Impacts of Phosphorus on Streams

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IMPACTS OF PHOSPHORUS ON STREAMS

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Final Report of the Phosphorus and High-Flow Field Studies

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ABSTRACT

The purpose of this study was to define methods of dealing with phosphorus in setting appropriate stream water quality goals or standards. In-stream and sediment nutrients were compared to rooted plant and attached algae growth in southern Wisconsin streams, 1981 and 1982. The resultant impacts on stream diel dissolved oxygen (DO) characteristics were also investigated. Three empirical models describing macrophyte biomass, tissue phosphorus content and in-stream phosphorus are presented. Results of the analyses suggest several different stream types, differing in the percent contribution of in-stream nutrients as opposed to sediment nutrients. Stream periphyton were also collected from glass slide and brick substrates. Models describing brick periphyton community biomass, tissue phosphorus content and in-stream phosphorus, similar to macrophyte models, are also presented. Single-station and double-station diel DO curve analyses as well as light/dark productivity studies are compared to in-stream primary producer biomasses. Maximum night-time DO deficit can be described as Respiration divided by Reaeration. This estimate, when combined with the ability to predict plant biomass from in-stream nutrients using the primary producer biomass models may allow prediction of the impact of changing phosphorus concentrations on small stream dissolved oxygen minima. In addition to investigating the impacts of phosphorus in small stream systems, the study also evaluates methods of documenting phosphorus impacts and recommends monitoring strategies.

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INTRODUCTION AND STUDY BACKGROUND

PHOSPHORUS WATER QUALITY STANDARDS

Any assessment of water quality must be made relative to water quality guidelines or criteria. Water quality standards and allowable concentrations of most chemical constituents have been developed and demonstrated based on toxicity or production of conditions unfavorable to public health or aquatic life. With the exception of the elemental form, no such Federal water quality standards have been set for phosphorus. Inability to define phosphorus as a pollutant in the conventional sense (e.g. toxicity, human health hazard) has hindered establishment of water quality phosphorus standards and related wastewater effluent limits. The U.S. EPA "Red Book" (1976) does, however, recognize phosphorus as a contributor to accelerated lake and stream eutrophication, and suggests levels of total and ortho-phosphorus which would slow enrichment of surface waters.

SOURCES OF PHOSPHORUS

Phosphorus is delivered to surface waters from both point and non-point sources. Generally, non-point source inputs are described by the total load of a particular pollutant. In terms of annual loading, storm events usually supply the greatest amount of nutrients. Nutrient concentrations as well as total loads surpass those of low-flow conditions. Lake or watershed management and planning are usually based on nutrient and sediment loadings. In most lake assessments, loadings are the criteria most often used to predict a lake's response to changes in nutrient inputs.

It is unlikely, however, that the forms of phosphorus in runoff are readily available to stream primary producers. Streambed scouring, light-limitation (increased turbidity), nutrient sorbtion on suspended solids and relatively short contact times between nutrients and stream primary producers would act to limit plant response to storm-induced nutrient delivery. Storm flows may remove stream periphyton and macrophytes, also reducing potential for nutrient uptake. In reach-specific analyses, stream primary production is probably influenced more by low-flow or sustained nutrient concentrations they experience rather than seasonal or event non-point source nutrient loadings.

Based on their ability to assimilate wastes, streams have served as natural treatment systems for wastewater. The rate of enrichment of a stream varies with the amount of dilution water available. The rate of biological response to introduced nutrients, in turn, is dependent on physical factors such as light, substrate type and stability, water temperature, depth and velocity.

Point source problems have usually been associated with inadequately treated wastewater or inadequate stream dilution. Excessive loads of unstabilized waste material results in lowered stream dissolved oxygen. Modern, efficient treatment plants are designed to reduce the high oxygen demand component, discharging mineralized, biologically available nutrients. Specific water quality standards have been applied to streams based on a particular stream's natural low-flow potential, using existing and potential aquatic life uses to determine acceptable levels of a particular waste discharged. Traditionally, the discharge constituents of concern are dissolved oxygen, ammonia, BOD, residual chlorine and solids.

STREAM PRIMARY PRODUCER COMMUNITIES

As the group directly able to use phosphorus, and responsible for many undesirable water quality conditions, primary producers offer the best opportunity to evaluate the impacts of phosphorus in stream environments. Streams are dynamic systems that support an extremely complex and variable biological community. Physical factors exert considerable influence on stream primary producers and modify this

community's response to nutrient enrichment. This makes measurement of stream biological response to nutrients difficult. Poor definition of water quality impairments relative to system productivity and nutrient inputs also impedes definition of optimum or desirable nutrient levels.

Phytoplankton have frequently been used to measure lake and large river response to nutrient enrichment. Deep, slow-moving stream reaches or backwater areas may support substantial short-term phytoplanktonic populations, which are probably flushed out by storm flows. While phytoplankton may be present in smaller streams, attached algae are predominant. Suspended algae in smaller rivers may actually represent detatched periphyton. Reliance on measurement of phytoplankton production to assess nutrients in small stream environments is therefore not dependable.

Because of their rapid turnover rates and recovery from catastrophic events, production of the periphyton component has frequently been used to study nutrients in streams. Unlike macrophytes, this group obtains all of its nutrients from the water, thus measurement of periphyton to assess the effects of nutrients is not complicated by estimating the contribution of sediment nutrients to community growth.

A variety of methods and approaches have been used in the assessment of periphyton communities. These include biomass measurements, indicator species, community diversity indices, tissue nutrient content and enzyme activity. Methods of measurement and environmental response of primary producers in streams have been reviewed as part of this study (Mace, et al. 1983).

For most macrophytes to become dominant in streams, the bottom substrate must be able to support growth, and the nutrient supply must be adequate. Other factors, such as light, water depth and velocity must also be conducive to growth at the proper time of year. Unlike periphyton, macrophyte growth nutrients may be obtained, to various degrees, from bottom sediments. This last factor complicates somewhat the use of macrophytes to determine the nutrient status of a particular stream. Macrophytes are also susceptible to tangling and breakage caused by stream current. Their large size, abundance and ability for producing "nuisance" conditions, however, increase their desirability as study subjects.

IMPACTS OF PRIMARY PRODUCERS ON STREAM ENVIRONMENTS

Macrophytes can, however, modify the stream environment to make it more hospitable for plant production. Primary producers impact the stream environment in a variety of ways. Excessive macrophyte growth can alter the stream channel by encouraging sediment deposition. By increasing sedimentation and water depth, macrophytes create more favorable conditions for growth. Through channel modification, perpetuation of macrophyte communities is almost assured. This may lead to a point where native fish and invertebrate populations lose quality habitat.

Filling of stream channels by macrophytes can cause flooding, navigational, aesthetic and dissolved oxygen (DO) problems. Macrophytes can, by retarding water flow and increasing the stream depth, lower stream re-oxygenation rates. Coupled with plant and animal respiratory oxygen demand, this could lower the night-time DO concentrations below the desirable level for a designated stream use.

Standing crop biomass does not necessarily reflect actual growth and production within a particular reach. Due to fragmentation or other causes, biomass export results in a substantial seasonal downstream loss of periphyton and macrophyte growth. Even if macrophyte growth (standing crop) does not cause severe problems within a particular reach, export of produced material can provide a considerable load of organic, oxygen-demanding materials to downstream lake, impoundment or riverine systems.

NEED FOR STUDY

O'Shaugnessy and McDonnell (1973) state that "trends to control discharges ... clearly dictate the need to establish effective procedures for identifying those elements primarily responsible for accelerating the rate of eutrophication in a given situation." The level of technical expertise is not always able to quantify observed or perceived problems and their causes. The public's perception of water quality conditions, especially use impairment, is also not necessarily directly related to empirical assessment of water quality. Basic to any classification or management scheme should be a definition of what constitutes an undesirable condition. Tools (such as models or indices) to evaluate nutrients in stream environments are also not always available.

The need for simple, reliable methods to evaluate the impacts of nutrient loadings to lakes and streams has led to a large number of approaches with an equally large number of methods and indicators. General agreements have emerged as to the methods of approaching and solving lake problems. Of particular significance are the relatively simple tools and procedures used for analysis and assessment of lake water quality -- usually water transparency, algal biomass, and the causative nutrient levels. Sufficient research, directed at solving specific problems, has been integrated into whole-lake approaches to be able to address lake water quality problems with some confidence.

Of most importance has been a further definition and refinement of lake use categories and classifications, nuisance conditions, and application and evaluation of various lake management and protective strategies. Many of the concepts developed and applied to lakes and ponds (lentic systems) are not directly applicable to streams (lotic systems). The predictive methodologies developed for lake systems are based on simple and useful parameters. No such tools are currently available for allocating phosphorus in a site-specific manner to flowing water.

If a defensible position and regulation of phosphorus discharges to small stream systems on a site-specific basis is to be taken, the following concerns must be addressed:

- What level of enrichment (or aquatic plant growth) is considered objectionable?

- Which community and level of function will provide the best tool to evaluate the impacts of increased or decreased phosphorus levels?

STUDY BACKGROUND

The overall purpose of the Phosphorus Assessment study is to define methods of dealing with phosphorus and non-point source pollution in setting appropriate water quality goals or standards. The study was conducted in phases, beginning with a review of past efforts to establish phosphorus and non-point source water quality standards or objectives. Reviews included other state's and agencies' approaches to defining phosphorus water quality standards and limiting non-point source impacts (Lewis 1980, Warn 1980), the ability of chemical and biological water quality indices to assess sediment and nutrient impacts on streams (Warn 1980, Chantry 1981, Schrank 1982, Wawrzyn and Randall 1983, Narf in prep.), lake and stream classification schemes (O'Flannigan 1980, Schuettpelz 1982, Ball 1982), and stream primary producer responses to nutrients (Mace et al. in prep.).

The conclusions and recommendations of other agencies provided a framework to assess the feasibility of establishing phosphorus water quality standards and non-point source control objectives in Wisconsin. The topical reviews formed the technical basis for selecting specific areas where more detailed investigation was required. These investigations constituted the second study phase. The direction given for the field studies then focused project efforts on the phosphorus control element (Mace et al. 1982).

In Wisconsin, water quality changes due to point source discharges have been evaluated through pre and post-operative surveys, basin-wide assessments, wasteload allocation studies and, recently, stream classifications. These studies document stream response to gross changes in wastewater characteristics of dischargers. These methods, however, are generally not adequate to evaluate stream response to changing phosphorus inputs, nor for allocating phosphorus to small stream systems. The sensitivity (or level of resolution) of such methods must be able to assess the existing situation, responsibly recommend phosphorus limits to the discharger, and reliably predict water quality improvements resulting from specified phosphorus removal recommendations. For a particular assessment or allocative method to succeed, it must be empirically developed, tested and defensible.

Currently a small stream, dissolved oxygen model is available for use to allocate certain wastewater constituents discharged to streams. These allocations are designed to maintain a specified dissolved oxygen criterion for a designated stream use under critical stream conditions (e.g. high temperature, low-flow). Either directly through respiratory demand, or indirectly through channel modification, macrophyte growth will impact stream dissolved oxygen (DO). Modification of DO models or development of similar mathematical expressions could establish a link between phosphorus, plant growth and the associated impacts on stream DO. This might allow allocation of phosphorus directly through the impacts on primary production and indirectly through this community's impacts on DO.

If the data support such, establishment of phosphorus uptake (P-decay) characteristics due to stream assimilation, much as BOD is now allocated, could be an important consideration in assigning phosphorus discharge limits. A second alternative for controlling phosphorus discharges is to define levels of acceptable or unacceptable primary producer growth, based on community response to phosphorus, aesthetics, or physical changes. There is little information available on the use of subjective limits based on plant density or aesthetic conditions.

The field program was designed then, to establish a basis for phosphorus control (water quality objective or standard), a methodology for applying standards (assigning effluent limits) and evaluating water quality impacts (monitoring requirements). In addition to providing a defensible, scientific basis for phosphorus control in small streams, the field study results should also assist in specifying receiving system classifications or categories, associated water quality criteria and methods of applying selected criteria.

FIELD STUDY DESIGN

Specific objectives of the Phosphorus Assessment field study included:

- Quantifying the relationship between phosphorus and plant biomass;
- Recognizing the factors which modify the response of stream plants to nutrients;
- Quantifying the impacts of plant biomass on stream systems;
- Determining a level of acceptable plant biomass within a particular system; and
- Evaluating and recommending monitoring strategies for use in small stream water quality investigations.

This study was conducted over a two year period. Due to the uncertainty of controlling factors in most small stream systems, an approach to evaluate a variety of factors in intensely studied reaches was implemented the first year. Study sites were selected which represented a variety of stream types and physical conditions, nutrient and flow regimes, and dominant biotic community.

Initially, it was also desirable to collect data from systems not impacted by wastewater discharges to document stream behavior in the absence of point sources. These streams would then serve as benchmark, or reference data to compare with stream behavior in reaches impacted by wastewater treatment plants. This would also provide comparative data on the importance of non-point vs. point source impacts in considering the relative contribution of a discharge to a particular situation in the presence of NPS inputs.

Based on analysis of the first year's data, the second year of data collection included a larger number of streams to further investigate relationships and answer specific questions. In order to isolate and deal effectively and responsibly with the phosphorus question, a number of other concerns needed to be resolved. These questions included the contribution of sediment nutrients to macrophyte nutrition in streams, the influence of bottom substrate type in macrophyte colonization and growth, seasonal weather influences, and physical changes the primary producers themselves impose on the system.

There were no tested approaches (methodologies) available to evaluate the impacts of phosphorus on small stream water quality. The field studies, then, were also designed to develop assessment methods that 1) identify actual or potential "problem" conditions; 2) estimate the potential for improvement or degradation; 3) by comparison or modeling methods, recommend actions to remedy or prevent water quality deterioration due to phosphorus discharges; 4) project stream response to phosphorus reduction; and 5) develop the ability to predict changes in water quality based on changes in phosphorus concentrations.

