
MEAN	5.1	1.3	.0	.0	.1	.2	93.4

Table V.8.4. Summary of community function according to feeding functional groups (%). * = percentage less than 0.1. Sand/silt samples not taken March UW and LW, sand May LW.

Month	Collector Gatherers	Collector Filterers	Shredders	Scrapers	Predators
UW - COBBLE/PEBBLE					
JAN	48.5	16.6	32.8	1.1	1.1
FEB	85.3	12.0	2.1	.6	-
MAR	94.8	2.5	.6	1.1	1.0
APR	61.4	32.2	.7	2.6	3.0
MAY	71.2	22.4	.4	2.9	3.1
JUN	94.9	4.1	-	.5	.6
JUL	95.5	4.2	-	-	.3
AUG	92.2	6.4	-	.1	1.3
SEP	91.3	7.5	-	*	1.1
OCT	88.8	9.5	-	-	1.7
NOV	85.0	11.8	1.7	.1	1.4
DEC	91.5	2.3	4.7	.1	1.5
MEAN	83.4	11.0	3.6	.8	1.3
LW - COBBLE/PEBBLE					
JAN	96.3	.6	.6	-	2.5
FEB	98.2	.5	11.0	-	.1
MAR	98.9	1.1	-	-	-
APR	93.8	5.3	.2	.5	.1
MAY	88.6	5.3	-	.4	5.7
JUN	91.6	5.5	-	.5	2.5
JUL	97.2	.4	-	*	2.4
AUG	91.0	.1	-	-	5.6
SEP	96.4	.2	-	-	3.4
OCT	96.7	-	-	-	3.3
NOV	97.8	-	.8	-	1.4
DEC	91.9	.2	7.3	-	.6
MEAN	94.9	1.6	1.7	.1	2.3

Table V.8.4. (continued)

Month	Collector Gatherers	Collector Filterers	Shredders	Scrapers	Predators
UW - SAND/SILT					
JAN	100.0	-	-	-	-
FEB	100.0	-	-	-	-
MAR		-	-	-	-
APR	100.0	-	-	-	-
MAY	100.0	-	-	-	-
JUN	100.0	-	-	-	-
JUL	87.0	-	-	-	13.0
AUG	76.9	1.9	-	-	21.2
SEP	60.4	-	-	-	39.6
OCT	29.4	*	-	*	70.5
NOV	88.9	.7	-	.2	10.3
DEC	99.4	.6	-	-	-
MEAN	85.6	.3	.0	.0	14.1
LW - SAND/SILT					
JAN	43.9	-	-	-	56.1
FEB	79.8	-	-	-	50.2
MAR		-	-	-	-
APR	100.0	-	-	-	-
MAY		-	-	-	-
JUN	100.0	-	-	-	-
JUL	16.9	-	-	-	83.1
AUG	21.1	-	-	-	78.9
SEP	48.2	-	-	-	51.8
OCT	32.0	-	-	.5	67.5
NOV	23.0	-	-	-	77.0
DEC	93.7	-	.4	-	5.9
MEAN	55.9	.0	.0	.1	47.1

There were some variations in community structure over time, but Diptera made up the greatest part of the community on each sampling date. On cobble/pebble substrates Plecoptera only accounted for more than 10% of the community in January at UW, but were an important part of the community at both UW and LW in December. During the summer, no Plecoptera were collected at either site. Trichoptera accounted for over 10% of the community only during October and November at UW and August, September, and October at LW on cobble/pebble substrates. During other months, Trichoptera were consistently present, but in less abundance in comparison to other groups. Ephemeroptera were also found as part of most cobble/pebble communities throughout the year. Ephemeroptera formed a major part of the cobble/pebble communities at both UW and LW in May (14.2% UW, 20.0% LW) and were also a major part of the UW community in October (14.4%). In the sand/silt communities, Diptera dominated throughout the year. Only in August and September at LW did another taxon, Oligochaeta, make up a major portion (26.7% August, 20.8% September) of the sand/silt community.

Community composition, according to trophic functional groups, is summarized in Table V.8.4. On cobble/pebble substrates, collectorgatherers were the dominant trophic group (83.4% UW, 94.9% LW). The taxa making up the largest percentage of the collector - gatherers were members of the Orthocladiinae/Chironominae group. Collector-filterers were

present in high numbers during the summer months, however, because Orthocladiinae/Chironominae numbers were so high during that part of the year, collector - filterers did not make up a very large percentage of the cobble/pebble community. Collector - filterers were found in high numbers during April and May at UW in comparison to other trophic groups. Shredders were important in the cobble/pebble community only during January at UW, although they were present in lesser numbers during the late fall and winter months. Scrapers (0.8% UW, 0.1% LW) and predators (1.3% UW, 2.3% LW) did not make up a large part of the cobble/pebble community.

In the sand/silt community only two trophic functional groups were important: collector-gatherers and predators. Collector - gatherers, which were mainly members of the Orthocladiinae/Chironominae group, were dominant at both sites but less dominant at LW (55.9%) than at UW (85.6%). Predators, belonging to the Tanypodinae subfamily, made up virtually all of the rest of the sand/silt community at each site (14.1% UW, 47.1% LW).

Production

Sixteen taxa occurred in sufficient numbers to estimate production by the size frequency method. Size-frequency distribution tables for each of the 16 taxa on each sampling date were compiled so that life history patterns could be determined

(Appendix A, Tables A8.4.1-34). An important part of the size frequency method is the accurate estimation of the cohort production interval or CPI. Where possible, CPI values were calculated from life history information from the Kanawha River. The CPI correction factors that were used for calculating production are recorded in Table V.8.5. In the case of Chironomidae, regression equations from Mackey (1977) were used in conjunction with the Winfield Pool mean annual temperature of 15°C to estimate a CPI of about 24.3 days. CPI's derived for the Kanawha River were compared with values from other production studies and found not to be exceptional. The dry weights for the size classes of all taxa are recorded in Appendix A, Table A8.5.

Production estimates are summarized in Table V.8.5. Total production of all taxa on cobble/pebble substrates was 43,838.4 mgDW/m^2/yr at UW and 16,552.7 mgDW/m^2/yr at LW. Production on sand/silt substrates was considerably lower 3,534.4 mgDW/m^2/yr at UW and 2,405.4 mgDW/m^2/yr at LW. The highest production for any taxon was 32,833 mgDW/m^2/yr for the Orthocladiinae/Chironominae group on cobble/pebble substrates at UW. The Orthocladiinae/Chironominae group was also the highest producer at LW, 13,582.5 mgDW/m^2/yr on cobble/pebble substrates. The lowest production value was for Caenis sp. at LW (2.5 mgDW/m^2/yr) on cobble/pebble substrates.

Table V.8.5. Summary of annual production by major aquatic insects in the Winfield Pool of the Kanawha River.

Organism - Location	Annual Production $mg/m^2/yr$	Average Biomass mg/m^2	Annual P/B Ratio	CPI correction factor
COBBLE/PEBBLE				
Isonychia bicolor-UW	1,646.9	127.4	12.9	2.4
Baetidae-UW	50.2	5.4	9.3	2.4
Baetidae-LW	20.4	2.3	8.9	2.4
Stenacron-UW	1,009.9	75.2	13.4	2.4
Stenacron-LW	186.7	15.3	12.2	2.4
Stenonema-UW	1,574.2	123.4	12.8	2.4
Stenonema-LW	702.7	49.5	14.2	2.4
Ephemerella-UW	335.5	46.4	7.2	2.4
Ephemerella-LW	83.8	7.0	12.0	2.4
Tricorythodes-UW	117.4	9.4	12.5	2.4
Tricorythodes-LW	162.0	11.5	14.1	2.4
Caenis-UW	6.9	1.4	4.9	1.1
Caenis-LW	2.5	.4	6.3	1.1
Argia-UW	200.0	42.9	4.7	1.0
Argia-LW	195.6	46.6	4.2	1.0
Taeniopteryx-UW	28.0	5.8	4.8	1.1
Taeniopteryx-LW	89.9	14.2	6.3	1.1
Strophopteryx-UW	112.0	15.0	7.5	1.1
Strophopteryx-LW	35.3	4.9	7.2	1.1
Cyrnellus fraternus-UW	62.4	6.6	9.5	2.4
Cyrnellus fraternus-LW	533.5	53.0	10.1	2.4
Cheumatopsyche-UW	4,519.7	567.8	8.0	2.3
Cheumatopsyche-LW	419.1	50.0	8.4	2.3
Hydropsyche morosa-UW	836.3	71.9	11.6	2.3
Hydropsyche morosa-LW	12.2	1.6	7.6	2.3
Orthocladiinae/ Chironominae-UW	32,833.5	482.1	68.1	15.0
Orthocladiinae/ Chironominae-LW	13,582.5	264.1	51.4	15.0
Tanypodinae-UW	396.0	8.6	46.0	15.0
Tanypodinae-LW	526.5	9.8	53.7	15.0
Simulium jenningsi-UW	109.5	11.0	10.0	2.5

Table V.8.5. (continued)

SAND/SILT

Orthocladiinae/ Chironominae-UW	2,198.7	49.0	44.9	15.0
Orthocladiinae/ Chironominae-LW	826.7	20.3	40.7	15.0
Tanypodinae-UW	1,335.7	19.9	67.1	15.0
Tanypodinae-LW	1,578.7	29.5	53.5	15.0

Of the total production by all taxa at UW on cobble/pebble substrates, Diptera accounted for 76.0%, Trichoptera for 12.4%, and Ephemeroptera for 10.8%, with the remaining 0.8% being accounted for by Plecoptera and Odonata. At LW, Diptera also accounted for most of the production (85.2%), with Trichoptera accounting for 5.8%, Ephemeroptera for 7.0%, Plecoptera for 0.8%, and Odonata for 1.2%. On sand/silt substrates virtually all production was accounted for by Diptera.

Production was divided unequally among trophic groups. Collector - gatherers accounted for 82.1% (35,990 mgDW/m^2/yr) at UW and 92.3% (15,274.1 mgDW/m^2/yr) at LW on cobble/pebble substrates. Collector-filterers were the next highest producers on cobble/pebble substrates, accounting for 16.2% of the production at UW (7,112.4 mgDW/mW/yr) but only 2.6% (431.3 mgDW/m^2/yr) at LW. Shredders, scrapers, and predators made up a minor part of the total production on cobble/pebble substrates at both sites. Sand/silt substrate community production at UW was divided between the collector-gatherers (62.2%) and predators (37.8%). At LW, however, predators (Tanypodinae) produced more than the collector-gatherers (Orthocladiinae/Chironominae) (65.6% versus 34.4%) on the sand/silt substrates.

In order to extrapolate the production for the entire Winfield Pool, it was necessary to estimate the extent of the riverine environment (similar to UW) and the lacustrine

environment (similar to LW) as well as the proportions of hard substrate (cobble/pebble) and soft substrate (sand/silt) in each environment. On the basis of a substrate survey (Huntington District Corps of Engineers, unpublished data) and field observations, it was estimated that approximately 30% of the pool length could be classified as "UW-like" and the remaining 70% "LW-like". In the UW section it was estimated that 28% of the river bottom was cobble/pebble, and the remainder consisted of sand/silt. It was estimated that only about 1% of the river bottom in the LW section was cobble/pebble. This 1% was located in a narrow strip on either shore. The remaining 99% of the river bottom in the LW section was classified as sand/silt substrate.

Total pool production was estimated to be 78,143 kgDW/yr, based on a total pool surface area of 1255 ha. Production in a hypothetical square meter section of pool bottom, made up of each of the substrate types in their appropriate percentages, was estimated to be 6,228.7 mgDW/m^2/yr. Approximately 71.4% of the annual production came from the UW section, with 59.1% being produced on cobble/pebble substrate in that section. Cobble/pebble substrate in the LW section was found to be rather unimportant in terms of overall pool production (1.9%). Sand/silt substrate accounted for about 70% of the substrate available in the pool but only produced 26.8% of the total pool production.

Trophic Basis of Production

Gut contents were analyzed in order to determine the feeding habits of the dominant taxa (Table V.8.6). Ephemerella spp., Stenonema spp., and Isonychia bicolor ingested only fine detritus. Taeniopteryx spp. and Strophopteryx spp. ingested mostly vascular plant detritus. Animal material was found in the gut of one Cyrnellus fraternus fifth instar larva along with detritus, however, most larvae (15/16) contained only detritus. For the purposes of this study, we have classified Cyrnellus fraternus as a collector-gatherer. Cheumatopsyche spp. larvae were found to contain algae and detritus, with algae only comprising a small percentage of the total gut contents. No animal material was found in the guts of Cheumatopsyche spp. The diet of Hydropsyche morosa was similar to that of Cheumatopsyche spp. Argia spp. guts contained predominantly animal material with small amounts of diatoms, other types of algae, and detritus.

In order to analyze the trophic basis of production, for the benthic macroinvertebrate community, it was necessary to estimate the feeding habits for all 16 taxa for which production was estimated (Table V.8.7). In cases where sufficient specimens were not obtained from the Kanawha River for gut analysis, feeding habits were estimated from published information. Literature values were also used to supplement data from the Kanawha River in cases where observed values differed greatly from published

Table V.8.6. Summary of food habits (%) of selected Kanawha River taxa. AN = animal matter; DI = diatoms; AL = other types of algae; DE = detritus; VP = vascular plant material. Numbers in parenthesis indicate number of guts with material which were analyzed.

Organism	AN	DI	AL	DE	VP
Ephemeroptera					
Isonychia (8)	-	-	-	100.0	-
Stenacron (9)	-	-	-	100.0	-
Stenonema (10)	-	-	-	100.0	-
Ephemerella (3)	-	-	-	100.0	-
Tricorythodes (2)	-	-	-	100.0	-
Odonata					
Argia (2)	77.3	1.3	0.1	21.2	-
Plecoptera					
Taeniopteryx (7)	-	-	-	87.0	13.0
Strophopteryx (4)	-	-	-	85.0	15.0
Trichoptera					
Cyrnellus (16)	4.4	-	0.1	95.5	-
Cheumatopsyche (26)	-	-	2.4	97.6	-

values. Most of the organisms that were used for diet analysis were obtained from the smaller PVC pipe artificial substrates. In retrospect, these artificial substrates, while easy to use, perhaps did not allow the colonizing organisms to obtain the full array of diet items that they would be able to ingest in the natural environment.

Diet analysis for Ephemeroptera were adjusted according to the results of Shapas and Hilsenhoff (1976) who investigated the diets of several of the same or closely related species. It was estimated that 5% of the diet of Isonychia bicolor was made up of diatoms, with the rest being fine detritus. The diet of Stenonema mediopunctatum, as reported by Shapas and Hilsenhoff (1976), was used for both Stenonema spp. and Stenacron sp. in the Kanawha River. Since Stenacron sp. is very similar to Stenonema spp. in its biology, their diets could be assumed to be the same. This same assumption was also made in the case of Caenis sp. and Tricorythodes sp. where both were considered to feed only on detritus. It was assumed that the detritus, diatoms, and other algae which were found in the guts of Argia spp. came from the guts of ingested prey. Therefore the production of Argia spp. was assumed to be based entirely on animal food (Corbet 1963). This assumption was also made for Tanypodinae larvae (Roback 1969). The diet of Cyrnellus fraternus was estimated to contain 5% animal material, slightly higher than that found in the Kanawha specimens, because some predation was reported by Wiggins (1977).

Table V.8.7. Summary of estimated feeding habits (%) of selected Kanawha River taxa. Percentages were estimated from analyses of Kanawha River specimens with modifications based on literature values. AN = animal matter; DI = diatoms; AL = other types of algae; DE = detritus; VP = vascular plant material.

Organism	AN	DI	AL	DE	VP
Ephemeroptera					
Isonychia	-	5	-	95	-
Baetidae	-	25	-	75	-
Stenacron	-	10	-	90	-
Stenonema	-	10	-	90	-
Ephemerella	-	15	-	85	-
Tricorythodes	-	-	-	100	-
Caenis	-	-	-	100	-
Odonata					
Argia	100	-	-	-	-
Plecoptera					
Taeniopteryx	-	-	-	-	100
Strophopteryx	-	-	-	-	100
Trichoptera					
Cyrnellus	5	-	-	95	-
Cheumatopsyche	1	5	2	92	-
Hydropsyche	8	4	1	87	-
Diptera					
Orthocladiinae/ Chironominae	-	5	5	90	-
Tanypodinae	100	-	-	-	-
Simulium	-	2	1	97	-

Cyrnellus fraternus ingested small amounts of algae (0.1%), but this was considered insignificant in supporting production. Diet analyses for Cheumatopsyche spp. were adjusted to include 5% diatoms, according to the results of Parker (1980) for Cheumatopsyche "notched". Cheumatopsyche spp. usually eat significant amounts of diatoms and the absence of diatoms from the guts of specimens from the Kanawha River was probably an artifact of the analytical procedures. The results of the diet analyses for Hydropsyche morosa were adjusted to the mean values for several species of Hydropsyche reported by Parker (1980). Several studies have shown the importance of animal materials in the diet of Hydropsyche species (Parker and Voshell 1983, Shapas and Hilsenhoff 1976), and so it was likely that animal material made up at least 8% of the Hydropsyche diet in the Kanawha River. The estimates for the diets of the Orthocladiinae/Chironominae subfamily and Simulium jenningsi were obtained from previous studies on the New River (Voshell 1985). In all cases the estimates for feeding habits were considered to be conservative, i.e., well within the previously reported ranges for consumption.

Assimilation and production efficiencies were obtained from the literature. Assimilation efficiencies (assimilation/ingestion) vary according to the caloric content and digestibility of the ingested material. Benke and Wallace (1980), in a study of filter-feeding caddisflies, estimated assimilation efficiency values of 70% for animal material, 30% for diatoms and

other algae, and 10% for detritus and vascular plant materials. A review of other literature indicated that these values were also acceptable for the other organisms in the Kanawha River (McDiffett 1970, Lawton 1971, McCullough 1975, Johannsson 1980, Giguere 1981). Net production efficiencies (NPE) (production/assimilation) were estimated to be 50%, based on studies by Benke and Wallace (1980), and Edington and Hildrew (1973), Otto (1974), and Sweeney and Vannote (1981).

The proportion of production attributable to each food type was determined for the sixteen taxa for which production was estimated and the results are summarized in Table V.8.8. In an average square meter of river bottom, 57.7% of production could be attributed to detritus consumption, with 23.8% of production attributed to animal material, 10.1% to diatom consumption, 8.4% to other types of algae, and only 0.2% to the consumption of vascular plant material. At UW, a greater amount of production could be attributed to the consumption of detritus and primary producer materials with 69.3% attributed to detritus consumption, and 22.2% to all types of algae. On cobble/pebble substrates, the trophic basis of production was very similar at UW and LW. On sand/silt substrates, however, production was based more on consumption of animal materials at LW than at UW. The amount of production attributed to consumption of animal material was less than half of the pool average, 8.7%. At LW, more than half (61.7%) of the production could be attributed to the consumption

of animal materials. The production attributed to detritus was only 28.8%, with production attributed to all types of algae only 9.1%.

From the estimates for production and feeding habits it was also possible to calculate the amount of each food type that would have to be consumed to account for production. There was over eleven times as much material consumed at UW (231,291 mgDW/m^2/yr) than at LW (20,766 mgDW/m^2/yr). The ratio of food consumed to production was higher at UW (15.6) than LW (8.2), indicating lower community assimilation efficiency at UW. The trophic group with the greatest consumption at both sites was the collector - gatherers (88.8% UW, 70.5% LW). On sand/silt substrates total consumption was more than twice as great at UW (40,461.4 mgDW/m^2/yr) than at LW (18,288.9 mgDW/m^2/yr) while production was only 1.5 times as great. This reflects the fact that predators (Tanypodinae) make up a greater percentage of the community at LW than UW. Animal materials only make up 9.4% of the material consumed at UW on sand/silt substates, while at LW they make up 24.7%. Since animal materials are more readily assimilated, not as much animal material needs to be consumed to produce an equivalent amount of production.

Effects of Tow Passage

Table V.8.8. Amounts (mg/m2/yr) and proportions (%) of annual production attributed to various food types and consumption necessary to account for production. AN = animal matter; DI = diatoms; AL = other types of algae; DE = detritus; VP = vascular plant material.

	Prod.	Animal	%	Diatom	%	OAlgae	%	Detritus	%	VPlant	%
COBBLE/PEBBLE - UW											
COLLECTOR FILTERERS											
Isonychia bicolor	1,646.9	.0	.0	224.6	13.6	.0	.0	1,422.3	86.4	.0	.0
Cheumatopsyche	4,519.7	263.6	5.8	565.0	12.5	226.0	5.0	3,465.1	76.7	.0	.0
Hydropsyche morosa	836.3	296.4	35.4	63.5	7.6	15.9	1.9	460.5	55.1	.0	.0
Simulium	109.5	.0	.0	6.2	5.7	3.1	2.8	100.2	91.5	.0	.0
Total	7,112.4	533.0	7.5	859.3	12.1	245.0	3.4	5,448.1	76.6	.0	.0
Consumption	117,924.3	1,600.2	1.4	5,728.3	4.9	1,633.2	1.4	108,962.6	92.4	.0	.0
COLLECTOR GATHERERS											
Baetidae	50.2	.0	.0	25.1	50.0	.0	.0	25.1	50.0	.0	.0
Stenacron	1,009.9	.0	.0	252.5	25.0	.0	.0	757.4	75.0	.0	.0
Stenonema	1,574.2	.0	.0	393.6	25.0	.0	.0	1,180.7	75.0	.0	.0
Ephemerella	335.5	.0	.0	116.1	34.6	.0	.0	219.4	64.4	.0	.0
Tricorythodes	117.4	.0	.0	.0	.0	.0	.0	117.4	100.0	.0	.0
Caenis	6.9	.0	.0	.0	.0	.0	.0	6.9	100.0	.0	.0
Cyrnellus fraternus	62.4	16.7	26.9	.0	.0	.0	.0	45.6	73.1	.0	.0
Orthocladiinae/ Chironominae	32,833.5	.0	.0	4,104.2	12.5	4,104.2	12.5	24,625.1	75.0	.0	.0
Total	35,990.0	16.8	.0	4,891.5	13.6	4,104.2	11.4	26,977.6	75.0	.0	.0
Consumption	599,570.3	48.0	.0	32,609.7	5.4	27,361.3	4.6	539,551.3	90.0	.0	.0
PREDATORS											
Argia	200.0	200.0	100.0	.0	.0	.0	.0	.0	.0	.0	.0
Tanypodinae	396.0	396.0	100.0	.0	.0	.0	.0	.0	.0	.0	.0
Total	596.0	596.0	100.0	.0	.0	.0	.0	.0	.0	.0	.0
Consumption	1,702.8	1,702.8	100.0	.0	.0	.0	.0	.0	.0	.0	.0
SHREDDERS											
Taeniopteryx	28.0	.0	.0	.0	.0	.0	.0	.0	.0	28.0	100.0
Strophopteryx	112.0	.0	.0	.0	.0	.0	.0	.0	.0	112.0	100.0
Total	140.0	.0	.0	.0	.0	.0	.0	.0	.0	140.0	100.0
Consumption	2,801.0	.0	.0	.0	.0	.0	.0	.0	.0	2,801.0	100.0
SUMMARY											
Total	43,838.4	1,145.8	2.6	5,750.8	13.1	4,349.2	9.9	32,425.7	74.0	140.0	.3
Consumption	721,998.4	3,351.0	.5	38,338.0	5.3	28,994.5	4.0	648,513.9	89.8	2,801.0	.4

Table V.8.8. (continued)

	Prod.	Animal	%	Diatom	%	OAlgae	%	Detritus	%	VPlant	%
COBBLE/PEBBLE - LW											
COLLECTOR FILTERERS											
Cheumatopsyche	419.0	24.4	5.8	52.4	12.5	21.0	5.0	321.3	76.7	.0	.0
Hydropsyche morosa	12.2	4.3	35.4	.9	7.6	.2	1.9	6.7	55.1	.0	.0
Total	431.3	28.7	6.7	53.3	12.4	21.2	4.9	328.0	76.0	.0	.0
Consumption	7,139.6	82.3	1.2	355.5	5.0	141.2	2.0	6,560.6	91.9	.0	.0
COLLECTOR GATHERERS											
Baetidae	20.4	.0	.0	10.2	20.0	.0	.0	10.2	50.0	.0	.0
Stenacron	186.7	.0	.0	46.7	25.0	.0	.0	140.0	75.0	.0	.0
Stenonema	702.7	.0	.0	175.7	25.0	.0	.0	527.0	75.0	.0	.0
Ephemerella	83.8	.0	.0	29.0	34.6	.0	.0	54.8	65.4	.0	.0
Tricorythodes	162.0	.0	.0	.0	.0	.0	.0	162.0	100.0	.0	.0
Caenis	2.5	.0	.0	.0	.0	.0	.0	2.5	100.0	.0	.0
Cyrnellus fraternus	533.5	146.6	26.9	.0	.0	.0	.0	389.9	73.1	.0	.0
Orthocladiinae/ Chironominae	13,582.5	.0	.0	1,697.8	12.5	1,697.8	12.5	10,186.9	75.0	.0	.0
Total	15,274.1	143.6	.9	1,959.4	12.8	1,697.8	11.1	11,473.3	75.1	.0	.0
Consumption	254,257.4	410.4	.2	13,062.6	5.1	11,318.8	4.5	229,468.6	90.3	.0	.0
PREDATORS											
Argia	195.6	195.6	100.0	.0	.0	.0	.0	.0	.0	.0	.0
Tanypodinae	526.5	526.5	100.0	.0	.0	.0	.0	.0	.0	.0	.0
Total	722.1	722.1	100.0	.0	.0	.0	.0	.0	.0	.0	.0
Consumption	2,063.2	2,063.2	100.0	.0	.0	.0	.0	.0	.0	.0	.0
SHREDDERS											
Taeniopteryx	89.9	.0	.0	.0	.0	.0	.0	.0	.0	89.9	100.0
Strophopteryx	35.3	.0	.0	.0	.0	.0	.0	.0	.0	35.3	100.0
Total	125.2	.0	.0	.0	.0	.0	.0	.0	.0	125.2	100.0
Consumption	2,504.0	.0	.0	.0	.0	.0	.0	.0	.0	2,504.0	100.0
SUMMARY											
Total	16,552.7	894.4	5.4	2,012.7	12.2	1,719.0	10.4	11,801.3	71.3	125.2	.8
Consumption	265,964.2	2,555.9	1.0	13,418.1	5.0	11,460.0	4.3	236,029.2	88.7	2,504.0	.9

Table V.8.8. (continued)

	Prod.	Animal	%	Diatom	%	OAlgae	%	Detritus	%	VPlant	%
SAND/SILT SUBSTRATES - UW											
COLLECTOR GATHERERS											
Orthocladiinae/											
Chironominae	2,198.7	.0	.0	274.8	12.5	274.8	12.5	1,649.0	75.0	.0	.0
Consumption	36,645.1	.0	.0	1,832.3	5.0	1,832.3	5.0	32,980.5	90.0	.0	.0
PREDATORS											
Tanypodinae	1,335.7	1,335.7	100.0	.0	.0	.0	.0	.0	.0	.0	.0
Consumption	3,816.3	3,816.3	100.0	.0	.0	.0	.0	.0	.0	.0	.0
SUMMARY											
Total	3,534.4	1,335.7	37.8	274.8	7.8	274.8	7.8	1,649.0	46.7	.0	.0
Consumption	40,461.4	3,816.3	9.4	1,832.3	4.5	1,832.3	4.5	32,980.5	81.5	.0	.0
SAND/SILT - LW											
COLLECTOR GATHERERS											
Orthocladiinae/											
Chironominae	826.7	.0	.0	103.3	12.5	103.3	12.5	620.0	75.0	.0	.0
Consumption	13,778.3	.0	.0	688.9	5.0	688.9	5.0	12,400.5	90.0	.0	.0
PREDATORS											
Tanypodinae	1,578.7	1,578.7	100.0	.0	.0	.0	.0	.0	.0	.0	.0
Consumption	4,510.6	4,510.6	100.0	.0	.0	.0	.0	.0	.0	.0	.0
SUMMARY											
Total	2,405.4	1,578.7	65.6	103.3	4.3	103.3	4.3	620.0	25.8	.0	.0
Consumption	18,288.9	4,510.6	24.7	688.9	3.8	688.9	3.8	12,400.5	67.8	.0	.0

The out-of-sailing-line (OSL) samplers were found resting on the bottom where they had been originally placed. The in-sailing-line (ISL) samplers were torn from their moorings and moved approximately 50m downstream, even though they had been staked to the bottom to prevent movement. Since every effort was made to place the two sets of samplers in areas of similar ambient flows, and since the ISL substrates were placed directly in the sailing line, it was concluded that the differences in the benthic macroinvertebrates during the month-long study were the result of tows passing directly over the ISL samplers.

Higher numbers of organisms occurred on the OSL samplers (mean = 943 organisms/sampler) than on ISL samplers (mean = 590 organisms/sampler) (Table V.8.9, Appendix A, Tables A8.6.1-2) however this difference was not significant ($.20<p<.10$). The samplers in the sailing line, however, had significantly higher diversity (Dbar 2.1) (Lloyd et al. 1968) than those outside of the sailing line (Dbar 1.7) ($p<.02$). In addition, a significantly greater number of taxa occurred in the sailing line (14 ISL, 10 OSL) ($p<.05$). No hydropsychid caddisflies were found on the OSL samplers, while they were quite abundant (105.6 organisms/sampler) on ISL samplers. Tanypodinae were present in significantly lower numbers on the ISL samplers than the OSL samplers ($p<.02$). Cyrnellus fraternus larvae were present in significantly higher numbers on OSL substrates than ISL ($p<.05$). There were no significant differences between densities of

Ephemeroptera or Orthocladiinae/Chironominae on OSL and ISL samplers.

In both communities, collector-gatherers were the dominant trophic group, making up 96% of the OSL community and 79% of the ISL community. Predators represented a minor portion of both the OSL (6%) and ISL (3%) communities. Collector-filterers were only present in the ISL community where they represented 18% of the fauna.

Head capsules of all taxa were measured in order to compare the presence of the various size classes on ISL and OSL samplers. No differences were apparent between ISL and OSL samplers in terms of the size of larvae present for Ephemeroptera or Orthocladiinae/Chironominae groups. Although Cyrnellus fraternus was represented on both ISL and OSL samplers no fifth instar C. fraternus were collected from the sailing line. No other organisms followed this pattern.

In an effort to detect other, more subtle, differences between the ISL and OSL communities that would be manifested in growth, groups of individual organisms in each size class were dried and weighed. Although there were some slight differences in average individual dry weights of organisms collected on ISL and OSL samplers, there were no consistent trends (Appendix A, Table A8.7). However, the total biomass was higher on cobble/pebble substrates in the sailing line than out of the sailing line.

Table V.8.9. Summary of results from tow passage experiment. ISL = in sailing line samplers; OSL = out of sailing line samplers; 260 tows transited study site during experiment.

	ISL	OSL
Mean Density (organisms/sampler)	591	944
Mean # taxa/sampler	10	6
Total # taxa/location	14	9
Mean diversity (Dbar)/sampler	2.1	1.7
Biomass (mg/sampler)	65.2	19.8

Total community biomass on ISL samplers was 65.2 mg/sampler while on OSL samplers it was only 19.8 mg/sampler. The difference in biomass was due to the presence of the hydropsychid caddisflies on the ISL substrates.

V.9. FISH

Total fish biomass for the Winfield pool of the Kanawha River was estimated from a series of five recent lock rotenone collections as 242 kg/ha. The total fish biomass observed in 12 recent lock rotenone surveys has fluctuated, but the variation appears random and does not suggest any consistent changes in overall standing stock.

Abundance by Species and Trophic Group

Tables V.9.1 and V.9.2 summarize estimates of absolute abundance in terms of kcal/m^2 and kg/ha for some of the more common fish species from UW and LW. Gizzard shad dominate the fish community at both sites. Common carp, channel catfish, and smallmouth buffalo are secondarily the most abundant species in the river at both sites. Detailed data on total catch and total weight are recorded in Appendix A, Tables A9.2.1-22 and A9.3.1-20, respectively.

Comparison of Fish Communities

Table V.9.1. Abundance of some common adult fish species from the Kanawha River, Upper Winfield pool.

Species	Abundance (kg/ha)	Abundance (kcal/m^2)
Gizzard shad	71.15	11.321
Common carp	45.49	6.553
Channel catfish	14.03	2.016
Smallmouth buffalo	27.83	4.004
Sauger	8.23	0.962
Silver redhorse	13.07	1.882
Golden redhorse	9.20	1.325
Shorthead redhorse	12.34	1.772
Freshwater drum	3.63	0.424
Smallmouth bass	9.20	1.076
Emerald shiner	3.87	0.557
Spotted bass	2.90	0.339
Largemouth bass	0.48	0.056
White bass	1.94	0.227
Bluegill sunfish	0.24	0.028
Spotted sucker	0.	0.
Longear sunfish	0.73	0.085
White crappie	0.39	0.046

Table V.9.2. Abundance of some common adult fish species from the Kanawha River, Lower Winfield pool.

Species	Abundance (kg/ha)	Abundance (kcal/m^2)
Gizzard shad	68.00	10.812
Common carp	37.27	5.372
Channel catfish	41.62	5.991
Smallmouth buffalo	36.78	5.300
Sauger	13.07	1.532
Silver redhorse	0.48	0.069
Golden redhorse	0.48	0.069
Shorthead redhorse	0.	0.
Freshwater drum	9.92	1.160
Smallmouth bass	0.15	0.018
Emerald shiner	2.90	0.418
Spotted bass	1.69	0.198
Largemouth bass	2.90	0.339
White bass	0.97	0.113
Bluegill sunfish	4.60	0.538
Spotted sucker	4.11	0.592
Longear sunfish	1.45	0.170
White crappie	1.21	0.141

The relative abundance of major fish species at each of the three sites is compared in Table V.9.3. Several species make up larger percentages of the total fish biomass at UW and UG than at LW, for example silver, golden, and shorthead redhorse, walleye, white bass, and smallmouth bass. The complimentary trend (a larger percentage at LW than UW or UG) is exemplified by freshwater drum, bluegill sunfish, black crappie, and spotted sucker. Gizzard shad and smallmouth buffalo comprised a greater proportion of the total at sites UW and LW than at UG. Channel catfish and largemouth bass progressively increase in relative importance from site UW to LW to UG.

Species which occurred in a consistent fashion throughout the year included the common carp, smallmouth buffalo, spotted sucker, sauger, and silver redhorse. Many species occurred more frequently during particular seasons. Species captured most commonly during the summer included white bass, white crappie, and black crappie. Other species occurred primarily in the summer and fall (catfishes from July-Nov., sunfishes from July-Aug.) or occurred commonly from spring through fall (gizzard shad, spotted bass, largemouth bass, drum, and gar). Smallmouth bass and emerald shiners were most prevalent in samples made during the colder months. The redhorses appeared consistently except for declines during spring samples.

The number of different taxa observed in a sampling period was relatively stable, but did decline during mid-winter. Total

Table V.9.3. Fish community composition by study site.

Species	Percentage of Total Biomass		
	UW	LW	UG
Gizzard shad	29.4	28.1	11.3
Common carp	18.8	15.4	14.8
Channel catfish	5.8	17.2	24.8
Smallmouth buffalo	11.5	15.2	3.9
Sauger	3.4	5.4	4.9
Silver redhorse	5.4	0.2	2.6
Golden redhorse	3.8	0.2	4.6
Shorthead redhorse	5.1	0.0	1.5
Freshwater drum	1.5	4.1	1.7
Smallmouth bass	3.8	0.0	1.1
Emerald shiner	1.6	1.2	3.0
Spotted bass	1.2	0.7	0.9
Largemouth bass	0.2	1.2	1.8
White bass	0.8	0.4	1.6
Bluegill sunfish	0.1	1.9	0.2
Spotted sucker	0.0	1.7	0.0
Longear sunfish	0.3	0.6	0.0
White crappie	0.4	0.5	0.0
Black crappie	0.0	0.3	0.0
Walleye	0.6	0.0	0.7

weight of fish captured in a sampling period varied throughout the year without any apparent seasonal trends.