METHODS

STUDY REACH SELECTION

Study reaches were selected which would best depict the impacts of nutrients on primary producer communities. Criteria used to select stream reaches were designed to minimize the effects of physical factors on the growth of primary producer communities in small streams. The criteria used to select the stream reaches were:

- Maximum reach depth 2-3 feet. Shallow depth would decrease the potential for light limiting plant growth;
- Mean annual flow less than 60 cfs. (small stream category);
- Maximum stream top width 60-70 feet (small stream category);
- Stream reach should be relatively unshaded and free of obstructions;
- Stream reach length should be a minimum of 300 feet to a maximum of 2,000 feet (1981 stream reaches were less than 300 feet).

Based on these criteria and existing water quality data 19 stream reaches were selected in southeastern and southern Wisconsin (see Appendix I). Seven stream reaches were intensive monitoring sites in 1981, four of which were expanded and monitored in 1982. Twelve synoptic stream reaches were selected and monitored in 1982. Sites were chosen which represented a variety of in-stream nutrient concentrations.

INTENSIVE STUDY REACHES

The major objective of monitoring the intensive study reaches was to evaluate the environmental factors impacting primary producer growth over the growing season. Water chemistry, macrophytes, periphyton, diel oxygen regime, substrate type and stream flow were monitored at the intensive study reaches.

Water Chemistry

Water chemistry samples were collected every two weeks from May through December

1981 and May through September 1982. Sampling times corresponded with primary productivity or diel DO surveys. During the diel surveys, samples were collected just prior to dawn, and again in late afternoon at the upstream and downstream limits of the reach. In 1982, samples were collected only at the downstream limit of the reach.

Chemistry samples were collected one foot below the water surface where possible, preserved according to Wisconsin State Lab of Hygiene (SLoH) and Standard Methods (APHA, et al. 1981) guidelines, iced and shipped within 24 hours to SLoH for analysis. Water chemistry parameters are presented in Table 1.

Table 1

Water Chemistry Analyses

Total Phosphorus (PTOT)
Ortho-phosphate (PO_4P)
Total Kjeldahl Nitrogen (TKN)
Nitrite-Nitrate Nitrogen (NO_2+NO_3N)
Ammonia Nitrogen (NH_3N)
Turbidity
Total Alkalinity*
Hardness*
pH*
Biochemical Oxygen Demand (BOD_5)
Chemical Oxygen Demand* (COD)
Total Non-Filtrable Suspended Solids
Total Volatile Non-Filtrable Suspended Solids

* 1981 analyses only.

Sediments

The purpose of sediment interstitial water and bulk sediment sampling in 1981 and spring 1982 was to characterize overall sediment nutrients within an entire reach, and to determine if differences in nutrient concentrations occurred between areas colonized and areas that were uncolonized by aquatic macrophytes. Bi-weekly stream mapping data were used to determine plant cover and associated bottom materials. Macrophyte and non-macrophyte sampling sites were chosen based on these occurrences. In 1982, surveys were also conducted on two streams that received wastewater treatment plant effluent. These surveys were designed to determine if nutrients were accumulating in the sediments and interstitial water at points downstream from the effluent outfalls.

Interstitial Water Analyses

Interstitial water (IW) nutrient concentrations within and outside of macrophyte beds were determined monthly in 1981 and once in the spring during 1982. In September 1982, the Bark River and White River were sampled upstream and at points downstream of the treatment plant outfalls. Sand-gravel substrates with visually similar composition were sampled at each location on these two rivers.

Samples were collected by vacuum from a 1/2" (1.25 cm) i.d., 9" (22.5 cm) long well-point in the substrate and filtered before contact with the atmosphere. Prior to collecting each sample, the well-point, suction lines, 0.45 um filter and collection flask were rinsed with 50-100 ml 10% HCl and twice with 50-100 ml distilled water. Sample blanks were collected at this time.

The well-point was inserted into the sediment deeply enough to avoid collecting the overlying stream water, yet within macrophyte rooting depth (ca. 6"). Slits for collection of the pore water were located in the terminal 2" (5 cm) of the well-point. Approximately 25-50 ml were collected by vacuum and discarded as rinse. Vacuum was reapplied to collect 50 ml of filtrate for the sample. Samples were iced and sent to SLoH for dissolved PO_4P (total dissolved P in 1982), NH_3N and NO_2+NO_3N determinations.

Bulk Sediment Analyses

Bulk sediment nutrient concentrations within and outside of macrophyte beds were also determined at monthly intervals in 1981. Cores collected in 1981 were from the predominant substrate in stream transects,

those from 1982 were collected from organic sediment deposits. Similar to the interstitial water sampling, sediment cores (composites) were collected up and downstream of treatment plant outfalls on the Bark River and White River in 1982.

To minimize variability between individual collections, sediment samples were collected from several different areas and composited from 2-4 cores taken from each transect. Two composite samples were collected, one representing bulk sediment nutrient content within macrophyte areas, the other nutrients from macrophyte-free areas. In the surveys conducted on the Bark River and White River in 1982, samples were composited from 5-10 cores collected at each site from areas of silt and organic sediment deposition.

Cores were 3.8 cm (1.5") in diameter and from 5-15 cm (2-6") in length, depending on substrate type. Grab samples were collected where substrate did not allow coring. Individual samples were mixed, sub-samples iced and sent to SLoH to be analyzed for TKN and PTOT.

Stream Mapping

Physical and biological stream characteristics were mapped every two weeks along established transects in each stream reach. In 1981, stream reach lengths were selected between 100-230 feet and transects were established at approximately 20 to 40 foot intervals. The reach lengths were expanded in 1982 to 800-1600 feet with 10-20 transects established per reach. One square-foot observation points were mapped at one to three foot intervals along each transect. These methods are a modification of a line-transect method (e.g. Kullberg 1974, Wong and Clark 1976, Wright et al. 1981). Depth, velocity, substrate type, percent macrophyte and/or periphyton coverage, and species abundance were recorded at each observation point.

Depth and velocity were measured using a wading rod and Marsh-McBirney^{R} Model 201 current meter. Stream discharge measurements were calculated from incremental depth and velocity measurements (Buchanon and Somers 1969). In 1982, velocity data were collected at one transect for discharge calculations.

Stream bottom type was recorded as percent of the various substrate size classes. Substrate was classified by visually estimating particle sizes using USGS (1978) guidelines (Table 2). A detritus class was added for incompletely decomposed organic material. Substrate data for each observation point were summarized by calculating a Substrate Index (SI). Weighting of the various size classes was conducted to provide a continuum of SI values. The SI was calculated from the following equation:

$$SI = \frac{(\%SILT) + 2(\%SAND) + 3(\%GRAV) + 4(\%RUBBLE) + 5(\%BOULDER)}{5}$$

Percent macrophyte coverage, periphyton coverage and type (filamentous or non-filamentous) were estimated. Macrophyte species abundance was given a rating corresponding to percent coverage (Table 3).

Macrophyte Harvesting

The objectives of the macrophyte harvesting surveys were to:

- Provide macrophyte biomass estimates at the time of diel oxygen surveys;
- Estimate macrophyte biomass accumulation throughout the growing season; and
- Determine maximum stable macrophyte biomass (summer standing crop) and when it occurred.

Macrophyte biomass samples were collected monthly, within 7 days following the diel surveys. Three to five sample quadrats were selected within 10 ft. up or downstream of each stream transect using random numbers tables; the first two digits specified the distance from the left streambank, the third the distance up (even number) or downstream (odd number) from the transect. Quadrats were not selected with-

in three feet of the transect to avoid sampling areas disturbed by the mapping activities.

Table 2

Substrate Size Classes
(based on USGS 1978 guidelines)

Class	Size mm (in.)
Boulder (Large Cobble)	256 (10")
Cobble (Rubble)	64 - 256 (2.5 - 10")
Gravel	2 - 64 (0.1 - 2.5")
Sand	.062 - 2 (.003 - .1")
Silt	.004 - .062
Detritus	-----

Table 3

Macrophyte Species Abundance Rating and
Corresponding Percent Coverage Estimate

Species Abundance Rating	Percent Coverage
5	80 - 100
4	60 - 80
3	40 - 60
2	20 - 40
1	1 - 20

In 1981, a record was kept of all previously harvested coordinates and samples were not collected within 2 ft. of a previously sampled location. After the first harvests in 1982, transects were moved slightly upstream or downstream of the original transects to avoid sampling previously harvested areas. Wright, et al. (1981) noted, however, that disturbance by mapping or harvesting did not result in changes in macrophytes or substrate.

Survey flags were used to locate quadrats in the stream. Quadrats were delineated using a one-square foot Surber sampler with the random number coordinate at its center. Percent macrophyte cover, species abundance ratings, water depth and velocity over the sampling point were measured before plant harvesting.

Macrophytes rooted within the Surber frame were harvested with roots, and vigorously washed with stream water. Plants from each quadrat were sorted into dominant and other species, bagged and transported to the lab on ice. Samples were briefly re-washed with tapwater in the lab, placed in dried pre-weighed paper bags, and dried to constant weight (24-48 hrs) at 65°C in a forced-air oven to determine sample dry weight.

Tissue nutrient analyses were conducted on dried macrophyte samples collected at each harvesting. Three to eight tissue samples were prepared, including one composite sample for each dominant species and/or three composite samples. Composite tissue samples were thoroughly mixed, bagged and sent to the University of Wisconsin Soil and Plant Analysis Lab, Madison, for analysis. Plant material was analyzed using inductively-coupled plasma spectroscopy for N, P, K, Ca, Mg, S, Zn, B, Mn, Fe, Cu, Al and Na.

Periphyton Harvesting

The primary objectives of the periphyton harvest were to characterize community growth at different nutrient concentrations and compare this with macrophyte growth characteristics and diel dissolved oxygen regimes.

Periphyton were collected every two weeks from glass-slide samplers (periphytometers) and bricks. In 1981, most reaches contained three periphytometers and six bricks. In 1982, as the reaches were lengthened, four periphytometers and eight bricks were harvested.

Bricks were scored into quarters for sample harvesting and placed on the stream bottom at periphytometer placement sites. Areas of each brick quadrant were measured for use in computing areal biomass estimates. After the first harvest in 1981, exposure periods for the bricks were effectively eight weeks. Each sampling date, one quarter of each brick was scraped into individual vials and iced for ash-free weight

determinations. Velocity over each brick was measured every two weeks, on placement and sampling dates.

In 1982, bricks were moved to locations in the stream to ensure minimal macrophyte shading. Exposure periods for bricks in 1982 were four weeks. Two brick quadrants were sampled for tissue nutrient analyses, and one quadrant each for chlorophyll and ash-free weight analyses. In the field, chlorophyll samples were washed directly into 50 ml centrifuge tubes with 90% acetone, packed on dry ice and shipped to SLoH for analysis. Tissue nutrient and ash-free weight samples were washed with distilled water into appropriate containers. Tissue nutrient samples were analyzed using inductively-coupled plasma spectroscopy at the UW Soils and Plant Analysis Lab, Madison. Ash-free weight was determined according to Standard Methods (APHA, et al. 1981). Water depth and velocity were measured at collection.

Periphytometers were exposed for two-week periods, the glass slides suspended 2.5 cm (1") below the water surface. At the time of placement and retrieval, velocity over the sampler was measured and sampler conditions noted. Slides were removed, drained of excess moisture, placed in foil-wrapped containers and frozen. Slides were selected for community composition, chlorophyll, ash-free dry weight and nutrient analyses.

A study comparing glass-slide periphyton parameters to those occurring on natural substrates was conducted in 1981. Periphytometers were treated similar to the routine placements. In addition, natural substrates (rocks) were sampled by placing a plastic cylinder over the substrate, and scraping the periphyton off using a stiff bristle brush and a razor blade. The collected material was handled in the same manner as the periphytometer and brick collections for chlorophyll and ash-free weight analyses.

Diel Studies

The primary objective of the diel studies was to investigate relationships between photosynthesis, respiration and plant biomass. Diel productivity studies were conducted monthly at each stream reach before macrophytes were harvested. Dissolved oxygen (DO) and temperature were measured at the upper and lower transects of each reach. In 1981, measurements were made at 2-3 hour intervals for 24-27 hrs. DO and temperature were continuously recorded in 1982. The DO and temperature data were collected using Yellow Springs Instrument^{R} (YSI) dissolved oxygen meters (Models 56, 57, 58). Meter calibration was checked against triplicate Winkler titrations at 3-6 hour intervals.

Light intensity data were collected using a Biospherical Instrument's^{R} Quantum Scalar Irradiance system, which measures Photosynthetically Active Radiation (PAR) in the 400-700 nm range. PAR data were collected at 2-3 hour intervals in 1981, and continuously recorded in 1982.

The light measurements represent instantaneous rates, and were measured in units of Quanta/sec $\cdot$ cm^{-2}. Light readings were taken submerged, approximately 15 cm from the stream bottom and at the surface (dry reference sensor). [Note: 6×10^{17} Q/sec $\cdot$ cm^{-2} = 1u Einstein/sec $\cdot$ cm^{-2}; 1 Watt/cm^2 = 4.6 uE/cm^2; 1 Klux = 18 uE/cm^2].

Water chemistry and suspended solids samples were collected twice during the diel period; just prior to sunrise and late afternoon.

SYNOPTIC SURVEY REACHES

The primary objective of conducting the synoptic surveys was to evaluate the relationship between mean summer in-stream nutrient concentrations and maximum stable summer macrophyte biomass. Twelve synoptic stream reaches were selected using the same criteria as were used for the intensive

monitoring sites, with macrophytes being the dominant primary producers.

Water chemistry collections were made at two week intervals for 6-8 weeks prior to macrophyte harvesting to characterize the mean summer in-stream nutrient concentrations. The synoptic reaches were sampled a minimum of three times with some reaches being sampled a fourth time. Collection methods and water chemistry parameters were the same as those used at the intensive study reaches in 1982. Samples were not analyzed for BOD at the Synoptic Sites.

Stream mapping and macrophyte harvests were conducted using the same methodologies as were used at the intensive monitoring sites. Periphyton harvests and diel surveys were not conducted at the synoptic monitoring sites.

Macrophyte harvests and stream mapping were conducted at the time judged to be maximum stable biomass. A number of investigators (e.g. Gerloff and Krombholz 1966, Caines 1965, Stake 1968, Ball et al. 1973) have recommended sample collection later in a growing season due to the stability of tissue nutrient concentrations. Prior to the actual harvest, all sites were visited frequently to judge the best harvesting time. The timing of the maximum macrophyte biomass was estimated from the data collected at the intensive monitoring sites in 1981 and field observations. This time period was found generally to be mid to late August. Macrophyte tissue sampling and analysis was identical to methods used for the intensive monitoring sites.

Photosynthesis/Respiration Studies

In 1982, recirculating light and dark chambers, and light bottle/dark bottle studies were conducted at stations on the Bark R. to determine in situ photosynthesis and respiration rates. Light, DO and temperature were continuously recorded for the duration of the experiments (usually 2-3 hours).

The recirculating plexiglass chambers had a volume of 80 liters, and used a pump capable of recirculating 90 liters/min. In the field, the boxes were placed over a macrophyte bed and sealed to the substrate using bentonite clay. The macrophytes were harvested at the end of the testing to determine the dry weight biomass. In all cases, macrophytes were dried and weighed in the same manner as the regular harvests.

Macrophytes were also placed in light and dark BOD bottles and incubated on the stream bottom. Incubation times were varied from 0.5 to 7.0 hours. As with the enclosure studies, light was continuously recorded. Initial and final DO concentrations in the bottles were determined by Winkler titrations. Macrophytes were removed from the bottles and dried at 65-70°C to constant weight.

Periphyton were filtered from a known volume of sample after the titration for dissolved oxygen was completed. The filters were dried at 100°C for 24 hours, then ashed at 500°C for 2 hours to provide dry weight and ash-free weights.

STUDY REACH CHARACTERISTICS

Nineteen stream reaches were selected to provide as wide a range of chemical conditions as possible, including reaches receiving wastewater treatment plant (POTW) effluent. To illustrate the range of parameter values represented in the data analyses, mean values were ranked, from lowest to highest. Occurrence of a particular study reach within the ranked hierarchy was used to describe or group streams with similar characteristics. This was done for chemical and physical parameters. For many parameters, divisions between groups was based on frequency distributions, in others, divisions were used which represented specific parameter ranges.

For the purpose of the following rankings, the 1981 data were treated as separate from the 1982 data, giving 22 data sets for com-

parison (7 sets in 1981 plus 15 sets in 1982). Data from the reaches represent mean growing season (June-August) values.

REACH CHEMICAL CHARACTERISTICS

Study reaches were ranked according to mean growing season (June-August) stream total phosphorus (TOTP) concentrations, inorganic nitrogen (NH_3 + NO_2 + NO_3N) concentrations and the N:P ratio. Although various in-stream levels have been suggested as limiting concentrations to primary producers, study reaches were grouped based on their distribution within the ranked hierarchy. Chemical characteristics of the study reaches are presented in Table 4.

Phosphorus

Study reaches, ranked by mean growing season in-stream total phosphorus concentrations, were separated into three groups representing "low" P (<0.05 mg/l TOTP), "medium" P (0.05-0.20 mg/l TOTP) and "high" P (0.20-0.50 mg/l TOTP) groups (Table 5). No streams receiving POTW effluent occurred in the low P group, which represented 30% of the study reaches. Fifty-six percent of the study reaches occurred in the medium P group, six of which (46% of this group)

Table 4

Summary of Stream Reach Chemical Characteristics, 1981-1982

Reach	POTW Impacted	Group* 1981	Group* 1982	Total P 1981	Total P 1982	Inor. N 1981	Inor. N 1982	Biomass 1981	Biomass 1982
Sugar	N	2	1	.122	.104	1.51	3.24	151.7	134.4
Ashippun-M	N	2	2	.139	.095	1.05	1.05	151.7	125.7
Ashippun-N	N	2	--	.137	---	1.08	---	99.0	---
Ashippun-S	N	2	--	.141	---	1.08	---	151.7	---
Kohlsville	N	2	--	.091	---	1.14	---	---	---
Bark-Wolf	N	2	2	.022	.021	.09	.11	45.2	25.6
Bark-Lurvey	Y	2	2	.244	.159	1.17	.74	249.6	289.0
Bark-Masonic	Y	--	2	---	.163	---	.72	---	187.5
SYNOPTIC SITES									
Bark-Wahl	N	--	3	---	.033	---	1.95	---	179.3
Mukwonago	N	--	3	---	.013	---	.03	---	57.8
Milwaukee									
- Campbellsport	Y	--	3	---	.493	---	.56	---	448.5
- East Br.	N	--	3	---	.040	---	.33	---	72.5
Suppernong	N	--	3	---	.027	---	.83	---	262.6
Pewaukee	Y**	--	3	---	.140	---	.48	---	213.8
Cedar	N	--	3	---	.050	---	.71	---	161.0
Mt. Vernon	N	--	3	---	.050	---	4.28	---	365.5
Black Earth	Y	--	3	---	.107	---	2.15	---	282.1
Fox-Portage	Y	--	3	---	.147	---	.62	---	146.9

*GROUP 1 = Intensive study site with primary producer harvest and mapping but without diel studies.
2 = Intensive study site with primary producer harvest and mapping, including diel studies.
3 = Synoptic survey, harvest and mapping.

**This site has received wastewater discharge from the City of Pewaukee up until the end of 1981.

were impacted by POTWs. [This group could be further sub-divided into medium-low and medium-high P ranges (0.05-0.12 and 0.12 - 0.20 mg/l TOTP respectively) with about 38% of the group occurring in the lower phosphorus group. This group roughly corresponds to most recommended levels of phosphorus necessary to control or slow eutrophication. All but one of the POTW-impacted reaches would occur in the medium-high range.] Two streams represented the high P group, both impacted by POTWs.

Nitrogen

The distribution of streams along the nitrogen gradient determined the group rank boundaries (Table 5). The groups represented "low" N (<1.00 mg/l inorganic nitrogen), "medium" N (1.00-1.9 mg/l inorganic nitrogen) and "high" N (>2.00 mg/l inorganic nitrogen). Almost 50% of the study reaches occurred in the low N group, 5 of which (45% of the group) were impacted by a POTW. Approximately 33% of the study reaches occurred in the medium N range, 2 of which (25% of the group) were impacted by POTWs. Four study reaches occurred in the "high N" range, only one of which received POTW effluent.

N:P Ratio

Grouping of reaches by N:P ratios followed somewhat the ranking of streams by nitrogen gradient (Table 5), those reaches with very high nitrogen having the highest N:P ratios. The N:P ratios ranged from approximately 1 to 86. Based on the ranked distribution of study reaches, reaches were grouped at N:P ratios of <5, 5-20, and >20:1.

Approximately 39% of the reaches had N:P ratios of less than 5:1, the majority of the group (67%) being impacted by POTWs. Forty-four percent of the reaches occurred in the middle ranking, two of which receive WWTP effluent, and 17% in the high N:P range, none of which receive an effluent.

REACH PHYSICAL CHARACTERISTICS

Stream physical characteristics may have a significant impact on plant growth. Substrate composition, water depth and velocity and other physical characteristics can modify plant response to nutrients, in addition to affecting diel DO changes. The study reaches represented a variety of morphometric conditions. Reach physical characteristics are presented in Table 6. For comparison, all streams were ranked by mean reach width, depth, cross-sectional area, predominant substrate and current velocity. For the most part, grouping of these parameters were determined by their frequency distributions rather than established criteria.

The data from 1981 reaches are not strictly comparable to their 1982 data. With the exception of Sugar Cr., all the 1981 reaches were lengthened in 1982.

Depth

Mean reach depths for all sites ranged from .15 m (0.50 ft) to .65 m (2.16 ft). Mean reach depths for those reaches continued from 1981 were generally higher than 1982 values. Of the 22 data points, approximately 26% were in the .15 -.2 m (0.50-0.75 ft) depth range, 61% in the 0.30-0.45 m (1.0-1.5 ft) depth range, and 13% in the 0.5-0.65m (1.65-2.15 ft) depth range (Table 6). Of those reaches in the shallow range, one is impacted by a POTW. Four of the middle range (or 28%) and two of the three in the deep range receive POTW effluent.

Width and Cross-sectional Area

Mean reach widths ranged from 3 to 20 m (10 - 67 ft) (Table 6). Approximately one-half (48%) of the reaches were 8 m (25 ft) or less across. The other half were 15 m (50 ft) or less, with only two reaches greater than 15 m across. Most of the POTW-impacted streams were in the 8-15 m (25-50 ft) range.

Ranking mean cross-sectional areas suggested almost equal division between the number of streams 2.3 square meters (25 square feet) or less and the number of streams 3-8 m^2 (35-85 ft^2) (Table 6). The range was .5-8 m^2 (5-85 ft^2). As with mean width, most of the POTW-impacted reaches were in the larger group.

Mean Velocity

The majority of streams (83%) were within a mean reach velocity range of 0.08-.20 m/sec (0.25-0.65 ft/sec) (Table 6). All but one of the POTW-impacted streams occurred in this group. The range of velocities was about .03-.3 m/sec (0.1-0.9 ft/sec).

Substrate

The majority of stream reach substrates were composed of gravel and sand. The two extremes, predominantly silt substrate and predominantly rubble substrate, were also represented in the study reaches.

Reaches were ranked by Substrate Index, and grouped by the SI values presented in Table 7. In this manner, approximately 30% of the reaches were represented by silt-sand substrates (SI = 20-40), 52% in the sand-gravel-rubble group (SI = 40-60) and 13% in the rubble-cobble group. Reaches receiving POTW effluent were present in each group. The majority, however, occurred in the sand-gravel-rubble group.

Table 5

Breakdown of Stream Reaches By Phosphorus and Nitrogen Characteristics

LOW P , LOW N	
Bark-Wolf - 81	(low N:P)
Bark Wolf - 82	" "
Mukwonago R.	" "
Cedar Cr.	(mid N:P)
Milwaukee R.-East Br.	" "
Scuppernong R.	(high N:P)
LOW P , HIGH N	
Bark-Wahlschlaeger	(high N:P)
Mount Vernon Cr.	" "
MID P , LOW N	
Bark Lurvey - 82	(low N:P)
Bark Masonic	" "
Fox at Portage	" "
Pewaukee R.	" "
MID P , MID N	
Ashippun Main - 81	(mid N:P)
Ashippun Main - 82	" "
Ashippun North	" "
Ashippun South	" "
Sugar Cr. - 81	" "
Kohlsville R.	" "
MID P , HIGH N	
Black Earth Cr.	(low N:P)
Sugar Cr. - 82	(high N:P)
HIGH P , LOW N	
Milwaukee Campbellsport	(low N:P)
HIGH P , MID N	
Bark Lurvey - 81	(low N:P)

Table 6

Summary of Stream Reach Physical Characteristics, 1981-1982

Reach	Length (m) 1981	Length (m) 1982	Width (m) 1981	Width (m) 1982	Depth (m) 1981	Depth (m) 1982	Velocity (m/sec) 1981	Velocity (m/sec) 1982	SI* 1981	SI* 1982
Sugar	60	60	7	7	.22	.35	.18	.24	53.6	52.2
Ashippun-M	40	235	5	6	.34	.39	.12	.27	34.0	37.0
Ashippun-N	26	—	4	—	.19	—	.16	—	48.2	—
Ashippun-S	32	—	3	—	.18	—	.19	—	45.0	—
Kohlsville	56	—	2	—	.19	—	.18	—	67.2	—
Bark-Wolf	68	303	9	11	.29	.49	.15	.16	44.4	46.8
Bark-Lurvey	67	480	12	12	.36	.65	.12	.11	45.2	50.0
Bark-Masonic	—	242	—	13	—	.45	—	.14	—	47.4
SYNOPTIC SITES										
Bark-Wahl	—	74	—	9	—	.36	—	.14	—	37.0
Mukwonago	—	300	—	16	—	.39	—	.22	—	43.0
Milwaukee - Campbellsport	—	211	—	9	—	.34	—	.08	—	84.2
- East Br.	—	91	—	12	—	.32	—	.08	—	30.4
Suppernong	—	91	—	5	—	.32	—	.12	—	36.4
Pewaukee	—	91	—	6	—	.15	—	.04	—	54.4
Cedar	—	86	—	5	—	.16	—	.20	—	64.2
Mt. Vernon	—	120	—	6	—	.35	—	.16	—	37.0
Black Earth	—	112	—	11	—	.43	—	.15	—	24.6
Fox-Portage	—	136	—	17	—	.44	—	.09	—	37.4

*Method of calculation in text.

Table 7

Substrate Index Substrate Type and Value Ranges

Substrate Class	SI Range
Silt	20 - 29
Sand-Silt	30 - 39
Sand-Gravel	40 - 49
Rubble	50 - 69
Boulder	90 - 100

STREAM PRIMARY PRODUCERS

MACROPHYTES

Introduction

The primary objective of the macrophyte surveys was to determine if a significant relationship exists between in-stream nutrient concentrations and late summer biomass (summer standing crop). Although several authors have suggested minimum nutrient concentrations which will stimulate maximum macrophyte growth (e.g. Gerloff 1969; Mulligan and Baranowski 1969; Pitcairns and Hawks 1973) little has been done to develop a usable predictive relationship defining growth using in-stream nutrient concentrations. These relationships have been quantified for lake phytoplankton (Jones and Bachmann 1976; Dillon and Rigler 1974; Hoyer and Jones 1983), but not for macrophytes.