Table V.9.4 gives absolute abundance of fish arranged by trophic group. When grouped by feeding guilds, herbivore/detritivores and omnivores constitute over two-thirds of the fish community. Crayfish/piscivores, benthic invertivores, and midwater invertivores are more abundant at UW than the LW. The remaining three groups (omnivores, piscivores, and herbivore/detritivores) exhibit the opposite trend. Table V.9.5 outlines the relative abundance of each trophic group, including the proportion of total fish biomass and number of taxa in each group. By far the greatest number of species feed on insects, although the proportion of total biomass represented by invertivores is small.

Autecology of Major Taxa

Gizzard Shad

Gizzard shad are, both numerically and by weight, one of the most common species found in large rivers in the midwest and southern U.S. waters. Hendricks and Noble (1979) studied the food habits of the gizzard shad in Trinidad Lake, Texas, and found detritus to be the predominant diet item along with various planktonic algae species. Pierce and Wissing (1981) found detritus taken from the surface layers of bottom sediments to be

Table V.9.4. Abundance of Kanawha river fishes by trophic group.

Trophic Group	kcal/m² UW	kcal/m² LW
Omnivores	11.240	13.433
Piscivores	2.015	2.788
Cray/Piscivores	1.548	0.572
Herb/Detritivores	15.480	16.275
Benthic Invertivores	4.258	2.213
Midwater Invertivores	0.648	0.498
	35.189	35.779

Table V.9.5. Fish community composition by trophic group.

Trophic Group	Biomass (g) UW	(%)	Biomass (g) LW	(%)	Taxa
Herb/Detritivore	165125.	(41)	154138.	(43)	4
Omnivore	123782.	(31)	125323.	(35)	10
Benthic Invertivore	53250.	(13)	28065.	(8)	19
Piscivore	28872.	(7)	34632.	(10)	9
Cray/Piscivore	21822.	(5)	7250.	(2)	4
Midwater Invertivore	7444.	(2)	5022.	(1)	20
	-----		-----		---
	400295.		354430.		66

the preferred food of this species in Acton Lake, Ohio. Baker and Schmitz (1971) also found adult gizzard shad from Ozark reservoirs utilizing benthic detritus, along with plankton, molluscs, and insect larvae. Feeding habits of gizzard shad in rivers have not been previously reported.

Smallmouth Buffalo

Smallmouth buffalo are common in the deeper,swifter waters of larger tributaries of the Mississippi River. Edwards and Twomey (1982) described the smallmouth buffalo as an opportunistic bottom feeder with a diet likely to include zooplankton, algae, or insect larvae in proportion to their availability. McComish (1967) also interpreted the smallmouth buffalo diet as indicative of benthic feeding habits due to the high frequency of insect larvae, attached algae, and sand observed in the gut. Zooplankton and algae were reported as the most important food types in the diet. Minckley et al. (1970) reported significant quantities of detritus and molluscs in the diet of smallmouth buffalo.

Channel Catfish

Channel catfish are opportunistic feeders demonstrating a wide range of feeding habits depending on food availability. Zuerlein (1982) investigated the food habits of the channel

catfish in the channelized Missouri River as well as the Niobrara River. Based on percent-frequency-of-occurrence methods, channel catfish fed primarily on insects (81%), crustaceans (33%), and plants (33%). The most important insects in the diet were also the most common items in the drift, including the orders Trichoptera, Diptera, and Ephemeroptera. Food habit studies of channel catfish from the Niobrara River demonstrated that insects, plants, and fish were the major food items occurring in the diet (95, 67, and 10% frequency-of-occurrence, respectively). Channel catfish from the Missouri River between Spencer Dam and the mouth of the Niobrara River consumed insects (99% occurrence), fish (11% occurrence), and plants (10% occurrence).

Freshwater Drum

Freshwater drum are found in a variety of habitats but seem to prefer large silty rivers and lakes. Summerfelt et al. (1972) found the primary foods (in decreasing order of importance) taken by freshwater drum in 4 Oklahoma reservoirs were fish, crayfish, organic detritus, small clams, aquatic insects, and aquatic oligochaetes. Fish or fish remains made up approximately 80% of the total volume of all contents. The authors mention that the drum is frequently described as molluscaphagous, but that several investigators have found gastropods and pelecypods to be only an insignificant part of the diet.

Redhorses(*Moxostoma* spp.)

Redhorses, as a group, are common inhabitants of larger streams with a moderate to swift current. Generally the redhorses ascend smaller tributaries to spawn in spring and move downstream during summer and fall and winter in larger streams. In the Kanawha River, we collected five species of redhorses (silver, golden, shorthead, river, and black), of which the silver, golden, and shorthead were dominant. Feeding habits of the river redhorse were examined in the Cahaba River (Alabama) by Hackney et al. (1966). The predominant food item was the Asiatic clam, Corbicula spp. Other food items found included the larvae of Ephemeroptera, Chironominae, and Trichoptera, although these made an insignificant contribution to the total diet. Yant et al. (1978) studied the diets of three species of redhorse (golden, black, and shorthead) in the middle reaches of the Wabash River, Indiana. The primary prey for all 3 species were chironomids, while simuliids, diptera pupae, hydropsychids, tricorithids, heptageniids, and potamanthids comprised the remaining important food groups. In Meyer's (1962) study of the golden, northern, and silver redhorse in the Des Moines River, the 3 species exhibited similar food habits by the frequency-of-occurrence method. Chironomidae larvae were found in 91% of the stomachs, emphemeroptera in 62%, and trichoptera in 18%. Bowman (1970) determined that black redhorse in the Niangua and Big Piney Rivers (Missouri Ozarks) also fed on aquatic insects, primarily diptera larvae. The diet of this species included Cladocera, Copepoda, and Nemathelminthes as well.

Common Carp

Common carp, a native of temperate Eurasia, were introduced to the U.S. in 1877 and are now widely distributed in many habitats in U.S. and southern Canada. The common carp's feeding habits were reviewed by Edwards and Twomey (1982), who described the species as capable of utilizing any available food source. Carp switch from a diet of zooplankton as fry to littoral fauna as they grow, and eventually forage upon bottom fauna. Oligochaetes and the larvae of aquatic insects are consumed in addition to seeds, algae, detritus, and other vegetable matter. Eder and Carlson (1977) studied carp food habits from the South Platte and St. Vrain Rivers (Colorado), characterizing the carp as an opportunistic feeder. Foods included aquatic insect larvae, terrestrial insects, seeds, aquatic plants, dissolved materials and detritus, and sand. On the basis of percentage of total volume in the gut, sand along with dissolved materials and detritus dominated the stomach contents. Chironomid larvae and pupae made up the majority of animal matter ingested by the carp.

Emerald shiner

Emerald shiners are common in large rivers and lakes where they swim in large schools in the mid- or near-surface waters. Food habit determinations for emerald shiners were conducted by

Fuchs (1965) in Lewis and Clark, South Dakota. Zooplankton were the primary diet item of adult emerald shiners, but insects were also consumed. Daphnia often made up over 90% of the total stomach contents, and were apparently selected for their large size relative to other zooplankton. Utilization of insects (immature and adult Diptera) peaked at a time which coincided with low Daphnia availability. Young-of-year emerald shiners fed primarily on blue-green algae, although rotifers and ciliated protozoa also appeared in the diet. With increasing size, the diet of the young of the year shifted gradually towards zooplankton. In a smaller stream, Minckley (1963) reported that emerald shiners ate mostly terrestrial insects in summer, and amphipods, mayfly nymphs and caddisfly larvae in winter.

Sauger

Sauger are commonly found in large, often turbid rivers, lakes and impoundments. Sauger are a popular game fish and are generally described as piscivorous. McBride and Tarter (1983) detailed the feeding habits of sauger from the vicinity of Gallipolis Locks and Dam on the Ohio River, and found fish as the main food type. Fish made up nearly 100% of the volume of all foods observed and were present in all stomachs containing food. Emerald shiners and gizzard shad were reported as the most important forage species. Wahl and Nielsen (1985), in a study of sauger feeding ecology conducted in the same area of the Ohio

River, also found fish dominating the sauger diet. Gizzard shad made up 42% of all food items, followed by emerald shiners (28%), freshwater drum (16%), and channel catfish (12%).

Growth and Mortality

Information on average length and weight at age for the common fish species are summarized in Appendix Tables A9.5.1-2. Fitted parameters for the von Bertalanffy growth equations are given in Table V.9.6 for weight (g) and Table V.9.7 for length. These fitted equations adequately describe the growth pattern of the fish populations in the Kanawha River for the range of ages sampled (Tables A9.5.1-2). Instantaneous annual mortality rates (Z) varied among fish species, ranging from 0.3002 for spotted sucker to 1.2959 for emerald shiner (Table V.9.8).

Diet Analysis by Species and Trophic Group

Diet composition was analyzed for the 17 fish species (Table V.9.9). Each species was characterized into one of six functional groups: omnivore, piscivore, crayfish/piscivore, herbivore/detritivore, benthic invertivore, or midwater invertivore.

Assignment to a trophic group was based on the relative contribution of various food types in the annual diet. Proportional annual consumption values for these species are also given in Table V.9.9. Species considered to be piscivores

Table V.9.6. Growth parameters for weight (g) for common adult fish species. K = coefficient for growth; W_∞ = asymptotic maximum weight; t_0 = time intercept for W = 0.

Species	K	W_∞	t_0
Gizzard shad	0.1750	2243.2	-0.344
Common carp	0.1308	12734.9	0.222
Channel catfish	0.1180	7340.3	-0.105
Smallmouth buffalo	0.2286	4326.6	-0.089
Sauger	0.1186	8953.5	-0.919
Silver redhorse	0.1705	3079.6	-1.267
Golden redhorse	0.2345	1571.3	-0.488
Shorthead redhorse	0.3820	950.0	0.118
Freshwater drum	0.3473	1067.3	0.563
Smallmouth bass	0.3853	1858.4	0.999
Emerald shiner	0.3051	31.1	0.054
Spotted bass	0.1580	2262.6	-0.398
Largemouth bass	0.4073	1350.9	0.629
White bass	0.5846	620.4	-0.283
Bluegill sunfish	0.5618	110.0	0.286
Spotted sucker	0.8839	900.0	1.500
Longear sunfish	0.3049	162.1	-0.153
White crappie	0.4223	502.0	0.261

Table V.9.7. Growth parameters for length (mm) of common adult fish species. K = coefficient for growth; L_∞ = asymptotic maximum length; t_0 = time intercept for L = 0.

Species	K	L_∞	t_0
Gizzard shad	0.2394	550.0	0.066
Common carp	0.0785	1239.0	-1.045
Channel catfish	0.1391	783.7	-0.383
Smallmouth buffalo	0.2521	664.3	0.124
Sauger	0.1448	836.6	-0.919
Silver redhorse	0.1595	646.6	-1.800
Golden redhorse	0.1034	673.7	-2.720
Shorthead redhorse	0.2755	525.4	-0.023
Freshwater drum	0.1860	573.4	-0.143
Smallmouth bass	0.1129	872.2	-0.123
Emerald shiner	0.2019	192.1	-0.158
Spotted bass	0.1184	662.1	-0.457
Largemouth bass	0.2957	448.6	0.291
White bass	0.5748	370.2	-0.092
Bluegill sunfish	0.2199	236.5	-0.817
Spotted sucker	0.2911	527.9	0.306
Longear sunfish	0.1959	233.0	-0.640
White crappie	0.3413	363.8	0.048

Table V.9.8. Estimated instantaneous mortality rates for common adult fish species.
Z = mortality rate; l_c = length (mm) at which species is considered first fully vulnerable to the gears.

Species	Z	age range	l_c
Gizzard shad	1.0992	2. - 4.	200.
Common carp	0.8553	3. - 5.	520.
Channel catfish	0.4422	1.5- 5.5	260.
Smallmouth buffalo	0.5864	4. - 6.	410.
Sauger	1.1617	2. - 6.5	350.
Silver redhorse	0.5473	2. - 3.	365.
Golden redhorse	0.9689	2. - 3.5	340.
Shorthead redhorse	1.0062	2. - 4.	300.
Freshwater drum	1.0211	2. - 5.5	200.
Smallmouth bass	1.2585	1.5- 4.5	160.
Emerald shiner	1.2959	1. - 2.5	55.
Spotted bass	0.8372	2. - 4.	170.
Largemouth bass	0.4574	1.5- 4.5	140.
White bass	0.8513	2. - 4.	250.
Bluegill sunfish	0.5666	1.5- 5.	105.
Spotted sucker	0.3002	1.5- 5.	115.
Longear sunfish	0.6825	2. - 4.	95.
White crappie	0.9980	2. - 5.	190.

Table V.9.9. Diet composition and trophic group classification for selected fish species. BI = benthic invertivores; CP = crayfish/piscivores; HD = herbivore/detritivores; MI = midwater invertivores; OM = omnivores; PI = piscivores.

Species (Trophic Group)	Percent of Diet				
	Fish	Cray.	Invert.	Plant	Detrit.
Gizzard shad (HD)	-	-	-	4.3	95.7
Common carp (OM)	-	0.2	65.9	23.9	10.0
Channel catfish (OM)	43.5	5.2	5.2	44.5	1.5
Smallmouth buffalo (HD)	-	-	5.0	62.9	32.1
Sauger (PI)	100.0	-	-	-	-
longnose gar (PI)	99.6	-	0.4	-	-
Silver redhorse (BI)	-	-	98.6	-	1.4
Golden redhorse (BI)	-	-	93.0	2.4	4.6
Shorthead redhorse (OM)	-	-	28.8	67.5	3.7
Freshwater drum (BI)	15.3	38.8	44.4	0.3	1.7
Smallmouth bass (CP)	29.9	59.9	10.2	-	-
Flathead catfish (PI)	98.4	1.6	-	-	-
Emerald shiner (MI)	-	-	83.0	17.0	-
Spotted bass (CP)	35.1	64.1	0.8	-	-
Largemouth bass (CP)	60.0	34.7	-	5.3	-
White bass (PI)	81.1	17.6	1.3	-	-
Bluegill sunfish (BI)	2.4	-	75.9	21.7	-

included sauger, longnose gar, flathead catfish, and white bass. The omnivores included shorthead redhorse, common carp, and channel catfish. Gizzard shad and smallmouth buffalo were categorized as herbivore/detritivores. The bluegill sunfish, freshwater drum, golden, and silver redhorses were categorized as benthic invertivores. Smallmouth, largemouth, and spotted bass were grouped under the crayfish/piscivores. The emerald shiner was considered to be a midwater invertivore. Each of the remaining 49 species were assigned to one of the six trophic groups based on inspection of all available data on stomach contents (Apendix A, Tables A9.4.1-35). These species occurred too infrequently for a reliable determination of proportional annual consumption, and in any case constitute only a small part of the observed biomass of their trophic groups (Table V.9.10). Tables V.9.11-16 provide the complete set of species in each group and their relative abundance within groups.

Proportional annual consumption data were then combined for species within each of the trophic groups. The data were weighted according to the relative abundance of each member species in the group. Table V.9.17 provides a summary of diet composition by trophic group for both sites. An analysis of each food type consumed by each trophic group was then conducted to determine the relative contributions of specific diet items in the annual ration. The results of these analyses are given in Tables V.9.18-23.

Production and Consumption by Species and Trophic Group

Table V.9.10. Proportion of biomass in each trophic group accounted for by the species analyzed for diet composition.

Trophic Group	UW	LW
Omnivores	.92	.84
Piscivores	.84	.97
Cray/Piscivores	.95	.97
Herb/Detritivores	.99	.99
Benthic Invertivores	.83	.67
Midwater Invertivores	.86	.84

Table. V.9.11. Relative abundance of fish species considered Midwater Invertivores.

Species	Biomass (g) UW	(%)	LW	(%)
Emerald shiner *	6370.	(86)	4209.	(84)
Mimic shiner	441.	(6)	46.	(1)
Spotfin shiner	103.	(1)	222.	(4)
Silver chub	60.	(1)	38.	(1)
River shiner	110.	(1)	119.	(2)
Bluntnose minnow	14.		313.	(6)
Sand shiner	7.		18.	
Steelcolor shiner	3.		30.	
Cyprinidae	2.		5.	
Mooneye	246.	(3)	0.	
Creek chub	51.	(1)	0.	
Striped shiner	20.		0.	
Notropis sp.	6.		0.	
Bullhead minnow	4.		0.	
Streamline chub	3.		0.	
Rosyface shiner	2.		0.	
Brook silversides	1.		0.	
Silver shiner	1.		0.	
Ghost shiner	0.		0.	
	7444.		5022.	

* analyzed for diet composition

Table V.9.12. Relative abundance of fish species considered Omnivores.

Species	Biomass (g) UW	(%)	LW	(%)
Common carp *	75145.	(58)	54399.	(39)
Channel catfish *	23372.	(18)	60983.	(44)
Shorthead redhorse *	20499.	(16)	0.	(1)
Quillback	2706.	(2)	1177.	(1)
River carpsucker	1912.	(1)	786.	
Goldfish	198.		0.	
Yellow bullhead	0.		1087.	(1)
Spotted sucker	0.		5998.	
Brown bullhead	0.		715.	
Black bullhead	0.		178.	
	123782. g		125323. g	

* analyzed for diet composition

Table V.9.13. Relative abundance of fish species considered Benthic Invertivores.

Species	Biomass (g) UW	(%)	LW	(%)
Silver redhorse *	21420.	(45)	541.	(4)
Golden redhorse *	15204.	(32)	521.	(4)
River redhorse	3487.	(7)	0.	
Black redhorse	3423.	(7)	0.	
Freshwater drum *	5908.	(5)	14635.	(10)
White crappie	1388.	(3)	1926.	(14)
Longear sunfish	1213.	(3)	2200.	(16)
Bluegill sunfish *	538.	(1)	6614.	(49)
N. hog sucker	533.	(1)	0.	
Log perch	77.		13.	
Redear sunfish	40.		185.	(1)
Dusky darter	13.		0.	
Banded darter	3		0.	
Bluebreast darter	2.		0.	
Etheostoma sp.	1.		0.	
Black crappie	0.		1039.	(8)
Pumpkinseed	0.		371.	(3)
Hybrid sunfish	0.		20.	
White sucker	0.		0.	
	53250.		28065.	

* analyzed for diet composition

Table V.9.14. Relative abundance of fish species considered Piscivores.

Species	Biomass (g) UW	(%)	LW	(%)
Sauger *	13617.	(47)	19159.	(55)
Longnose gar *	4463.	(15)	2550.	(7)
White bass *	3334.	(12)	1321.	(4)
Flathead catfish *	3024.	(10)	10816.	(31)
Walleye	2277.	(8)	0.	
Skipjack herring	1745.	(6)	786.	(2)
Muskellunge	189.		0.	
American eel	142.		0.	
Ohio lamprey	81.		0.	
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	28872.		34632.	

* analyzed for diet composition

Table V.9.15. Relative abundance of fish species considered Herb/Detritivores.

Species	Biomass (g) UW	(%)	LW	(%)
Gizzard shad *	117808.	(71)	99659.	(65)
Smallmouth buffalo *	46117.	(28)	53779.	(34)
Black buffalo	1200.	(1)	700.	
Brook lamprey	0.		0.	
	165125.		154138.	

* analyzed for diet composition

Table V.9.16. Relative abundance of fish species considered Crayfish/Piscivores.

Species	Biomass (g) UW	(%)	LW	(%)
Smallmouth bass *	15388.	(71)	226.	(3)
Spotted bass *	4609.	(21)	2486.	(34)
Rock bass	1206.	(5)	178.	(2)
Largemouth bass *	619.	(3)	4360.	(60)
	21822.		7250.	

* analyzed for diet composition

Production-biomass ratios ranged from 0.367 for spotted sucker to 1.578 for emerald shiner (Table V.9.24).

Annual production was estimated for each of the eighteen fish species listed in Table V.9.25. Gizzard shad had the highest production of any species (11.762 kcal/m^2/yr at UW). Table V.9.26 lists annual production arranged by trophic groups. Production by herbivore/detritivores represented over half of total fish community production, omnivores comprised nearly one quarter, and benthic invertivores, midwater invertivores, and piscivores comprised the remainder.

Annual consumption was estimated indirectly as the consumption required to support measured production. Thus the consumption data reflect the same trends noted above for production. Table V.9.27 summarizes consumption estimates for individual species. Consumption arranged by trophic groups is given in Table V.9.28. The species for which production and consumption estimates were estimated represented at least 59% of the biomass of each trophic group (Table V.9.29).

V.10. LARVAL FISHES

Distribution

Larvae were identified into 10 taxonomic groups (Appendix Tables A.10.1.1-2). Several Catostomidae species are common in

Table V.9.17. Diet composition by food type for each trophic group.

Trophic Group	Percent of Diet				
	Fish	Cray.	Invert.	Plant	Detrit.
SITE UW					
Omnivore	8.7	1.2	47.4	35.5	7.2
Piscivore	97.3	2.4	0.3	-	-
Cray/Piscivore	32.4	60.0	7.6	-	-
Herb/Detritivore	-	-	1.4	20.7	77.9
Benthic Invertivore	2.2	5.4	88.7	1.1	2.6
Midwater Invertivore	-	-	83.0	17.0	-
SITE LW					
Omnivore	23.1	2.9	33.7	34.8	5.5
Piscivore	98.7	1.2	0.1	-	-
Cray/Piscivore	52.5	47.2	0.3	-	-
Herb/Detritivore	-	-	1.8	24.8	73.4
Benthic Invertivore	10.5	24.5	56.9	6.8	1.3
Midwater Invertivore	-	-	83.0	17.0	-

Table V.9.18. Major food types by diet item for Omnivores.

Food type	Diet items	Percent of Food Type UW	LW
Fish	Cyprinidae	21.0	21.0
	Gizzard shad	79.0	79.0
Invertebrates	Oligochaeta	0.7	0.0
	Amphipoda	0.4	0.3
	Ephemeroptera	1.9	5.0
	Odonata	0.5	1.2
	Hemiptera	0.5	1.2
	Trichoptera	2.7	0.0
	Coleoptera	0.5	1.2
	Diptera	21.3	12.0
	Pelecypoda	43.2	27.8
	Terrestrial	28.3	51.3
Plants	Filamentous algae	61.0	68.1
	Allochthonous	39.0	31.9

Table V.9.19. Major food types by diet item for Benthic Invertivores.

Food type	Diet items	Percent of Food Type UW	LW
Fish	Cyprinidae	94.5	94.5
	Catfish	5.5	5.5
Invertebrates	Oligochaeta	6.4	0.4
	Amphipoda	0.5	2.7
	Ephemeroptera	1.9	8.7
	Odonata	4.0	18.6
	Trichoptera	0.1	0.8
	Diptera	18.4	14.0
	Gastropoda	0.2	1.0
	Pelecypoda	66.0	25.9
	Terrestrial	2.5	27.9
Plants	Allochthonous	100.0	100.0

Table V.9.20. Major food types by diet item for Crayfish/Piscivores.

Food type	Diet items	Percent of Food Type UW	LW
Fish	Cyprinidae	78.1	42.3
	Catfish	21.5	50.8
	Gizzard shad	0.4	6.9
Invertebrates	Ephemeridae	12.0	50.0
	Odonata	13.3	50.0
	Hemiptera	74.7	0.0

Table V.9.21. Major food types by diet item for Herb/Detritivores.

Food type	Diet items	Percent of Food Type UW	LW
Invertebrates	Zooplankton	69.3	69.3
	Amphipoda	11.5	11.5
	Diptera	11.5	11.5
	Pelecypoda	7.7	7.7
Plants	Filamentous algae	33.2	39.5
	Allochthonous	6.0	5.6
	Diatoms	60.8	54.9

Table V.9.22. Major food types by diet item for Midwater Invertivores.

Food type	Diet items	Percent of Food Type UW	LW
Invertebrates	Oligochaeta	4.2	4.2
	Trichoptera	15.7	15.7
	Diptera	44.9	44.9
	Pelecypoda	35.2	35.2
Plants	Filamentous algae	100.0	100.0

Table V.9.23. Major food types by diet item for Piscivores.

Food type	Diet items	Percent of Food Type UW	LW
Fish	Lamprey	2.5	2.6
	Gizzard shad	2.0	0.8
	Cyprinidae	80.9	74.7
	Catfish	1.0	1.0
	Centrarchidae	4.9	12.1
	Freshwater drum	8.7	8.8

Table V.9.24. Production-biomass ratios (P/B) of common adult fish species.

Species	P/B
Gizzard shad	1.039
Common carp	0.453
Channel catfish	0.547
Smallmouth buffalo	0.528
Sauger	0.776
Silver redhorse	0.403
Golden redhorse	0.435
Shorthead redhorse	0.711
Freshwater drum	0.833
Smallmouth bass	0.953
Emerald shiner	1.578
Spotted bass	0.813
Largemouth bass	0.662
White bass	0.741
Bluegill sunfish	0.619
Spotted sucker	0.367
Longear sunfish	0.659
White crappie	0.889

Table V.9.25. Production of common adult fish species from the Kanawha River.

Species	kcal/m²/yr UW	kcal/m²/yr LW
Gizzard shad	11.762	11.234
Common carp	2.969	2.434
Channel catfish	1.103	3.277
Smallmouth buffalo	2.114	2.798
Sauger	0.747	1.189
Silver redhorse	0.758	0.028
Golden redhorse	0.576	0.030
Shorthead redhorse	1.260	0.
Freshwater drum	0.353	0.966
Smallmouth bass	1.025	0.017
Emerald shiner	0.879	0.660
Spotted bass	0.276	0.161
Largemouth bass	0.037	0.224
White bass	0.168	0.084
Bluegill sunfish	0.017	0.333
Spotted sucker	0.000	0.217
Longear sunfish	0.056	0.112
White crappie	0.041	0.125

Table V.9.26. Production of adult Kanawha river fishes by trophic group.

Trophic Group	kcal/m²/yr UW	LW
Omnivores	5.796	6.661
Piscivores	1.551	2.158
Cray/Piscivores	1.408	0.414
Herb/Detritivores	14.016	14.174
Benthic Invertivores	2.024	1.643
Midwater Invertivores	1.022	0.786
	25.817	25.836

Table V.9.27. Annual consumption by common adult fish species from the Kanawha River.

Species	kcal/m²/yr UW	LW
Gizzard shad	117.620	112.340
Common carp	19.793	16.227
Channel catfish	3.939	11.704
Smallmouth buffalo	14.093	18.653
Sauger	4.669	7.431
Silver redhorse	3.610	0.133
Golden redhorse	2.743	0.143
Shorthead redhorse	8.400	0.
Freshwater drum	1.471	4.025
Smallmouth bass	4.271	0.071
Emerald shiner	6.279	4.714
Spotted bass	1.150	0.671
Largemouth bass	0.154	0.933
White bass	1.050	0.525
Bluegill sunfish	0.081	1.586
Spotted sucker	0.000	1.206
Longear sunfish	0.267	0.533
White crappie	0.171	0.521

Table V.9.28. Annual consumption by adult Kanawha River fishes by trophic group.

Trophic Group	kcal/m²/yr UW	LW
Omnivores	34.926	32.738
Piscivores	9.693	13.485
Cray/Piscivores	5.868	1.726
Herb/Detritivores	133.043	132.316
Benthic Invertivores	9.374	7.156
Midwater Invertivores	7.301	5.612
	200.205	193.033

Table V.9.29. Proportion of biomass in each trophic group analyzed for annual production and consumption.

Trophic Group	UW	LW
Omnivores	.92	.89
Piscivores	.59	.59
Cray/Piscivores	.95	.97
Herb/Detritivores	.99	.99
Benthic Invertivores	.89	.97
Midwater Invertivores	.86	.84

the Winfield Pool, but only larvae of Ictiobus-Carpiodes were collected in the samples. This is not unexpected as larvae of Moxostoma spp. are not usually abundant in ichthyoplankton samples from the Ohio River due to their photophobic nature and preference for well-oxygenated interstitial waters of gravel-rock substrates (Pearson and Krumholz 1984). Centrarchids were identified to three lower taxa: Lepomis spp., Pomoxis spp., and Ambloplites rupestris. No Micropterus were collected, likely indicating a different spawning/nursery habitat. Pearson and Krumholz (1984) report that centrarchids other than Pomoxis spawn most successfully in embayments, and suggest that the disturbances created by towboats reduce successful spawning in shallow waters bordering the mainstream. Clupeids were easily identified and assumed to be predominantly gizzard shad based on adult abundance. Of the several cyprinid species present, common carp was the only one that could be consistently identified. The bulk of the unidentified cyprinids were presumed to be emerald shiner based on larval characteristics and adult abundance. Ictalurids are rare in larval samples from the Ohio River (Pearson and Krumholz 1984) and the same appears to be true for the Kanawha River; only one channel catfish was captured. Percidae were not identified below the family level because of difficulty and the presence of species whose larval forms remain to be described. Freshwater drum were common and easily identified.

Larval fish were present on all sampling dates with highest densities in June and early July. Densities were lower than expected in late May and late July, probably as a result of high river discharge and falling water temperatures (Fig. V.10.1). Unidentified cyprinids were the most abundant taxa (especially along the shoreline) followed by clupeids and freshwater drum. These three taxa composed nearly 98% of the 22,136 larvae collected. Pearson and Krumholz (1984) report that emerald shiner, gizzard shad, and freshwater drum have pelagic or semi-pelagic eggs and/or larvae, and appear to have successful reproductive strategies in the Ohio River. For the following discussions on each taxonomic group, refer to Figs. V.10.2-8 and Appendix Tables A10.2.1-6 and A10.3.1-9.

Ictiobus-Carpiodes were present May-August with the majority collected in June. Peak abundance on the Ohio River occurs in May (Pearson and Krumholz 1984); it is likely that the high discharge and drop in water temperatures immediately prior to the late May sampling resulted in the low catch observed. Abundance in June did not differ with depth or distance across river, nor between UW and LW; however, densities at LW were significantly higher at night ($p < .001$; Fig.V.10.2). The mean length of Ictiobus-Carpiodes larvae in June was 7.2 mm.

Lepomis were not abundant at any time but were present June-August. The majority (85%) were captured along the shoreline

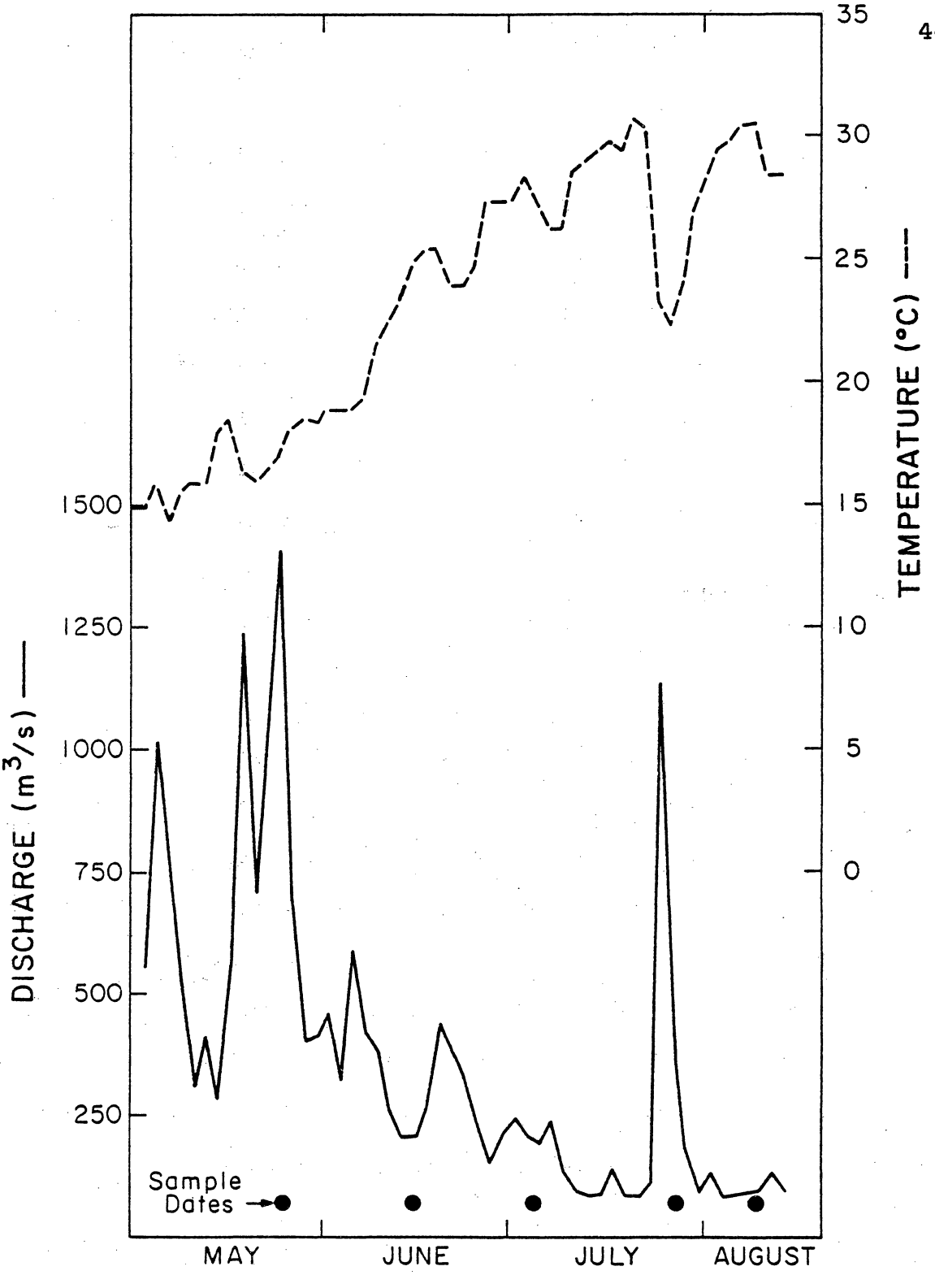


Figure V.10.1. Temperature and discharge of the Kanawha River at Charleston, West Virginia, during 1983 larval fish sampling.

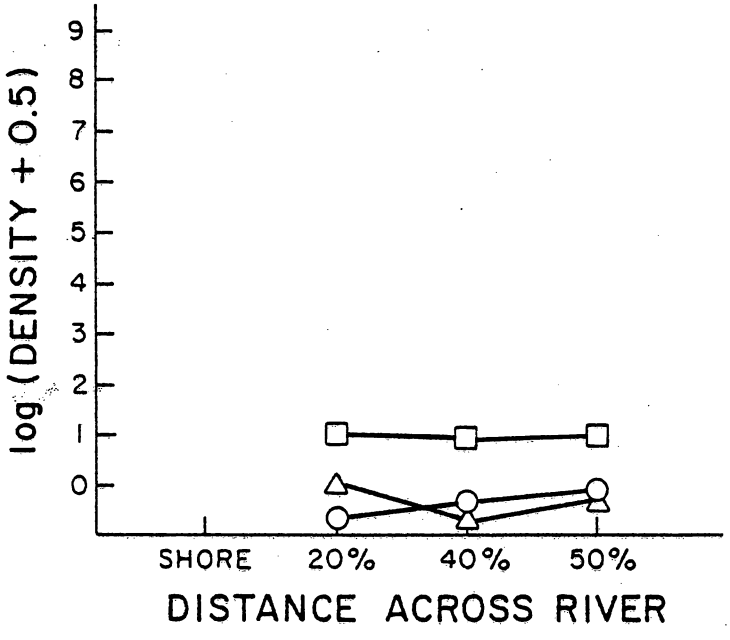


Figure V.10.2. Horizontal distribution of larval Ictiobus-Carpiodes in the Kanawha River in June.

where densities were significantly higher than those in the river on several occasions (Fig.V.10.3): early July LW night ($p < .001$), late July LW day ($p = .009$), late July LW night ($p < .001$), and August UW day ($p < .001$). Lepomis captured along the shoreline had a mean length of 6.7 mm, while river larvae averaged 4.5 mm. Shoreline densities did not vary between sites except in August when UW had higher densities than LW ($p = .003$). Diel differences in abundance were not significant. Pomoxis were collected in June and early July, almost exclusively at UW (12 of the 13 specimens). Only one Ambloplites rupestris was captured (LW in June).

Clupeids, a relatively abundant taxon, were present June-August with the peak densities occurring in June. Clupeids used the shoreline zone extensively as indicated by the numerous times shoreline densities significantly exceeded those of the river (Fig.V.10.4): early July LW day ($p < .001$), early July UW day ($p < .001$), late July LW day ($p < .001$), late July UW day ($p < .001$), and August LW night ($p < .001$). Diel variations in catch were apparent on two sampling dates. In early July, significantly more clupeids were captured in the river during daylight than at night ($p < .001$). Shoreline samples collected at the same time show a similar trend, although their densities did not differ statistically at the .01 level ($p = .011$). Graser (1979) reported this diel variation in catch of clupeids from Barkley Reservoir on the Cumberland River. Tuberville (1979)

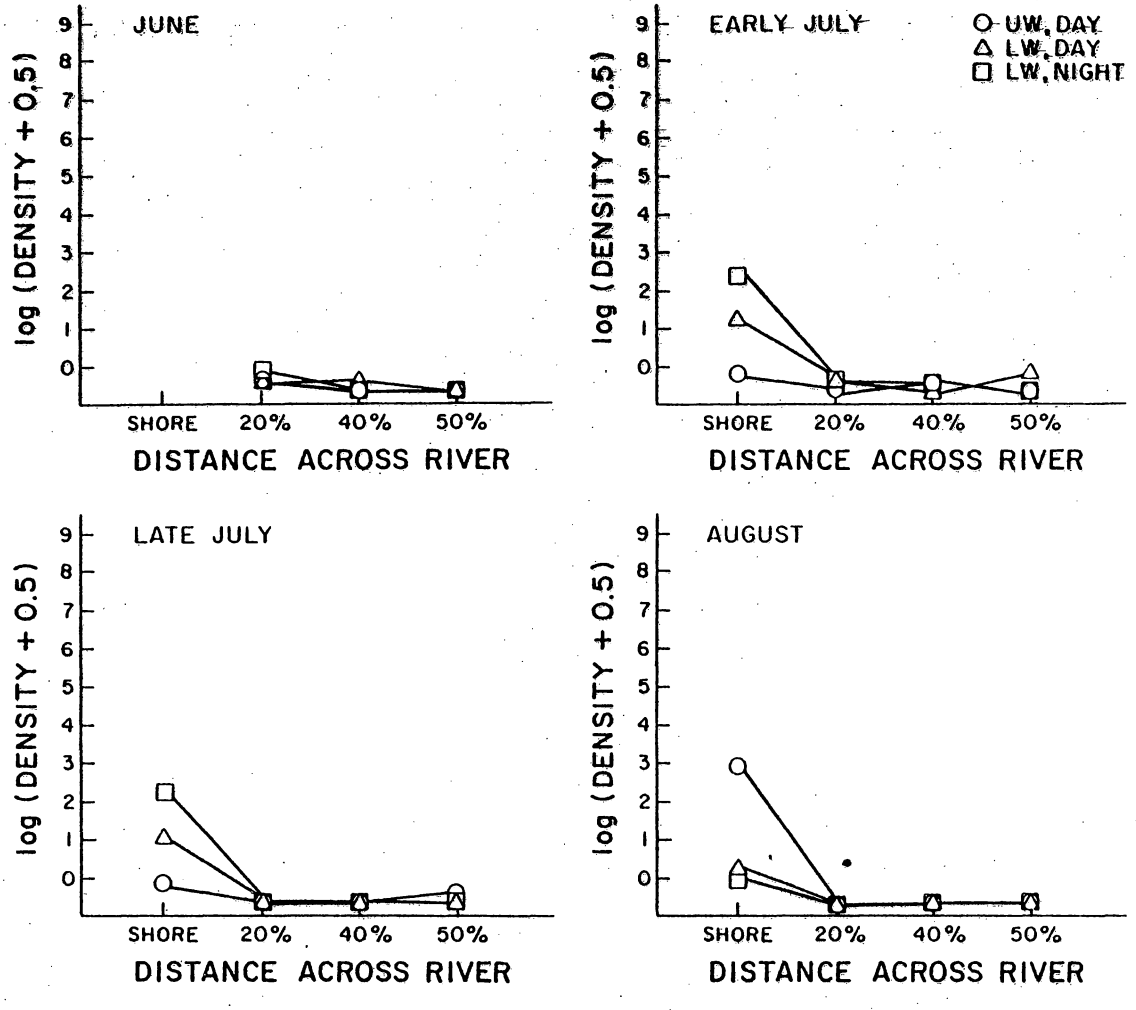


Figure V.10.3. Horizontal distribution of larval Lepomis spp. in the Kanawha River.

observed this trend on an impounded portion of the Tennessee River, but only for clupeids less than 10 mm in size; for larvae exceeding 10 mm, the reverse was true. The mean lengths of clupeids captured in the Kanawha River in early July were less than 10 mm (Table V.10.1). In August the mean length of clupeids captured along LW shoreline was closer to 10 mm, and shoreline densities were significantly higher at night ($p < .001$). In late July mean lengths far exceeded 10 mm, but nocturnal shoreline densities were not significantly higher than diurnal densities ($p = .100$). River densities in late July and August were too low to discern any trends. In June, diel variations in catch did not differ statistically. This is likely an artifact of inadequate sampling design; too few samples were collected from middepth (where clupeids were concentrated during daylight in June) for adequate statistical analysis. In early July, clupeid catch was significantly higher at LW than UW for both river ($p < .001$) and shoreline samples ($p < .001$). Densities did not differ significantly between sites in June or late July; however, river densities in August were significantly higher at UW ($p = .004$) while shoreline densities were not ($p = .027$). Vertical trends in clupeid distribution were evident at LW in June and early July. During daylight, clupeid densities were highest at middepth in June ($p < .001$) and near surface in early July ($p < .001$). At night the larvae appeared uniformly distributed (June) or concentrated near the bottom (early July, $p = .009$). Both

Graser (1979) and Tuberville (1979) observed pronounced surface orientation of clupeids during the day, but a uniform distribution (Tuberville 1979) or preference for the bottom at night (Graser 1979) at night.

Common carp were present late May-late July with 97% of the 155 specimens captured in June and early July. In June, carp were not captured at LW during daylight, yet they were a consistent component in the night samples and in UW daylight samples (Fig.V.10.5). In early July, daylight abundance did not differ between UW and LW, but nocturnal densities at LW still exceeded daylight densities for the river samples ($p = .001$) and shoreline collections (abundant in nocturnal samples but absent from daylight collections). In early July, the nocturnal shoreline catch was significantly higher than the corresponding river catch ($p = .009$). Vertical trends in abundance were evident in the LW river samples. In the June nocturnal collections, surface samples had the highest abundance ($p = .002$), but this nocturnal trend was not evident in early July. During daylight in early July, densities were higher in the bottom samples ($p = .001$). The mean length of carp larvae collected was 6.6 mm in June and 7.3 mm in early July.

Unidentified cyprinid, the most abundant taxon collected (80% of the total catch), were present from late May through August. Highest densities occurred in June and early July.

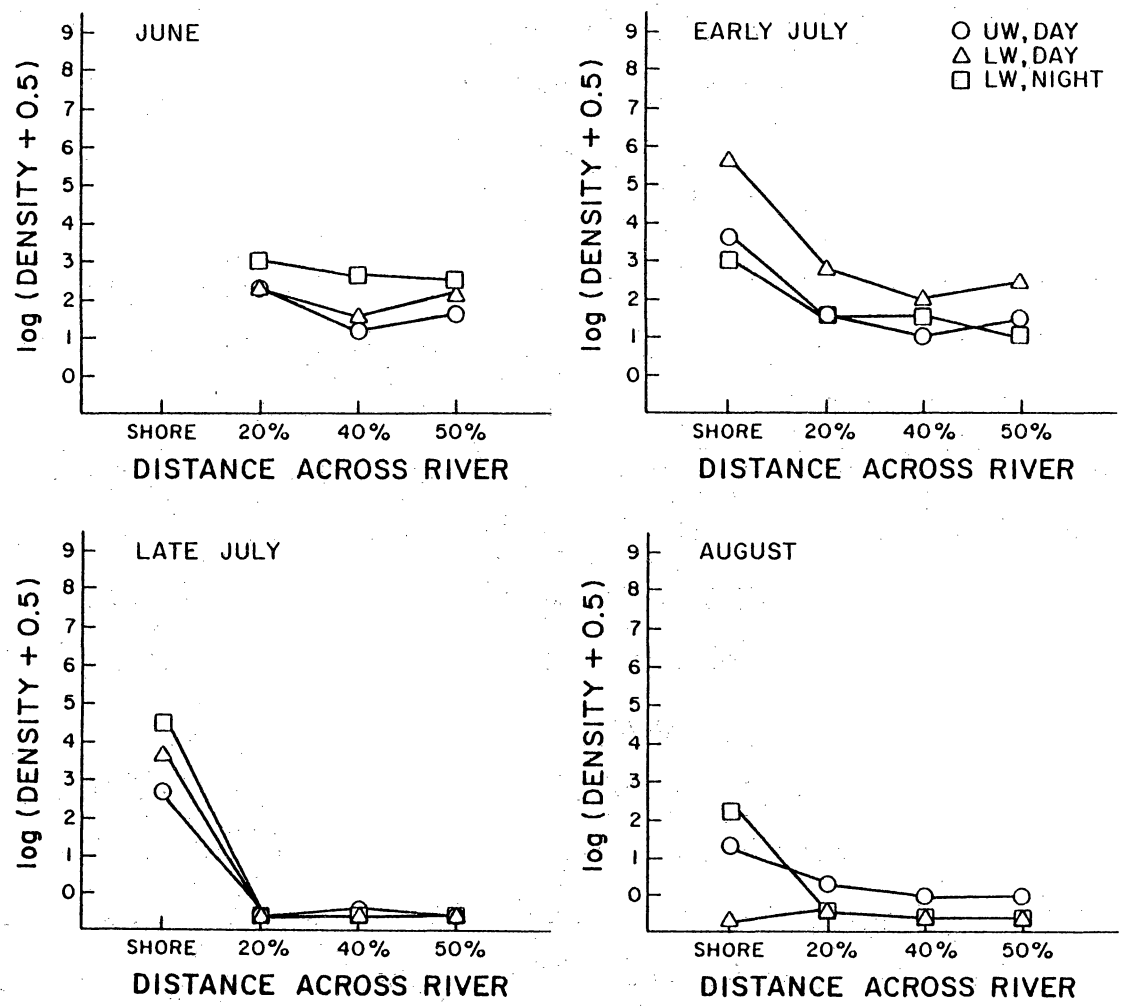


Figure V.10.4. Horizontal distribution of larval Clupeidae in the Kanawha River.

Table V.10.1. Mean lengths (mm) of Clupeidae in distribution samples collected from Winfield Pool. UW = Upper Winfield; LW = Lower Winfield.

June			Early July			Late July			August		
UW	LW	LW	UW	LW	LW	UW	LW	LW	UW	LW	LW
Day	Day	Night	Day	Day	Night	Day	Day	Night	Day	Day	Night
River surface											
5.3	5.3	5.2	4.8	4.9	6.0	-	12.1	-	7.3	8.6	-
River bottom											
5.1	5.1	5.2	4.9	5.4	5.2	14.2	-	-	8.0	-	9.6
Shoreline											
*	*	*	4.8	5.2	8.6	15.3	12.6	13.9	9.9	-	9.4

* - No samples taken.

\- - No larvae captured.

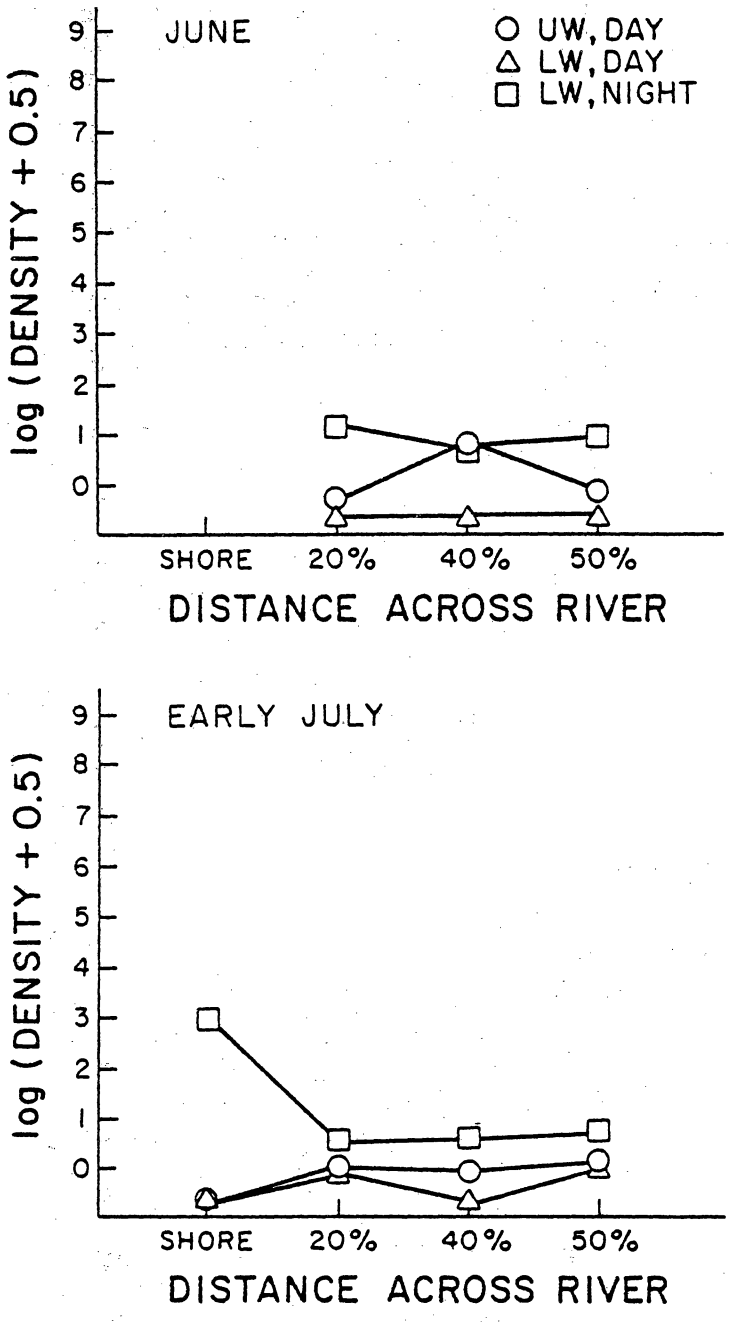


Figure V.10.5. Horizontal distribution of larval Cyprinus carpio in the Kanawha River.

Nearly 92% of these cypinids were captured along the shoreline, where densities were significantly higher than river densities ($p < .001$) on all occasions. The mean length of cyprinids collected along the shoreline usually exceeded that of larvae collected in the river (Table V.10.2). Shoreline densities at LW were significantly higher than those at UW in early July ($p < .001$), but not on other sampling dates (Fig.V.10.6). River densities were also higher at LW in early July ($p < .001$), but higher at UW in June ($p = .003$) and August ($p = .007$). For the river samples, diel variation in abundance was evident only in June when LW densities were higher at night than during the day ($p < .001$). Shoreline densities at LW were significantly higher at night in August ($p < .001$), but not in early or late July ($p = .015$ and $p = .370$, respectively). Vertical trends were not evident except at LW in June when densities were significantly higher near the bottom during both day ($p < .001$) and night ($p < .001$).

Percidae were present early May-August, but were never an abundant taxa. Densities in late May were lower than anticipated; this is likely a result of the high stream flows that occurred just prior and during sampling (Fig.V.10.1). The majority of the percids (70%) were collected along the shoreline where densities exceeded those in the river on several occasions (Fig.V.10.7): early May LW night ($p < .001$), early May UW day ($p = .004$), early July LW night ($p < .001$), and late July LW night ($p = .002$). In early May no percids were captured in the LW

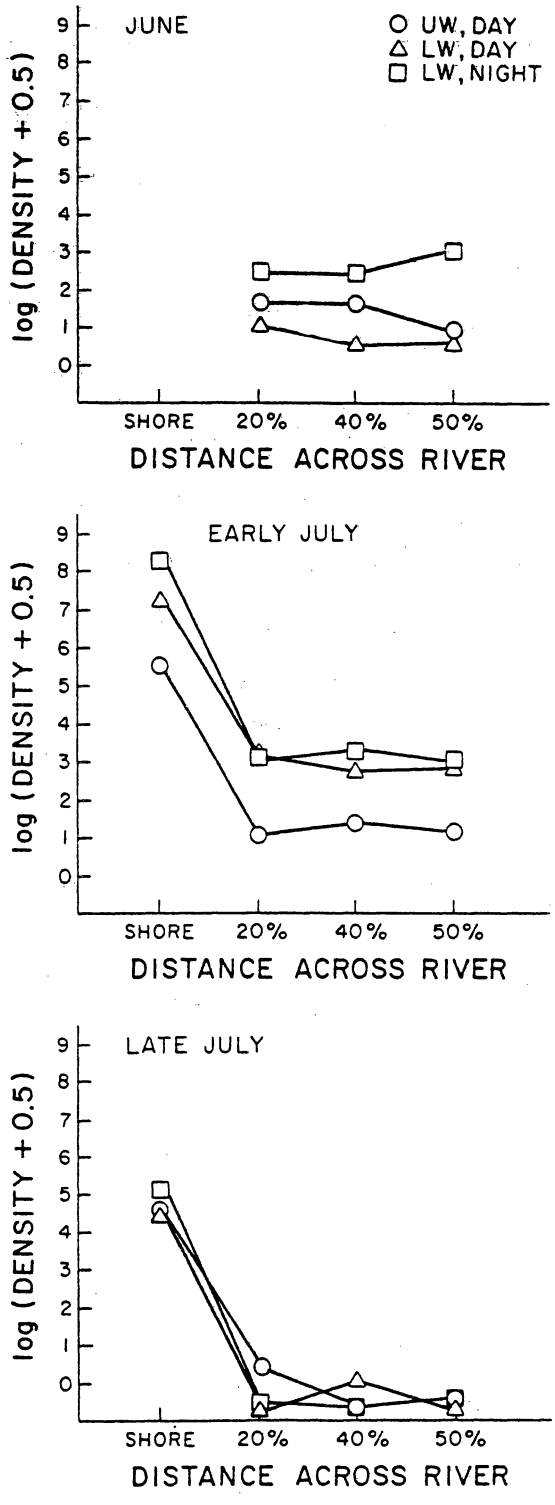


Figure V.10.6. Horizontal distribution of unidentified Cyprinidae larvae in the Kanawha River.

Table V.10.2. Mean lengths (mm) of Unidentified Cyprinidae in distribution samples collected from Winfield Pool. UW = Upper Winfield; LW = Lower Winfield.

June			Early July			Late July			August		
UW	LW	LW	UW	LW	LW	UW	LW	LW	UW	LW	LW
Day	Day	Night	Day	Day	Night	Day	Day	Night	Day	Day	Night
River surface											
4.7	5.1	4.6	4.7	4.9	4.7	7.5	6.1	6.3	4.6	5.7	9.1
River bottom											
4.7	4.8	4.8	4.6	4.7	4.6	8.2	5.8	7.3	4.7	5.6	6.4
Shoreline											
*	*	*	5.3	6.7	6.4	9.4	7.8	8.2	7.8	6.7	8.0

* - No samples taken.

diurnal shoreline samples, but they were relatively abundant in the nocturnal shoreline samples and along the shore at UW. Diel or site variations in shoreline abundance were not evident on the remaining sampling occasions. Densities in the river were too low to discern any vertical distribution trends.

Freshwater drum, a relatively abundant taxon, were present June-August with peak densities occurring in early July. Densities in late July were too low (due to high stream flows) to be useful in interpreting distribution trends. Densities differed between sites in June (higher at UW, $p < .001$), but not on other sampling dates (Fig.V.10.8). Diel variations in shoreline catch were significant during peak abundance in early July. Although drum were abundant in the daylight river samples at both sites, they were absent from the diurnal shoreline samples. At night, however, shoreline densities far exceeded river densities ($p < .001$), and the mean length of larvae captured along the shoreline was higher than that of drum collected in the river (Table V.10.3). In June and August, drum were more abundant in the bottom samples during daylight hours: June LW day ($p < .001$), June UW day ($p = .003$), August LW day ($p = .003$) and August UW day ($p = .010$). This trend was also evident at night during June ($p = .003$). In early July during peak densities, this diurnal preference for the bottom was not evident. Mathews (1984) provided evidence that vertical preference of drum larvae is linked to light intensity. He

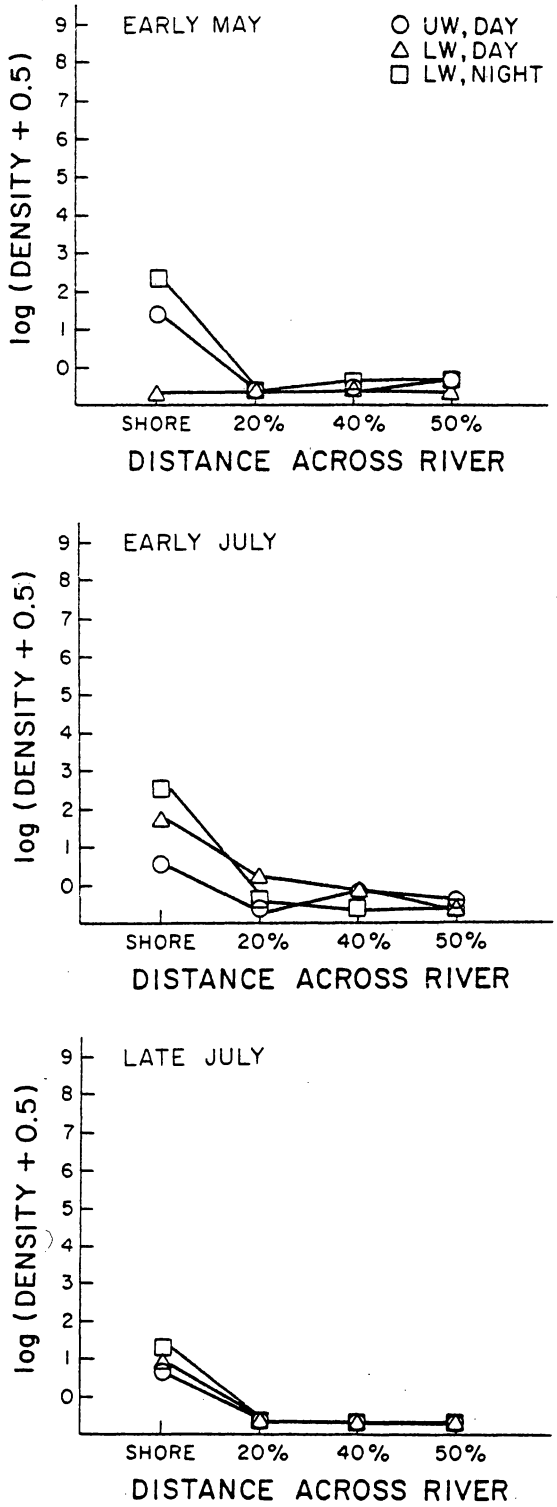


Figure V.10.7. Horizontal distribution of larval Percidae in the Kanawha River.

observed that drum normally concentrate near the bottom, but that increased turbidity resulted in higher abundance at middepth and surface. Although not significantly different at the .01 level ($p = .038$), the diurnal middepth samples at LW in early July contained more larvae than corresponding surface or bottom samples. Light penetration may have been reduced in early July, thereby altering the vertical patterns of drum observed at other times. The mean lengths of larvae did not appear to vary with depth (Table V.10.3).

Densities were too low in late July and August for adequate comparisons of gear types. In early July, gear selectivity was evident and varied with taxa (Table V.10.4). Bongo net samples consistently contained drum larvae, but push nets captured none. Gallagher and Conner (1983) noted similar differences in catch between surface collections made with 0.5 m push nets and a 1.0 m plankton net towed behind a boat. They attributed the observed differences to a greater concentration of drum larvae at a depth of 50-100 cm. Although the bongo nets used in this study were 0.5 m in diameter, the manner in which they were towed resulted in them sampling a greater depth strata than what the push nets could. An opposite trend was observed in cyprinid (excluding carp) catch. Push net samples had both higher densities and larger larvae than the bongo samples. Clupeid catch appeared to be higher in the push nets also, but mean lengths did not differ. This agrees with the findings of Gallagher and Conner (1983)

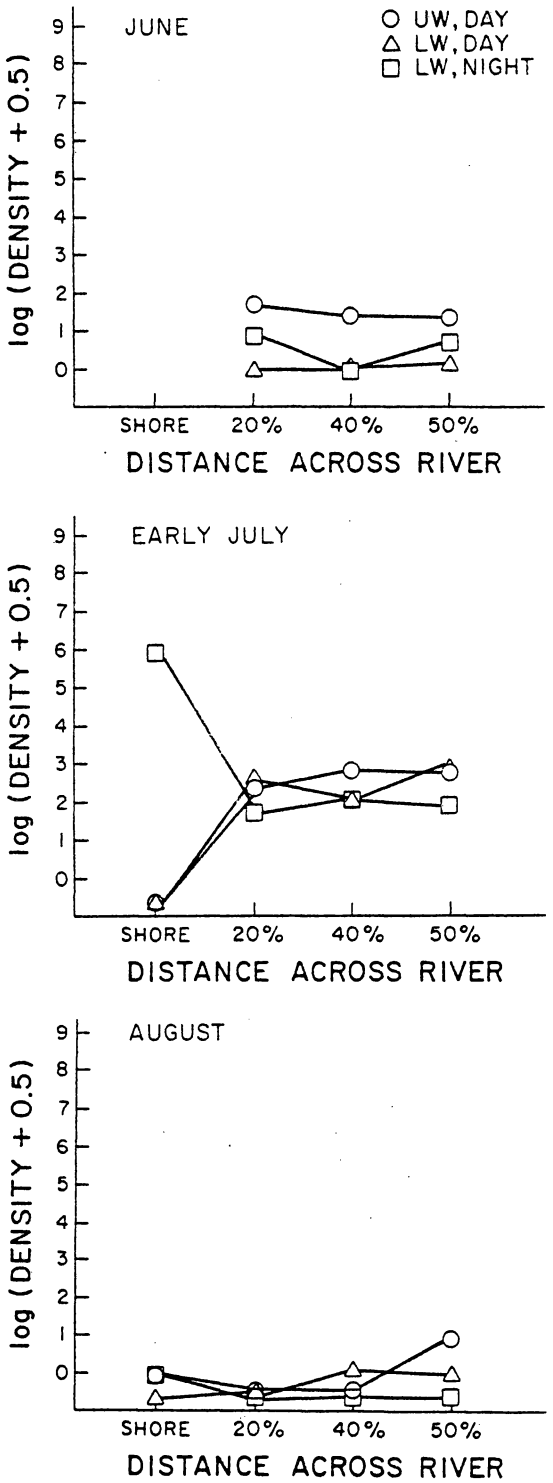


Figure V.10.8. Horizontal distribution of larval Aplodinotus grunniens in the Kanawha River.

Table V.10.3. Mean lengths (mm) of Freshwater Drum in distribution samples collected from Winfield Pool. UW = Upper Winfield; LW = Lower Winfield.

June			Early July			Late July			August		
UW	LW	LW	UW	LW	LW	LW	UW	LW	UW	LW	LW
Day	Day	Night	Day	Day	Night	Day	Day	Night	Day	Day	Night
River surface											
4.5	-	3.2	4.1	4.5	4.5	-	4.2	-	4.8	-	-
River bottom											
4.7	3.9	4.3	4.2	4.7	4.5	-	-	-	4.6	5.6	-
Shoreline											
*	*	*	-	-	5.6	-	-	8.5	5.0	-	6.5

* - No samples taken.

\- - No larvae captured.

which they attributed to the concentration of clupeids close to the surface.

Barge-Induced Mortality

The percentage of live larvae in samples collected "before" and "after" barge passage events did not differ at the .05 level for either June or July (Table V.10.5). In June, the mean percent living larvae in the "before" samples was 31.9%, while 18.3% in the "after" samples. The larvae collected were primarily clupeids (81%), unidentified cyprinids (11%) and freshwater drum (3%). These three taxa were the dominant taxa in July as well, but of different proportions (22, 30, and 45% respectively). "Before" samples in July had a mean of 21.5% living larvae, while "after" samples had 31.0%.

Laboratory Experiments

Mortality due to experimental turbulent high velocity water flows varied among species ($p < .001$). Bluegill larvae (mean length 5.6 mm) had higher mortality rates than the other species ($p < .001$); carp and walleye (mean length 6.6 and 8.2 mm respectively) did not differ from each other ($p = .334$); channel

Table V.10.4. Mean densities (#/100 cubic meters) and lengths (mm) of larvae collected with bongo and push nets near the surface at Upper Winfield in early July.

	Clupeids	Unidentified Cyprinids	Freshwater Drum
Bongo net samples n = 6			
Density	3.62	2.39	11.31
Mean length	4.8	4.7	4.1
Push net samples n = 2			
Density	6.98	12.80	0.00
Mean length	4.9	5.6	-

Table V.10.5. Two-sided Wilsoxon Rank Sum tests on barge-induced larval fish mortality data.

	June	
	Before	**After**
# of tows:	11.0	9.0
mean % live larvae:	31.9	18.3
P - value:	.0742	

	July	
	Before	After
# of tows	21.0	13.0
mean % live larvae	21.5	31.0
P-value	.1231	

catfish (mean length 14.5 mm) exhibited no fatalities. Bluegill mortality varied with water velocity ($p = .001$); mortality averaged 20% at 1000 cm/sec inflow velocity, while only 6% at 800 cm/sec (Table V.10.6). There was no difference in mortality between the two sets of 800 cm/sec runs despite the difference in water volumes pumped through the stress chamber.

The presence of 220 mg/l of sediment increased bluegill mortality significantly (Run #1, $p = .007$; Run #2, $p = .006$). The mean difference in percent mortality between runs with and without sediment was 12.9%. Sediment concentrations up to 880 mg/l did not influence channel catfish survival (100% survival in all experiments).

V.11. ENERGY FLOW MODEL

Estimates of Standing Stocks

Average standing stock estimates for the 19 trophic groups of the model are given in Table V.11.1. The largest estimates are for the particulate organic matter groups. Phytoplankton biomass exceeds periphyton biomass at both sites. Both primary producer groups are more abundant at LW than at UW, particularly phytoplankton. Zooplankton biomass exceeds phytoplankton biomass at each site. The collector gatherers dominate the other insect

Table V.10.6. Mortality of larval fish subjected to experimental turbulence.

	Inflow Velocity and Volume of Water Passing Through Chamber		
	1000 cm/sec 1.27 l/sec	800 cm/sec 1.02 l/sec	800 cm/sec 2.12 l/sec
Bluegill			
Mean % mortality	20.13	6.10	6.00
Standard deviation	11.87	1.54	1.61
# of replicates	9	6	6
Walleye			
Mean % mortality	0.35	0.00	0.00
Standard deviation	0.40	-	-
# of replicates	6	6	6
Common carp			
Mean % mortality	0.25	0.00	0.06
Standard deviation	0.28	-	0.14
# of replicates	6	6	6
Channel catfish			
Mean % mortality	0.00		0.00
Standard deviation	-	*	-
# of replicates	6		6

* No runs made.

groups at both UW and LW. Collector filterers are second in abundance among the insects at UW, but third behind predators at LW. Scraper grazers and shredders are the least abundant insects at both sites (scraper grazer biomass assumed, not measured).

Both mollusc and crayfish abundances are assumed based on trophic demand for these organisms. Herbivore/detritivores and omnivores dominate the fish community at both sites, followed by midwater invertivores. Benthic invertivores, piscivores, and crayfish/piscivores are the least abundant fishes. Both benthic invertivores and crayfish/piscivores are more abundant at UW than LW, while the reverse is true for piscivores.

Estimates of Energy Flows

The estimates for all inter-component energy transfers are shown in Table V.11.2 for UW and Table V.11.3 for LW. Energy flows in the two reaches are sufficiently similar that only those for UW are reviewed below. One meaningful difference between reaches involves the flow of energy representing resuspension of C.P.O.M., X(4) to S.P.O.M., X(3). This flow in the upper, more lotic reach is estimated to be approximately twice the same flow in the lower reach. By far the most thoroughly integrated groups (highest connectivity) are C.P.O.M. X(4), and F.P.O.M. X(5). The highest volumes of energy flow (> 1000 kcal/m^2/yr) exclusively involve organic matter groups. Flows in the range of 500 to

Table V.11.1. Model values for standing stock of trophic groups. (Units are kilocalories per average square meter.)

Trophic Group	Name	River segment UW	River segment LW
x(1)	Phytoplankton	1.797	3.688
x(2)	Periphyton	1.084	1.154
x(3)	SPOM	75.000	100.000
x(4)	CPOM	450.000	350.000
x(5)	FPOM	500.000	500.000
x(6)	Zooplankton	18.690	18.690
x(7)	Coll/Gatherers	2.734	6.064
x(8)	Coll/Filterers	1.197	0.203
x(9)	Scraper Grazers	0.010	0.120
x(10)	Invt Shredders	0.029	0.025
x(11)	Invt Predators	0.153	0.328
x(12)	Molluscs	9.750	9.750
x(13)	Crayfish	3.313	2.994
x(14)	B. Invertivores	3.823	1.535
x(15)	Omnivores	11.240	13.433
x(16)	Cray/Piscivores	0.862	0.217
x(17)	Herb/Detritivores	16.275	15.480
x(18)	M. Invertivores	6.480	4.980
x(19)	Piscivores	1.569	2.143

1,000 kcal/m^2/yr also include organic matter groups as either a donor or recipient. Thus the majority of all energy being transferred directly includes organic matter.

The importance of organic matter is further confirmed by the major role it plays at all of the higher trophic levels. Among the insect invertebrates, the major energy transformation is between collector gatherers X(7) and F.P.O.M. X(5). For noninsect invertebrates molluscs' X(12), utilization of S.P.O.M. X(3) and subsequent conversion to F.P.O.M. X(5) are the major energy flows. The dominant flow among fish groups involves the herbivore/detritivores X(17) and F.P.O.M. X(5).

Of the two primary producer groups, phytoplankton X(1) is utilized more by intermediate trophic level groups (e.g. zooplankton X(6), collector gatherers X(7), collector filterers X(8), molluscs X(12)), whereas periphyton X(2) is more commonly utilized by higher trophic level groups (e.g. crayfish X(13), omnivores X(15), herbivore/detritivores X(17), midwater invertivores X(18)).

Tables V.11.4 and V.11.5 give the energy flows entering from outside the modeled system for UW and LW, respectively. Energy flows exiting the system at UW and LW are listed in Tables V.11.6 and V.11.7, respectively. Primary production is greater at UW than at LW. Leaf inputs at LW are over twice this same flow at UW, and utilization of terrestrial insects by fish is also

Table V.11.2. UW matrix of energy flows (kcal/m^2/yr) between model components.

F(i,j) = flow from donor X(i) to recipient X(j) where i = row and j = column.

	X1	X2	X3	X4	X5
X1	-	-	569.234	-	-
X2	-	-	-	92.722	27.120
X3	-	-	-	42,344.695	10,000.000
X4	-	-	10,592.034	-	10,592.034
X5	-	-	10,000.000	-	-
X6	-	-	171.745	-	-
X7	-	-	-	33.972	965.250
X8	-	-	-	4.894	131.835
X9	-	-	-	0.005	0.140
X10	-	-	-	0.027	3.176
X11	-	-	-	0.943	5.611
X12	-	-	-	10.914	443.978
X13	-	-	-	0.828	8.282
X14	-	-	-	1.306	2.689
X15	-	-	-	3.937	20.263
X16	-	-	-	0.393	1.124
X17	-	-	-	8.711	265.559
X18	-	-	-	5.870	19.846
X19	-	-	-	0.603	1.723

Table V11.2. (Cont'.) UW matrix of energy flows (kcal/m^2/yr) between model components.

F(i,j) = flow from donor X(i) to recipient X(j) where i = row and j = column.

	X6	X7	X8	X9	X10	X11	X12
X1	165.344	41.310	10.604	-	-	-	38.274
X2	-	-	-	0.200	-	-	-
X3	210.436	-	137.293	-	-	-	459.288
X4	-	55.385	-	-	3.529	-	-
X5	-	997.615	-	-	-	-	-
X6	-	0.078	2.599	-	-	-	12.758
X7	-	-	-	-	-	15.055	-
X8	-	-	-	-	-	2.300	-
X9	-	-	-	-	-	-	-
X10	-	-	-	-	-	0.037	-
X11	-	-	-	-	-	1.310	-
X12	-	-	-	-	-	-	-
X13	-	-	-	-	-	-	-
X14	-	-	-	-	-	-	-
X15	-	-	-	-	-	-	-
X16	-	-	-	-	-	-	-
X17	-	-	-	-	-	-	-
X18	-	-	-	-	-	-	-
X19	-	-	-	-	-	-	-

Table V.11.2. (cont'.) UW matrix of energy flows (kcal/m^2/yr) between model components.