The Interim Technical Report of the Phosphorus Assessment Study (Mace, et al. 1982) reported that a significant relationship did exist between late summer biomass and mean summer phosphorus concentrations at seven study reaches in four southeastern Wisconsin streams. The 1981 samplings, however, involved too few data points to develop a substantiated predictive model. Sampling in 1982 included the 1981 sites and 11 additional stream reaches representing a wider range of in-stream nutrient concentrations.

Results and Discussion

While correlation coefficients are indicators of relationships, they are not necessarily indicators of cause and effect relationships. Least squares regression equations were calculated for paired parameters having significant correlation coefficients.

The strength of the regression models was assessed using R-square values, mean residual error and confidence limits (p=.05) expressed as percent of the predicted values. R-square is the proportion of the total variance in the dependent variable that may be attributed to the regression on the independent variable. Mean residual error is the absolute difference between the observed and predicted values as a percentage of the predicted values. The confidence intervals (p=.05) for the predicted values are given as the difference between the predicted value and the confidence limit (p=.05) value expressed as a percentage of the predicted value.

These parameters describe how well a regression model fits a particular data set. These parameters do not test a model to determine how well it will work as a management tool. An independent data set is used to test empirical models and verify their predictive capabilities.

Stream Type Determination

The stream reach mapping data suggested that two basic types of stream reaches were surveyed. Criteria related to macrophyte distribution and dominant substrate type were used to classify the stream reaches as Type I or Type II. Stream reaches were classified as Type I if macrophyte populations were relatively homogenously distributed and secondarily if the reach had substrate dominated by sand, gravel or rubble. Stream reaches were classified as Type II if macrophyte distribution was patchy and secondarily was limited to areas of silt and silt deposition. Based on these criteria, Type I streams included all stream reaches except Mount Vernon Creek, Black Earth Creek and Scuppernong River which were classified as Type II.

The mapping data were evaluated using frequency analysis to assess edaphic impacts on macrophyte occurrence. The frequency analysis compared the dominant bottom substrate of each stream reach Type (I or II) and the substrate size over which macrophytes were growing. Larger size substrate classes ($SI > 40$) were dominant in Type I stream reaches. Smaller size substrate classes ($SI \leq 40$) were dominant in Type II stream reaches (Table 8).

Table 8

Summary of Stream Reach Mapping Data

Stream Reach Type	Number of Reaches	[1]Percent Occurrence On SI>40	Percent Occurrence On SI<40	[2]Percent Substrate SI>40	Percent Substrate SI<40
I	16	77	23	73	27
II	3	28	72	31	69

[1]Values calculated from observation points where macrophytes occurred.
[2]Values calculated from all observation points.

Macrophytes occurred at approximately the same percentage of sample points (81%-83%) in both Type I and Type II stream reaches (Figure 1). The important difference between the stream Types, however, was that in Type I stream reaches macrophytes occurred on sand, gravel and larger substrates (SI> 40) and in Type II stream reaches macrophytes were found to occur on silt dominated substrates (SI< 40).

All stream reaches were selected using identical criteria, designed to standardize or minimize the physical impacts exerted on macrophyte growth. By limiting the physical impacts on macrophyte growth the amount of available nutrients would be the dominant factor controlling macrophyte growth.

The data presented in Figure 2 indicate that both Type I and Type II streams are capable of producing a high macrophyte biomass but that Type II streams can produce high macrophyte biomass at relatively low in-stream phosphorus concentrations.

It has been established that macrophytes can absorb nutrients from either the ambient water or from the sediment through roots (Carignan and Kalff 1980; McRoy and Barsdate 1970; McRoy, et al. 1972; Walsel and Shapira 1971).

A predictive empirical relationship was developed by Carignan (1982) which indicates the probable source of macrophyte nutrients. Carignan's (1982) model estimates the percentage of plant tissue phosphorus taken up by the roots. This model suggests that macrophytes obtain the nutrients they require from the most plentiful and readily available source.

Figure 1

Frequency of occurrence of macrophytes on substrate sizes in Type I and Type II Stream Reaches

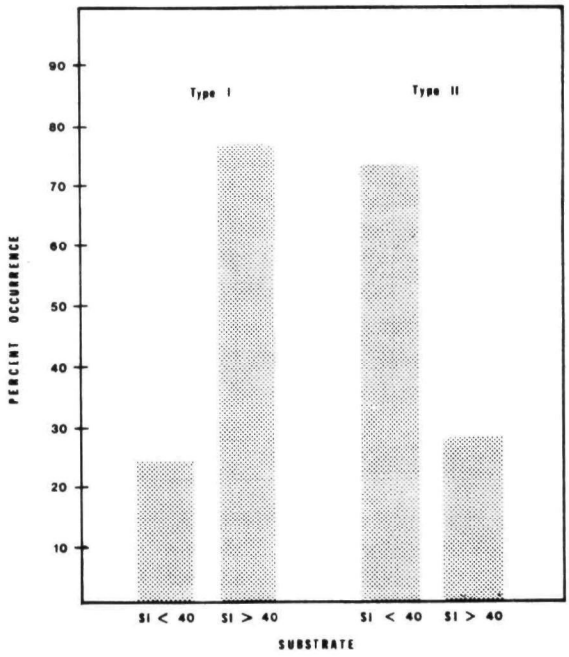
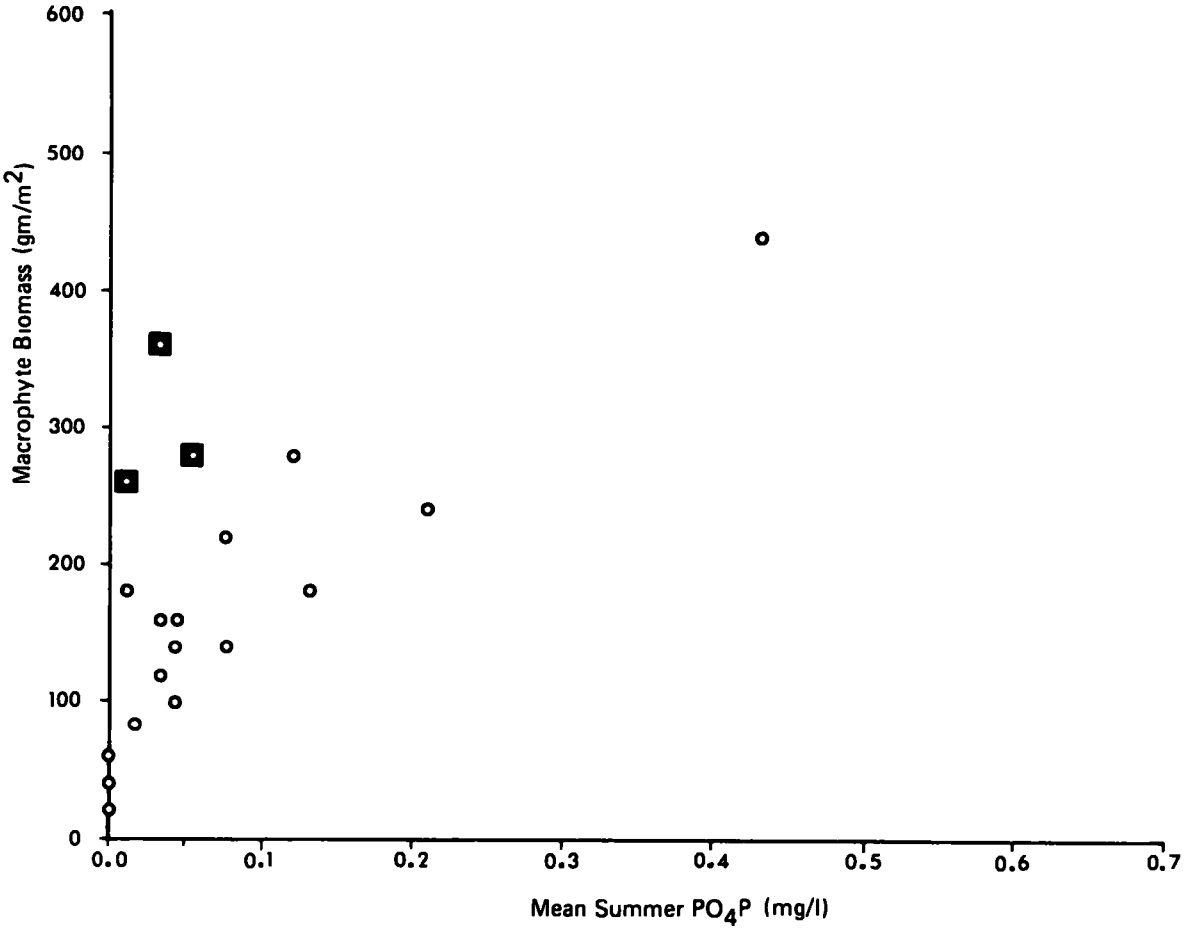


Figure 2

Plot of late summer macrophyte stable biomass (gm/m^2) against the mean summer phosphate-phosphorus conentration (PO_4 in mg/l). Open circles are Type I stream data points and boxes are Type II stream data points



Vaux (1962, 1968) studied the interchange of intragravel (interstitial) water with the overlying water in streams. He found this interchange is affected by stream bed permeability (substrate size), the depth of the material, the configuration of the stream bed surface and channel slope.

If the permeability changes (e.g. decreases) down-welling of surface water will occur immediately downstream of a low-permeability area. This occurs on small (point measurements) as well as larger scales. Examples of localized exchange were illustrated by the interchange resulting from placement of rocks on the streambed (Vaux 1968).

Stream channels, especially those with larger bottom substrate sizes, usually contain alternating channel slopes due to geomorphic factors, irregular substrate size and distribution, shifting of substrates and animal activities. This would result in rapid and extensive exchange of intragravel and overlying surface water throughout a stream reach in larger substrate areas. The larger substrates in Type I streams would then permit intragravel flow as well as rapid interchange between overlying and interstitial waters.

Based on the premise that macrophyte growth is a function of nutrient availability, it appears, then, that in those streams clas-

sified as Type I, macrophytes are essentially offered overlying stream water as the primary nutrient source, either through shoot absorption or due to intimate root contact with percolating overlying water. These data also suggest that macrophytes in Type II reaches are utilizing an alternate nutrient source. It seems probable then, that sediments can fulfill a significant portion of macrophyte phosphorus nutrition in these streams which produce a large biomass at low in-stream phosphate phosphorus concentrations.

Evaluation of the stream reach mapping data and the pore water/ambient water relationship indicated that two distinct Types of streams were surveyed. Type I stream reaches were found to have substrate dominated by larger particle sizes (SI > 40), macrophytes were relatively homogenously distributed and the ambient water is the probable primary nutrient source being utilized by the macrophytes. Type II stream reaches had small size dominated substrate (SI < 40), macrophyte occurrence was often limited to zones of silt or silt deposition and the sediments are the probable primary nutrient source.

Sediment Nutrients

The 1981 sediment data were analyzed to determine if significant relationships existed between sediment nutrients and stream macrophyte biomass. Multiple correlation analyses including pore water and bulk sediment phosphorus and nitrogen concentrations, macrophyte biomass, macrophyte percent coverage, macrophyte tissue nitrogen and phosphorus, and macrophyte tissue nitrogen to phosphorus ratio were conducted.

The only sediment parameters to correlate significantly with macrophyte biomass measurements were interstitial phosphorus in non-macrophyte areas with macrophyte biomass per square meter ($r = .638$, $p = .002$) and mean reach percent coverage ($r = .836$, $p = .0001$). In-stream PO4P concentrations also correlated with macrophyte area and non-macrophyte area interstitial phosphorus concentrations ($r = .798$, $p = .0001$; and $r = .859$, $p = .0001$ respectively). Reviewing the discussion of sediment interstitial water and overlying water exchange (Vaux 1962, 1968) it is probable that this relationship is responsible for the correlation between in-stream phosphorus and interstitial water phosphorus concentrations.

In general, sediment nitrogen did not correlate well with either in-stream nitrogen or plant biomass parameters. Exceptions were the correlations between stream inorganic nitrogen with macrophyte and non-macrophyte area interstitial NO_2-NO_3N concentrations ($r = .959$, $p = .0001$; and $r = .675$, $p = .0009$ respectively). The relationship between stream water and interstitial water nutrients again is a probable cause for these correlations.

Sediment nutrient data were also evaluated to determine if there were significant differences between sediment nutrient concentrations in macrophyte-populated areas and non-macrophyte areas. The purpose of this analysis was to determine if macrophytes were colonizing areas that had higher concentrations of sediment phosphorus and nitrogen or if the macrophytes had an obvious impact on pore water nutrient concentrations. If macrophytes were found to colonize areas of higher nutrient concentrations, this would have provided supportive evidence that the sediments may have been the dominant macrophyte nutritional source.

Growing season (June-September) mean sediment nitrogen and phosphorus concentrations were computed for each reach from monthly samples collected in and out of macrophyte areas in 1981. Mean sediment phosphorus and nitrogen concentrations within macrophyte (MSEDP and MSEDN) and out of macrophyte areas (NSEDP and NSEDN) were compared using a t-test (significance level .05). T-test values, sediment nitrogen and phosphorus concentrations in and out of macrophytes are given in Table 9. No significant differences in nutrient concentrations within and outside of macrophyte beds within each stream were indicated by the analyses.

Mean sediment interstitial water (IW) nutrient concentrations within and outside of macrophyte beds are given in Table 10. Correlations between interstitial water and bulk sediment parameters also showed no clear relationships between these two sediment measurements.

The results of the sediment nutrient analysis provided no clear relationships between bulk sediment nutrients and stream macrophytes. This suggests that in the reaches studied macrophyte nutritional needs are satisfied either directly through shoot absorption or indirectly through water percolating through the substrate and into the root system.

Macrophyte Biomass and In-stream Nutrients

Based on the concepts of agriculture and horticulture, that plant growth is proportional to the amount of nutrients available for growth, it would be expected that macrophyte growth could be modeled most accurately for stream reaches where the primary nutrient source has been quantified. That is, the relationship between macrophyte biomass and available nutrients can be defined best for Type I streams where the amount of available nutrients has been quantified (i.e. ambient water), and not in Type II streams where the primary nutrient source was not quantified (i.e. sediments).

Table 9

Bulk Sediment Nutrient Concentrations. All Concentrations are Annual Means, Expressed in mg/kg Sediment Dry Weight

Stream	*MSEDP	*NSEDP	†P(0.5)	*MSEDN	*NSEDN	†N(0.5)
Ashippun-Mainstem	565	487	0.58	3725	2225	1.63
Ashippun-North Branch	507	325	1.44	2825	1127	1.31
Ashippun-South Branch	452	332	1.11	2200	1330	1.11
Bark-Lurvey	110	133	-0.71	356	246	0.72
Bark-Wolf	185	157	0.64	1280	925	0.51
Sugar	572	374	1.17	2610	1556	0.89

* MSEDP and MSEDN represent phosphorus and nitrogen concentrations within macrophyte areas.
NSEDP and NSEDN represent phosphorus and nitrogen concentrations outside of macrophyte areas.

Table 10

Sediment Interstitial Water Nutrient Concentrations. All Concentrations are Expressed in mg/l

Stream	MIWP	NIWP	MNH3	NNH3	MNO3	NNO3
Ashippun-Mainstem	.003	.019	.500	.250	.530	.760
Ashippun-North Branch	.049	.024	.220	.090	.760	.900
Ashippun-South Branch	.012	.180	.107	.065	.850	1.040
Bark-Lurvey	.950	.250	.075	.075	.380	1.020
Bark-Wolf	.368	.027	.017	.017	.013	.660
Sugar	.026	.032	.176	.176	1.410	1.210

A preliminary correlation analysis including all streams (data in Tables 11 and 12) indicated that significant relationships existed between late summer macrophyte biomass and mean summer (June-August) in-stream nutrient concentrations. Total phosphorus (TOTP) and phosphate-phosphorus (PO4P) concentrations had correlation coefficients of .642 (p=.003) and .686 (p=.001), respectively, with macrophyte biomass (SQMBIO). The natural log of SQMBIO correlated significantly with the natural logs of TOTP (r=.633, p=.004) and PO_4P (r=.783, p=.0001). Total kjeldahl nitrogen (TKN) and inorganic-nitrogen (INORN) were insignificantly correlated with SQMBIO, (r=.219, p=.367 and r=.403, p=.088, respectively). The logarithmic transformation of INORN and SQMBIO, however, did correlate significantly (r=.689, p=.001).

The macrophyte biomass/in-stream nutrient concentration relationship improved when only the Type I streams were included in the correlation analysis. This analysis was conducted under the premise that if the two Types of streams (Type I and Type II) exist, the relationships between plant biomass and in-stream nutrients will improve when the streams were categorized. This analysis is also used as supportive evidence for classifying the streams as Type I or Type II. With Type II stream reaches deleted (Mount Vernon Creek, Black Earth Creek and Scuppernong River), the correlation between SQMBIO, TOTP and PO4P increased to .889 (p=.0001) and .901 (p=.0001), respectively. The highest correlation was found between the logarithmic transformations of SQMBIO and PO4P (r=.907, p=.0001). Although the correlation between SQMBIO and total kjeldahl nitrogen improved (r=.624, p=.01), the relationship between SQMBIO and INORN did not.

The correlation analysis indicated that significant relationships existed between in-stream phosphorus concentrations and late summer biomass in Type I streams. Predictive equations were developed from these relationships, by regressing TOTP, PO4P and their logarithmic transformations against macrophyte biomass. The most statistically significant least squares regression model (R-square=.823) was developed by regressing the natural log of the maximum summer biomass on the natural log of the mean summer PO4P concentration (Figure 3). The equation describing this relationship is:

Model I

$$SQMBIO = 546.8\ (PO4P)^{.415}$$

where: SQMBIO = Late summer biomass (grams per square meter)

PO4P = Mean summer (June-August) phosphate-phosphorus (milligrams per liter)

The equation was developed from the data in Tables 11 and 12 with a PO4P concentration range of .002 to .430 milligrams per liter and macrophyte biomass from 25.6 to 448.5 grams per square meter. The mean residual error for this regression is 24.7 percent and ranged from 1.8 to 87.0 percent of the predicted values. The ninety-five percent confidence limits for the predicted values ranged from 53 to 114 percent of the predicted values. This equation appears to be a good predictive tool for the assessment of macrophyte communities in stream reaches where macrophytes derive phosphorus from the water.

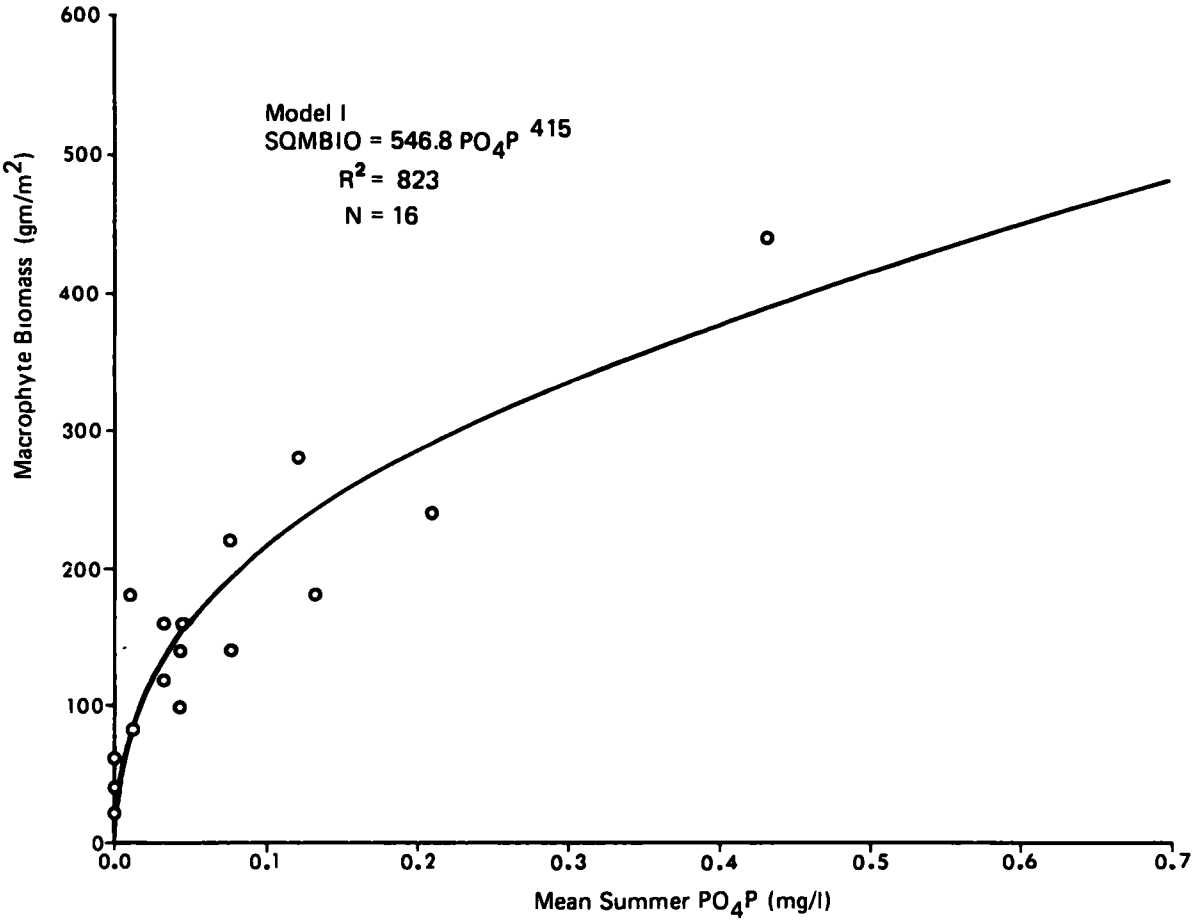
Macrophyte Tissue And In-Stream Nutrients

It has been shown that macrophyte growth is dependent upon tissue phosphorus concentrations (Gerloff and Krombholtz 1966; Wilson 1972). If the relationship between macrophyte tissue nutrients and in-stream nutrient concentrations can be quantified, it will provide supportive evidence that macrophyte growth may be limited by controlling in-stream nutrient concentrations. This relationship would only be quantifiable for streams where the amount and source of available nutrients (i.e. Type I streams) has been determined.

Macrophyte tissue phosphorus (PHOS) and nitrogen (N) concentrations were highly correlated with in-stream phosphorus and

Figure 3

Regression line of late summer biomass (gm/m^2) on mean summer phosphate-phosphorus concentration (PO_4 in mg/l) for Type I streams



nitrogen concentrations in Type I streams. The highest correlations occurred between the logarithmic transformations of tissue phosphorus and PO4P (r=.959, p=.0001) and TOTP (r=.930, p=.0001). The correlation of the natural logarithmic transformations of tissue-nitrogen with inorganic nitrogen was .826 (p=.0001) and tissue nitrogen with total kjeldahl nitrogen was .397 (p=.17).

The relationships between macrophyte tissue nutrients and in-stream nutrients led to the development of a predictive equation. This model resulted from the least squares regression of the natural log of the macrophyte tissue phosphorus concentration on the natural log of the mean summer phosphate-phosphorus concentration in Type I stream reaches (Tables 11 and 12). The equation describing the relationship is:

Model II

$$PHOS = 9.469\ (PO4P)^{.310}$$

where: PHOS = Macrophyte tissue phosphorus concentration (grams per kilogram)

PO4P = Instream phosphate-phosphorus concentration (milligrams per liter)

Table 11

Mean Summer (June-August) Water Chemistry Parameter Values. Concentrations are Presented in Milligrams per Liter. Given for Each Parameter are the Mean Value (Mean), Standard Deviation (Std. Dev.), and Number of Samples (N).

Stream	Year	Stream Type	Total Phosphorus mg/l			Phosphate Phosphorus mg/l			Total Kjeldahl Nitrogen mg/l			Inorganic Nitrogen mg/l		
			Mean	Std.Dev.	N	Mean	Std.Dev.	N	Mean	Std.Dev.	N	Mean	Std.Dev.	N
Ashippun River	82	I	.095	.026	7	.032	.011	7	1.03	.21	7	1.05	.26	7
Bark River-Lurvey	82	I	.159	.053	7	.125	.055	7	.79	.02	7	.74	.27	7
Bark River-Masonic	82	I	.163	.061	5	.128	.064	5	.78	.11	5	.72	.34	5
Bark R.-Wallschlaeger	82	I	.033	.012	3	.016	.006	3	.60	.20	3	1.95	.26	3
Bark River-Wolf	82	I	.021	.004	7	.003	.002	3	.74	.10	3	.11	.05	3
Cedar Creek	82	I	.050	.017	3	.029	.012	2	.50	.10	3	.71	.11	3
Fox River-Portage	82	I	.147	.025	3	.078	.018	3	1.10	.17	3	.62	.21	3
Milw. R.-Campbellsport	82	I	.493	.186	3	.430	.174	3	1.67	.31	3	.56	.24	3
Milw. R.-East Branch	82	I	.040	.020	3	.015	.007	3	.80	.20	3	.33	.07	3
Mukwonago River	82	I	.013	.006	3	.002	.000	3	.57	.06	3	.03	.01	3
Pewaukee River	82	I	.140	.069	4	.074	.048	4	1.05	.30	4	.48	.19	4
Sugar Creek	82	I	.104	.026	7	.041	.021	7	.81	.26	7	3.24	.80	7
Ashippun River	81	I	.139	.085	21	.050	.031	21	1.15	.34	21	1.07	.28	21
Bark River-Lurvey	81	I	.244	.051	7	.214	.051	7	.83	.11	7	1.18	.38	7
Bark River-Wolf	81	I	.022	.004	7	.002	.000	7	.69	.14	7	.09	.04	7
Sugar Creek	81	I	.122	.024	6	.043	.008	6	.78	.09	6	1.51	.13	6
Black Earth Creek	82	II	.107	.025	3	.052	.011	3	.43	.15	3	2.15	.12	3
Mount Vernon Creek	82	II	.050	.010	3	.037	.008	3	.20	.00	3	4.28	.12	3
Scuppernong River	82	II	.027	.012	3	.009	.004	3	.83	.45	3	.83	.05	3

Table 12

Summer Macrophyte Biomass (Grams Per Square Meter) and Macrophyte Tissue Nutrient Concentrations (Grams per Kilogram). Given are the Mean Value (Mean), Standard Deviation (Std. Dev.), and Number of Samples (N).