F(i,j) = flow from donor X(i) to recipient X(j) where i = row and j = column.

	X13	X14	X15	X16	X17	X18	X19
X1	-	-	-	-	-	-	-
X2	4.969	-	8.571	-	60.648	9.170	-
X3	-	-	-	-	-	-	-
X4	2.485	0.092	5.479	-	3.872	-	-
X5	0.828	0.217	2.850	-	242.805	-	-
X6	-	-	-	-	3.024	-	-
X7	6.667	1.063	2.788	0.004	0.228	2.666	0.001
X8	1.019	0.057	0.496	0.002	0.252	0.612	0.001
X9	-	0.015	-	-	-	-	-
X10	0.016	-	0.066	0.043	-	-	-
X11	0.580	0.247	0.316	0.008	0.023	2.522	-
X12	-	4.886	8.104	-	0.336	15.760	-
X13	-	0.451	0.475	2.249	-	-	0.138
X14	-	-	-	-	-	-	0.391
X15	-	0.010	-	0.261	-	-	0.279
X16	-	-	-	-	-	-	-
X17	-	-	2.720	0.005	-	-	2.403
X18	-	0.174	0.723	0.948	-	-	2.515
X19	-	-	-	-	-	-	-

Table V.11.3. LW matrix of energy flows (kcal/m^2/yr) between model components.

F(i,j) = flow from donor X(i) to recipient X(j) where i = row and j = column.

	X1	X2	X3	X4	X5
X1	-	-	509.201	-	-
X2	-	-	-	42.748	20.800
X3	-	-	-	32,122.137	10,000.000
X4	-	-	5,357.001	-	10,714.002
X5	-	-	10,000.000	-	-
X6	-	-	460.657	-	-
X7	-	-	-	9.564	928.356
X8	-	-	-	0.228	19.954
X9	-	-	-	0.032	0.420
X10	-	-	-	0.021	2.532
X11	-	-	-	4.428	15.612
X12	-	-	-	22.511	443.978
X13	-	-	-	0.749	7.486
X14	-	-	-	0.419	1.719
X15	-	-	-	4.672	22.356
X16	-	-	-	0.084	0.241
X17	-	-	-	2.415	250.277
X18	-	-	-	1.488	15.254
X19	-	-	-	0.830	2.369

Table V.11.3 (cont.). LW matrix of energy flows (kcal/m^2/yr) between model components.

F(i,j) = flow from donor X(i) to recipient X(j) where i = row and j = column.

	X6	X7	X8	X9	X10	X11	X12
X1	146.102	17.730	1.792	-	-	-	38.274
X2	-	-	-	0.600	-	-	-
X3	827.908	-	20.664	-	-	-	459.288
X4	-	20.612	-	-	2.813	-	-
X5	-	1,001.202	-	-	-	-	-
X6	-	0.048	0.322	-	-	-	12.758
X7	-	-	-	-	-	-	18.136
X8	-	-	-	-	-	-	-
X9	-	-	-	-	-	-	-
X10	-	-	-	-	-	0.005	-
X11	-	-	-	-	-	7.858	-
X12	-	-	-	-	-	-	-
X13	-	-	-	-	-	-	-
X14	-	-	-	-	-	-	-
X15	-	-	-	-	-	-	-
X16	-	-	-	-	-	-	-
X17	-	-	-	-	-	-	-
X18	-	-	-	-	-	-	-
X19	-	-	-	-	-	-	-

Table V.11.3. (cont') LW matrix of energy flows (kcal/m^2/yr) between model components.

F(i,j) = flow from donor X(i) to recipient X(j) where i = row and j = column.

	X13	X14	X15	X16	X17	X18	X19
X1	-	-	-	-	-	-	-
X2	4.492	-	10.722	-	69.588	7.050	-
X3	-	-	-	-	-	-	-
X4	2.246	0.335	5.026	-	4.128	-	-
X5	0.748	0.064	2.489	-	218.175	-	-
X6	-	-	-	-	3.708	-	-
X7	5.211	0.152	0.975	-	0.116	3.128	0.001
X8	0.007	0.031	-	-	0.307	1.050	-
X9	-	0.028	-	-	-	-	-
X10	-	-	0.119	-	-	-	-
X11	2.268	0.280	0.856	-	0.191	0.280	-
X12	-	0.726	4.241	-	0.412	12.110	-
X13	-	1.206	1.314	0.379	-	-	0.095
X14	-	-	-	-	-	-	0.543
X15	-	0.028	-	0.214	-	-	0.392
X16	-	-	-	-	-	-	-
X17	-	-	8.257	0.029	-	-	3.353
X18	-	0.489	2.195	0.178	-	-	3.510
X19	-	-	-	-	-	-	-

greater at LW than UW. Utilization of drifting aquatic insects by fish is generally greater at UW than LW, the sole exception being for herbivore/detritivores.

Respiration losses of energy dominate the energy flows leaving the system at both sites. In comparison, emergence of adult insects and fish harvest are relatively minor sinks of energy. Phytoplankton and zooplankton export energy downstream, while S.P.O.M. relies on importation of energy from upstream sources. This import is the largest flow of energy at both sites in the model.

Flow Analysis Results

The results of the flow analyses conducted on the energy transfers of the model are summarized in Tables V.11.8-11. As discussed previously, the N^{**} matrix (Tables V.11.8 and V.11.9) provides the fate of energy stored temporarily in any trophic group. The more efficiently the energy is conducted from lower to higher trophic levels, the greater the percentage of flow through the donor that will reach the recipient. The following review pertains specifically to results observed for the LW segment of the model; the results for the upper Winfield segment are similar. The major entry points of energy into the system are x(1), x(2), x(3) (phytoplankton, periphyton, and S.P.O.M.). The following compares these entry points to determine which one

Table V.11.4. UW matrix of extrasystem energy flows ($kcal/m^2/yr$).

Trophic Group	Name	Flows into System: GPP	leaf fall	terr insects	drift insects
x(1)	Phytoplankton	1,640.000	-	-	-
x(2)	Periphyton	271.200	-	-	-
x(3)	SPOM	-	26.660	-	-
x(4)	CPOM	-	-	-	-
x(5)	FPOM	-	-	-	-
x(6)	Zooplankton	-	-	-	-
x(7)	Coll/Gatherers	-	-	-	-
x(8)	Coll/Filterers	-	-	-	-
x(9)	Scraper Grazers	-	-	-	-
x(10)	Invt Shredders	-	-	-	-
x(11)	Invt Predators	-	-	-	-
x(12)	Molluscs	-	-	-	-
x(13)	Crayfish	-	-	-	-
x(14)	B. Invertivores	-	-	0.184	0.950
x(15)	Omnivores	-	-	5.403	1.586
x(16)	Cray/Piscivores	-	-	-	0.228
x(17)	Herb/Detritivores	-	-	-	0.501
x(18)	M. Invertivores	-	-	-	23.200
x(19)	Piscivores	-	-	-	0.015

Table V.11.5. LW matrix of extrasystem energy flows ($kcal/m^2/yr$).

Trophic Group	Name	Flows into System: GPP	leaf fall	terr insects	drift insects
x(1)	Phytoplankton	1,438.000	-	-	-
x(2)	Periphyton	208.000	-	-	-
x(3)	SPOM	-	54.769	-	-
x(4)	CPOM	-	-	-	-
x(5)	FPOM	-	-	-	-
x(6)	Zooplankton	-	-	-	-
		-	-	-	-
x(7)	Coll/Gatherers	-	-	-	-
x(8)	Coll/Filterers	-	-	-	-
x(9)	Scraper Grazers	-	-	-	-
x(10)	Invt Shredders	-	-	-	-
x(11)	Invt Predators	-	-	-	-
x(12)	Molluscs	-	-	-	-
x(13)	Crayfish	-	-	-	-
x(14)	B. Invertivores	-	-	0.782	0.804
x(15)	Omnivores	-	-	8.007	1.054
x(16)	Cray/Piscivores	-	-	-	0.002
x(17)	Herb/Detritivores	-	-	-	0.616
x(18)	M. Invertivores	-	-	-	17.832
x(19)	Piscivores	-	-	-	0.005

Table V.11.6. UW matrix of extrasystem energy flows ($kcal/m^2/yr$).

Trophic Group	Name	Flows out of System			
		resp	emergence	harvest	net transport
x(1)	Phytoplankton	410.000	-	-	+405.234
x(2)	Periphyton	67.800	-	-	-
x(3)	SPOM	767.700	-	-	-32,559.739
x(4)	CPOM	21,254.910	-	-	-
x(5)	FPOM	11,244.315	-	-	-
x(6)	Zooplankton	160.386	-	-	+25.190
x(7)	Coll/Gatherers	58.201	8.493	-	-
x(8)	Coll/Filterers	7.804	1.224	-	-
x(9)	Scraper Grazers	0.040	-	-	-
x(10)	Invt Shredders	0.157	0.007	-	-
x(11)	Invt Predators	6.906	0.236	-	-
x(12)	Molluscs	26.342	-	-	-
x(13)	Crayfish	4.141	-	-	-
x(14)	B. Invertivores	3.960	-	-	-
x(15)	Omnivores	13.514	-	1.313	-
x(16)	Cray/Piscivores	1.837	-	0.394	-
x(17)	Herb/Detritivores	32.291	-	-	-
x(18)	M. Invertivores	23.854	-	-	-
x(19)	Piscivores	2.814	-	0.603	-

Table V.11.7. LW matrix of extrasystem energy flows ($kcal/m^2/yr$).

Trophic Group	Name	Flows out of system: resp	emergence	harvest	net transport
x(1)	Phytoplankton	359.500	-	-	+365.401
x(2)	Periphyton	52.000	-	-	-
x(3)	SPOM	3,212.000	-	-	-30,260.549
x(4)	CPOM	16,106.163	-	-	-
x(5)	FPOM	11,222.678	-	-	-
x(6)	Zooplankton	415.724	-	-	+80.793
x(7)	Coll/Gatherers	71.560	2.393	-	-
x(8)	Coll/Filterers	1.144	0.057	-	-
x(9)	Scraper Grazers	0.120	-	-	-
x(10)	Invt Shredders	0.131	0.005	-	-
x(11)	Invt Predators	19.158	1.109	-	-
x(12)	Molluscs	26.342	-	-	-
x(13)	Crayfish	3.743	-	-	-
x(14)	B. Invertivores	2.244	-	-	-
x(15)	Omnivores	16.035	-	1.558	-
x(16)	Cray/Piscivores	0.393	-	0.084	-
x(17)	Herb/Detritivores	32.910	-	-	-
x(18)	M. Invertivores	18.336	-	-	-
x(19)	Piscivores	3.870	-	0.830	-

most efficiently provides support to the other trophic groups in in the model.

Energy flowing through phytoplankton X(1) flows to zooplankton X(6), invertebrate collector-filterers X(8), and molluscs X(12) more efficiently than energy originating through periphyton X(2) or S.P.O.M. X(3). These recipients all function by straining or sorting through the water column for food, and would generally be considered as among the middle trophic level.

Energy flowing through S.P.O.M. X(3) is more efficiently passed to S.P.O.M. itself, (via settling and subsequent resuspension), C.P.O.M. X(4) (via settling), and invertebrate shredders X(10) than energy flowing through either of the primary producers. These recipient groups would generally be included in the lower trophic level.

Energy more efficiently reaches F.P.O.M. X(5), invertebrate collector gatherers X(7), scraper-grazers X(9), predators X(11), crayfish X(13) and virtually all of the fish groups X(14-19) from periphyton X(2) than from the other two major entry points. Both in terms of number and position in the ecosystem, then, it appears that energy once present as periphyton reaches the highest trophic levels more efficiently than either of the other major sources. Energy originating through periphyton is also more efficiently passed out of the system as emerging insects or through fish harvest than energy originating via phytoplankton or S.P.O.M..

Table V.11.8. UW N** matrix. Elements give the percentage of energy flow through X(j) destined to support flow in X(i), rounded to first decimal place. tr = trace (<.1%).

N**(i,j) = element in row(i),column(j)

	C1	C2	C3	C4	C5	C6	C7	C8	C9	C10
R1	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R2	0.0	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R3	67.7	46.3	161.6	59.2	75.7	77.3	69.6	69.5	57.1	70.4
R4	53.5	72.2	127.1	146.6	59.7	60.9	58.3	58.4	48.8	56.7
R5	33.6	63.5	66.7	50.8	136.5	36.5	123.7	123.4	100.8	126.2
R6	10.3	0.2	0.6	0.2	0.3	100.3	0.3	0.3	0.2	0.3
R7	4.1	2.9	3.1	2.4	6.1	1.7	105.6	5.5	4.5	5.7
R8	0.9	0.1	0.4	0.2	0.2	0.9	0.2	100.2	0.1	0.2
R9	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R10	tr	tr	tr	tr	tr	tr	tr	tr	tr	100.0
R11	0.1	tr	0.1	tr	0.1	tr	1.6	1.7	0.1	1.2
R12	3.3	0.4	1.4	0.5	0.7	4.1	0.6	0.6	0.5	0.6
R13	tr	1.9	tr	tr	0.1	tr	0.7	0.8	tr	0.5
R14	tr	0.1	tr	tr	tr	tr	0.2	0.1	7.5	tr
R15	0.1	3.5	0.1	tr	0.1	0.1	0.4	0.5	0.1	2.0
R16	tr	0.3	tr	tr	tr	tr	0.1	0.1	tr	1.3
R17	0.5	23.1	0.7	0.6	1.5	1.2	1.3	1.5	1.1	1.4
R18	0.1	3.4	0.1	tr	tr	0.1	0.5	0.7	tr	0.2
R19	tr	0.4	tr	tr	tr	tr	tr	0.1	0.4	tr

Table V.11.8. (cont') UW N** matrix (cont). Elements give the percentage of energy flow through X(j) destined to support flow in X(i), rounded to first decimal place. tr = trace (<.1%).

N**(i,j) = element in row(i),column(j)

C11	C12	C13	C14	C15	C16	C17	C18	C19	
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R1
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R2
36.0	69.4	47.2	35.0	45.1	28.9	66.8	36.9	28.9	R3
36.2	57.5	44.5	43.7	45.6	33.3	55.6	40.8	33.3	R4
60.5	123.4	81.1	54.1	75.6	46.3	118.7	59.9	46.3	R5
0.1	0.3	0.2	0.1	0.2	0.1	0.3	0.1	0.1	R6
2.7	5.6	3.7	2.5	3.4	2.1	5.3	2.7	2.1	R7
0.1	0.2	0.1	0.1	0.1	0.1	0.2	0.1	0.1	R8
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R9
tr	tr	tr	tr	tr	tr	tr	tr	tr	R10
107.6	0.1	0.1	tr	0.1	tr	0.1	tr	tr	R11
0.3	100.6	0.4	0.3	0.4	0.3	0.6	0.3	0.3	R12
3.4	tr	100.0	tr	tr	tr	tr	tr	tr	R13
1.6	1.0	2.7	100.0	tr	tr	tr	0.3	tr	R14
2.1	1.7	2.9	tr	100.0	tr	1.0	1.4	tr	R15
0.8	0.1	13.6	tr	0.7	100.0	tr	1.8	tr	R16
0.8	1.4	1.0	0.6	0.8	0.5	101.3	0.7	0.5	R17
14.5	3.1	tr	tr	tr	tr	tr	100.0	tr0	R18
0.8	tr	1.0	4.7	0.7	tr	0.8	4.7	100.0	R19

Table V.11.9. LW N** matrix. Elements give the percentage of energy flow through X(j) destined to support flow in X(i), rounded to first decimal place. trace (<.1%).

N**(i,j) = element in row(i),column(j)

	C1	C2	C3	C4	C5	C6	C7	C8	C9	C10
R1	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R2	0.0	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R3	63.2	41.8	150.5	48.6	70.5	72.3	64.4	64.7	53.4	65.6
R4	43.8	50.7	103.8	133.5	48.7	49.9	45.6	45.9	42.7	46.5
R5	34.1	74.1	71.8	58.2	138.8	36.1	126.3	126.8	103.0	128.7
R6	11.3	0.7	2.7	0.9	1.3	101.3	1.1	1.1	0.9	1.1
R7	2.8	3.3	3.3	2.7	6.2	1.6	105.7	5.7	4.6	5.8
R8	0.2	tr	0.1	tr	tr	0.1	tr	100.0	tr	tr
R9	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R10	tr	tr	tr	tr	tr	tr	tr	tr	tr	100.0
R11	0.1	0.1	0.1	0.1	0.1	tr	2.2	0.1	0.1	0.3
R12	3.4	0.4	1.5	0.5	0.7	2.0	0.6	0.6	0.5	0.7
R13	tr	2.2	tr	tr	tr	tr	0.6	0.1	tr	0.1
R14	tr	0.2	tr	tr	tr	tr	0.1	0.2	4.7	tr
R15	0.1	6.5	0.1	0.1	0.1	0.1	0.3	0.4	0.1	4.3
R16	tr	0.1	tr	tr	tr	tr	tr	tr	tr	tr
R17	0.4	34.2	0.7	0.6	1.4	0.7	1.3	2.6	1.0	1.3
R18	0.1	3.4	tr	tr	tr	0.1	tr	4.6	tr	tr
R19	tr	0.8	tr	tr	tr	tr	0.1	0.4	0.5	0.1

Table V.11.9. (Cont') LW N** matrix. Elements give the percentage of energy flow through X(j) destined to support flow in X(i), rounded to first decimal place. tr = trace (<.1%).

N**(i,j) = element in row(i),column(j)

C11	C12	C13	C14	C15	C16	C17	C18	C19	
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R1
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R2
33.5	64.7	44.6	31.6	40.2	26.3	61.2	32.5	26.3	R3
33.8	49.3	37.8	31.5	38.2	28.6	43.5	27.7	28.6	R4
62.0	125.6	85.2	58.7	75.3	47.8	120.0	62.1	47.7	R5
0.6	1.1	0.8	0.6	0.7	0.5	1.1	0.6	0.5	R6
2.8	5.6	3.8	2.6	3.4	2.2	5.4	2.8	2.1	R7
tr	tr	tr	tr	tr	tr	tr	tr	tr	R8
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R9
tr	tr	tr	tr	tr	tr	tr	tr	tr	R10
117.8	0.1	0.1	0.1	0.1	tr	0.1	0.1	tr	R11
0.3	100.7	0.4	0.3	0.4	0.3	0.6	0.3	0.3	R12
5.2	tr	100.0	tr	tr	tr	tr	tr	tr	R13
1.1	0.2	8.1	100.0	0.1	tr	tr	1.2	tr	R14
2.5	1.0	8.8	tr	100.0	tr	2.8	5.3	tr	R15
0.1	tr	2.6	tr	tr	100.0	tr	tr	tr	R16
1.0	1.3	0.8	0.6	0.7	0.5	101.2	0.6	0.5	R17
0.7	2.4	tr	tr	tr	tr	tr	100.0	tr	R18
0.2	0.2	1.6	11.0	0.9	tr	1.2	8.7	100.0	R19

Table V.11.10. UW N* matrix. Elements give the percentage of energy flow through X(i) dependent upon flow from X(j), rounded to first decimal place. tr = trace (< .1%).

N*(i,j) = element in row(i),column(j)

	C1	C2	C3	C4	C5	C6	C7	C8	C9	C10
R1	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R2	0.0	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R3	2.1	0.2	161.6	46.7	31.6	0.5	1.4	1.9	tr	tr
R4	2.1	0.5	161.2	146.7	31.6	0.5	1.5	0.2	tr	tr
R5	2.5	0.8	160.0	96.1	136.5	0.6	6.0	0.8	tr	tr
R6	45.2	0.1	90.5	26.2	17.7	100.3	0.8	0.1	tr	tr
R7	6.1	0.7	154.0	95.0	126.0	0.6	105.6	0.8	tr	tr
R8	9.7	0.2	149.0	43.1	29.1	2.2	1.3	100.2	tr	tr
R9	0.0	100.0	0.0	0.0	0.0	0.0	0.0	0.0	100.0	0.0
R10	2.1	0.5	161.2	146.6	31.6	0.5	1.5	0.2	tr	100.0
R11	6.6	0.7	153.4	88.2	113.0	0.8	91.6	13.9	tr	0.2
R12	10.5	0.2	147.7	42.7	28.9	3.0	1.3	0.2	tr	tr
R13	3.7	30.4	108.9	70.9	68.0	0.5	46.3	7.0	tr	0.1
R14	7.5	2.5	124.8	48.5	44.5	1.9	19.8	1.7	0.2	tr
R15	3.5	23.9	89.5	50.7	38.8	0.9	10.2	1.7	tr	0.2
R16	3.5	24.3	89.2	53.7	49.6	0.6	31.3	4.9	tr	1.2
R17	2.4	20.1	127.1	77.1	107.0	1.5	4.8	0.7	tr	tr
R18	3.8	17.1	59.6	21.8	20.3	1.0	9.9	1.9	tr	tr
119	3.4	18.0	95.2	49.3	60.2	1.2	9.3	1.5	tr	tr

Table V.11.10. (Cont') UW N* matrix. Elements give the percentage of energy flow through X(i) destined to support flow in X(j), rounded to first decimal place. tr = trace (<.1%)

N*(i,j) = element in row(i),column(j)

C11	C12	C13	C14	C15	C16	C17	C18	C19	
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R1
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R2
tr	0.7	tr	tr	tr	tr	tr	tr	tr	R3
tr	0.7	tr	tr	tr	tr	0.4	0.1	tr	R4
0.1	2.8	0.1	tr	0.1	tr	1.6	0.1	tr	R5
tr	0.4	tr	tr	tr	tr	0.2	tr	tr	R6
tr	2.6	0.1	tr	0.1	tr	1.5	0.1	tr	R7
tr	0.6	tr	tr	tr	tr	0.4	tr	tr	R8
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R9
tr	0.6	tr	tr	tr	tr	0.4	tr	tr	R10
107.6	2.3	tr	tr	tr	tr	1.4	0.1	tr	R11
tr	100.6	tr	tr	tr	tr	0.4	tr	tr	R12
3.8	1.4	100.0	tr	0.1	tr	0.8	tr	tr	R13
3.5	60.1	5.4	100.0	0.2	tr	0.5	2.1	tr	R14
1.0	21.8	1.2	tr	100.0	tr	7.3	1.9	tr	R15
3.8	9.9	60.1	tr	7.0	100.0	1.2	25.5	tr	R16
tr	2.3	tr	tr	0.1	tr	101.3	0.1	tr	R17
5.0	29.6	tr	tr	tr	tr	0.2	100.0	tr	R18
2.6	19.1	2.9	6.8	4.9	tr	42.9	44.1	100.0	R19

Table V.11.11. LW N* matrix. Elements give the percentage of energy flow through X(i) dependent upon flow from X(j), rounded to first decimal place. trace (<.1%).

N*(i,j) = element in row(i),column(j)

	C1	C2	C3	C4	C5	C6	C7	C8	C9	C10
R1	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R2	0.0	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R3	1.9	0.2	150.5	33.5	33.9	1.5	1.4	tr	tr	tr
R4	2.0	0.3	150.3	133.5	33.9	1.5	1.5	tr	tr	tr
R5	2.2	0.7	149.2	83.6	138.8	1.6	5.8	0.1	tr	tr
R6	16.7	0.2	127.9	28.5	28.8	101.3	1.2	tr	tr	tr
R7	3.9	0.7	146.7	83.1	134.3	1.5	105.7	0.1	tr	tr
R8	9.9	0.2	138.3	30.8	31.2	2.8	1.3	100.0	tr	tr
R9	0.0	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R10	2.0	0.3	150.3	133.5	33.9	1.5	1.5	tr	tr	100.0
R11	1.6	0.3	60.2	34.1	55.2	0.6	43.4	tr	tr	tr
R12	9.7	0.2	138.7	30.9	31.3	3.9	1.3	tr	tr	tr
R13	2.0	30.4	90.2	58.3	67.2	0.9	43.9	0.1	tr	tr
R14	2.7	10.0	69.4	35.5	34.9	1.2	17.7	0.9	0.6	tr
R15	1.9	29.9	68.6	39.2	40.1	1.2	6.2	0.2	tr	0.3
R16	2.3	27.0	77.4	43.9	50.7	1.1	24.5	0.7	tr	0.1
R17	1.9	23.9	113.6	63.7	102.9	2.4	4.4	0.2	tr	tr
R18	3.4	17.1	55.5	16.3	20.4	1.3	8.7	2.6	tr	tr
R19	2.6	20.3	82.2	39.4	58.0	1.8	7.8	1.3	tr	tr

Table V.11.11. (cont') LW N* matrix. Elements give the percentage of energy flow through X(i) destined to support flow in X(j), rounded to first decimal place. tr = trace (<.1%).

N*(i,j) = element in row(i),column(j)

C11	C12	C13	C14	C15	C16	C17	C18	C19	
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R1
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R2
tr	0.7	tr	tr	tr	tr	0.4	tr	tr	R3
0.1	0.8	tr	tr	0.1	tr	0.4	tr	tr	R4
0.1	2.9	0.1	tr	0.2	tr	1.6	0.1	tr	R5
tr	0.6	tr	tr	tr	tr	0.3	tr	tr	R6
0.1	2.8	0.1	tr	0.1	tr	1.5	0.1	tr	R7
tr	0.7	tr	tr	tr	tr	0.4	tr	tr	R8
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	R9
0.1	0.8	tr	tr	0.1	tr	0.4	tr	tr	R10
117.8	1.1	tr	tr	tr	tr	0.6	tr	tr	R11
tr	100.7	tr	tr	tr	tr	0.4	tr	tr	R12
17.9	1.4	100.0	tr	0.1	tr	0.8	0.1	tr	R13
11.2	18.4	24.5	100.0	0.6	tr	0.1	10.0	tr	R14
2.8	11.6	2.9	tr	100.0	tr	18.7	4.9	tr	R15
9.4	10.4	48.1	tr	26.7	100.0	9.1	23.5	tr	R16
0.2	2.3	tr	tr	0.1	tr	101.2	0.1	tr	R17
0.8	29.6	tr	tr	tr	tr	0.2	100.0	tr	R18
1.6	16.0	3.1	6.9	5.1	tr	44.0	45.4	100.0	R19

It is important to note that transfer efficiency does not necessarily correlate with absolute throughput in the ecosystem. Far more energy at one time flows through the suspended particulate matter group than either of the primary producer groups (roughly 50,000 kcal per square meter per year through S.P.O.M. compared to 1500 kcal through phytoplankton or 250 kcal through periphyton). Thus while the system may more efficiently transfer energy originating as periphyton, even with less retentiveness the large volume of S.P.O.M. ultimately makes it the more important food source.

We also consistently observe that for any donor group, a large percentage of the energy processed is ultimately destined to support one of the particulate organic matter groups X(3-5) rather than other groups. This means that nonliving organic matter, and not production, is the fate of most of the energy retained in the ecosystem.

The conclusions drawn above are born out more strongly by the second energy flow analysis. As discussed in the methods, the N^* matrix (Tables V.11.10 and V.11.11) provides a measure of the degree to which a trophic group depends on other trophic groups for energy flow. In arriving at the final percentage dependence figure, the analysis integrates all avenues (direct as well as indirect) of energy flow to each trophic group. For

example, the sixteenth row and third column of Table V.11.11 gives the percentage (77.4%) of flow through the crayfish/piscivore component which once passed through the S.P.O.M. component. Note that in Table V.9.13 the crayfish/piscivores do not directly consume detritus at all. However, the invertebrates, crayfish, and other fish which are consumed by the crayfish/piscivores themselves consume a large proportion of detritus in their own diets. Since the middle trophic level (crayfish, invertebrates, and other fish) which supports the higher trophic level (crayfish/piscivores) is itself supported by detritus, so then is the higher trophic level, albeit indirectly.

Since as demonstrated above, direct and indirect energy pathways are confounded, the most appropriate conclusions to draw from this analysis pertain to the ultimate sources of energy supporting each trophic group. In this case we compare dependence of the various trophic groups on autochthonous sources of energy (matter at some point in the form of phytoplankton or periphyton) versus allochthonous sources (mainly S.P.O.M., C.P.O.M., and F.P.O.M.). The results clearly demonstrate a higher dependence on allochthonous energy sources. Energy exiting the ecosystem through sixteen of the nineteen trophic groups originates predominantly in the form of S.P.O.M.. In approximately half of these cases, the element of the matrix exceeds unity, indicating that the energy on which these groups

depend has cycled through the suspended matter class at least one time. In contrast, dependence upon primary producers is substantially less. Most trophic groups (exception zooplankton) rely on phytoplankton for less than ten percent of the energy flowing through them. Dependence on energy flow which had passed through periphyton is greater (up to 27%), but still not of the same magnitude as dependence on organic matter sources. Phytoplankton, periphyton, and the invertebrate scraper grazers (with token representation) are the only trophic groups which depend more on autochthonous than allochthonous energy sources.

One notes that different trophic levels exhibit differential dependence upon the two groups of primary producers. Aquatic macroinvertebrate insects rely more on phytoplankton-based energy sources than periphyton-based sources. In constrast, all of the fish trophic groups show a greater dependence on energy channeled through periphyton than through phytoplankton.

Zooplankton are not significantly relied upon as a source of energy for any other trophic group in the model. The key trophic group among the insect invertebrates is the collector gatherers X(7). Collector gatherers consistently mediate more energy flow to invertebrate insect predators, crayfish, and the fish groups than other insects.

Tow Traffic Simulation Results

Four different levels of tow traffic were tested for their effects on the biota of the Kanawha River. These included simulations of traffic corresponding to baseline (1982-83), future (2040) without project, Winfield replacement, and both Marmet and Winfield replacements. The experimental simulation results are summarized in a separate section.

The reductions in standing stocks of the nineteen trophic groups of the model are listed in Tables V.11.12 and V.11.13 for UW and LW, respectively. All trophic groups responded to the perturbation with declines in standing stock. Overall these are not large reductions, even in Scenario #4. The largest single declines were predicted for phytoplankton X(1) at UW and for periphyton X(2) for LW, although the percentage declines are similar for both groups at either site. The least affected trophic groups were the cray/piscivores X(16), followed closely by the benthic invertivores X(14) and piscivores X(19) in both reaches.

If the trophic groups are aggregated into general classes, the greatest declines were predicted for primary producers, followed by zooplankton, insect and non-insect invertebrates, and lastly fish groups. Comparing the two reaches, it appears that declines in standing stock of most trophic groups are more often greater at LW than UW. Consistent reverses of this trend include the collector gatherers X(7), scraper grazers X(9), and invertebrate predators X(11).

Table V.11.12. Percentage declines in standing stock of UW trophic groups.

Trophic Group	Name	UW Tow Traffic Scenarios S1	S2	S3	S4	S5
x(1)	Phytoplankton	0.00	4.93	5.94	15.17	100.00
x(2)	Periphyton	0.00	4.97	5.58	14.77	99.77
x(3)	SPOM	0.00	0.08	0.09	0.24	1.59
x(4)	CPOM	0.00	0.08	0.08	0.23	1.49
x(5)	FPOM	0.00	0.06	0.06	0.16	1.08
x(6)	Zooplankton	0.00	0.95	1.02	2.82	18.49
x(7)	Coll/Gatherers	0.00	0.24	0.27	0.73	4.73
x(8)	Coll/Filterers	0.00	0.42	0.46	1.27	8.30
x(9)	Scraper Grazers	0.00	1.97	2.08	5.83	37.01
x(10)	Invt Shredders	0.00	0.06	0.06	0.18	1.15
x(11)	Invt Predators	0.00	0.23	0.25	0.68	4.49
x(12)	Molluscs	0.00	0.35	0.37	1.03	6.77
x(13)	Crayfish	0.00	0.19	0.20	0.56	3.56
x(14)	B. Invertivores	0.00	0.01	0.01	0.03	0.18
x(15)	Omnivores	0.00	0.10	0.10	0.29	1.85
x(16)	Cray/Piscivores	0.00	0.01	0.01	0.02	0.16
x(17)	Herb/Detritivores	0.00	0.40	0.42	1.18	7.47
x(18)	M. Invertivores	0.00	0.17	0.18	0.52	3.31
x(19)	Piscivores	0.00	0.01	0.01	0.03	0.21

Table V.11.13. Percentage declines in standing stock of LW trophic groups.

Trophic Group	Name	LW Tow Traffic Scenarios S1	S2	S3	S4	S5
x(1)	Phytoplankton	0.00	6.84	9.28	17.52	99.97
x(2)	Periphyton	0.00	7.28	10.14	19.00	98.79
x(3)	SPOM	0.00	0.10	0.14	0.27	1.51
x(4)	CPOM	0.00	0.09	0.13	0.24	1.31
x(5)	FPOM	0.00	0.06	0.08	0.15	0.86
x(6)	Zooplankton	0.00	0.80	1.13	2.07	11.63
x(7)	Coll/Gatherers	0.00	0.16	0.22	0.40	2.28
x(8)	Coll/Filterers	0.00	0.59	0.83	1.53	8.62
x(9)	Scraper Grazers	0.00	0.72	1.02	1.87	10.22
x(10)	Invt Shredders	0.00	0.07	0.09	0.17	0.95
x(11)	Invt Predators	0.00	0.05	0.07	0.14	0.76
x(12)	Molluscs	0.00	0.44	0.62	1.14	6.42
x(13)	Crayfish	0.00	0.22	0.32	0.58	3.15
x(14)	B. Invertivores	0.00	0.01	0.01	0.02	0.10
x(15)	Omnivores	0.00	0.12	0.18	0.32	1.77
x(16)	Cray/Piscivores	0.00	0.01	0.01	0.02	0.12
x(17)	Herb/Detritivores	0.00	0.55	0.78	1.43	7.70
x(18)	M. Invertivores	0.00	0.22	0.31	0.56	3.05
x(19)	Piscivores	0.00	0.01	0.02	0.03	0.21

Experimental Simulation Results

In an experiment designed to test the importance of autochthonous sources of energy, gross primary production by phytoplankton and by periphyton were arbitrarily set to ten percent of their original values. The model responded with larger standing stock reductions than those observed in the various tow traffic scenarios, but the effects at the upper trophic levels remained slight. Significant reductions in standing stock were predicted for zooplankton (18.49% at UW) and scraper-grazers (37.01% at UW). The simulation time required to bring the system to a new equilibrium exceeded that available. Thus the simulation was conducted with the same duration as the traffic scenarios (2 months) in order to compare the response of the model up to that point.

Primary producers were driven essentially out of the system when primary production rates were reduced to 10% of their former value. Even in the effective absence of autochthonous sources of energy, the remaining groups in the ecosystem did not respond with substantial declines, although it is important to recognize that the remaining groups may not have settled to an equilibrium state.

The results of the experiment will be reviewed for three categories of trophic groups; invertebrates, vertebrates, and organic matter groups. The largest declines among invertebrate groups were, as in the traffic simulations, for zooplankton (UW 18.49%, LW 11.63%), collector filterers (UW 8.3%, LW 8.62%), scraper grazers (UW 37.01%, LW 10.22%), and molluscs (UW 6.7%, LW 6.42%). Other invertebrate groups, for example the collector gatherers, shredders, invertebrate predators, and crayfish, demonstrate declines of less than five percent in all cases at either pool site. Declines in standing stock of the vertebrate trophic groups ranged from 0.16 to 3.31% at UW and 0.12 to 3.05% at LW. The single exception to the above is for the herbivore/detritivores with declines of 7.47 and 7.7% at UW and LW respectively.

Suspended particulate, benthic coarse, and benthic fine organic matter decline, but still by small amounts: 1.59, 1.49, 1.08% at UW, and 1.51, 1.31, 0.86% at LW, respectively.

VI. DISCUSSION

VI.0 WATER QUALITY

Temperature and Specific Conductance

Water temperature followed the expected seasonal trend but deviated occasionally for short periods (Appendices A0.1.1-13 and A0.2.1-12 due to large differences in air temperature or precipitation. The effect of local weather may also be responsible for the apparent differences between Upper Winfield and Lower Winfield. The sampling dates with large differences in water temperature were also noted for parallel differences in air temperature.