Stream	Year	Stream Type	Maximum Biomass gm/m^2			Tissue Phosphorus gm/Kg			Tissue Nitrogen gm/Kg		
			Mean	Std.Dev.	N	Mean	Std.Dev.	N	Mean	Std.Dev.	N
Ashippun River	82	I	125.7	147.7	44	3.37	.18	5	29.22	2.69	5
Bark River-Lurvey	82	I	289.0	240.9	67	4.83	.45	7	31.66	3.04	7
Bark River-Masonic	82	I	187.5	108.3	36	4.63	.87	5	32.08	1.66	5
Bark River-Wallschlaeger	82	I	179.3	217.3	36	2.75	.41	5	27.58	2.73	5
Bark River-Wolf	82	I	25.6	41.7	44	1.42	.20	3	21.37	3.01	3
Cedar Creek	82	I	161.0	85.3	44	2.40	.22	4	24.57	2.77	4
Fox River-Portage	82	I	146.9	121.0	50	4.47	.41	6	26.88	1.65	6
Milwaukee River-Campbellsport	82	I	448.5	508.9	50	6.68	.25	3	30.93	2.03	3
Milwaukee River-East Branch	82	I	72.5	60.0	48	3.73	.92	5	26.12	2.79	5
Mukwonago River	82	I	57.8	58.3	55	1.13	.52	3	15.67	3.93	3
Pewaukee River	82	I	213.8	317.2	40	4.01	.31	4	23.80	1.78	4
Sugar Creek	82	I	134.4	130.6	24	4.10	.70	3	29.83	.97	3
Ashippun River	81	I	98.9	74.5	26	3.81	.41	6	28.47	2.48	6
Bark River-Lurvey	81	I	249.8	253.1	34	5.41	.96	7	27.03	1.85	7
Bark River-Wolf	81	I	45.9	86.5	30	1.38	.25	4	20.05	3.06	4
Sugar Creek	81	I	151.4	132.4	24	3.60	.24	4	25.92	1.03	4
Black Earth Creek	82	II	282.1	334.5	46	5.81	.96	6	26.17	2.75	6
Mount Vernon Creek	82	II	365.5	503.8	44	5.48	.33	4	34.12	.64	4
Scuppernong River	82	II	262.6	252.8	48	2.44	.58	5	22.14	.58	5

This equation (Figure 4) has an R-square value of .921 and a mean residual error of 9.5 percent of the predicted value. The ninety-five percent confidence intervals for the predicted values range from 30 to 43 percent of the predicted values.

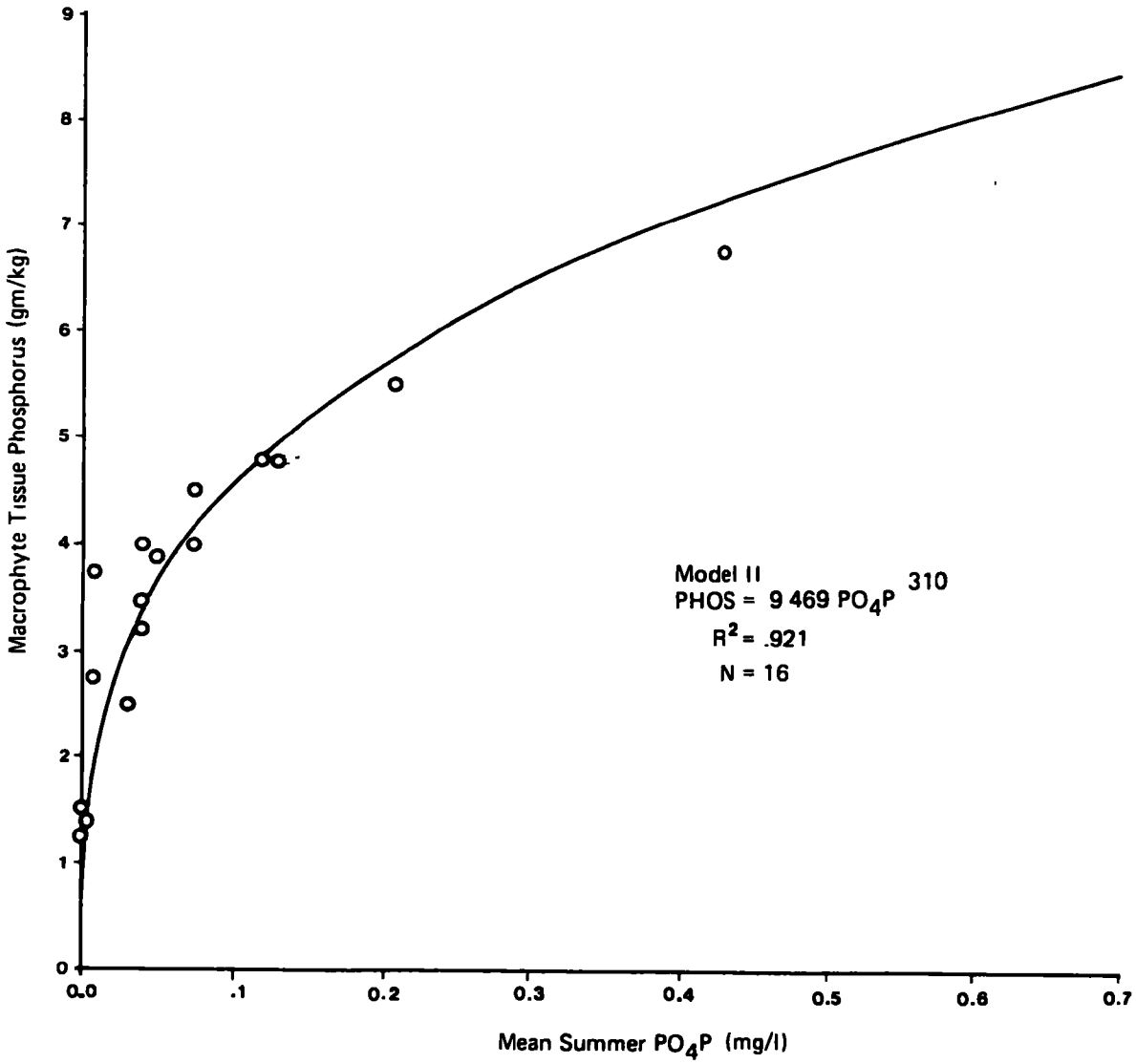
Model II indicates that macrophyte tissue phosphorus is a function of the in-stream phosphate-phosphorus concentration in Type I streams. Based on the concept that macrophyte growth is dependent upon macrophyte tissue nutrient concentrations, this model provides supportive evidence that in Type I streams macrophyte growth is a function of in-stream phosphate-phosphorus concentrations.

Macrophyte Biomass and Tissue Nutrients

Establishing an empirical relationship between late summer macrophyte biomass and

Figure 4

Regression line of the macrophyte tissue phosphorus concentration (gm/kg) on the mean summer phosphate-phosphorus concentration (PO_4 in mg/l) for Type I streams



macrophyte tissue nutrient concentrations could provide a methodology to estimate macrophyte biomass independent of nutrient source. This type of model would be of significant utility in streams where it would be difficult to determine the sources and amounts of nutrients available for macrophyte growth.

Macrophyte tissue phosphorus and tissue nitrogen correlated significantly with macrophyte biomass. The correlation analysis evaluating the relationships between macrophyte tissue phosphorus, macrophyte tissue nitrogen, and plant biomass included the data from all streams surveyed (Table 12). The correlation coefficients are .798 (p=.0001) and .634 (p=.004) for the logarithmic transformations of SQMBIO:PHOS and SQMBIO:N relationships, respectively.

Based on these correlations, an equation was developed to describe the relationship between macrophyte tissue phosphorus concentration at late summer biomass and summer maximum biomass. All the data points in Table 12 were included in this least squares regression analysis. The equation describing this relationship is (Figure 5):

Model III

$$SQMBIO = 36.06\ (PHOS)^{1.161}$$

where: SQMBIO = Late summer macrophyte biomass (grams per square meter)

PHOS = Macrophyte tissue phosphorus concentration (grams per kilogram dry weight)

The R-square for the model is .637 and the mean residual error is 36.4 percent of the predicted values. The mean ninety-five percent intervals for the predicted values were 63.7 to 175.9 percent of the predicted values. This model has the widest confidence limits of the three equations developed by these analyses. A probable cause for the wide confidence limits for this model may be that macrophyte tissue phosphorus concentrations rapidly decrease in senescing macrophytes. It has been determined that from 20 to 50 percent of the tissue phosphorus can be rapidly lost from decaying macrophytes, and 65 to 85 percent may be lost over longer periods (Nichols and Keeney 1973; Solski 1962, in Wetzel). It was noted in sampling that a few of the macrophyte populations were in the process of senescence at the time of harvest. This model may have improved substantially if all harvesting had been conducted before any macrophyte populations began to deteriorate.

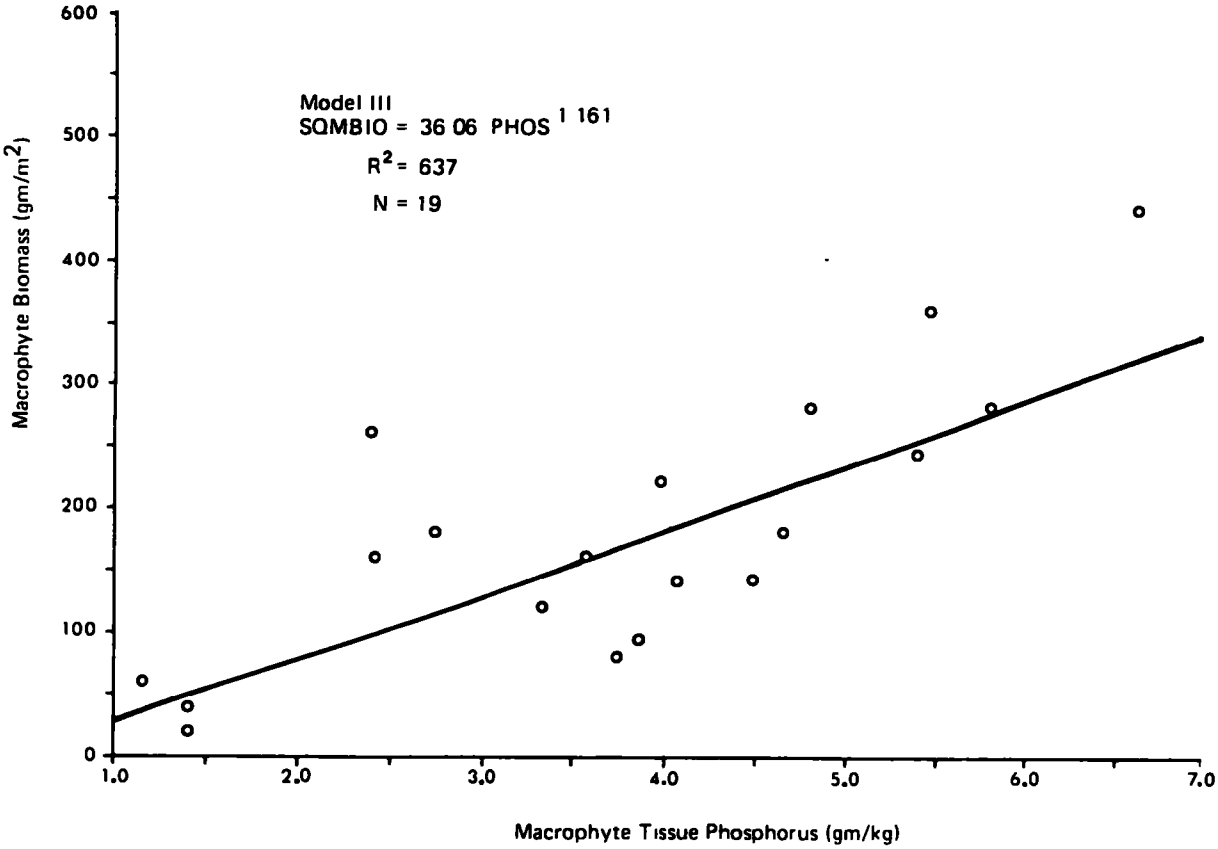
A number of investigators (e.g. Gerloff 1975, Gerloff & Krombholz 1966) have suggested tissue nutrient concentrations at which a particular nutrient becomes limiting to growth. Gerloff (1975) has suggested approximately .1% tissue phosphorus as limiting. Schmidt and Adams (1981) have reported P limitation at about .3%. The tissue nutrient concentrations reported here are somewhat above the .1 level, even though significant relationships are described between water P, tissue P, and plant biomass. This may be due to differences in the tissue nutrients of different parts of the plant. Gerloff's (1975) values are taken from apical meristem tissue whereas this study used whole plants (including roots) for analysis.

Model Selection

The primary objective of this portion of the study was to develop an empirical relationship predicting macrophyte biomass (summer standing crop) in small streams. This objective was accomplished with the development of Models I and III. The different variables used to derive these models make their applicability dependent upon stream Type (Type I or Type II). Model I was developed from the relationship between in-stream phosphate-phosphorus and late summer biomass. Model III is derived from the relationship between macrophyte tissue phosphorus and plant biomass, and is independent of nutrient source (i.e. water or sediment). Model I should be used whenever it is applicable as it is a much more

Figure 5

Regression line of summer macrophyte biomass (gm/m^2) on the macrophyte tissue phosphorus concentration (gm/kg) for Type I and Type II streams



accurate model than is Model III. It is obvious that Model I will most accurately predict macrophyte growth in Type I streams (Figure 3) and Model III (Figure 5) is the best available method to evaluate biomass in Type II streams.

For Model I or Model III to become accepted stream management tools, a methodology must be developed which determines the primary nutrient source for stream macrophytes. The best methodology for determining primary nutrient source would be to use the model developed by Carignan (1982). This method though has not been evaluated for streams and would require verification before it could be widely used.

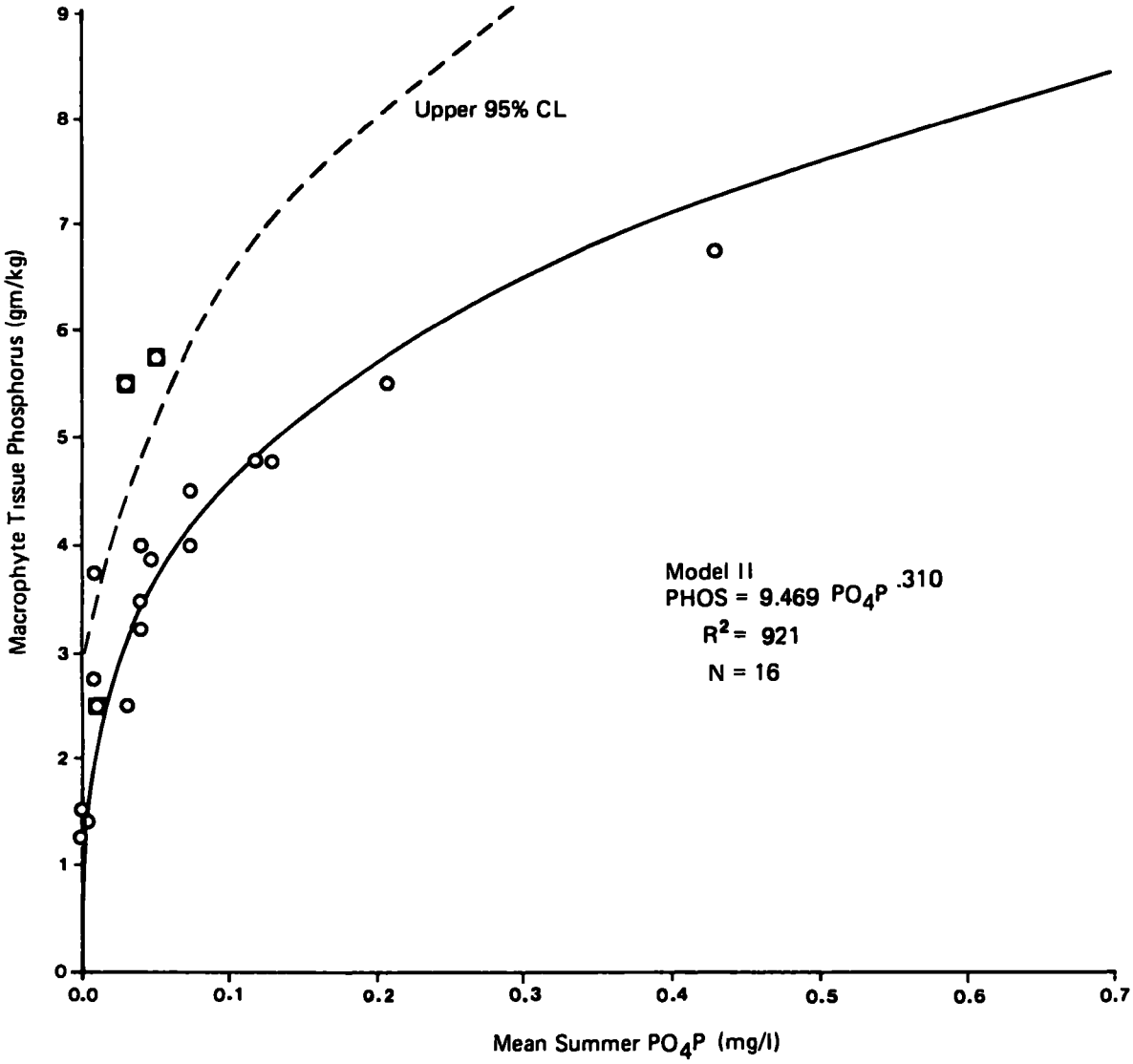
Field observations describing the distribution of macrophytes could also be used to indicate the most probable macrophyte nutrient source. Macrophyte populations with a relatively homogeneous distribution in a stream reach having a high percentage of large size bottom substrates (SI>40) are believed to be indicative of Type I streams. Streams having macrophyte populations limited to zones of silt deposition are characteristic of Type II streams.

Tissue analysis is based on the assumption that nutrient concentrations in an organism are representative of the amount of nutrients available to the organism for growth. The theory of tissue analysis has been discussed by several authors (Lundegardh 1951; Ulrich 1952; Bould, et al. 1960; Smith 1962; Chapman 1966). Based on these assumptions, the results of this study may provide a suitable alternative method for

determining the primary macrophyte nutrient source. Model II indicates that macrophyte tissue phosphorus concentration (PHOS) is dependent upon the in-stream mean summer phosphate-phosphorus concentration (PO4P) in Type I streams. This relationship suggests that Model II may be used to identify the conditions where the ambient water is the primary nutrient source. Figure 6 indicates that in Type II streams, macrophytes can have high tissue phosphorus at low stream PO4P, which suggests they are obtaining tissue phosphorus from the sediments. If macrophytes are using ambient water as a primary nutrient source they should belong to the relationship described by Model II. Stream data points lying near or outside of the average upper confidence limits of Model II would be a strong indication that these streams do not belong to this relationship and the ambient water is not the major source of tissue phosphorus.

Figure 6

Regression line of Model II including the estimated upper ninety-five percent confidence limit. Open circles are Type I stream data points and boxes are Type II stream data points



The above methodology can be used to determine if Model I or Model III would be the best management tool to predict late summer macrophyte biomass for a given stream reach. If the data for a given stream reach, when plotted on Figure 6, fall within the upper confidence interval established for Model II and macrophyte distribution and substrate type are characteristic of Type I streams, then it would be strong evidence that Model I would be the best model available to predict macrophyte biomass. If the conditions for Model I are not met, then Model III should be used to predict macrophyte biomass, as it was developed under the premise that biomass is predictable independent of nutrient source (i.e. water or sediment). The confidence intervals for Model III, however, are much wider than those for Model I and the predicted values from this model would have a lesser degree of accuracy associated with them.

Summary and Conclusions

Empirical relationships were developed that describe the responses of macrophyte communities to a range of in-stream phosphorus concentrations. The primary purpose for quantifying these relationships was to develop a predictive tool to estimate late summer macrophyte biomass in selected stream reaches.

It was evident from the data analysis that macrophyte growth could not be predicted from in-stream nutrient concentrations for all stream reaches that were surveyed. Streams were classified by substrate type, macrophyte distribution and apparent macrophyte nutrient source. Type I stream reaches are characterized by having sand-gravel-rubble bottom substrate, shallow depth, and relatively homogenous macrophyte distribution. Macrophytes in Type I streams are believed to be utilizing the ambient water as their primary nutrient source. Type II stream reaches had silt-sand substrate, shallow depth and macrophyte occurrence was limited to zones of silt deposition. Bottom sediments are believed to be the primary nutrient source in Type II streams. All streams received little or no shade.

The results of the 1981 sediment surveys suggest that interstitial water nutrients are closely related to the nutrient concentrations in the overlying water. This relationship has been demonstrated by Vaux (1962) for streams with large substrates.

The analyses of the 1981 sediment and interstitial water data indicated that there were not significant differences in the phosphorus and nitrogen concentrations in and out of areas colonized by macrophytes. This analysis suggests that for Type I streams macrophytes are not colonizing areas of nutrient rich substrate. These findings are supportive of the hypothesis that for Type I streams macrophytes are utilizing the ambient water as the primary nutrient source.

It was apparent that macrophyte growth responses could be modeled best for situations where the major nutrient source was quantified. Model I and Model II were developed with the data collected from the Type I stream reaches. Model I is linear regression equation that estimates macrophyte biomass from mean summer in-stream phosphate-phosphorus concentrations. The equation describing Model II was developed by regressing the average macrophyte tissue phosphorus concentration on the mean summer in-stream phosphate-phosphorus concentration. Model I and Model II fit the data sets they were developed from very well, with R-square values of .823 and .921, respectively.

Model III is a linear regression equation developed to describe the relationship between macrophyte biomass and macrophyte tissue phosphorus concentrations. Data from both Type I and Type II stream reaches were used to calculate this model as this relationship was considered to be unaffected by the source of nutrients that the macrophytes were utilizing. Model III does not fit the data as well as Model I or Model II, it has an R-square value of .638.

Model I and Model III are predictive equations that estimate late summer biomass in selected small stream reaches. Model II may be used along with stream reach mapping data to determine which model will provide the best estimate of macrophyte biomass. Model I will provide the best biomass estimates for stream reaches classified as Type I stream reaches. Model III can provide a methodology to estimate biomass in Type II stream reaches.

In order to provide useable management tools, we wanted to derive empirical models which were at least as statistically significant as models currently being used by lake managers. An analysis of the precision of various lake phosphorus loading models was conducted by Canfield and Bachmann (1981) using a data set of 704 lakes. Models evaluated were Canfield and Bachmann (1981), Larsen and Mercier (1976), Jones and Bachmann (1976), Reckhow (1979), Kirchner and Dillon (1975), Chapra (1975) and Vollenweider (1975).

The most precise model evaluated in this group was that of Canfield and Bachmann (1981) which had an R-square of .69, average residual error of 38 percent and confidence limits (p=.05) of 31-288 percent. The range of the average error for the rest of the models was 42 to 63 percent with confidence limits (p=.05) ranging from 15 to 599 percent. The precision of the models developed from the preceding data analyses compares favorably with lake models currently being used.

Of the models developed from this study, Model I may have the most significant impact on the management of water quality in small streams. This model has the predictive capabilities to estimate changes in macrophyte biomass when mean summer in-stream phosphate-phosphorus concentrations are changed. At this point however, Model I as well as the other models are untested and therefore their application should be limited to the stream reaches that the models were developed from. Before these models can become accepted water quality management tools, they should be substantiated by applying them to independent data sets. The test data set must be collected using identical criteria as were used to collect the data set the models were developed from.

PERIPHYTON

Introduction

Research directed at defining periphyton response to stream enrichment has utilized a variety of approaches and methods. These include plant pigment, gravimetric and enzyme analyses, cell counts, community species composition, and occurrence of indicator species. Employing many of these types of analyses in routine water quality management activities is usually not practical, due to time or budget constraints and the complexity of many of the analyses.

Periphyton analyses such as chlorophyll-a and gravimetric (e.g. ash-free weight biomass) estimates have been used in routine monitoring programs. These analyses have the advantage of being relatively inexpensive and commonly used. Collection and analysis techniques have also been developed to the extent of providing relatively uniform sample quality assurance and comparability of data. This includes sampling equipment (such as glass-slide samplers), incubation (exposure) times and sample handling and preservation. Data evaluation and interpretation, however, is still dependent on the investigator's skill, experience and personal preferences.

The influence of physical factors such as light, temperature and water velocity on the resulting biomass estimates is, however, poorly documented. This could result in inaccurate or poor correlation with growth nutrients.

The purpose of the periphyton element in the Phosphorus Assessment study was mainly to characterize growth of periphyton communities in streams and compare this to stream chemical and physical characteristics. Specific objectives included:

- Augmenting macrophyte collections to estimate stream primary production (including photosynthesis and respiration estimates)(see diel section); and

- Evaluating the ability of sampling methodologies and conventional parameters as tools to assess stream nutrient status. This included glass-slide samplers as well as samples collected from brick substrates.

As with the macrophyte data, correlation matrices and least squares regression models were calculated to evaluate which physical and chemical parameters correlated best with the periphyton data. Regression equations were only calculated for paired parameters having significant correlation coefficients.

The data used in the development of the periphyton models (brick collections) are presented in Table 13.

Results and Discussion

Periphyton Biomass and In-stream Nutrients

Periphytometers

Monthly means were calculated from the 1981 and 1982 periphytometer data. Periphytometer chlorophyll-a concentrations correlated positively with in-stream phosphorus and nitrogen concentrations. These relationships improved with transformation to their natural logs (Table 14). The natural log (ln) of periphytometer chlorophyll-a was correlated most strongly with ln TOTP (r=.635, p=.0001, n=76) and

Table 13

Brick Data Used to Calculate Models IV, V and VI
PO_4P Values are in mg/l, Brick Chla in mg/m^2, and
Brick Tissue Phosphorus in mg/gm Dry Weight

Stream	Month	PO_4P	Chl-a	Tissue-P	(n)
Bark - Wolf	6	.003	20	.67	2
	7	.002	13	.51	2
	8	.002	13	.46	3
	9	.002	18	.39	2
	10	.002	15	.28	1
Bark - Lurvey	6	.095	33	.96	2
	7	.116	68	1.10	2
	8	.119	63	1.83	3
	9	.202	96	1.84	2
	10	.260	140	1.20	1
Bark - Masonic	7	.108	121	2.22	1
	8	.110	224	2.28	3
	9	.208	146	2.56	2
	10	.237	400	1.50	1
Kohlsville	6	.040	54	1.06	2
	7	.045	60	1.37	2
Sugar	6	.053	103	1.57	2
	7	.050	62	1.65	2
	8	.043	59	1.42	3
	9	.031	55	1.41	1

$lnPO_4P$ (r=.603, p=.0001, n=76). Periphytometer ln chlorophyll-a : ln inorganic nitrogen and ln total nitrogen correlation coefficients were r=.485 (p=.0001, n=79) and r=.337 (p=.003, n=76) respectively.

Periphytometer Dry Weights and Ash-free Weights did not significantly correlate (p .05, n=76) with in-stream nutrient values.

Bricks

Monthly mean values were calculated from the 1981 and 1982 data. Chlorophyll-a and nutrient data were collected only in 1982. In general, brick periphyton biomass estimates exhibited higher correlation coefficients with in-stream nutrients than periphytometer estimates. As with the periphytometer values, correlations improved with natural log transformations. The natural log of brick chlorophyll-a collections with lnTOTP and $lnPO_4P$ correlates were r=.879 (p=.0001, n=21) and r=.875 (p=.0001, n=21) respectively (Table 14). The natural logs of brick chlorophyll correlated significantly with ln inorganic nitrogen (r=.648, p=.004, n=21), but not with ln total nitrogen (r=.090).