Specific conductance is a measure of the capacity of water to carry a current, is related to the concentration of ionized substances, and is an estimate of the dissolved solids content. The conductivity in the Kanawha River appears to vary seasonally with the discharge. It was lowest in the spring during high discharge, and it was highest in the summer during low discharge. Changes in conductivity can be attributed to dilution during high discharge and to evaporation during low discharge. An increase in specific conductance occurs downstream, as well as evaporation, and is a common occurrence in many rivers (Hynes 1970).

pH and Alkalinity

The pH was consistently measured at levels acceptable to the biota. The higher values of July and August were probably due to the high photosynthetic rate of the phytoplankton which probably removed most of the available dissolved carbon dioxide from the water. The pH values measured were all within the W.Va. State Standard Range of pH 6 to 9 (West Virginia Department of Natural Resources 1980).

Alkalinity is a measure of the capacity to neutralize acids. At the pH range of the Kanawha River, bicarbonate is probably the major inorganic ion controlling the buffering capacity. Sources of buffering ions include precipitation, dissolution, and erosion within the watershed, and the final concentration in the river is subject to chemical equilbria, dilution, and evaporation. The biota, particularly photosynthesizers, may affect the chemical equilbria between bicarbonate and carbon dioxide during uptake of the latter, but this was not apparent with the wide variability of alkalinity measured. The buffering capacity is also an important concern when considering the impact of acid rain. The alkalinity values measured show a fairly well buffered river.

Dissolved Oxygen

Dissolved oxygen concentration was close to saturation for most of the year with the exception of the summer values which

were attributed to higher photosynthetic rates (supersaturation) and higher respiration rates (lower saturation levels) in the plankton. The aeration occurring below a dam maintains oxygen saturated conditions year-round, while the heterotrophic metabolism of the river tends to deplete the oxygen while on the journey to the lower pool. The increasing depth of the pool downstream from a dam also places a larger percentage of the water column in low light conditions where photosynthesis is not possible, but where respiration continues. All dissolved oxygen measurements were well above the W.Va. State Standard of 4 mgO_2/l (West Virginia Department of Natural Resources 1980).

Transparency and Light Penetration

Light extinction is a result of the absorption of light by the water, any colored constituents, and any particulate matter (which may also reflect and scatter light). The depth of 1% incident light is the depth above which photosynthesis is most probable, and is thus called the photic zone. The photic zone occasionally exceeded the maximum depth at UW but it usually did not include the deeper benthos at that station or downstream depths. Light extinction appears to be related to seston concentration (see Section VI.6). The higher extinction values for June reflected the muddy state of the river, as evidenced by seston concentrations, even though discharge rates were low.

VI.1. TERRESTRIAL INPUTS

The total amounts and temporal distribution of terrestrial inputs into the Kanawha River were very similar to the results of several other studies. In the Kanawha River, the total annual leaf fall (also called meteorological input or litter fall) was 371.2 gDW/m^2, blow-in (also called lateral transport) was 68.4 gDW/m, and insect fall was 28.3 gDW/m^2. At several sites within the Matamek River watershed in Quebec, Conners and Naiman (1984) measured annual leaf fall ranging from 235-593 gDW/m^2 (assuming conversion factor of DW= AFDW/0.9) and blow-in ranging from 28-78 gDW/m. Other measurements of annual leaf fall include 362 gDW/m^2 in North Carolina (Webster and Patten 1979) and 458 gDW/m^2 in New Zealand (Winterbourn 1976). Winterbourn's measurement of blow-in was 110 gDW/m. In the only comparable study of insect fall that could be found in the literature, Mason and MacDonald (1982) reported an annual total of 11-36 gDW/m^2 into a stream in England.

When the values measured in the riparian zone were extrapolated to the UW and LW pool segments according to canopy density and lateral extent of riparian area that would contribute terrestrial inputs, the annual amounts for an "average" square meter of river surface were: 1.04 gDW/m^2 insect fall into UW, 0.47 gDW/m^2 insect fall into LW, 11.88 gDW/m^2 total leaf input (leaf fall and blow-in) into UW, and 5.78 gDW/m^2 total leaf input

into LW. The extrapolation to the entire river produces values that are considerably lower than in the immediate riparian area. Terrestrial inputs are thought to have less influence on ecosystem processes in large rivers than in headwater streams because of the lower shoreline to surface area ratio in large rivers (Vannote et al. 1980). It is likely that these methods underestimated the amount of terrestrial insects entering the pool, because flying insects would fall onto the water's surface in mid-channel in addition to the insects that fell from trees along the edges.

VI.2. BENTHIC DETRITUS

It was not possible to obtain a quantitative estimate of the organic material in the sediments because the phleger sampler failed to retain a solid core of the loose, coarse substrate in the Kanawha. The data that were obtainable on percent composition of organic material (Figs. V.2.1-2) indicate rather low concentrations of detritus in the sediments. Few comparisons are available, but in the upper Mississippi River Carlson (1968) reported "The bottom sediments at all areas were classified as silt-loam rich in organic matter." In the Kanawha River there was a general trend for higher percentages of organic matter at LW than UW, both for the annual average as well as for most months. The lowest percentages of organic matter occurred at both sites in April and the highest percentages occurred in June.

The most likely explanation for these trends is that the Kanawha River provides a more riverine physical environment, even with the present locks and dams, than larger rivers in other parts of North America (e.g. Mississippi River). Because of the physical characteristics of the channel, there are sufficient erosional forces to prevent the deposition of most of the fine particulate organic matter. The lack of organic detritus in the bottom sediments is probably not due to the present levels of tow traffic because there are low amounts at LW where the water is too deep for the bottom to be disturbed by tows. There is probably sufficient ambient current, or frequent enough high discharges, to keep the bottom reasonably free of organic deposits. This is supported by the fact that the highest and lowest percentages of organic matter (June and April, respectively) occurred during periods of low and high discharge, respectively (see Section VI.3.). In addition, data on the physical composition of the bottom materials (Huntington District Corps of Engineers, unpublished data) is also indicative of a riverine habitat with very little fine material. At UW, 90-95% of the sediments were found to be sand or larger particles, while at LW 75-80% of the sediments were sand or larger particles.

VI.3. SESTON

The broadest definition of seston includes all particulate matter suspended in the water. The demarcation between the suspended and dissolved states is usually 0.45µm. Although the inorganic components of the seston were measured and reported in Appendix Tables A3.4.1 - 24, the energy flow model only considers the organic components. Organic seston contains living and non-living components. The living components are collectively referred to as plankton which may include microscopic crustaceans and rotifers (zooplankton), algae (phytoplankton), and unattached bacteria. The non-living components (tripton) are fine particles of detritus; however, the detrital particles always occur with a layer of living microbiota (fungi and bacteria). For analyses of trophic relationships, both the quantity and quality of seston are important. Various aspects of seston quality (bacterial density; chl *a* concentration; relative proportions of tripton, bacteria, phytoplankton, and zooplankton) have been reported and discussed in Sections V.4 and VI.4 on Bacteria. In this section the discussion will be restricted to the quantity of organic seston.

The concentration of total organic seston in the Kanawha River is within the range reported for other rivers of similar size. Webster et al. (1979) provide a summary of organic seston concentrations that have been reported from flowing waters, according to stream order. In stream orders 6-7, annual mean

concentrations have been reported as low as 0.000742 gAFDW/l and as high as 0.0072 gAFDW/l. Individual measurements of organic seston concentration have been reported to range as much as 0.00209 to 0.01983 gAFDW/l during one year. In the Kanawha River the annual means at the different sites and depths were between 0.00221 and 0.00322 gAFDW/l and the annual range was 0.00093 to 0.01143 gAFDW/l (Table V.3.1.).

The particle size distribution in the Kanawha River was also similar to other flowing waters, with most material occurring in the 0.45 - 25 µm size class (Table V.3.1.). In two 5th order streams in Virginia, Voshell and Parker (1985) found that 80-85% of the organic seston was less than 25 µm. Similar findings have also been reported by Sedell et al. (1978) and Naiman and Sedell (1979). The predominance of UF seston probably results from its having several different sources: death of algae cells, breakdown of coarse particles, and flocculation of dissolved organic matter.

There was a spatial trend for higher concentrations of total organic seston at UW than LW (Table V.3.1.), presumably because of higher current velocity at the more riverine UW site. There was also a spatial trend for higher concentrations in the upper water column at UW and for higher concentrations in the lower water column at LW. Under the more lacustrine conditions at LW, the concentrations were probably higher in deeper water because of settling.

The temporal differences that were observed (Fig. VI.3.1.) appear to be caused by changes in discharge (Fig. VI.3.2.). The highest total organic seston concentrations were measured at UW in December in the Top samples and at LW in March in the Top samples (Table V.3.1.). On both occasions swift current prevented taking Bottom samples, hence only the data from Top samples were plotted in Fig. VI.3.1. The information obtained during December and March is particularly interesting because the simultaneous discharge conditions at the two study sites were drastically different during both sampling periods (Fig. VI.3.2.). On December 16 samples were collected at LW when the mean daily discharge was 13,145 cfs. On December 17 samples were collected at UW immediately following a large release of water from upstream reservoirs, and the discharge suddenly increased from 10,700 to 43,300 cfs. The difference in discharge accounts for the 9x higher concentration of seston at UW than LW in December (0.01143 and 0.00134 gAFDW/l, respectively). Opposite conditions existed during the March sampling period (24th-25th). A few days earlier discharge exceeded 60,000 cfs at UW and 70,000 cfs at LW. When seston was sampled on March 24 at LW discharge was still reasonably high (46,740 cfs), but by March 25 when seston was sampled at UW discharge had dropped considerably (28,800 cfs). The 3x higher seston concentration at LW (0.00755 gAFDW/l) than UW (0.00234 gAFDW/l) in March is in agreement with the discharge conditions that were encountered. The lowest

organic seston concentrations were measured at both sites in January during relatively low discharge. However, lower discharges and higher seston concentrations occurred during several summer and fall months. Apparently the low organic seston concentration in January was due to a combination of low discharge and low phytoplankton biomass brought about by cold temperatures (see Section V.5.).

One of the most important questions to be addressed by the energy flow model was the relative importance of allocthonous inputs (primarily organic seston) in comparison to autochthonously produced biomass (primarily phytoplankton). The seston results were extrapolated to the entire pool for a 1-year period by multiplying the all-time mean discharge at each site by the average organic seston concentrations measured during 1982-1983. The Top and Bottom samples were each considered to represent one half of the water column. The estimates for the amount of organic seston transported by the Kanawha River during one year are enormous. It is estimated that 30,426,837 and 35,057,119 kgAFDW are transported each year at UW and LW, respectively.

VI.4 BACTERIA

Density and Relative Abundance

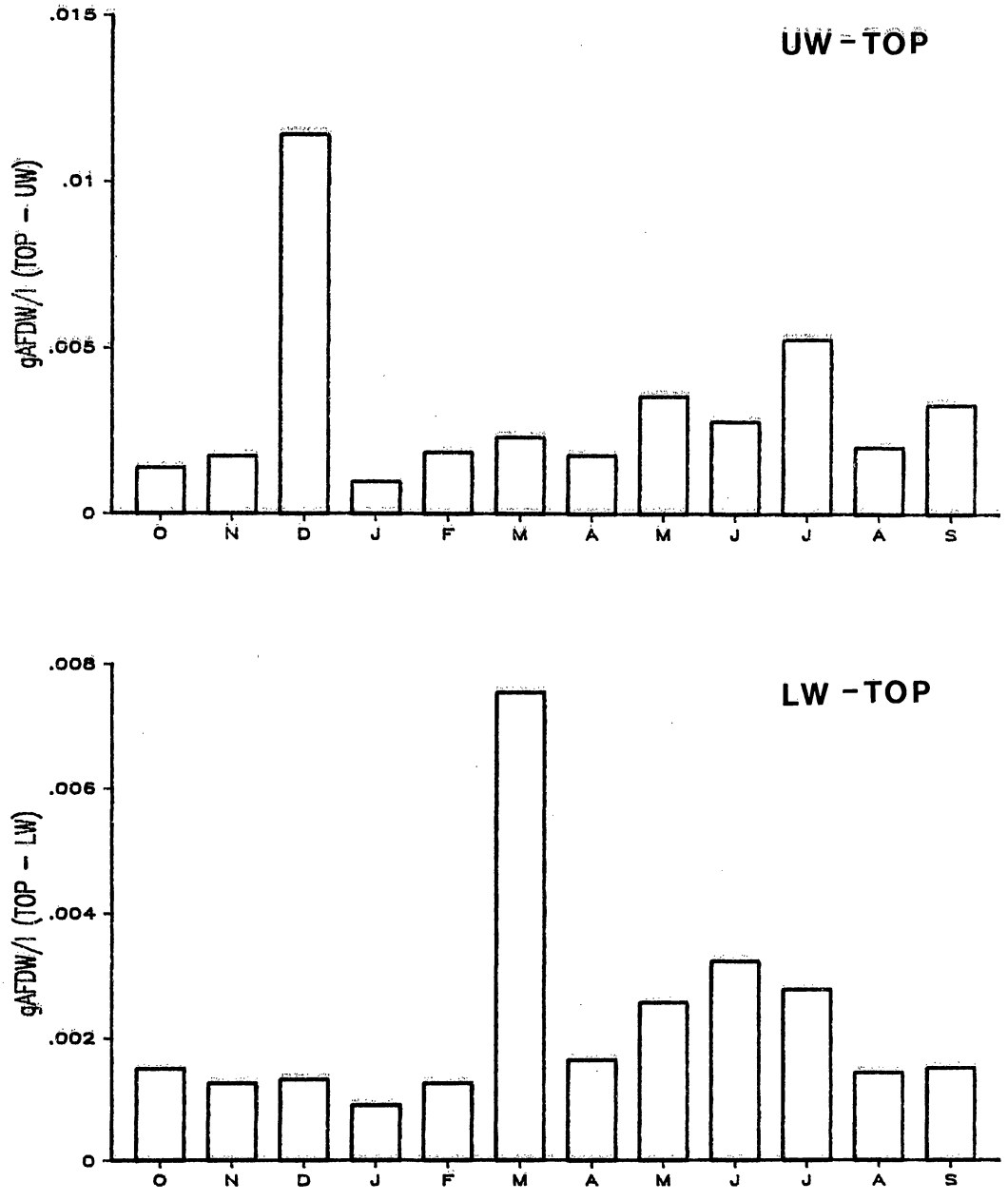


Fig. VI.3.1. Total organic seston concentrations (gAFDW/l). Upper graph: UW, Top samples. Lower graph: LW, Top samples.

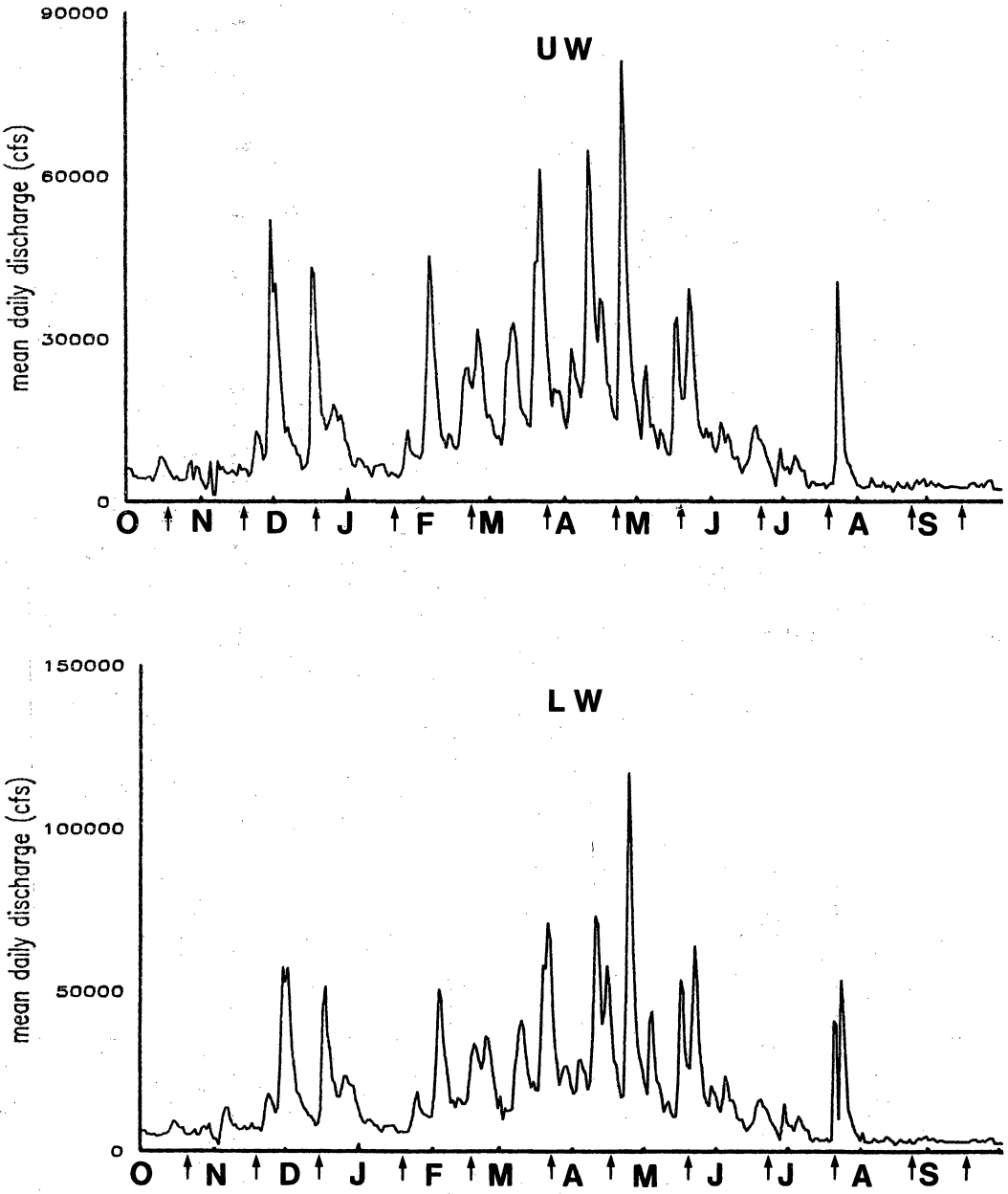


Fig. VI.3.2. Mean daily discharge (cfs) recorded at gauging stations from Oct. 1, 1982 to Sept. 30, 1983. Upper graph: gauge on Kanawha R. at Kanawha Falls, assumed to approximate Upper Winfield (UW). Lower graph: gauges on Kanawha R. at Charleston and Coal R. at Tornado assumed to approximate Lower Winfield (LW). Arrows denote sampling dates.

In comparing the density of bacteria in the UF size fraction as seen in Figs. V.4.1-4, it is important to recognize that the UF size fraction contained most of the unattached bacteria, whereas, the bacteria associated with the other size fractions were attached forms. Figs. V.4.1-4 show that the number of attached bacteria in the S, F, and VF size classes was about 10^3 cells/ml. If the same relationship held true in the UF size class (10^3 cells/ml attached), then the number of free living bacteria would be approximately 10^3 cells/ml. Therefore, approximately 99.9% of the bacterial population would be unattached and 0.1% attached.

Based upon a review of other studies, there appears to be a wide variation in the proportion of unattached versus attached forms in aquatic systems. Kondratieff and Simmons (in press) found the ratio was slightly greater than 50% for unattached cells in a free flowing mountain stream in size fraction < 25µm whereas, in the impounded portion of the river and downstream environs, the percentage of unattached forms greatly increased (68-80%). Porter and Feig (1980) reported that approximately 90% of the bacteria in their samples from eutrophic, seston-rich waters were unattached. Spencer and Ramsy (1978) found a high percentage of unattached bacteria (88-99%) in three New Zealand rivers. Rieman (1978) found between 50-90% unattached bacteria in four euthrophic Danish lakes. Even though a major portion (≤

99%) of bacteria in the Winfield Pool appear to be unattached, the importance of the attached forms should not be underestimated. Kirchman and Mitchell (1982) found that while the number of particle-bound bacteria in their study was low (< 10%), their heterotrophic activity was greater than the unattached forms.

The purpose in presenting Figs V4.5-8 is to show that there was a distribution pattern of bacteria in the size fraction > 25μm. The pattern seems to be that all size classes at the upper stations had some bacteria each month. At LW, for example, few bacteria were present at the lower depths until May. The other interesting feature was that each station had, at least, one pulse of elevated density, but shared little correlation with each other. At UW-Top, the pulse was in December, whereas, at UW-Bottom, the major pulse was in May, but there were also smaller pulses in February, March, August, and September. At LW-Top, the major pulse was in March, and at LW-Bottom, the major pulse was in May which corresponded with the upper station.

The UW site was characterized by greater turbulence and this may have accounted for a more even bacterial distribution through all size classes for nearly each month. The LW site tended to be more lentic in nature, and the larger seston particles probably settled, except when under the influence of high flows or phytoplankton blooms.

Several investigators have shown an inverse relationship between microbial density and particle size (Boling et al. 1975, Fenchel 1970, Hargrave 1972, Kondratieff and Simmons in press, Odum and de la Cruz 1967, Olah 1972, Wallace et al. 1982). This may be true for particles $\leq$ 25 µm but, Fig. 5-8 show this relationship was not true for particle sizes > 25µm. During some months (UW-Top, May) there was a clear inverse relationship, but during most months, the distribution seemed to reflect a bell-shaped curve skewed toward the F size class. Because we are suggesting that the bacterial densities in the L to VF size classes were primarily attached bacteria, we believe that it is important to recognize that there may be cycles of activity among the attached forms which would be masked by looking at the total bacterial community.

Figs V.4.9-10 show that bacterial density increased during mid-winter to levels that would remain for the study period. The density, at an order of magnitude of 10^6 cells/ml, is a density that has been measured by other recent investigators in a variety of natural aquatic habitats (Rieman 1978, Ferguson and Palumbo 1979, Geesey and Costerton 1979, Kirchman et al. 1982, Porter and Feig 1980; Kondratieff and Simmons in press, Lizotte 1984). The interesting observation is the change in density occurred at a time when the water was the coldest and respiration rates were the lowest (Lizotte and Simmons 1984). Geesey and Costerton (1979) found some of the higher bacterial concentrations in the

Athabasca River in northeastern Alberta to occur in December and February. They also noted a high correlation between bacterial numbers with total organic carbon concentrations in the river. Wetzel (1983) stated that planktonic bacterial biomass is generally lower during winter than summer in temperate lakes which is correlated with low winter temperatures and reduced loading of particulate and dissolved organic matter. Campbell (1983), however, states that bacterial numbers vary mostly in relation to the amount of organic matter present.

While several investigators have attributed the seasonal fluctuation of bacterial densities to temperature, particulate organic matter, dissolved organic matter (Wetzel 1983, Campbell 1983) and total dissolved carbon (Geesey and Costerton 1979), such fluctuations may be more complex in a river, such as the Kanawha, that flows through an industrialized, metropolitan area. In addition to the natural factors of leaf breakdown, phytoplankton wax and wane, scouring of river banks and bottoms during spates, there are also important anthropogenic factors to consider. These include stormwater runoff, flushing of sewage treatment plants, and the addition of chemical waste products which may enhance or reduce bacterial numbers at the time of collection.

Figs. VI.4.1-3 illustrate our interpretation of the two factors which seem to have the greatest influence on the

fluctuation of bacterial densities at LW. We found only a small correlation between bacterial densities and tripton availability in the ash-free dry weight form ($^{+}$0.3455, UW; $^{+}$0.1636, LW). This may be related to the fact that most of the bacteria in the Kanawha River in the vicinity of Charleston were probably unattached. We also found no relationship between temperature or changes in phytoplankton density except for a three month period in the latter case. Using the Spearman Rank-Correlation Coefficient (Woolf 1968), there was an inverse correlation between discharge rate (CFS) and conductivity (umhos $\cdot$ cm^{-1}) (Fig. VI.4.1). If the months of February and March are dropped because of abnormally high flow rates (> 30,000 CFS), and the months of May and June are dropped because of abnormally low conductivity readings (probably due to a malfunctioning meter), the correlation coefficient (r_s) is -0.7440 which is significant at p = .05. If the month of October is excluded because of the exceptionally low conductivity reading for the discharge rate, the r_s value is -0.9643. It appears that the two variables which exert the strongest influence on fluctuations of bacterial densities at LW were discharge rate and conductivity. Fig.VI.4.2 shows the relationship between discharge rate and bacterial density, and the Spearman Rank-Correlation Coefficient (r_s) is -0.7667 which is significant at p = 0.05. Western (1984) also found the highest correlation of zooplankton density fluctuations to be related to conductivity. With respect to bacterial

density, conductivity appears to have the second most important influence. The Spearman - Rank Correlation Coefficient with this variable was -0.7500 and is significant at $p = 0.05$. Figure VI.4.3 illustrates the relationship between the two variables of bacterial density and conductivity. Because conductivity is a measure of total dissolved solids (Cole 1983), the conductivity values may indirectly relate to the availability of total dissolved carbon as discussed by Geesey and Costerton (1979).

When discharge rates are low, particulate and dissolved compounds have time to accumulate from both natural and anthropogenic sources, and bacterial densities rise. When discharge rates are high, bacterial densities probably increase temporarily as sewage treatment facilities, streets, and streams are flushed. The populations are probably then reduced as a result of dilution (Hynes 1963).

These data are similar to those reported by Daubner (1969) on the Danube River and Smiddy (1974) on the Ohio River. It appears that the bacteria community undergoes a seasonal succession of species much like any other biological community. In the winter, psychrophilic species would be dominant, and as the river water warmed, mesophilic species would predominate. Smiddy (1974) found that many of the bacteria in the water column were of soil or benthic origin. This origin would allow such organisms to be swept into the water column during turbulent

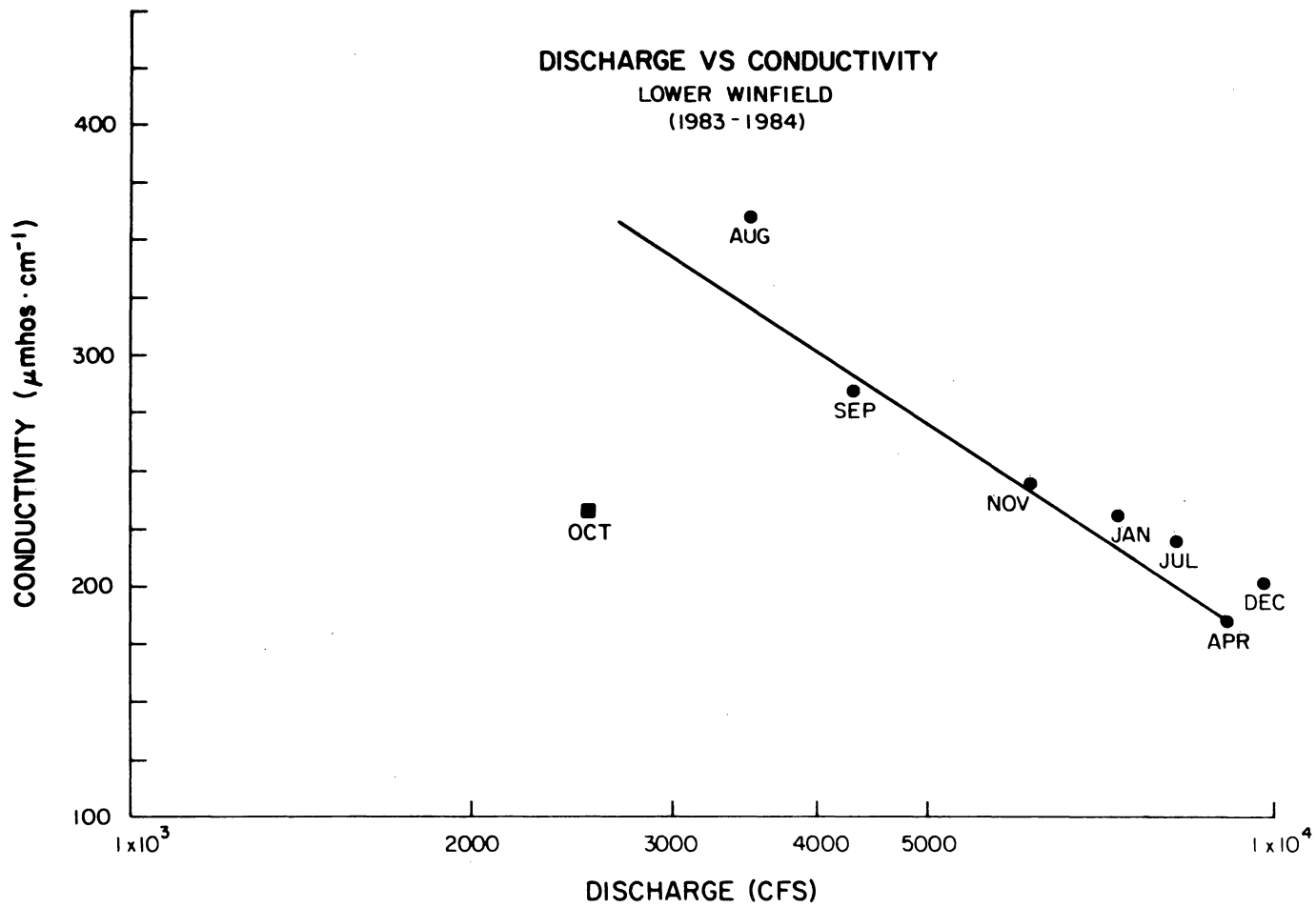


Fig. VI.4.1. Relationship between conductivity and discharge at LW.

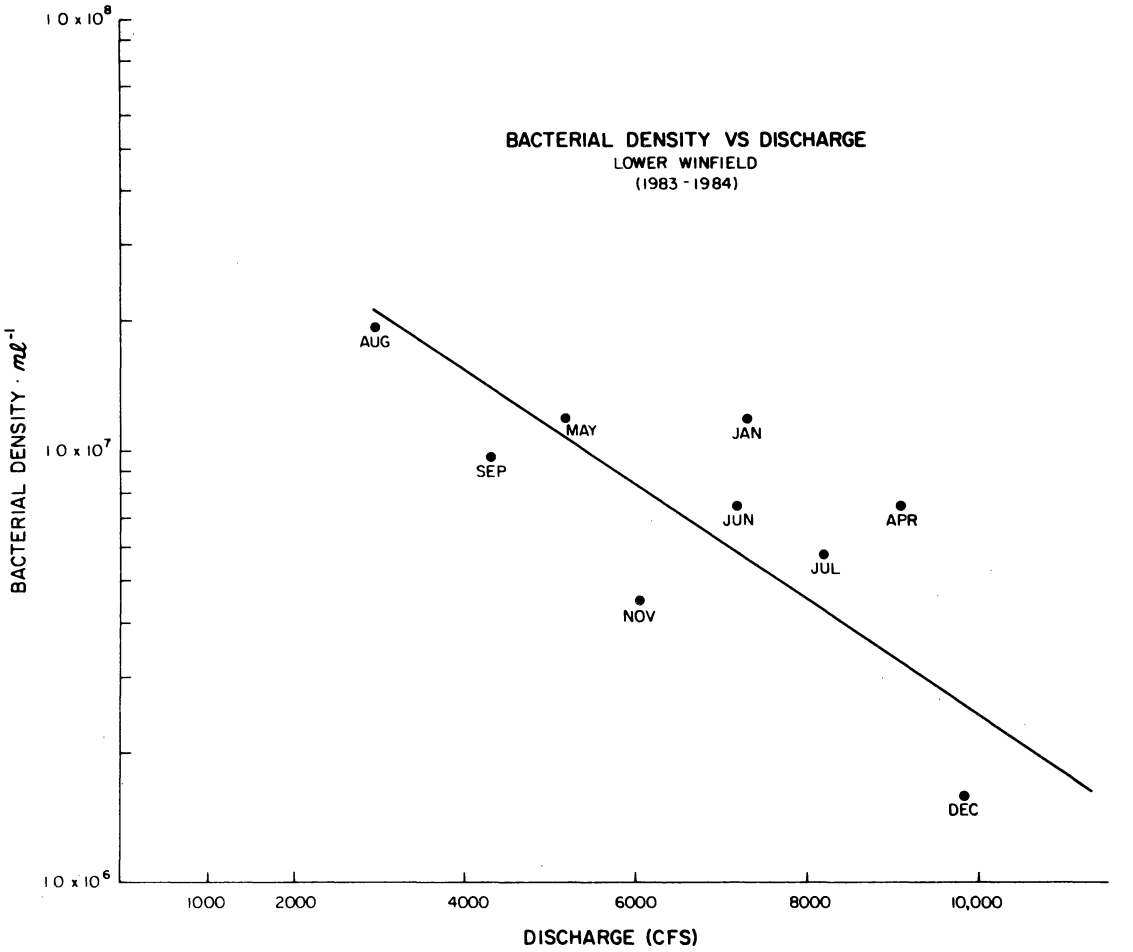


Fig. VI.4.2. Relationship between bacterial density and discharge at LW.

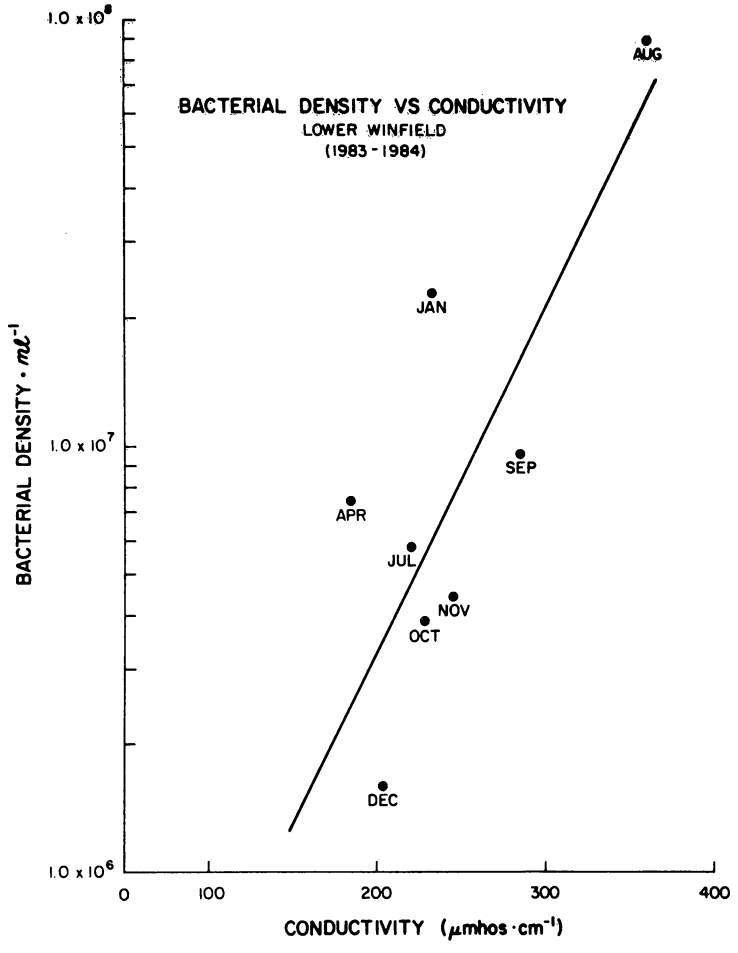


Fig. VI.4.3. Relationship between bacterial density and conductivity at LW.

flow. However, under extended high flow conditions, a dilution and reduction in density should occur.

Similar correlations did not apply at the UW station. There was a very low correlation between discharge and conductivity (r_s = +0.1190), and bacterial density and discharge (-0.2848). Bacterial density and conductivity showed a higher correlation (r_s = +0.5715), but the correlation was positive rather than inverse as observed at the LW station. Any relationship between bacterial density and any other property at the UW station was probably masked by the combination of turbulence, scouring, and release of limnetic water from the Marmet Pool.

One of the more common generalizations about the seasonal distribution of bacteria in lakes is the correlation between density changes of bacteria and phytoplanktonic algae (Wetzel 1980). Changes in bacterial densities often lag behind phytoplankton maxima by as much as 10 days (Straskrabova and Komarkova 1979). Even though the frequency of our sampling schedule was far greater than the generation time of the bacteria, a few trends seem evident in Fig.V.4.12 which compares changes in bacterial density with algae biomass as measured by chl a in the surface samples from LW, the more lentic of the two stations. While bacterial density increased independently of the phytoplankton during the early part of the year, there was a decline in July when the phytoplankton showed their first big

bloom. In August when the phytoplankton declined almost to winter density levels, the bacterial density had increased again. In September when the phytoplankton density increased the second time, bacterial density decreased. If sampling periods are sufficiently close, the investigator may see a time lag of days. Otherwise, the correlation may appear to be inverse.

Fig.V.4.11 shows the correlation between bacterial density and algae biomass at UW where the water was shallower and more turbulent. These waters not only contained populations from the river, but also from Marmet Pool immediately above UW. Water turbulence and mixing probably accounted for the lack of any apparent seasonal cycles of the bacteria in relation to changes in phytoplankton density as measured by chl a at this station.

VI.5 PHYTOPLANKTON

Taxonomic Composition and Cell Density

The phytoplankton of the Kanawha River have been enumerated in at least two national surveys which included monthly sampling at Winfield, WV. The National Water Quality Network (NWQN) survey from July 1960 to July 1961 found an annual median of 242 algae/ml (Williams and Scott 1962) and singled out this river for having excessive organic industrial pollution and a very low density and diversity of plankton (Williams 1964). The National

Stream Quality Accounting Network survey for the year beginning October 1974 recorded a range of phytoplankton densities from 300 to 2500 cells/ml and a mean of 934 cells/ml (Briggs and Ficke 1977), and placed this river in the lowest 25% of stations sampled. The results of our monthly study at the same LW station were similar (range = 16 to 5,770 cells/ml; mean = 952 cells/ml) to those found in the 1974-75 survey. The median value was 141 cells/ml, and would be a lower value if the counts had been in algae/ml. This would represent a decrease of more than 50% from the median reported for 1960-61.

Williams (1964) calculated a "diversity density value" for 100 stations of the NWQN based on a sum of the geometrically scaled densities of the species having more than 150 cells/ml during the productive seasons. The species diversity is inversely related to this value. The values reported for 1961-62 ranged from 3 to 450 and the Kanawha River had a value of 119. The diversity density value calculated from our data was 56 which indicated significant increase in diversity. This change may reflect improvements in water quality over the past 20 years.

Downstream environmental changes appeared to be reflected by changes in the populations of the dominant plankters rather than by a change in the type or dominants (Williams 1964). For almost every taxon composing more than 10% of the total phytoplankton population at the UW station, either cell density or percent of

total was lower at the downstream station. The exception was Phormidium angustissimum, a filamentous blue-green alga which may have been recruited from the periphyton community, in December and February.

Differences in cell density and composition between stations were greatest during the summer, the season of lowest flow. Under these flow conditions the aquatic biota is most likely to be stressed by polluted effluent discharges. Changes in phytoplankton populations may be due to this "pseudo-seasonal" variation (Brinley and Katzin 1942). The passage downstream involves moving through the vast industrial complex of Charleston-Nitro, WV, a highly concentrated setting for various industries. Sewage treatment facilities for a population of approximately 150,000 is predominately secondary, but some of the smaller communities may still discharge primary treated or untreated waste into the river (Dames and Moore, Inc. 1975). There are three major tributaries between UW and LW, the Elk, Coal, and Pocatalico Rivers. These inflows may alter water quality, change concentrations, or introduce phytoplankton competitors to the Kanawha River.

Another important result from decreased flow volume is a decreased flow velocity. During high flow conditions, the 35 miles between the stations may be covered by a parcel of water in a single day; during low flow conditions, the velocity of the

water may only be 1-5 miles per day (Dames and Moore, Inc. 1975). Successional changes may occur during the days or weeks in transit, particularly when populations are reproducing at peak rates.

The various species of phytoplankton have vastly different physiological tolerances and capabilities (Whitton 1975). Moreover, the rate of production is not always proportional to the biomass present for a given taxon (Lizotte 1984). The small cells which appear to dominate because of high cell densities (Actinastrum, Cyclotella, Skeletonema, Agmenellum) may truly dominate with a high potential metabolism in spite of their low biomass. The biomass of these small cells may be limited or decreased with downstream flow because of selective predation by attendant rotifer populations (Western 1984).

Seasonal succession showed several episodes similar to those seen in other temperate rivers (Lack 1971, Wager and Schumacher 1970): low cell densities in the winter, a diatom dominance in the spring, and increased green and blue-green populations as summer proceeded. UW was more similar in population characteristics to the River Thames (Lack 1971) and Susquehanna River (Wager and Schumacher 1970) because of the almost year-round dominance by diatoms. The more lentic LW showed a summer dominance by the green algae, primarily flagellates, and a winter dominance by filamentous blue-green algae. The sewage input

between the sites may also be a factor in that diatoms appear to be less favored by sewage than blue-green or green algae (Wager and Schumacher 1970).

The reported water quality preference for algae taxa collected in phytoplankton and periphyton samples from the Winfield Pool of the Kanawha River are summarized in Table VI.5.1.

Biomass

The size fractioning of phytoplankton biomass should reflect the size classes of the species present. In the Kanawha River, most algae were less than 25µm diameter, and this was supported by the biomass estimates (as chl a). Large portions of the total biomass that occurred in larger size fractions can be attributed to single, larger taxa: Euglena sp. in the 25-43 µm fraction at LW Bottom in January; Diatoma vulgare in the 43-105 µm fraction at UW Bottom in January; Melosira varians in the 24-43 µm, 43-105 µm, and 105-243 µm fractions at both sites in May and June; and Melosira granulata in the 43-105 µm and 105-243 µm fractions at LW in September. The biomass of filamentous diatoms (e.g. Melosira), filamentous blue-green algae (e.g. Phormidium, Oscillatoria), and large colonial green algae (e.g. Eudorina, Pandorina, Platydorina, Errerella, Pediastrum) may be retained in larger size classes even though the individual cells are all less

Table VI.5.1. Reported water quality preference (Lowe 1974) for algal taxa collected in phytoplankton and periphyton samples from the Kanawha River.

	pH	Nutrient	Saprob.	Specific habitat	Palmer
Achn c	I		Sx		
*Achn l	Alp		O	P	
Achn m	I		M->O	P	
Aster f	Alp	E->M	O->bM	E	
Atth z	Alp	E	O	E	
*Bacil p	I		O	P	
Carp c					
*Cocc pe	Alp		Sx->bM	PEph	
*Cocc ph	Alp		Spo->aM	P+PEph	Poll
*Cyclo m	Alp		aM	P,T,E	Poll
Cymb a	Alp->Alb		Spo->O		
*Cymb t	Alp->Alb		O->Sx	P	
*Diat v	Alb->Alp	E	bM->O	P+PEph	Poll
Eun p m	I->Acp		O	P	
Frag c	Alp	E->M	O->bM	E	
Frag v	Alp	E	bM	P	
Frust r a	Acp->I		Sx		
*Gompha o	Alb	E	O->M	P	Poll
*Gompha p	I		M	P	
Hann a	Alp->I		Sx		
Melos d	Acp	O->D	Sx	P	
*Melos g	Alp	E	O->M	E	Poll
Melos g a	Alp	E	M	E	
*Melos v	Alp	E	bM	P	Poll
Merid c	Alp	E	O	P,T	
*Navic c	Alp	E	bM		Poll
*Navic d	Alp		O		
*Navic mi	Alp	E	O	P	
*Navic mu	I->Alp			P,T	
*Navic t	Alp	E	O	P	
*Navic v	Alp	E	O->M	P	Poll
Nitz a	Alp->I				
Nitz f	I		Spi		
Nitz p	I	E	M->P	T,P	
Pinn q	Acp	E->O	O->bM	P	
Rhizos e	Alp	E			
Rhoic c	Alp	E	M	PEph	
*Skelet p	Alp	E		E	
surir o	Alp		M		
Syn u	Alp	E	O->bM	E	
Tab f	Acp	M->O->D	aM->Sx	P,T	

* = important species by pop. density

pH: Indifferent = 3, Alkaliphilous = 13, Alkalibiontic = 2
Nutrient: Eutrophic = 9
Halobion (Oligohalobous): halophilous = 2
(Indifferent) = 14
Saprobien: Mesosaprobic = 7, Oligosaprobic = 9,
Gen. Habitat: Lakes and Ponds = 3, Streams and Rivers = 8
Spec. Habitat: Euplanktonic = 3, Tychoplanktonic = 0,
Periphytic = 13
Seasonal Distr.: Winter = 1.5, Spring = 1.3, Summer = 4.8,
Fall = 4.3
Temp.: Mesothermal = 1, Oligothermal = 7
Eurythermal = 5, Methathermal = 1, Stenothermal = 1

P = periphyton
T = tychoplankton
E = euplankton

than 25 μm in diameter. Other sources of chl a in the larger size fractions could be zooplankton which have been feeding on the phytoplankton or terrestrial leaf detritus.

Gross Primary Production and Respiraton

Phytoplankton gross primary production did not become significant in the lower Kanawha River until July, and net primary production in the plankton community at the 1 m depth was positive only during July and August (Figs. VI.5.1-2). Knopp (1960, cited by Hynes 1970) showed the river phytoplankton lagged about one month behind the predicted potential production from increased daylight. This lag time was thought to be necessary for the phytoplankton population to increase in the river. A similar population lag was seen in the Kanawha River (Lizotte 1984).

The nannoplankton (< 25 μm) were found to be the most important contributors to phytoplankton biomass and gross primary production during the summer, as noted for similar fractions of marine plankton (Durbin et al. 1975, Ibarra 1981). The biomass of the net plankton (> 25 μm) appears to be more variable than that of the nannoplankton fraction (Gelin 1975, Malone 1980). The water passing a 3 μm filter contained less than 1% of the total chl a concentration or was negligible. The only significant contribution measured for the < 3μm size fraction in this study was 11% of the total oxygen production. The

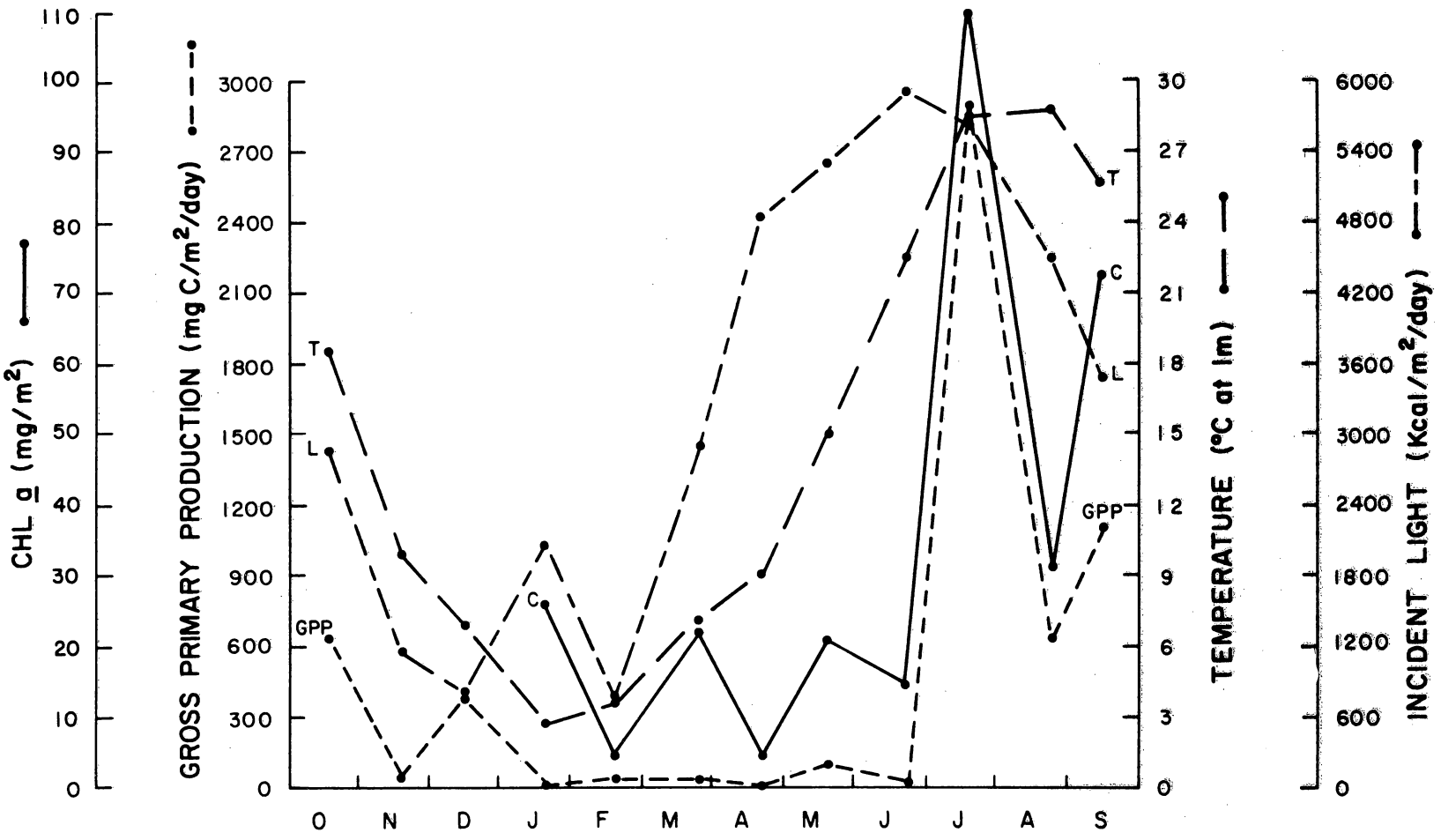


Fig. VI.5.1. Seasonal wax and wane of phytoplankton biomass (Chl a) and gross primary production (GPP) in relation to incident light and temperature at UW.

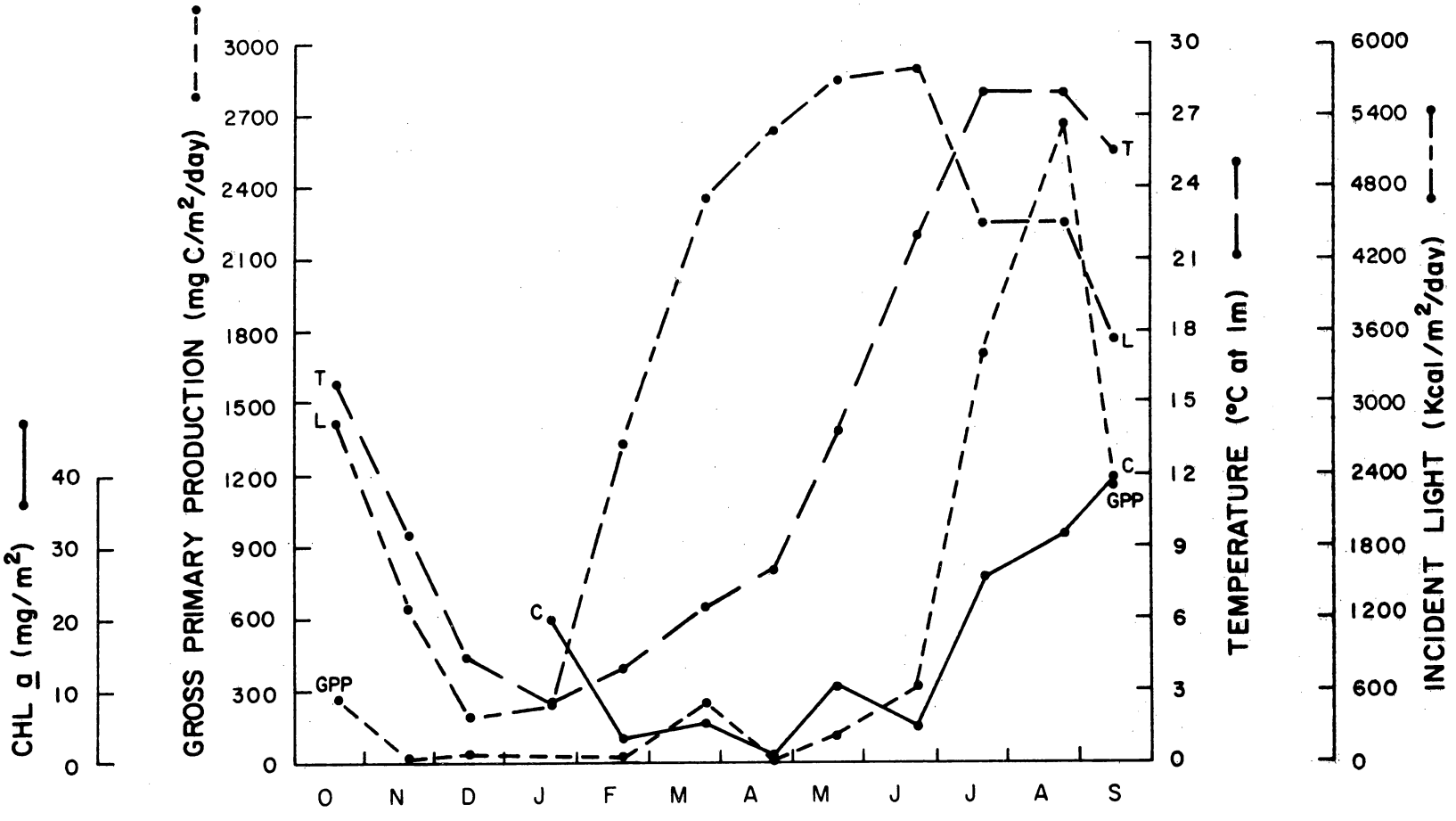


Fig. VI.5.2. Seasonal wax and wane of phytoplankton biomass (Chl a) and gross primary production (GPP) in relation to incident light and temperature at LW.

importance of phytoplankton cells this small has been reported for marine ecosystems (Berman 1975, Herbland and LeBouteiller 1981, Ibarra 1981, Larsson and Hagstrom 1982) and for freshwater ecosystems (Paerl and Mackenzie 1977, Sicko-Goad and Stoermer 1984, Tison and Wilde 1981).

A proportional comparison of the size fractionated phytoplankton can be made by placing the gross primary production estimates on a biomass basis. This ratio is known as the photosysnthetic capacity or assimilation number when derived at the optimum light intensity. Because we do not know if our experiments were carried out under optimum light conditions, we will refer to production in gO2/gchl*a*/hr as the productivity index. The productivity indices for the 3-25µm phytoplankton and the unfiltered, "total" samples were in the same range as those reported for the River Thames and the River Kennet (Hammer 1980). Hammer also listed 29 lakes with photosynthetic capacities ranging from 1.2 to 120. Values for the size fractions < 3 µm (lower range of nannoplankton) and 25-43 µm (net plankton) were measured which far exceeded 120 (Table V.5.8).

Those who would generalize about an inverse relationship between cell size and photosynthetic capacity have contended that the cells are limited by uptake rates, and a greater surface area: volume ratio increases photosynthetic capacity (Taguchi 1976). This theory is supported by some comparative studies of nannoplankton versus net plankton (Gelin 1975, Malone 1971, 1980,

Pollingher and Berman 1982) and disputed by others (Durbin et al. 1975, Furnas 1983). There may be a species specific response of the photosynthetic capacity to nutrient limitation (Glover 1980) and atypically high community assimilation numbers have been observed during dinoflagellate blooms (Harrison and Platt 1980). The high productivity indices recorded for phytoplankton < 3µm and 25-43 µm in the Kanawha River were probably due to one or several species with very rapid high growth rates, and perhaps at the beginning of a bloom. The larger algae were usually colonies or filaments of cells with a diameter <25 µm, so differences in potential metabolism from the 25 µm size class were probably not significant (Banse 1976). The high productivity index observed in the < 3 µm phytoplankton may be a result of decreased population density (Findenegg 1971) for the small blue-green alga Dactylococcopsis raphidioides Hansgirg.

DCMU inhibits photosynthesis by blocking electron transfer in Photosystem II, preventing oxygen production. If oxygen concentrations are kept low in algae cells, photorespiration will not be initiated. Even though the use of DCMU allows incubation of samples in light, it interrupts one of the greatest potential light-mediated oxygen uptake mechanisms. Gross primary production will probably be underestimated if it is calculated from the difference between natural and DCMU-treated samples.

Only during July was a significant amount of respiration attributable to the phytoplankton-dominated 3-25 µm size class.

This was also the month which had the highest recorded gross primary production. It should be noted that because size fractions were sequentially removed, the difference attributed to the phytoplankton may include an increased respiration by bacterioplankton (< 3 µm) in the presence of the algae. Extracellular release by phytoplankton can stimulate bacterial growth (Cole 1982), although with DCMU-treatment, glycollate release from photorespiration would be inhibited. The fact that respiration was greater for night samples than for day samples in the 3-25 µm size class, while the opposite was found for the 0-3 µm size class, suggests different pathways of oxygen use. A significant amount of gross primary production in the 0-3 µm size class for July may also be confounding community respiration with algae-bacterial interactions. The assumption of the < 3 µm size class representing only bacterial respiration may not apply for the July sample. For the other three months with significant respiration, the 0-3 µm size class was the only significant contributor to community respiration, and from cell counts, the bacteria appear to be the dominant organisms responsible.

The bacterial population was relatively constant from June-September. Population densities were similar to ranges reported for other freshwater habitats (Coveney et al. 1977, Riemann 1983). Retention of bacterial cells by the 3 µm filter was low, and approximately 90% passed through the filter. Similar screening efficiencies have been noted elsewhere (Larson and

Hagstrom 1982, Salonen 1974). These bacteria were primarily free cells. Bacteria attached to particulate seston was a small fraction of the population and apparently contributed little to community respiration. Free bacteria are often dominant in oxygen uptake (Straskrabova 1979) and related heterotrophic activity (Azam and Hodson 1977, Azam and Holm-Hansen 1973, Berman 1975, Berman and Stiller 1977, Sepers et al. 1982, Spencer 1979). The specific oxygen consumption (mg/day/10^6 bacteria) ranged from 0.05 to 0.74. Values reported for other freshwater populations include 0.01 - 1.78 (Straskabova 1979) and 0.04 - 0.45 (Shtevneva and Sudakova 1981).

Differences between night and day respiration rates in the < 3 μm size class might be related to the light-mediated substrate uptake reported elsewhere (Ellis and Stanford 1982, Spencer 1979) and considered a response to extracellular release by photosynthesizing organisms. Spencer (1979) stated that respiration rates were not large enough to explain the increased uptake and were not very different for dark incubations. This suggests an uptake and storage of substrate. Given the small "storage capacity" of microbial cells (Odum et al. 1963), the incubation time used for radioisotope uptake studies (1-2 hrs.) may not be long enough to measure subsequent respiration of the substrate. The differences we found between night and day bacterial respiration were determined after physical separation from the phytoplankton, suggesting a reserve of algae release

products in the filtrate. Spencer (1979) noted a lag time of 10 - 15 minutes before the uptake rates in 3 μm pre-filtered samples tapered off. Respiration may continue longer if the allochthonous dissolved organic carbon of a large heterotrophic river is of greater importance as a source of substrate to the bacterioplankton than the algae release products implicated in the previously studied lentic ecosystems. This point may be supported by the relatively constant bacterial population in the Kanawha River and its lack of correlation with the variable phytoplankton population (Bowen and Simmons 1984). A relatively constant bacterial population does not translate to a steady respiration rate, and there is the possibility that differences between night and day respiration are due to changes in substrate concentration or species composition between sampling times.

The greater oxygen uptake for a night sample versus a day sample in the size fraction attributed to phytoplankton may be due to a photosynthetic inhibition of respiration. Ried et al. (1973) hypothesized that photosynthetic inhibition of respiration was caused by Photosystem I dependent processes (which are not inhibited by DCMU treatment) and found that the inhibition of respiration processes became saturated at light intensities greater than 1 to 5 x 10^{-9} $E/cm^2/sec$. The light intensities at 1 m in the Kanawha River probably exceeded this level for the entire productivity season. There is also strong evidence that light deactivates key enzymes of the oxidative phosphorylation

cycle and the glycolytic pathway for starch metabolism (Buchanan 1980). Inhibition of respiratory pathways in the light allows the photosynthetic pathways to more efficiently utilize the many enzymes these antagonistic processes share.

VI.6 PERIPHYTON

Taxonomic Composition and Cell Density

Attached algae often dominate algae biomass in small streams and shallow lakes (Wetzel 1983), and they no doubt play an important role in large rivers and lakes. This is because the attached algae community serves as a source of food for grazing organisms. The seasonal growth patterns in a river like the Kanawha River is more complex than in smaller rivers or lakes of various size because of turbidity, scour, and fluctuating water levels. Light, in particular, has been cited as one of the major environmental controlling factors for periphyton communities (Whitton 1975, Wetzel 1975, Wetzel 1983).

Fig.V.6.1 shows that the Bacillariophyceae dominated the periphyton community for most of the year except late summer when the Cyanophyceae were dominant. Members of the Chrysophyceae and Chlorophyceae were present at various times during the year but never existed as dominants in the periphyton community. This pattern is not unlike that found by other investigators (Round 1964, Sheath and Hellebust 1978, Romagoux 1979), except that the change in biomass preceded changes in phytoplankton biomass.

Biomass

Figs.VI.6.1-2 show that the periphyton biomass increased to a maximum level early in the spring several months before the phytoplankton biomass showed significant changes. At UW, by the time the phytoplankton biomass had increased to seasonal maximal levels, the periphyton biomass was on the decline to the preceding fall levels. In contrast, the growing period was sustained for most of the summer at LW. The fact that the periphyton exhibited a marked change in biomass and cell density before the phytoplankton suggests that the periphyton community was a source of the plankton community in the Kanawha River. Changes in cell density of the periphyton community followed changes in chl a levels measured in the periphyton community and was similar to that reported from other aquatic habitats (Wetzel 1983).

Gross Primary Production and Respiration

In terms of productivity, the periphyton community on the glass slide substrates exhibited higher rates of GPP per unit area than the phytoplankton community below an equivalent area of water (Tables V.5.4 and V.6.1). Figs. VVI.6.1-2 shows that GPP rates followed changes in chl a very closely and began several months before positive changes could be detected in the phytoplankton. The close similarity between changes in chl a and primary production has been observed by other investigators

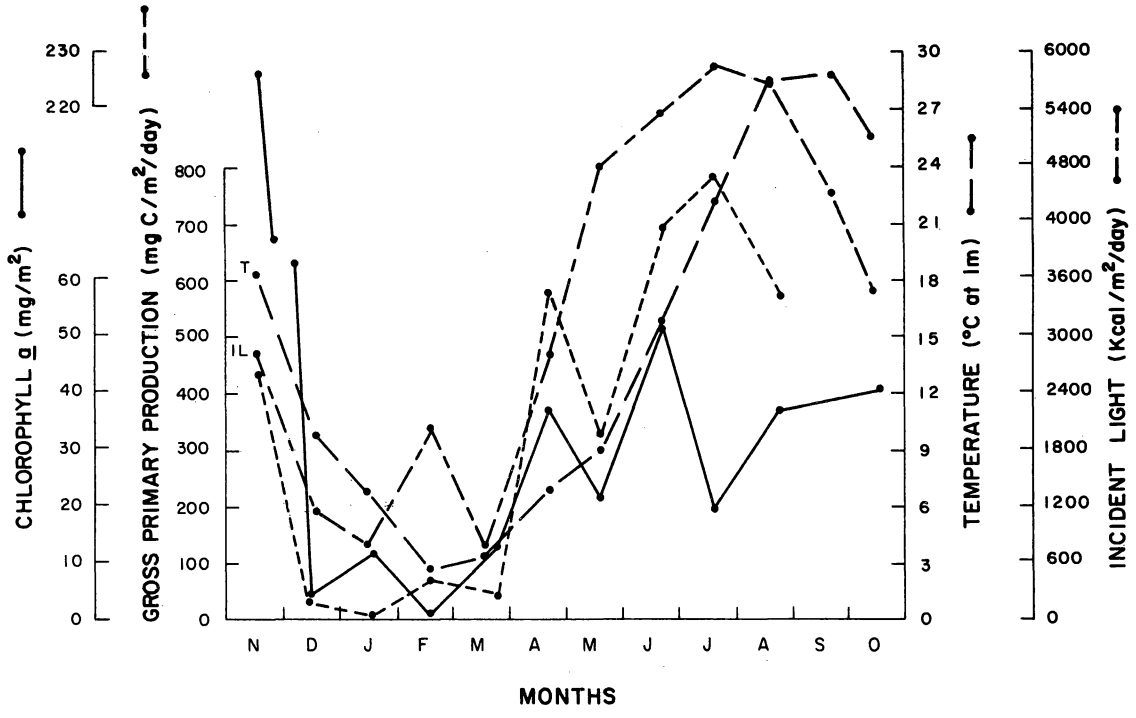


Fig. VI.6.1 Seasonal wax and wane of periphyton biomass (Chl a) and gross primary production (GPP) in relation to incident light and temperature at UW.

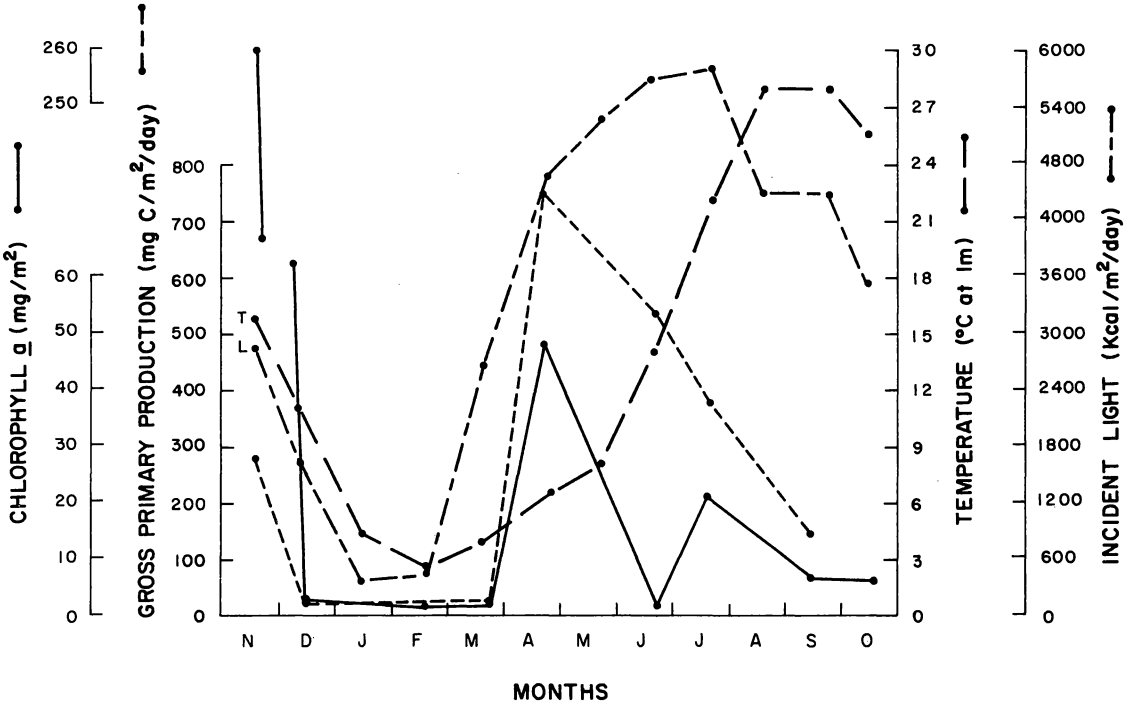


Fig. VI.6.2. Seasonal wax and wane of periphyton biomass (Chl a) and gross primary production (GPP) in relation to incident light and temperature at LW.

(Pieczynska and Szczepanska 1966, Allen 1971, Hunding and Hargrave 1973, Hickman 1971).

Even though the GPP rates of the periphyton community were high on an area basis (m^2), when expand to a "pool" value (Table V.6.2), their rate diminished considerably. This latter value does not diminish their importance, however, as a source of food for grazing organisms. As seen in other studies, the real importance of the periphyton community in the Kanawha River may be as a food source and as a source of organisms for the plankton community. The fact that the periphyton biomass increased rapidly in early spring would suggest the latter relationship.

VI.7 ZOOPLANKTON

Seasonal

Seasonal trends of total rotifer concentration in the Kanawha River were characterized by a single summer population pulse which is similar to trends shown in other rivers, e.g. the Ocqueoc River System in Michigan, USA, (Beach 1960) and the Sokoto River in Nigeria (Holden and Green 1960). Studies on the Illinois River, Illinois, USA, (Kofoid 1908) and Motala River in Sqeden (Carlin 1943) showed an early spring pulse in addition to a summer pulse. Pulses have been noted in winter months in the Yamuna River in India (Rai 1974) in addition to a vernal-early summer pulse. Lack of a winter or spring rotifer bloom at UW may

be attributed to low temperature, high river discharge, and indirectly to low specific conductance. Low temperature decreases reproductive rates of planktonic rotifers (Edmondson 1964). High discharge lowers rotifer populations by dilution and by physically removing the rotifers (Williams 1966, Hutchinson 1939, Kofoid 1908, Rai 1974). Specific conductance decreased during high water probably as a result of reduced mineralization at low water temperatures (Holland et al. 1983) and dilution. Lack of winter or spring rotifer blooms at LW may be attributed to low phytoplankton concentration, since LW species are primarily herbivores or carnivores of herbivores, and low temperature. Although some species found in the Kanawha River in winter are recorded elsewhere as cold stenotherms, such as Filinia terminalis (Hutchinson 1967), none of these or any other rotifer exhibited a winter population peak. Even though cold stenotherms have shorter egg development time at low temperatures than do warm stenotherms (Hofmann 1977), high flow rates throughout the winter and spring may not allow these cold adapted species to remain present long enough to capitalize on their developmental advantage and attain large densities.

Seasonal trends of rotifer densities at UW did not correlate well with chl a but did correlate with specific conductance. Specific conductance in freshwater systems shows a relationship with total dissolved solids (TDS); total dissolved solids is a measure of inorganic salts and dissolved organics (Cole 1979).

Direct utilization of dissolved organic substances by rotifers has not been demonstrated, but the possibility cannot be excluded (Pourriot 1977). More likely, however, high concentrations of dissolved organics benefits bacterial growth, evident from UW bacterial counts. Bacteria are an important food source in the absence of phytoplankton (Pedros-Allio and Brock 1983). Seston ash-free dry weight (AFDW), a measure of particulate organics (often backerial ladened), increases with fall leaf input and also when flood waters pick up detritus. In both fall and spring, rotifer concentrations increased marginally, positively correlated with seston AFDW. Lack of a greater rotifer density increase in fall was apparently due to falling temperatures, whereas, lack of greater rotifer density increase in spring was apparently due to high flow conditions. Many of the rotifers present at UW are listed by Pourriot (1977) as exclusively or frequently bacteriophageous and/or detritvores. These include Hexarthra mira, Keratella cochlearis var. cochlearis, and Lecane sp. H. mira and Lecand sp. were found in appreciable quantities (up to 1.48×10^5 m^{-3}) at UW only. Polyarthra remata and Synchaeta stylata, which can feed on bacteria (Pourriot 1977, Gliwicz 1969), were found at UW in quantities up to 2.33×10^5 M^{-3} and 7.37×10^5 m^{-3}, respectively. At UW, however, neither species showed significant correlation with chl a while both showed significant correlation with specific conductance ($r = .462$ and $.486$, $p = .01$ respectively). At LW both P. remata and S.

stylata showed significant correlation r^2 = .523 and .567, p = .01 respectively) with chl a, but not significant correlation with conductivity. It should be noted that the majority of chl a was found in the < 25 µm size class at UW thus the rotifers had a choice of food items in an edible size range. These data indicate that bacteria, detritus and/or dissolved organics were playing a significant role in the nutrition of some rotifers at UW. Similar phenomena have been reported by Johansson (1983) who noted Synchaeta spp. correlated with chl a one season, but not another season even though the phytoplankton biomass and species present were the same.

Longitudinal

Plankton densities in general are expected to increase downstream (Cushing 1964, Greenberg 1964) due to the river becoming more lentic with increased watershed. Dams creating impounded areas have been shown to increase rotifer densities, but rotifer density quickly drops when lotic conditions recur below a dam (Beach 1960, Cushing 1964, Reif 1939, Whitton 1975). While a general downstream increase in rotifer concentration was evident from station 8 to station 1, the downstream and dam effects were overshadowed by what appeared to be increased nutrient loading from municipal and/or industrial wastewater (Fig. VI.7.1).

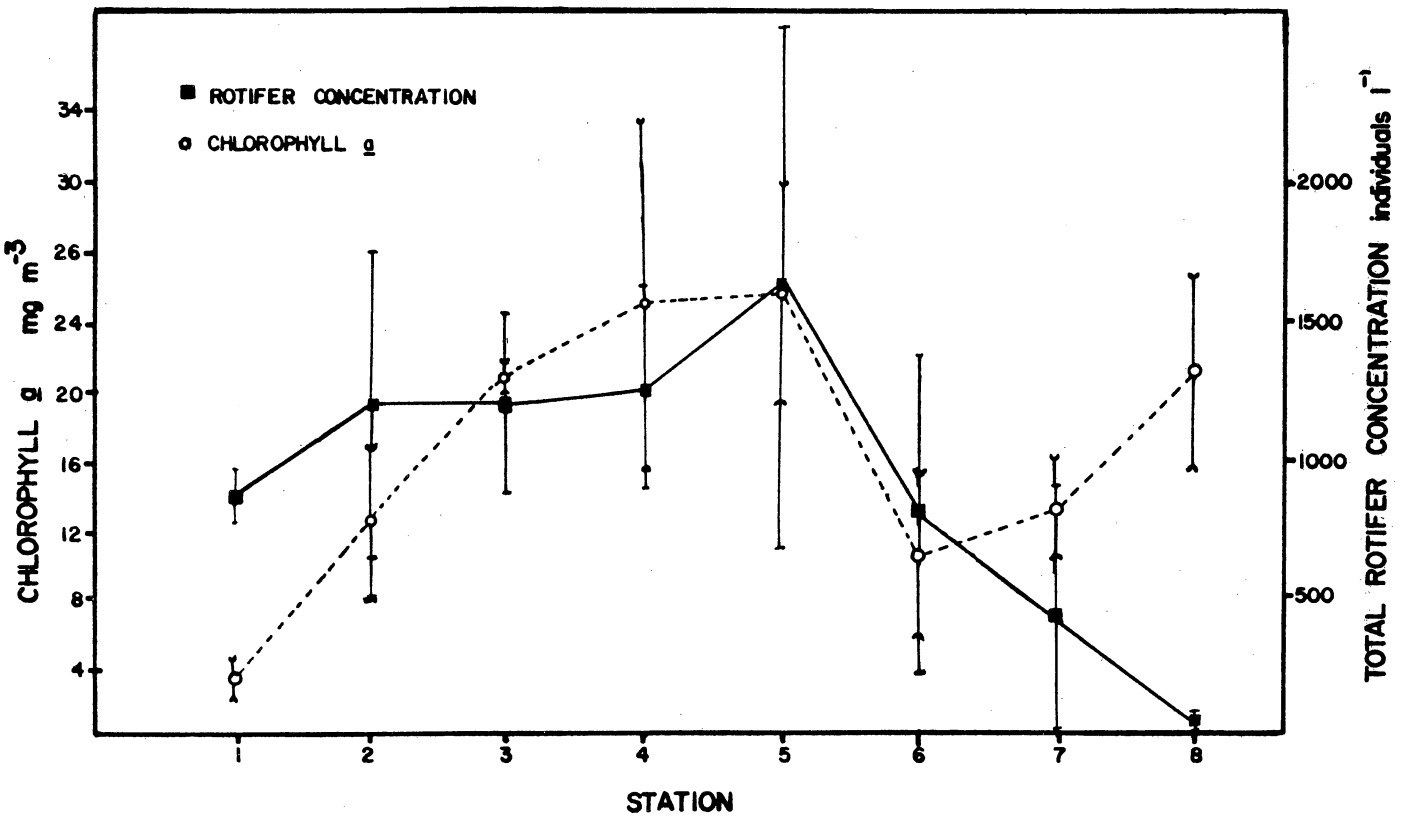


Fig. VI.7.1. Mean total rotifer concentration and mean chlorophyll a concentration in Top samples on three dates.

Total rotifer concentrations at stations 1 to 8 did not correlate well with chl a concentrations at stations 1 to 8 (r^2 = .08), however, when stations 7 and 8 were excluded from the analysis then total rotifer concentrations at stations 1 - 6 did correlate well with chl a (r^2 = .67). Lack of a strong positive correlation when all stations were analyzed may be attributed to the lower concentrations of the bacteriophageous and detritivorous species at stations 7 and 8. Correlations of species with physical, chemical and biological parameters showed the herbivorous Brachionus calyciflorus, B. havanaensis, B. quadridentata and Polyarthra vulgaris were negatively correlated with chl a (i.e. as the number of herbivores increased, the amount of chl a decreased). This may be attributable to the tremendous filtering and feeding rates of species such as B. calyciflorus -- on the average 40 to 50 cells/ animal but up to 4000 cells/animal/hr (Starkweather and Gilbert 1977, Pourriot 1977).

Total rotifer concentrations in the longitudinal profile correlated well with pH and alkalinity. It should be noted that the range of allkalinity was 36 to 52 mg $CaCO_3/l$ and the pH range was 6.1 to 7.5. Species such as B. calyciflorus normally associated with alkaline waters were found in circumneutral water (pH 6.7 to 7.3). Although total rotifer concentration correlated with pH and alkalinity, no particular species showed the same correlation trend on all sampling dates. Thus, total rotifer

concentrations may not be directly associated with pH and alkalinity.

In the longitudinal profile, lack of correlation of species composition with temperature was attributed to the homogenous nature of the temperatures. The greatest temperature range on any sampling date was 1.5° C.

There is a gradual shift in the number of species indicative of eutrophy, mestrophy, and oligotrophy (indicator species according to Sladecek (1983) and Pawlowski (1973)) along the longitudinal transect sampled. The shift is from more oligotrophic upstream to more eutrophic downstream. A coefficient of association cluster analysis showed that when stations are clustered by species composition, stations in the lower reaches (1-3) were similar to each other and stations in the upper reaches (5-8) were more similar to each other on the majority of sampling dates (Fig. VI.7.2).

Significantly higher concentrations of rotifers at 1 m than at 4 m in the lower reaches of the Kanawha River may be attributable to the dominance of herbivorous species. Herbivorous rotifers would be expected to be concentrated in the photic zone (0 to 3 m) where their food is concentrated. No significant difference in concentrations of rotifers at 1 m and 4 m at station 8 was probably due to the bacteriophageous species present there. Bacterial counts showed that bacterial densities

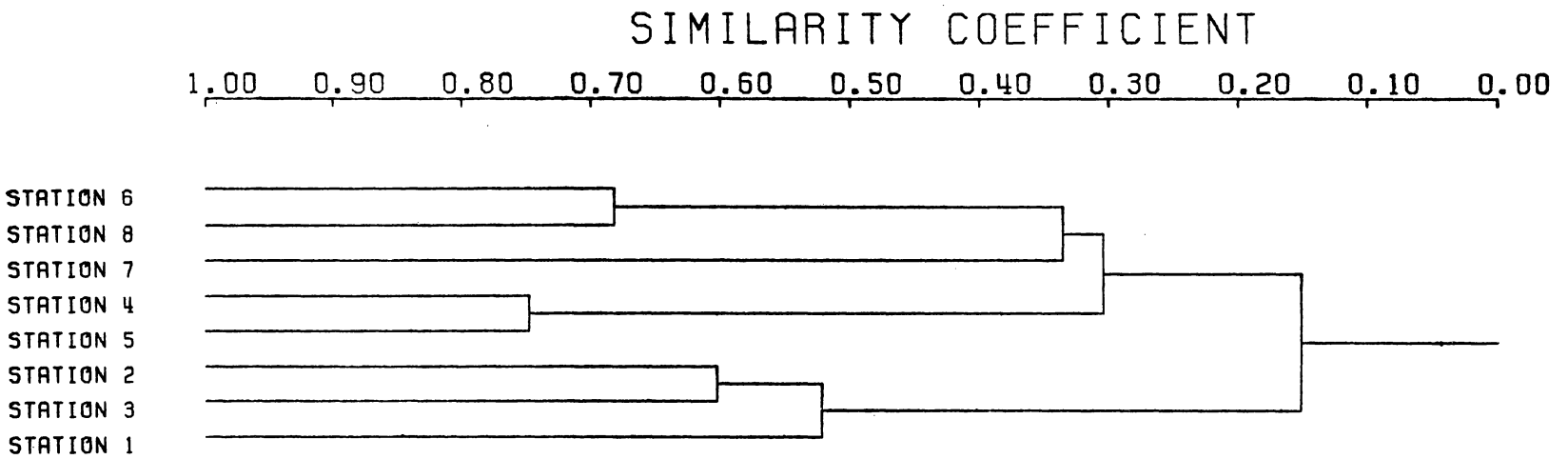


Fig. VI.7.2. Dendrogram plot of Pinkham-Pearson similarity coefficients of stations clustered by species composition.

were similar throughout the water column in this season. Because vertical variation in physical and chemical parameters was slight in the longitudinal profile (with the exception of chl *a* and vertical light intensity), it is reasonable that species composition was very rarely affected by depth. In the few instances species composition was statistically different between depths it could be attributed to species such as *Trichocerca agnata* staying in the top of the water column near their food source.

Entrainment Study

Zooplankton collected behind the James K. Ellis and the Interstate showed no more physical damage than zooplankton collected elsewhere. This does not mean that the zooplankton, particularly rotifers, were not damaged. Increased hydrostatic pressure and turbulence are often detrimental to rotifers (Beach 1960). Towboat propellors generate both turbulence and high hydrostatic pressure in the micro environment of rotifers. It is therefore not clear what direct effect towboats are having on the rotifer fauna.

VI.8. BENTHIC MACROINVERTEBRATES

Community Structure and Function

The benthic macroinvertebrate communities in the lower reaches of the Winfield Pool have been studied since the 1960's as part of several projects that assessed the impacts of the industries around Charleston. A review of these data indicates that there have been definite changes in benthic macroinvertebrate community structure. Studies conducted by the United States Environmental Protection Agency (EPA) (Mason et al. 1971) from 1963 to 1967 reported "odors of hydrogen sulphide and other chemicals," "water... black with coal dust," and "coal fines and tar-like deposits" covering the river bottom with the sand/silt fauna consisting of only oligochaetes with a few chironomids (99% oligochaetes, 1% other taxa). Studies conducted during the 1970's (Benfield et al. 1974, Benfield et al. 1975, Maciorowski et al. 1977, and Hendricks et al. 1979) indicated that although oligochaetes were still the dominant taxon, chironomids made up approximately 7% of the sand/silt community near LW. At present, oligochaetes no longer make up a majority of organisms, but rather Chironomidae are now the dominant organisms in the sand/silt community. A similar shift in community composition has also occurred in the Ohio River where oligochaetes make up a smaller proportion of the community than

they did during the 1960's. EPA sampling of the Ohio River conducted from 1963 to 1967 revealed that over 80% of the community was composed of oligochaete worms (Mason et al. 1971). Sampling conducted in 1978, however, showed that at that time oligochaetes only made up 21.2% of the community with the rest of the community composed mainly of various Trichoptera (43.7%) and chironomids (29.0%) (U.S. Army Corps of Engineers 1980).

The artificial substrates which were used by the EPA (Mason et al. 1971) during the 1960's at LW were comparable to those used in this study, therefore, it is also possible to analyze changes in the cobble/pebble community over the past 20 years. The only major difference between the EPA study and the present study was that the EPA samplers were hung adjacent to the lock walls. The proximity of the EPA samplers to the lush growths of periphyton on the lock walls led to an abundance of snails that did not occur on the suspended baskets used in the present study. The same shift in community structure which occurred in the sand/silt community appears to have occurred on cobble/pebble substrates. Except for snails, almost all of the organisms collected in the 1960's were aquatic worms (mainly oligochaetes with nematodes dominating on one sample date). At present, the dominant organisms on the cobble/pebble substrates are Chironomids (89.0% UW, 89.1% LW). The same shift in community composition has also occurred in the Ohio River. Samples taken during the 1960's revealed a dominance of oligochaetes on hard

substrates (multiplate samplers) (Mason et al. 1971), however in 1978 oligochaetes only made up about 9.7% of the hard substrates while Chironomidae accounted for 65.4% (U.S. Army Corps of Engineers 1980). The decrease in dominance of oligochaetes in the Kanawha River since the 1960's is probably indicative of the decrease in organic enrichment and overall improvement of water quality. This improvement appears to have also occurred in the Ohio River.

Comparison to Other Large Rivers

In the Kanawha River the average annual density on sand/silt substrates at UW was estimated to be 2,058.5 organisms/m^2 and at LW, 691.0 organisms/m^2 with standing stocks of 68.9 mgDW/m^2 at UW and 49.8 mgDW/m^2 at LW. Berner (1951) estimated the standing stock of benthic macroinvertebrates in the soft sediment of the Missouri River to be 4.3 mgDW/m^2 with a density of about 20 organisms/m^2, which are much lower than densities and standing stocks found in the Kanawha. Average annual densities in the Kanawha River were also higher than sample densities reported for the Ohio River (301 organisms/m^2) (U.S. Army Corps of Engineers 1980). In the Satilla River, however, benthic macroinvertebrate average annual densities on sand substrates were much higher than in the Kanawha River (49,046 and 20,477 organisms/m^2) at both sites sampled (Benke et al. 1979). The higher average annual

density of organisms in the Satilla probably results from higher temperatures which allow greater turnover of fast growing organisms, particularly Chironomidae.

The average summer densities on the soft sediment of the Upper Mississippi River (2,924 organisms/m^2) were somewhat higher than in the Kanawha River, however, the community structure was much different than that of the Kanawha (Carlson 1968). Over 75% of the organisms collected were Sphaerium transversum (fingernail clams); Hexagenia mayflies (7.5%), various species of Chironomidae (6.9%) and Oligochaeta (2.9%) made up most of the rest of the community (Carlson 1968). In the Kanawha River almost all of the organisms collected on sand/silt substrates were chironomid larvae (99.4% UW, 96.5% LW). The second most numerous organisms were the oligochaete worms. This same pattern was reported for sand substrates in the Satilla River where chironomids made up 95.3% and 87.9% of the organisms collected at two sites, and oligochaetes made up only 2.2% and 8.4% (Benke et al. 1979). Some of the differences between the Upper Mississippi River and the Kanawha River are probably related to the composition of the sediments. Carlson (1968) described the sediments of the Upper Mississippi as "silt-loam, rich in organic matter", whereas the Kanawha is mostly sand bottom with very little organic matter (Huntington District Corps of Engineers, unpublished data). Fingernail clams, such as those found in the Upper Mississippi River would not occur in high numbers in the Kanawha River because of the shifting sand habitat (Pennak 1978).

Chironomidae made up the greatest proportion of organisms on cobble/pebble substrates in the Kanawha River (89.0% UW, 89.1% LW). The Ohio River hard substrate communities were also dominated by chironomidae (U.S. Army Corps of Engineers 1980). However, in the Satilla River Chironomidae only made up 22.7% and 26.8% of the snag community at two sites (Benke et al. 1979). The dominant organisms in the Satilla appeared to be Simuliidae (60.6%, 27.9%) and various net-spinning Trichoptera (14.2%, 40.1%). This represents a difference in functional groups represented on cobble/pebble substrates in the Kanawha and Ohio Rivers (collector-gatherers) and the snag substrates present in the Satilla River (collector-filterers). This major difference is probably due to the series of dams on the Kanawha River. During the summer and fall there is not enough current over much of the length of the Kanawha for efficient filter feeding, and the reduced currents allow a great accumulation of detritus for collector-gatherers.

Production

Estimates of production in the Kanawha are quite similar to the few production estimates that are available for large rivers. The only studies of total community production that have been conducted on large rivers were done on the Thames and the Satilla

Rivers. Total benthic production in the Thames River was estimated to be 66,200 mgDW/m^2/yr (Mann 1972). However, this number includes production by bivalves, sponges and bryozoans along with a correction factor for chironomid production, which was believed to be severely underestimated. If all bivalves, sponge, and bryozoa are discounted (no production estimates were made for any of these taxa in the Kanawha River), then insect production is estimated to be about 5,800 mgDW/m^2/yr in the Thames. In the Kanawha River, production on an "average" square meter of river bottom, is 6,228.7 mgDW/m^2/yr. Production on an average square meter of UW substrate is almost double the amount in the Thames (14,819.5 mgDW/m^2/yr), while production on an average LW square meter is much less (2,546.9 mgDW/m^2/yr).

In the Satilla River, production was estimated on three different types of habitats: snags, sand, and mud (Benke et al. 1979). By taking the proportions of these types of substrates in the river (Benke et al. 1979) and multiplying them by the production estimates, the "average" total commmunity production was estimated to range from 14,748 to 29,336 mgDW/m^2/yr in the Satilla River. Hence the production at UW in the Kanawha River fell within the range for the Satilla River, even though the Satilla is considered to be "subtropical" and higher production might be expected. It is informative that production on an average square meter of bottom in the Kanawha River is not very different than production in either the Satilla or the Thames Rivers, neither of which are affected by commercial navigation.

Production in the Kanawha River was distributed unequally among taxonomic groups. The greatest amount of production came from the Chironomidae at both sites (93.2% UW, 99.9% LW). Chironomidae have been shown to dominate benthic macroinvertebrate production in other large river environments. Production of chironomids is consistently high because of their multivoltine life cycles, short generation times, high densities, and ability to colonize a variety of substrates. Mann (1972) reported that Chironomidae accounted for a large proportion of the benthic macroinvertebrate production in the Thames (60% of all benthic production including bryozoans, sponges, etc). In the New River, Voshell (1985) reported that Diptera made up 76.4% of total benthic macroinvertebrate production on rock outcrops, with Chironomidae making up 69.4% of total production. Chironomidae also made up a large proportion of total production (83.8%, 73.8%) in the Satilla River on sandy substrates but were less important on snag habitat (13.4%, 17.5%) (Benke et al. 1979).

In the Satilla River on snag habitat the highest producers were those taxa which are filter feeders (Benke et al. 1979). At the upper Satilla site Simuliidae accounted for 62.7% of snag community production and were also important in the snag cummunity at the lower site (24.5%). In the Kanawha River Simuliidae only accounted for 0.2% of production at UW on cobble/pebble substrates and no Simuliidae were found at LW.

Trichoptera were also very important in terms of the proportion of total community production on snag habitat in the Satilla River (16.8%, 47.3%). In the Kanawha River, while the Trichoptera contributed a greater proportion of the total community production than Simuliidae, they contributed a smaller proportion of the cobble/pebble community production than in the Satilla River (12.4% UW, 5.8% LW). Hydropsychid caddisfly production in the Kanawha River on cobble/pebble substrates (5,356 $mg/m^2/yr$ UW, 431.3 $mg/m^2/yr$ LW) was substantially less than hydropsychid production on either the Satilla (11,614 - 25,607 $mg/m^2/yr$) or Savannah River (10,979 - 38,402 $mg/m^2/yr$) snag substrates. This difference in production of various taxonomic groups is also reflected in differences in production among the various functional groups.

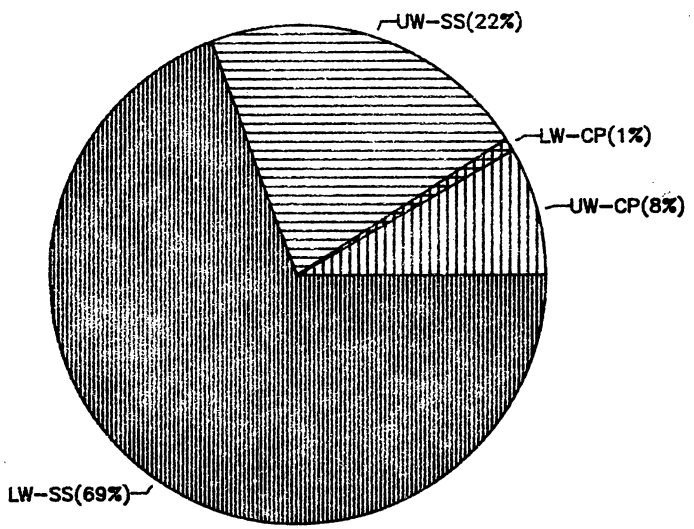
When adjusted for the relative abundance of habitat types in the Winfield Pool, the distribution of production was 67.1% collector - gatherers, 9.6% collector - filterers, 23.1% predators, and 0.2% shredders. The Orthocladiinae/Chironominae subfamily group was responsible for the high production by collector-gatherers. Since no production estimate was made as part of this study for Oligochaeta, which are also collector - gatherers, production for collector - gatherers at each site and on each substrate is probably underestimated. However, since comparatively few oligochaetes were collected throughout the year from the Kanawha River, this underestimate is probably not very

great. The dominance of collector - gatherers was also noted in studies of other medium and large rivers. In the New River, Voshell (1985) found that 70% of the production came from collector - gatherers while 30% came from collector - filterers (production was not estimated for predators). In the Satilla River, Benke et al. (1984) estimated that the distribution of production was 71% collector - gatherers, 12% collector - filterers, and 17% predators. It is interesting to note that the percentage of production attributed to collector - gatherers in the New, Satilla and Kanawha was very similar even though the rivers themselves are quite different.

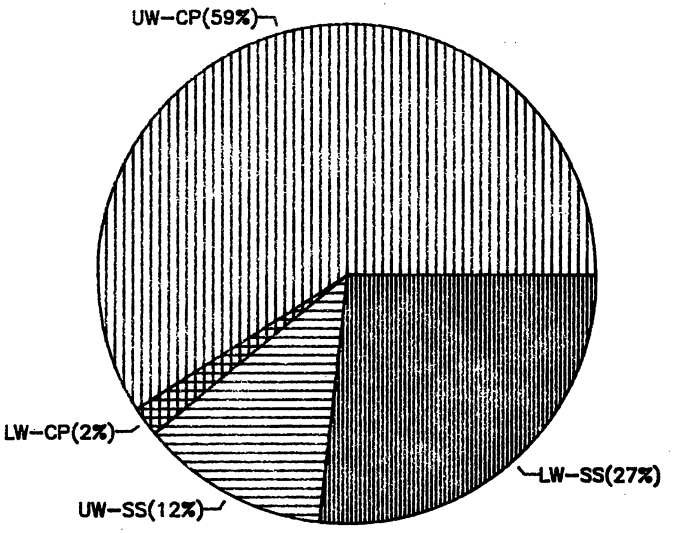
Production of the various functional groups is indicative of the food and substrates available in the Kanawha. Very few scraper - grazers were collected, less than the number needed for a reliable production estimate. The apparent absence of scraper - grazers from the benthic community could be the result of several factors. Because the Kanawha is a turbid river, periphyton production is low and occurs in a relatively narrow area along the shoreline. Periphyton produces a relatively minor percentage of the total energy available in the pool when compared to other sources. The amount of scraper-grazers collected could have been underestimated by the sampling methods used: artificial substrates hung below the photic zone and Ponar grabs of soft sediments. Shredders are also not a very productive group in the Kanawha River. Because of the low ratio of shoreline

to surface area in a large river, the inputs of leaves are low in comparison to the abundnace of fine detritus in the system. In addition there are few structures to retain the leaves in "packs" that are preferred by shredders. The collector-filterer functional group requires hard substrate for attachment and constant currents for filtering. Hence, the production of collector-filterers was highest at UW on cobble/pebble substrates and was limited elsewhere by a lack of firm substrates. The collector-gatherer trophic group, however, is made up of a relatively diverse group of macroinvertebrates including the Orthocladiinae/Chironominae subfamilies, several mayfly taxa and Cyrnellus fraternus. The diversity of this group allows it to be very productive in any of the habitats found in the Kanawha River.

Production in the Winfield Pool was highest on cobble/pebble substrate at UW, amounting to 59.1% of the total production, even though this substrate only makes up 8.6% of the available substrates in the Winfield Pool (Fig. VI.8.1). Production at UW (14,819.5 mgDW/m^2/yr), including production on the sand/silt substrate was much higher than at LW (2,546.9 mgDW/m^2/yr). The ability of certain substrates and reaches of rivers to produce more than others was also noted in the Satilla River (Benke et al. 1979). Production on snag habitat amounted to 9% of the total production while snags only provided 2.4-2.6% of the total substrate area available. Modde and Schmulbach (1973) showed that



Substrate Availability



Total Pool Production

Fig. VI.8.1. Availability of natural substrates in the Winfield Pool and percent of total pool production on each substrate type. UW-CP = Upper Winfield cobble/pebble; LW-CP = Lower Winfield cobble/pebble; UW-SS = Upper Winfield sand/silt; LW-SS = Lower Winfield sand/silt.

benthic standing stocks in the Missouri River on sand and silt substrates were about 100 mgDW/m^2, while Nord and Schmulbach (1973) indicated that colonization of macroinvertebrates on multiplate samplers was 1000 to 3000 mgDW/m^2/yr. All of these studies, as well as the work in the Kanawha River, indicate the importance of hard substrate to benthic macroinvertebrates in large rivers. Indeed, several studies have suggested that in areas of sufficient current, suitable substrates appear to be the limiting factor in large river benthic production (Fremling 1960, Nelson and Scott 1962, Nilsen and Larimore 1973, Benke et al. 1979, Cudney and Wallace 1980).

Production on cobble/pebble substrates, although greater than production on sand/silt substrates in the Kanawha River, falls short of production on snag habitat in the Satilla River (57,415 - 72,236 mgDW/m^2/yr) (Benke et al. 1979), snag habitat in the Savannah River (12,600-22,700 mgDW/m^2/yr, caddisflies only) (Cudney and Wallace 1980), or the rock-outcrop community studied by Voshell (1985) in the New River (427,625 mgDW/m^2/yr). The actual difference in production between the snag and rock outcrop habitats in these rivers, and the cobble/pebble habitat in the Kanawha is greater than is readily apparent. While each artificial substrate basket used in this study represented approximately 0.93m^2 of river bottom, if all the cobbles and slab were to be measured for surface area, a single basket might contain up to 10 times that amount of total surface area.

Production values reported for snag and rock outcrop habitats, reflect production per square meter of surface area, not per square meter of river bottom. The difference in production is apparently not caused by temperature. In the Kanawha River the number of degree-days between May and October (above 0°C) is estimated to be 4050. This value is considerably above the 3580 degree-days reported for the New River (Voshell 1985) and should have the effect of stimulating increased benthic macroinvertebrate production. As will be discussed in the next section, the difference in production between the New River rock-outcrops and the Satilla snag habitats and the Kanawha River is very likely related to food quality in the Winfield Pool.

Trophic Basis of Production

In an average square meter of the Winfield Pool, approximately 57.7% of the total production can be attributed to the consumption of detritus, with 18.5% to diatoms and other types of algae and 23.8% to animal matter (Fig. VI.8.2). These values are influenced heavily by production of collector-gatherers at UW where 69.3% of production can be attributed to detritus consumption. In the New River, Voshell (1985) estimated that 53% of benthic macroinvertebrate production could be attributed to the consumption of fine detritus, with diatoms and other algae making up a slightly lesser percentage (44%) and

animal food contributing about 3%. Algae would be expected to contribute more to production in the New River because the study site was just below a large impoundment.

At LW a much lower percentage of the total production could be attributed to detritus consumption (28.7%) than UW, with the main source of energy for production being animal matter (61.7%) (Fig. VI.8.2). This is the case because of the large numbers of Tanypodinae which were found in the sand/silt substrates at LW. Total consumption by predators at LW exceeded the production of other trophic groups for which production was estimated and which would serve as prey. Tanypodinae larvae probably also consume smaller tanypodinids and other chironomid larvae, oligochaetes, and microinvertebrates which are detritus feeders but for which no production estimates were made. Based on this assumption, the trophic basis of production for the sand/silt community at LW is also the consumption of detritus.

In the Kanawha River at UW only 10.4% of net-spinning caddisfly production can be attributed to animal foods. At LW this figure is even lower, 6.6%. These values are much lower than were reported in the Satilla River, where 80% of production was attributed to the consumption of animal matter (Benke et al. 1984). However, in the New River the amount of net-spinning caddisfly production attributed to production is similar to the Kanawha, 13% (Voshell 1985). This would indicate that netspinning

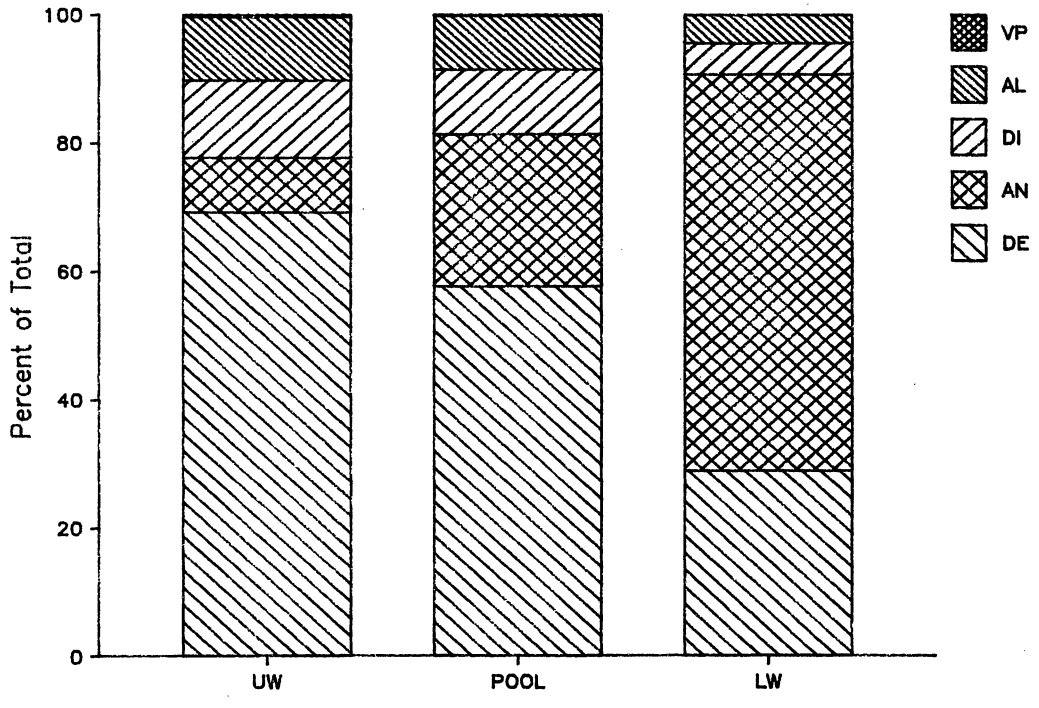


Fig. VI.8.2. Proportion of production (%) attributed to various food types on an "average" square meter of substrate at UW, LW and the entire pool. AN = animal matter; DI = diatoms; AL = other types of algae; DE = detritus; VP = vascular plant material.

caddisflies in the Kanawha and New Rivers are forced to rely on a much less nutritive food source for their production than in a river like the Satilla.

Role of Benthic Macroinvertebrates

Benthic macroinvertebrates can affect the rate at which energy flows through a system such as the Winfield Pool. Energy in the form of particulate organic matter can be obtained either from the water column (collector - filterers) or bottom deposits (collector - gatherers, shredders) and made available to higher trophic levels. One indication of the amount of energy which passes through the Winfield Pool is the amount of seston, or suspended organic matter, which is carried by currents down the river. Seston concentrations (as ash-free dry weight/liter) were measured at UW and LW on a monthly basis for one year (see Section V.3.). These concentrations can be extrapolated to annual seston transport by multiplying concentration by discharge (see Section VI.3.). Seston concentration was measured according to several size classes: LA (large, >1000 µm), ML (medium-large, 234-1000 µm), SM (small, 105-234 µm), FI (fine, 43-105 µm), VF (very-fine, 25-43 µm), and UF (ultra-fine, 0.45-25 µm). The size class of seston which is available to filter feeding insects depends on the particular taxon. At LW the only filter feeders were caddisflies, which because of the mesh size of their nets

could probably only take advantage of the ML and SM size classes. At UW there are three different types of filtering organisms: the net-spinning caddisflies, Isonychia bicolor, a filtering mayfly, and various black fly species. The caddisflies could utilize the ML and SM size classes. Isonychia bicolor would be able to adjust the size of particles filtered by adjusting the position of its front legs when held in filtering position and probably utilize the SM, FI, VF and part of the UF size class (Wallace and O'Hop 1979). The Simulium black flies would probably utilize several smaller size classes but mostly material from the UF size class (Voshell 1985).

By comparing the amount of organic material consumed by filter-feeding organisms with the amount that is being transported down the river in one year, the ability of the benthic community to slow downstream energy transport can be estimated. Approximately 30,427,000 kgAFDW of seston of all size classes passes through UW during one year with 3,127,000 kgAFDW/yr of falling within the size class range of ML and SM. If the consumption by the filter feeders in a one meter wide strip across the pool at UW is compared with the total organic seston which passes through the area, only 1.5×10^{-5}% of the total seston would be removed during one year's time by the filter feeders (Fig. VI.8.3). If only the ML and SM size classes are considered accessible, then 1.4×10^{-4}% of the seston would be removed of the total from those two size classes. At LW, the consumption of

filter feeders in relation to the total pool bottom is much less than at UW while the amount of seston is greater (35,057,000 kgAFDW/yr). Thus the percentage of seston consumed at LW would be much lower than at UW.

If the total amount of consumption of the two major primary consumer feeding functional groups, collector-gatherers and collector-filterers, is compared with the energy contained in the seston which passes through the pool, less than 0.0001% of the energy passing over a square meter of river bottom in the Winfield Pool is being consumed during one year (UW). This value is far below previously reported values for energy removal in other stream systems. Voshell (1985) estimated that total community consumption was equivalent to about 0.03% of all seston passing though a New River section. McCullough et al. (1979a, 1979b) estimated that in a small stream about 0.01% of all seston was removed by collector-filterers during one year. The Kanawha, when compared to these other stream systems appears to be at least two orders of magnitude less efficient in being able to capture and utilize the energy flowing downsteam. Most of this inefficiency probably comes during periods of high flows, when enormous amounts of suspended organic materials are transported downstream. In the Kanawha River, because of increased depth and limited hard substrate for collector-filterer colonization, the probability that a seston particle will be filtered out is much less than in a more shallow, rocky-bottom stream or river. In

conclusion macroinvertebrates only have a miniscule effect on the downstream transport of energy through the pool.

Although benthic macroinvertebrates only remove a very small portion of the energy which passes through the pool, they still serve a very important function in the Winfield Pool ecosystem. Benthic macroinvertebrates form an energy flow pathway between "lower" (detritus, primary producers) and "higher" (fish) trophic levels. This pathway consists of the capture of low energy organic materials and the concentration of the energy into comparatively energy rich animal tissue. Without macroinvertebrates occupying the intermediate trophic levels, many important members of higher trophic levels (e.g. sport and commercial fishes) would not have access to the major energy inputs into the system.

A conceptual model of energy flow through the macroinvertebrate components of the Winfield Pool ecosystem at UW is shown in Fig. VI.8.3. Benthic macroinvertebrates consume only 0.0001% of the energy which flows through UW as seston each year, with diatom and other algae production included in the passage of seston in the river. Most of the energy consumed by the collectors (both gatherers and filterers), however, is returned to the river as egestion or lost from the system as respiration. Total collector production can either be consumed by macroinvertebrate predators, consumed by other invertivores

(fish, etc.) or leave the system as insect emergence (estimated at 20% of total production). Predator production can be lost from the system through emergence, or it can be consumed by invertivores. Only a small part of the energy consumed by the macroinvertebrate primary consumers is actually made available (through production) to higher trophic levels (3.4% at UW). Major limitations on the amount of energy which is consumed by macroinvertebrates are the lack of suitable substrate (cobble/pebble) and the depth of the pool, which affects the percentage of seston which is available for consumption at any one time. Therefore, the most significant function of benthic macroinvertebrates in the Kanawha River ecosystem appears to be in retarding the downstream transport of a small proportion of the total energy that would otherwise not be available to higher trophic levels.

Effects of Tow Passage

The density, diversity, and production of benthic macroinvertebrates are all very important factors in the overall function of a river ecosystem. Because so little research has been conducted on the possible effects of tow traffic on macroinvertebrates, it is difficult to define the magnitude of the impact on these three factors. The tow-passage experiment conducted in the Winfield Pool suggests that the center channel

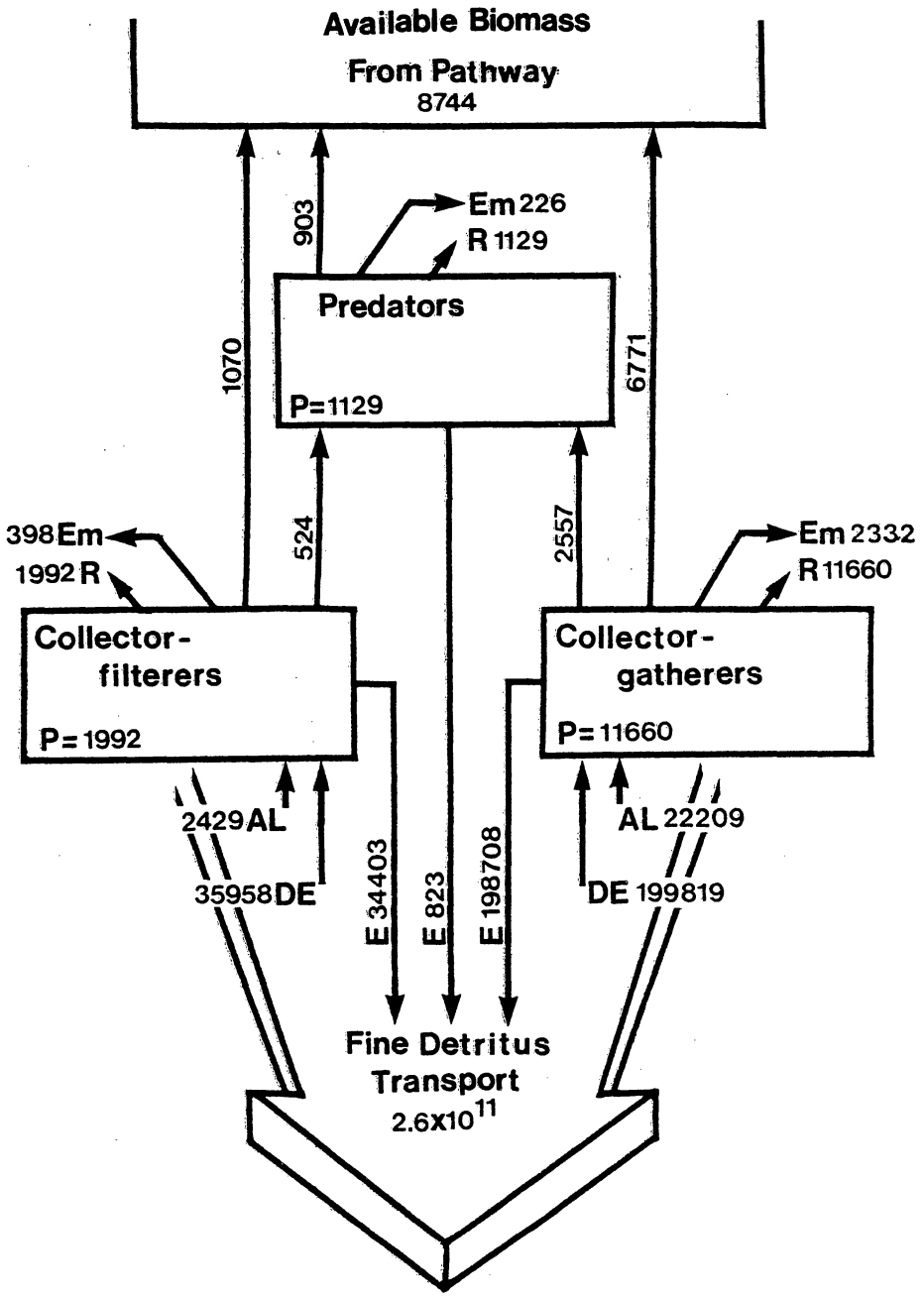


Fig. VI.8.3. Conceptual model of benthic macroinvertebrate energy flow pathway. Available biomass from pathway is the amount of biomass that passes through the pathway and is available to higher trophic levels. All values are expressed as mg dry weight per square meter at UW. DE = detritus; AL = all types of algae; P = production; Em = emergence; R = respiration; E = egestion.

area is certainly not devoid of macroinvertebrate life. This conclusion was also reached by researchers studying the Ohio River, although their studies were conducted in deeper water (U.S. Army Corps of Engineers 1980).

In the Winfield Pool, the effects of any increase in commercial tow traffic would probably be greatest in the shallow, cobble/pebble areas near UW. The tow passage experiment was conducted during "worst-case" conditions when any tow passage impact, if it was to occur, would be evident. These conditions included: relatively shallow water (4 m at sailing line), a high number of tows during the month (260) relative to the annual average, low natural flows (magnifying the impact by increasing the difference between ambient flows and tow induced flows) and the placement of the artificial substrates in an area where they would be affected by any backflow conditions which might occur as tows tried to increase speed while travelling downstream. Even with these factors which could serve to magnify tow traffic impacts on any organisms, the in-sailing-line (ISL) substrates had significantly more taxa and higher diversity than out-of-sailing-line (OSL, substrates. Even more important, however, was the higher biomass on ISL than OSL substrates. Although this experiment was limited in scope to one site and one sampling event, there was no indication that tows had any significant adverse effects on the benthic macroinvertebrate community. Since the biomass of ISL substrates was greater than that of OSL

substrates, it appears that benthic macroinvertebrate production is not being directly affected by present tow traffic levels. In downstream areas of the Winfield Pool, deeper water would probably eliminate any possibility of impacts from tow passage.

In the Winfield Pool of the Kanawha River, production appears limited at this time more by the amount of cobble/ pebble substrate, than by tow passage. The structure, function, and production of the benthic macroinvertebrate community has become adjusted to the present levels of tow traffic. An increased increment of tow traffic would probably have no adverse effects on benthic macroinvertebrate communities.

VI.9 FISH

The results presented here represent the first reported estimates of basic population parameters for 18 fish species from a large river system. Comparisons with published data from similar systems, though desirable, are not possible. Lotic systems in general have not been studied to the same degree as lentic systems. Few published accounts of fish production in warmwater streams are available in the literature (Neves 1981). Data from large rivers are scant. Mann (1964, 1965, 1967, 1975) and Mann et al. (1972) have provided a thorough investigation of the River Thames, but this system is noted for an unusually high abundance and production of fish (Burgis and Dunn 1978, Chapman,

1978). Furthermore, the species studied in the Thames are not found in the Kanawha River. More general comparisons with published data will be made where possible.

The fish community of the Kanawha River appears typical of a large river, both in terms of total biomass per unit area and its distribution among species and trophic groups. Total biomass density (242 kg/ha) is similar to that observed from upper Ohio river fish surveys (Pearson and Krumholz 1984). The fish community of the Kanawha River is dominated by bottom feeders as is typical of reservoir fish communities (Jenkins 1975) Gizzard shad, common carp, channel catfish, and smallmouth buffalo together account for about 70% of total fish biomass. The first three of these species are, by weight, ranked in the same order of importance in both the Kanawha and the Ohio Rivers (Preston and White 1976). These observations suggest a high degree of dependence on detritus and other allochthonous sources of energy as predicted by Cummins (1977). In this case we would not expect significant changes in the fish community associated with increased tow traffic, since tow traffic density is unlikely to decrease these sources of energy.

Gizzard shad are by far the most abundant and productive fish species in the Winfield pool, accounting for approximately one-quarter of total fish biomass and one-half of all fish production. Although gizzard shad are reported to feed

extensively as adults on phytoplankton in some systems (Cramer and Marzolf 1970), their diet in the Kanawha River is based almost entirely on benthic detritus. Similar observations were reported by Jude (1973) and by Pierce and Wissing (1981). Gizzard shad appear to benefit from a short and direct food chain. As mentioned, detrital sources of energy are not likely to be affected by increased navigational use of the river. Zooplankton have been shown to be an important forage item of young of the year and juvenile shad (Kutkuhn 1957, Cramer and Marzolf, 1970). But the nearly complete absence of planktivory among any of the fish species analyzed for diet suggests that this is not the case for adult fish in the Kanawha. However, the larval stages of most species probably rely on small zooplankton.

Gizzard shad and smallmouth buffalo are considered both herbivore/detritivores. This group consists of only 4 taxa in all. The common carp and channel catfish are classified as omnivores together with 8 other species. Thus herbivore/detritivores and omnivores, 14 out of the 66 taxa observed in the study (21%), account for 75% of the total biomass. Ths remaining one-quarter of total fish biomass is divided more or less evenly between 39 species feeding primarily on invertebrate insects and 13 species utilizing other fish and crayfish for food.

For the purposes of this study specific diet items were arranged into 5 food types; fish, crayfish, invertebrates, plants, and detritus. Only a few of the fish species studied for diet compositon fed exclusively on a single food type. Sauger, longnose gar, and flathead catfish fed almost exclusively on fish. The remaining 14 species of Table V.9.9 each included 2 or more food types in their diet. While the ability to forage on a variety of food types may provide these species with alternative sources of energy, each species exhibited a preferred food type. Declining availabiity of such a preferred food type would in all probability decrease productivity and/or abundance of the species, and not be completely offset by a shift to other foods.

The proportional annual consumption indices confirmed our general expectations regarding food habits of Kanawha River fish species, with 2 notable exceptions. Both the emerald shiner and the gizzard shad were expected to rely at least in part on plankton as a food source, but neither phyto- nor zooplankton figured significantly in the diet of either fish species. Both fish are abundant throughout the study area, and this suggests that phytoplankton and zooplankton do not represent as important an energy pathway as was anticipated. Gizzard shad were observed to feed on periphyton, but benthic detritus constituted the bulk of their diet. Emerald shiners fed primarily on invertebrates, mainly larval insects and small bivalves.

The trophic group composite diet summary in Table V.9.17 revealed that food habits varied considerably between study sites. There was a general increase in the utilization of forage fish at LW by those trophic groups which consume fish, and a concordant decrease in invertebrate utilization across trophic groups. This shift took place despite an apparent decrease in abundance of midwater invertivores, important forage fish. These observations suggest diminished availability of invertebrate insects at LW, a conclusion also supported by the apparent decrease in relative abundance of benthic invertivores there as well.

Specific diet consumption, that is the relative importance of specific food items of each food type in the "diet" of a trophic group, also exhibited some differences between LW and UW. Small cyprinids were the main fish-type forage (80.9%) of piscivores at both sites, but centrarchids became increasingly important at LW. This probably reflects an increased availability of sunfish and the decreased availability of shiners in this portion of the pool as indicated in Table V.9.3. The specific components of fish-type forage in the diet of crayfish/piscivores also shifted between the two sites. Cyprinids were the main diet item at UW, but were replaced by young catfish at LW. Benthic invertivorous fishes consumed a significantly higher proportion of pelecypoda (bivalves) at UW than LW, where Odonata and terrestrial insects were the more

important forms of invertebrate forage. The same kind of shift was reflected by omnivorous fishes; decreasing utilization of bivalves and increasing use of terrestrial invertebrates from UW to LW. Differential availability of these foods between sites is implicated by the consistent response of different trophic groups.

VI.10 LARVAL FISHES

Distribution

Larval fish were collected on all sampling dates with peak densities occurring in late spring - early summer. Species composition in the Kanawha River is similar to that of the Ohio River, with species having pelagic eggs and/or larvae being the most abundant: unidentified cyprinids (majority presumed to be emerald shiners), clupeids (primarily gizzard shad), and freshwater drum. Species compositions were very similar at UW and LW except Pomoxis spp. were found almost exclusively at UW. Several taxa were more abundant at UW than LW at the onset of their spawning season, but as the season progressed, densities at LW equalled or exceeded those at UW.

Abundance did not vary appreciably with distance across the river (20, 40, and 50% distance across from one bank) at either UW or LW. However, shoreline samples usually had densities

several times higher than those of the main river channel. Differential gear susceptibility was observed in the gear comparison analysis and may be responsible for at least some of the observed differences between shoreline and river densities. Lepomis, spp. clupeids, percids, and especially cyprinids (excluding carp) often had significantly higher densities in shoreline samples during both day and night. Common carp and freshwater drum were essentially absent from the diurnal shoreline samples, but at night, their densities were significantly higher in the shoreline samples than in the river samples. This heavy use of shoreline habitat was not evident for Ictiobus-Carpiodes or Pomoxis. Those taxa that were abundant in the river samples were also the same that were abundant along the shoreline.

Abundance did not vary with depth at UW for any species except freshwater drum. This may likely be a result of the greater mixing of water in this portion of the river. Freshwater drum larvae showed the same preference for the bottom of the river at UW as they did at LW. Vertical differences were apparent for several taxa at LW, where depth was much greater and water velocities slower. Freshwater drum were generally more abundant near the bottom at LW, especially during the day. Common carp displayed a similar pattern, tending to be more abundant in the bottom samples during daylight, but equally dispersed or near the surface at night. Cyprinids (excluding

carp) were more abundant near the bottom during both day and night in June, but showed no vertical preferences on other sampling dates. Clupeids were more abundant at middepth or near the surface during daylight, while being equally dispersed or near the bottom at night. This is similar to vertical patterns observed by Graser (1979) which he attributed to their positive phototaxic response.

Diel changes in catch varied with specie and sampling date, but the general trends were that nocturnal densities either equalled or exceeded those of daylight samples. Clupeids were an exception to this. In early July, diel variations in clupeid catch were similar to those reported by Graser (1979), with daylight densities exceeding those at night. It is not clearly understood to what extent these observed diel differences in catch were influenced by changes in gear avoidance, microdistribution patterns (Gallagher and Conner 1983), and diel movements of larvae between the main river channel and the shoreline habitat.

On the basis of distribution patterns alone, impacts of barge traffic on larval fish would probably be greatest when barges disturb shallow nearshore areas. Larval fish are present from May through August with greatest densities likely in June and early July.

Barge-Induced Mortality

The findings of this study are similar to those reported by Holland and Sylvester (1983b). They observed a significant increase in the percentage of damaged freshwater drum eggs following barge passage (mean increase of 14%), but could not discern a similar trend in larvae. In this study percent of live larvae (all species combined) after barge passage decreased in June samples (31.9% to 18.3%) and increased in July samples (21.5% to 31.0%) though not significantly due to variability among samples.

Three explanations for those findings may be plausible. First, barges may not kill or damage significant numbers of larvae. Morgan et al. (1976) reported that for striped bass, larvae could survive shear forces better than eggs, but the eggs and larvae of white perch did not differ. Unfortunately, similar information for species common to the Mississippi River drainage is not available. Second, Holland and Sylvester (1983b) suggest that the mixing of nonimpacted larvae with impacted larvae with the barge wake may dilute the damaged larvae to an imperceptible percentage. This implies that larvae have a lower barge-associated mortality than eggs; otherwise, Holland and Sylvester (1983b) would not have found a significant increase of damaged eggs following barge passage. Third, the sampling methodology used in this study, as well as by Holland and Sylvester (1983b) may have inherent shortcomings in assessing barge-associated

mortality of larvae. Avoidance of nets by larvae is believed to be triggered by visual clues and pressure waves moving ahead of towed nets. The "before" samples were collected in relatively calm water; whereas the "after" samples were collected in the turbulent wake of barges. Turbidity levels may increase following barge passage reducing visibility, and the turbulence within the wake may disguise the pressure waves of an oncoming net. Reduced avoidance capability could result in a higher percentage of live larvae being captured than if the water's physical conditions matched those existing when the "before" samples were taken. Though not significantly different ($p = .123$), it is curious to note that in July the mean percent of living larvae was higher in the "after" samples.

Laboratory Experiments

Larval mortality varied with inflowing velocity and not with the volume of water passing through the chamber. This suggests that the major cause of mortality was associated with the inflowing jets of water and not impingement against the Nitex divider.

It appears that there may be a relationship between size of larval fish and survival after exposure to high velocity, turbulent water flow. The smallest larvae tested (bluegill) had the lowest survival, while the largest larvae (channel catfish)

had 100% survival. Carp and walleye (intermediate in size) had some mortality but considerably less than that experienced by bluegill. The presence of suspended sediments (125-150 microns) decreased bluegill survival but had no effect on channel catfish, indicating that mortality in fragile species will likely increase if suspended sediment concentrations become elevated.

Hochstein and Adams (1985) report that the highest velocities resulting from towboats will normally occur in the midchannel region (sailing line) and in shallow water along the shoreline. Equations presented in their paper predict that propeller jet velocities for a 5500 horsepower towboat with twin 183 cm props may exceed 650 cm/sec within a radial distance of 91.5 cm of the propeller axis. However, as radial distance doubles, maximum propeller jet velocity decreases by 50%. This limits the high velocities of the prop wash to a narrow zone within the river cross section. Hochstein and Adams (1985) predict that these midchannel velocities are sufficient to temporarily increase suspended sediments, especially in the shallow portions of the river. Their predictions of tow-generated diverging waves indicate maximum wave heights of 30.5 cm, which will create bottom velocities of 290, 119, and 31 cm/sec at depths of 15.2, 30.5, and 61 cm respectively, as they break along the shoreline. Hochstein and Adams (1985) predict that wave action will not significantly increase suspended sediments, but their predictions are based on bottom wave

velocities occurring at a depth of 61 cm. They failed to discuss what the bottom wave velocities occurring in water less than 61 cm will do to suspended sediment concentrations. These higher velocities will likely increase suspended sediments temporarily.

It is impossible to say to what extent conditions in the stress chamber simulate the forces larvae in a river may be exposed to; however; inflow velocities were 800 cm/sec and higher in the stress chamber. These far exceed those predicted for the shoreline, but are not much higher than potential propeller jet velocities. Based on survival rates observed in the stress chamber, it is likely that some direct mortality will occur for small pelagic larvae entrained through the propellers or subjected to high velocity propeller wash, especially if abrasive suspended sediments are present (high ambient river velocities or tow operation in shallow water). The pelagic larvae likely to be affected the most are the yolk sac larvae of the three most common taxa in the Kanawha River: clupeids, (primarily gizzard shad), freshwater drum, and unidentified cyprinids (the majority of which are likely emerald shiners). Direct mortality along the shoreline is likely to be negligible except in the very shallow edges less than 15.2 cm deep. Larval fish distribution data show the shoreline to be an important nursery habitat in the Kanawha River. However, all shoreline distribution samples were collected from areas where water depth exceeded 55 cm; the use of habitats less than 15.2 cm in depth is unknown at present and deserves further study.

VI.11. ENERGY FLOW MODEL

It is apparent from all results that impacts to the biota of the Kanawha River associated with energy flow and its interuption by tow traffic are minimal. This is the result of the small contribution autochthonous sources of energy make in the overall functioning of an ecosystem dominated by allochthonous sources. The following discussion includes the results predicted for the worst-case tow traffic projection, the experimental reduction of autochthonous energy sources, and the energy flow analysis. Limitations and shortcomings of the model are also discussed.

Kanawha River Energy Flow Pathways

The results of the flow analysis suggest that the Kanawha River ecosystem is supported primarily on allochthonous sources of energy. Given that the ecosystem is supported almost entirely by detrital energy sources, one would expect minimal impacts from increased navigation use on energy flow to the higher trophic levels. This expectation is confirmed by each of the simulations performed by the model.

The major energy flow pathways throughout the ecosystem were determined by comparing the magnitude of total energy flows through each of the trophic groups (Table VI.11.1), and then

observing the relationships between the flow-dominant groups. Flow dominance as a trophic group is in some way a measure of importance or success in the ecosystem. The most significant flows within similar taxonomic groups are all clearly related to detritus, either as suspended material or as benthic materials. The fluxes through SPOM are the largest flows in the entire system, implying that the ecosystem is highly dependent upon allochthonous materials originating from upstream areas. Among the invertebrate insects, the dominance of collector gatherers as processors of energy is apparent. It would also appear that collector filterers make a significant contribution as a channel of energy flow, especially at UW where solid substrate and water currents are not as limiting. Invertebrate shredders play only a minor role in the Winfield Pool ecosystem. Unlike a headwater system in which terrestrial plant litter provides the major source of energy (Fisher and Likens 1973), the pattern observed among the invertebrate groups in the Kanawha suggests that organic materials which have already undergone appreciable processing (Cummins 1977) are the main energy source. This would explain the relatively large role of filtering molluscs, as well as the great success of the herbivore/detritivorous fish, both of which subsist largely on fine particulates. The other abundant fish groups such as the omnivores and midwater invertivores are successfully functioning as trophic generalists which utilize detritus. The more specialized feeders which do not directly

consume detritus (e.g. benthic invertivorous, crayfish/piscivores, and piscivores) are not as abundant. Throughout all trophic levels, then, a consistent trend is observed in which organisms which are able to take direct advantage of allochthonous inputs dominate the community.

These findings concur with those of Mann (1964, 1965) and Mann et al. (1972) for the Thames River, England. Like the Kanawha, the Thames River carries waste discharges and has been developed for navigation. While phytoplankton productivity is lower in the Kanawha than in the Thames, energy processing predominantly involves allochthonous materials in both systems. Benthic invertebrates of the Thames are mainly detritus-feeding organisms, and the fish community is both directly and indirectly supported by detritus. Both rivers can be described as heterotrophic systems.

From a management perspective, the energy pathways leading to the harvestable fish groups become important. Omnivores (including channel catfish) are directly dependent on periphyton, molluscs, C.P.O.M., herbivore/detritivores, and terrestrial insects for 76.5 and 80.1% of energy consumed at UW and LW, respectively. The major flows are from periphyton and molluscs at UW, and from periphyton and herbivore/detritivores at LW. Crayfish, omnivorous fish, and midwater invertivores support 92.3 (UW) and 96.1% (LW) of the energy flow to the crayfish/piscivore

Table VI.11.1. Total energy flux through trophic groups.

	(kcal/m²/yr¹) UW	LW
x(1) phytoplankton	1640.	1438.
x(2) periphyton	271.2	208.
x(3) spom	53919.412	46641.997
x(4) cpom	42509.820	32212.326
x(5) fpom	22488.630	22445.356
x(6) zooplankton	375.780	974.010
x(7) coll/gatherer	1094.388	1039.592
x(8) coll/filterer	150.496	22.778
x(9) scrap/grazer	0.200	0.600
x(10) shredders	3.529	2.813
x(11) invt. predators	18.702	52.040
x(12) molluscs	510.320	510.320
x(13) crayfish	16.564	14.972
x(14) b.invertivore	8.346	4.925
x(15) omnivore	39.577	45.255
x(16) cray/pisciv	3.748	0.808
x(17) herb/detrit	311.689	297.241
x(18) m.invertivore	53.930	41.450
x(19) piscivore	5.743	7.899

group, which includes the smallmouth and largemouth basses. Crayfish mediate the most important energy pathway to these popular sportfish. Herbivore/detritivores (e.g. gizzard shad) and midwater invertivores (e.g. various shiners) supply 85.6 and 86.8% of the energy flow directly supporting the piscivore group (including walleye, sauger, and white bass) at UW and LW, respectively.

Note that benthic macroinvertebrates do not directly provide major support to any of the harvestable fish groups. This suggests that enhancement of invertebrate insect habitat through addition of rubble or snags would not (at least from an energy flow perspective) dramatically increase fish production. Hard substrates placed in the photic zone might increase periphyton abundance, however, which could in turn increase the energy available to herbivore/detritivores and omnivores.

Traffic Scenarios

Four different levels of tow traffic were tested for their effects on the biota of the Kanawha River. These included simulations of traffic corresponding to baseline (1982-83), future (2040) without project, Winfield replacement, and both Marmet and Winfield replacements. In as much as the various tow traffic simulations differed only in the frequency of passage, the following comments pertain to the results of the worst-case scenario in which both Winfield and Marmet locks are upgraded.

Other traffic levels tested (future without-project conditions, replacement of Winfield only) demonstrated intermediate results between baseline 1982-83 and the worstcase simulation.

Even under highest projections of future traffic conditions, the reductions in standing stocks predicted by the model were relatively minor. The greatest reductions in standing stock were, as would be expected, for the primary producers (phytoplankton and periphyton). Periodic interuptions of photosynthesis caused by tow passage reduced phytoplankton biomass by 15.17% at UW and 17.52% at LW. Periphyton were predicted to decline by 14.77% and 19.00% for UW and LW, respectively. The larger reductions predicted for LW were a result of greater tow frequencies simulated over this portion of the pool. The direct effects of tow traffic on primary producers, while in and of themselves appreciable, were predicted to result in only slight declines in biomass for other trophic groups of the ecosystem. These indirect effects will be discussed for three categories of trophic groups; invertebrates, vertebrates, and organic matter groups. The greatest reductions in biomass of invertebrates were for zooplankton (UW 2.82%, LW 2.07%), collector filterers (UW 1.27%, LW 1.53%), scraper grazers (UW 5.83%, LW 1.87%), and molluscs (UW 1.03%, LW 1.14%). These reductions were expected in that these are the organisms whose diet includes primary producer organisms. Other food sources are both available and utilized ,however, and these continued to

support the above-mentioned groups at near baseline levels. Other invertebrate groups, for example the collector gatherers, shredders, invertebrate predators, and crayfish, did not depend on authochthonous food sources to a large extent, and consequently did not experience significant declines in standing stock under the heaviest traffic conditions.

The same kind of results were predicted among the vertebrate groups, where those organisms most directly dependent on autochthonous sources of energy were the most effected. All but one fish trophic group, the herbivore/detritivores, remained within one-percent of baseline abundances (most within one-half percent); herbivore/detritivores were predicted to decline by 1.18% at UW and 1.43% at LW. Their decline resulted from a greater utilization of periphyton than other fish groups.

Suspended particulate, benthic coarse, and benthic fine organic matter all declined by quite small amounts (< 0.27%). Since only a small fraction of the total available organic matter transported downstream is captured and utilized to begin with, these reductions are inconsequential with respect to ecosystem functioning. Overall then, trophic groups which directly depend on phytoplankton and/or periphyton were predicted to decline by a small amount, while the majority of the trophic groups demonstrated insignificant declines. Such minor impacts result from a ecosystem which functions almost entirely on allochthonous

sources of energy that are transported from upstream reaches. As a detritus-based system, then, the Kanawha River is relatively insensitive to reductions in the rates of photosynthesis by primary producer organisms. These organisms exist in the river, but do not mediate critical energy pathways which are essential to maintain the integrity of the present community. It is illuminating to note that 1) fish species such as gizzard shad which are physiologically adapted to utilize phytoplankton (Hendricks and Noble 1979, Pierce and Wissing 1981) opt instead to feed primarily on benthic deposits of organic matter, and 2) scraper grazer organisms which would feed upon periphyton were the least common of the benthic macroinvertebrates collected.

At present the community is adapted to function by processing allochthonous sources of energy. As such it is not surprising that there are no major changes in the community when autochthonous sources of energy are reduced by tow traffic. It is important at this point to distinguish between a periodic interuption of photosynthesis caused by traffic on the river and the complete absence of primary producers in the system. Should the physical conditions of the river continuously and severely reduce photosynthesis, some amount of non-photosynthesizing phytoplankton would still continue to be transported from upstream reaches and would still be available for organisms to consume. In this regard the model's predictions are conservative, in that no such exogenous supply is explicitly modeled as being

available for consumption. Another conservative feature of the model is the nature of the interruption to photosynthesis. The model assumes that no photosynthesis would occur for a period approximately twice the length of time Hochstein and Adams (1985) predicted for materials suspended by tow passage to settle out (i.e. return to ambient pre-passage conditions). Furthermore, the zone of the river channel which experiences temporary pulses of increased turbidity has been estimated as about a third of the total river's width (Hochstein and Adams 1985), while the model assumes complete and homogenous cessation of primary production throughout the reach of river recently affected by a passing tow. In light of the above, there is little concern that the potential impact of additional tow traffic on energy flow in the ecosystem has been underestimated.

One possible shortcoming of the model is its use of annual average flows of energy applied as constants throughout the simulations. The program which runs the model includes mechanisms to accomodate time dependence, but such refined data are not presently available for all energy flows. Seston availability peaks in February-March and reaches a minimum during October-November when total leaf inputs are at their greatest. Primary production peaks during the June-August period. One might hypothesize that the importance of primary production and the potential for impacts might be greater during the summer growing season when river flows are within their lowest and most

stable range. This is a valid criticism, but given the availability of particulate organic matter during this period we do not believe this to be a serious error.

A second possible shortcoming of the model is the grouping of all life stages of organisms into a common trophic entity. It is known that the feeding habits of most organisms vary with their age and size, but determination of food habits at all life stages for the major organisms found in the river was beyond the scope of the study. In particular, larval fish were not explicitly represented in the model. We offer that the perturbation of concern is the increment to existing levels of tow traffic, under which the nutritional or other dietary needs of immature forms are apparently met. Recruitment is sufficient as the system presently functions, and no change is anticipated in the functioning of the ecosystem. Given that the ecosystem is essentially driven on detritus, and that detrital sources are not reduced by increased traffic, this shortcoming of the model does not affect the predictions made.

Experimental Simulation

The experimental simulation was intended to determine if primary production played any significant role in the functioning of the ecosystem. The results suggest that it does not. Apparently primary production contributes only incidentaly to the

overall pattern of energy flow in the Kanawha River. The few heterotrophic groups for which autochthonous sources of energy are critically important (zooplankton and scraper grazers) are neither themselves abundant nor are they depended upon to any appreciable degree by other trophic groups. The experimental simulation demonstrates that the model is capable of responding (with meaningful changes in abundance of sensitive trophic groups) to reductions in autochthonous inputs. However, the sensitive groups do not mediate important flows of energy in the ecosystem. Consequently reductions in these flows of energy have a minimal effect on the integrity of the present-day ecosystem.

VII. CONCLUSIONS

1) The Kanawha River has a history of having some of the most atrocious water quality in the United States. However, since the 1960's water quality has improved. During 1982-83, the water quality at the upper and lower ends of the Winfield Pool did not present conditions what would be deleterious to aquatic biota. Temperature, pH, and dissolved oxygen were well within West Virginia water quality standards, and the alkalinity indicated a fairly well buffered river.

2) The terrestrial inputs (leaves and insects) are within the range of values reported for various rivers around the world. A substantial amount of allocthonous detritus (5.78 - 11.88 gDW of leaves and 0.47 - 1.04 gDW of insects per m^2 of total surface area) is added to the Winfield Pool from the riparian zone.

3) There are relatively low percentages of organic detritus in the bottom sediments. Apparently the Kanawha River has sufficient ambient current or frequent enough high discharge events to keep most particulate detritus in transport. The bottom is composed primarily of coarse materials (sand or larger), even in the downstream sections of the Winfield Pool. This suggests that the Winfield Dam does not produce a habitat that is typical of a reservoir, but rather that the lower reaches more closely resemble a pool of a large river.

4) Organic seston concentrations are within the range reported for other rivers of similar stream order. Most of this suspended detritus consists of particles < 25 μm in diameter. Temporal and spatial patterns indicate that seston concentration varies according to discharge. The total amounts of organic seston transported annually are tremendous (30,426,837 - 35,057,119 kgAFDW).

5) Bacterial densities are on the order of 10^5 - 10^6 cells/ml. Unattached bacteria account for > 95% of the total bacterial population. The density of attached bacteria is greatest on the smallest sizes of detritus particles. Bacteria achieve greatest densities in mid-winter, with discharge and conductivity exerting the greatest influence. There is little relationship between seasonal distribution of bacteria density and phytoplankton biomass.

6) Concomitant with improved water quality, the biological communities have become more normal for a river the size of the Kanawha. Phytoplankton diversity is reasonably good, but the density of organisms is somewhat low in comparison to other similar rivers.

7) Measurements of primary production illustrated the heterotrophic nature of the system. Net primary production occurred (photosynthesis exceeded respiration) in only a few

summer months. For the phytoplankton, when the production rates were integrated over the water column, respiration was always greater than photosynthesis. The periphyton community exhibited higher rates of gross primary production per unit area than the phytoplankton community below an equivalent area of water. Th production: respiration ratio for periphyton communities was usually one. The levels of primary production observed in the Kanawha River are not exceptional for a large river in a municipal/industrial watershed.

8) Zooplankton densities have increased markedly since the 1960's. The zooplankton community consist almost entirely of rotifers, as is typical for large rivers. Total rotifer concentrations are characterized by a single summer population peak. There is no evidence that entrainment of rotifers in towboat propellers causes mortality, because samples collected directly behind passing tows contained no more damaged rotifers than samples collected away from the sailing line.

9) The benthic macroinvertebrate community found in the Winfield Pool of the Kanawha River has changed since the 1960's. Whereas oligochaetes were extremely dominant, now the dominant organisms are members of the family Chironomidae, and oligochaetes are not found in high abundances. This is probably related to an improvement in water quality over the 20 year period. The composition and densities of the macroinvertebrate

communities of the Winfield Pool of the Kanawha River are similar to other large rivers.

10) Annual production of benthic macroinvertebrates is intermediate among values reported in several rivers. Differences in production levels among these rivers appear to be related to quality of food and physical habitats. Production in the Kanawha River is heterotrophically (detritus) based and is limited primarily by the availability of firm (cobble/pebble) substrate.

11) Although the present levels of tow traffic may influence the structure of the benthic macroinvertebrate community in the sailing line at the shallow end of the pool, the macroinvertebrates have adjusted to the present levels of traffic and produce as much as macroinvertebrates outside of the sailing line. Hence, an increased increment of traffic would not likely have an adverse impact on benthic macroinvertebrate communities.

12) A total of 66 fish species were collected during the study. Age, growth, and mortality data were reported for the 18 most common species. These were (in decreasing order of abundance at UW): gizzard shad, common carp, smallmouth buffalo, channel catfish, shorthead redhorse, golden redhorse, smallmouth bass, sauger, emerald shiner, freshwater drum, spotted bass, white bass, longear sunfish, largemouth bass, white crappie, bluegill, and spotted sucker.

13) Total fish biomass based on recent lock-rotenone collections was estimated at 242 kg/ha, which was similar to the estimate made in the upper Ohio River.

14) Gizzard shad, common carp, channel catfish, and smallmouth buffalo were the dominant fish species by weight; together they comprised 70% of the total fish biomass.

15) Trophic structure of the fish community was dominated by herbivore-detritivores, which made up 44.0 to 45.5% of the total biomass, and omnivores, which made up 31.9 - 37.5% of the total biomass. Piscivores, benthic invertivores, crayfish-piscivores, and midwater invertivores together comprised the remaining 17.0 to 24.1% of the total. Omnivores and piscivores were more abundant at LW than at UW where crayfish-piscivores, benthic invertivores, and midwater invertivores were more abundant at UW than at LW.

16) Total fish production was estimated at 25.8 kcal/m^2/yr at both sites. Gizzard shad alone accounted for 43 to 46% of the total fish production. Diet of gizzard shad in the Kanawha River was based almost entirely on benthic detritus.

17) Small differences in fish species composition between UW and LW were noted. Silver, golden, and shorthead redhorse, walleye, white bass, and smallmouth bass comprised a greater proportion of total biomass at UW and upper Gallipolis than at

LW, while freshwater drum, bluegill, black crappie, and spotted sucker comprised a greater proportion of the total at LW than at UW and upper Gallipolis.

18) Larval fishes were collected on all sampling dates from May through August with greatest densities in June and early July. Cyprinids (excluding carp), clupeids, and freshwater drum were the dominant taxa, and comprised nearly 98% of the 22,136 larvae collected. These three species all have pelagic or semi-pelagic eggs and/or larvae.

19) Species composition and phenology of larval fish in the Kanawha River was similar to that previously reported in the Ohio River. Species composition was similar between UW and LW except for Pomoxis spp. which were found only at UW.

20) Densities for several taxa were greater at UW than at LW early in the season, but as the spawning season progressed densities at LW equalled or exceeded those at UW.

21) Larval fish abundance did not vary significantly across the main channel (20, 40, and 50% of width) at either LW or UW. However, for most taxa shoreline samples usually had densities several times higher than those of the main river channel.

22) Freshwater drum and common carp larvae showed a preference for near-bottom areas of the river, especially during the day, at both sites. Clupeids were more abundant at middepth

or near surface during daylight, while being equally dispersed or near the bottom at night.

23) The percentage of live fish larvae (all species combined) was not significantly different between samples taken in the sailing line before and after barge passage. Evidence from other rivers indicates that the egg stage of freshwater drum is more vulnerable to damage than the larvae.

24) Bluegill, common carp, walleye, and channel catfish larvae were exposed to turbulent water flow in experimental stress chambers. Bluegill larvae had higher mortality rates (6-20%) than other species, whose mortality rates were less than 1%. Mortality was related to size of fish larvae. In experiments in which 220 mg/l of sediment were added to the chamber, bluegill larvae mortality increased (difference: 12.9%) while channel catfish was unchanged.

25) Direct impacts of towboats on larval fish are expected only in the narrow prop wash zone in shallower portions of the river channel and in the shallow shoreline zone. Smallest larvae (centrarchids, clupeids, freshwater drum, and cyprinids) are expected to be most fragile.

26) Energy flows involving particulate organic matter dominate all other energy flows within the Kanawha River ecosystem.

27) The most abundant organisms at each trophic level in the ecosystem directly utilize suspended or benthic sources of detritus.

28) The major effect of tow passage on the ecosystem is a temporary reduction in the rates of photosynthesis by primary producers.

29) Projected increases in navigation use are not predicted to significantly reduce the standing stocks or production of river biota.

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