Table 14

Correlation Coefficients for Monthly Mean Water Chemistry Values and Periphyton Biomass Estimates

	lnTOTP	$lnPO_4P$	lnTKN	lnINORN
lnPAFWT	.453	.413	.273	.413
lnPUCCHLA	.635	.603	.343	.479
lnBAFWT	-.203	-.060	-.405	.387
lnBUCHLA	.879	.875	.036	.648

Periphytometer Values included 1981-1982 Data and Brick Values included 1982 Data

Periphyton parameters are noted as: Periphytometer Ash-free Weight (PAFWT), Periphytometer Chlorophyll-a (PUCCHLA), Brick Ash-free Weight (BAFWT), and Brick Chlorophyll-a (BUCHLA)

Model Development - Periphyton Biomass and In-Stream Nutrients

The correlation analyses indicated that a significant relationship existed between in-stream PO_4P and periphyton chlorophyll-a concentrations. Between the periphytometer and brick harvests, the brick chlorophyll a: PO_4P relationships were most significant. A least squares regression equation was then calculated for brick chlorophyll-a and stream PO_4P. The curve representing this relationship is presented in Figure 7. The equation describing this model is:

MODEL IV

$$\text{BRICK CHLOROPHYLL-A} = 258.68(PO_4P)^{.455}$$

where: BRICK CHLOROPHYLL-A is in mg/m^2

PO_4P = stream PO_4P concentration in mg/l

This model has an R-square of .766 (p=.0001, n=20). Mean residual error is 36.24%, and ranged from .60 to 214.40% of the predicted values. The mean upper and lower 95% confidence limits are 267% and 37% respectively.

Periphyton Tissue Nutrients and In-stream Nutrients

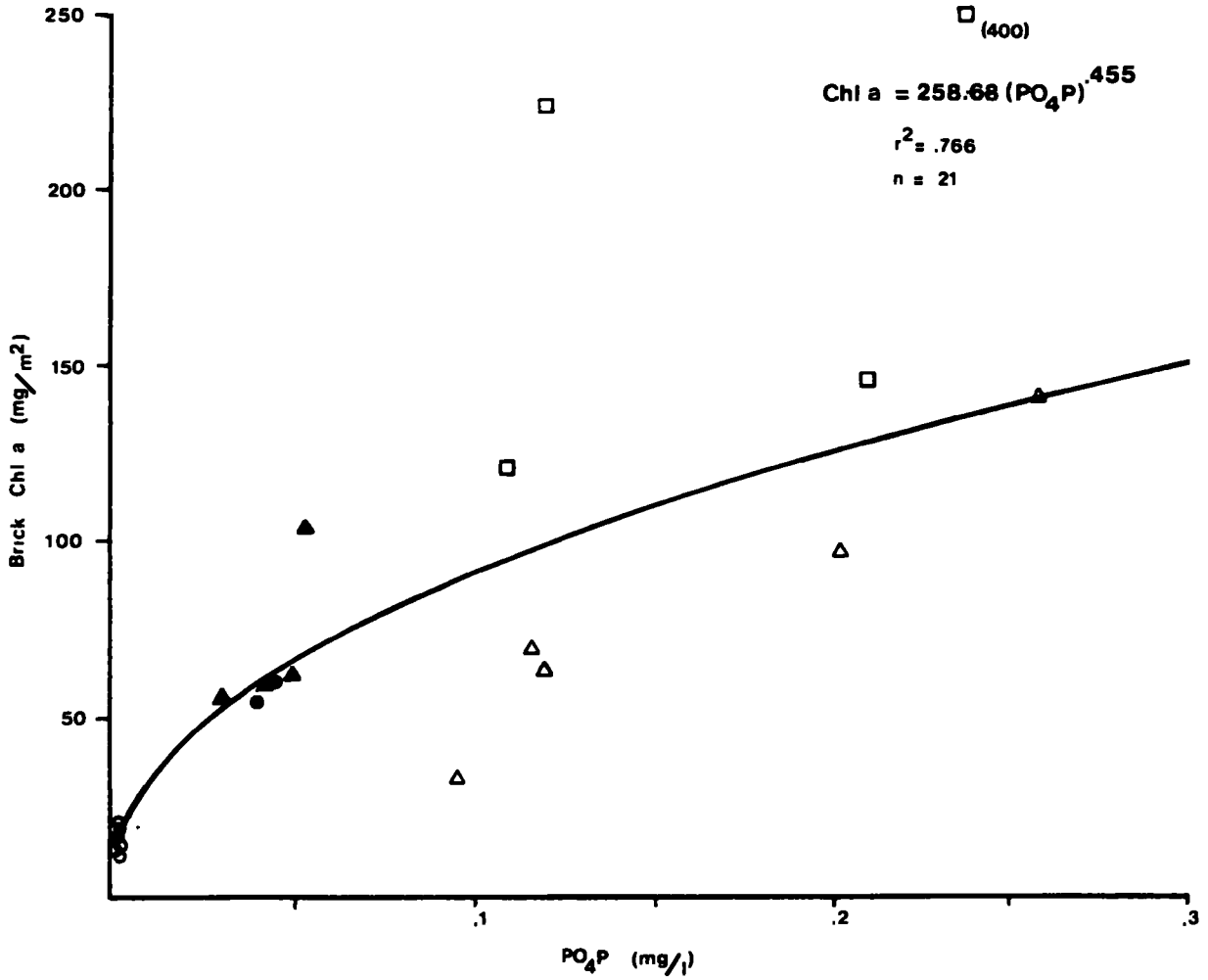
Periphytometers

In general, periphytometer tissue nutrients were positively correlated with in-stream phosphorus and poorly correlated with in-stream nitrogen. Relationships improved when natural log transformations were made.

The best correlations occurred between periphytometer tissue phosphorus with lnTOTP (r=.375, p=.0006, n=74) and $lnPO_4P$ (r=.352, p=.0008, n=74). Periphyton tissue N was not significantly correlated with either in-stream nitrogen or phosphorus values (Table 15).

Figure 7

Regression line of monthly mean brick periphyton chlorophyll-a (mg/m^2) on monthly mean stream PO_4P concentrations (mg/l)



The correlations between periphytometer tissue nutrient concentrations (mg P/g AFWT) and stream nutrient concentrations are generally poorer than those derived for bricks or macrophytes. One possible explanation for this result lies in the methodology used to determine periphytometer tissue nutrient concentration. Ash-free weight (mg/sq meter) and "periphyton nutrient content" (mg P/sq meter) were determined from separate slide collections. Unless periphyton growth on both of the slides was very similar, the resultant tissue nutrient concentration (as mgP/g AFWT) would be inaccurate. (Brick and macrophyte tissue nutrient concentrations were determined by the UW Soils Lab as a percentage of the actual sample dry weight).

Bricks

Tissue nutrient concentrations from brick periphyton were positively correlated with in-stream nutrient concentrations. Correlation coefficients were generally much higher than those of corresponding periphytometer data.

Table 15

Correlation Coefficients for Monthly Mean Water Chemistry Values and Periphyton Tissue Nutrient Concentrations

	lnTOTP	$lnPO_4P$	lnTKN	lnINORN
lnPCAFW	.375	.352	.184	.194
lnNCAFW	.226	.179	.092	-.019
lnBRIKN	.848	.868	.130	.738
lnBRIKP	.839	.864	.286	.819

Periphyton parameters are as noted: Periphytometer Tissue Phosphorus Concentration (PCAFW), Periphytometer Tissue Nitrogen Concentration (NCAFW), Brick Tissue Nitrogen Concentration (BRIKN) and Brick Tissue Phosphorus Concentration (BRIKP)

Tissue phosphorus and nitrogen had higher correlation coefficients with in-stream phosphorus than in-stream nitrogen. Brick tissue phosphorus correlated best with $lnPO_4P$ (r=.864, p=.0001, n=20), lnTOTP (r=.839, p=.0001, n=20) and lnINORN (r=.819, p=.0009, n=20). Similarly, Brick tissue N correlated best with lnPO4P (r=.868, p=.0001, n=20), lnTOTP (r=.848, p=.0001, n=20) and lnINORN (r=.738, p=.01, n=20).

Based on these correlates a least squares regression equation was calculated describing the relationships between brick tissue phosphorus and in-stream PO4P concentrations (Figure 8). This equation is:

MODEL V

$$\text{BRICK TISSUE PHOSPHORUS} = 3.07(PO_4P)^{.230}$$

where: TISSUE PHOSPHORUS is in mgP/g dry weight

PO_4P = instream PO_4P concentrations in mg/l

This model has an R-square of .747. Mean residual error is 26.76%, ranging from 6.25 to 53.27% of the predicted values. The mean upper and lower 95% confidence limits are 203.66 and 49.03% respectively.

Periphyton Biomass and Tissue Nutrients

The correlation coefficients of the periphyton biomass: tissue nutrient concentration parameters are listed in Table 16. The correlation coefficients derived from brick parameters were substantially higher than those derived from periphytometers. The rather large negative correlations between nutrient concentrations derived from periphytometer and brick ash-free weights are anomolies. The negative correlations could be due to shading of the bricks by macrophytes or, as mentioned above, inaccurate approximation of the periphyton nutrient concentrations.

Table 16

Correlation Coefficients for Periphyton Nutrient Concentration and Biomass Estimates

	lnPCAFW	lnNCAFW	lnBRIKP	lnBRIKN
lnPAFWT	.335	.258	.234	.109
lnPUCCHLA	.529	.415	.637	.540
lnBAFWT	-.869	-.930	.713	.531
lnBUCHLA	.793	.652	.831	.826

Periphytometer data included 1981-1982, Brick data included 1982 only.

Periphyton parameters are noted as those in Tables 14 & 15.

The best reasonable (i.e. positive) relationship between a tissue nutrient concentration and a biomass measurement was between brick chlorophyll-a and brick tissue phosphorus. The model calculated for this relationship is (Figure 9):

Figure 8

Regression line of monthly mean brick periphyton tissue P on monthly mean stream PO_4P concentrations (mg/l)

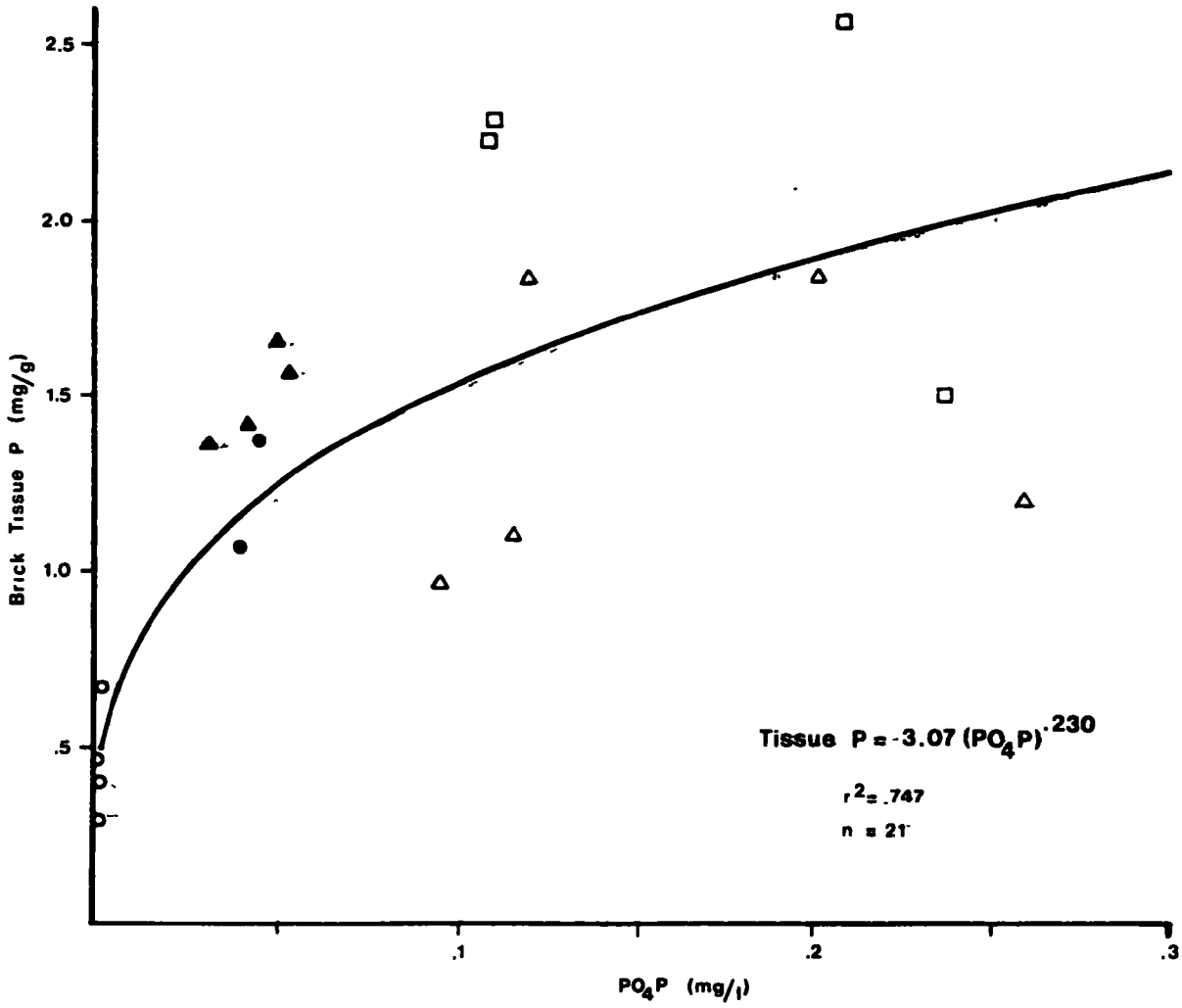
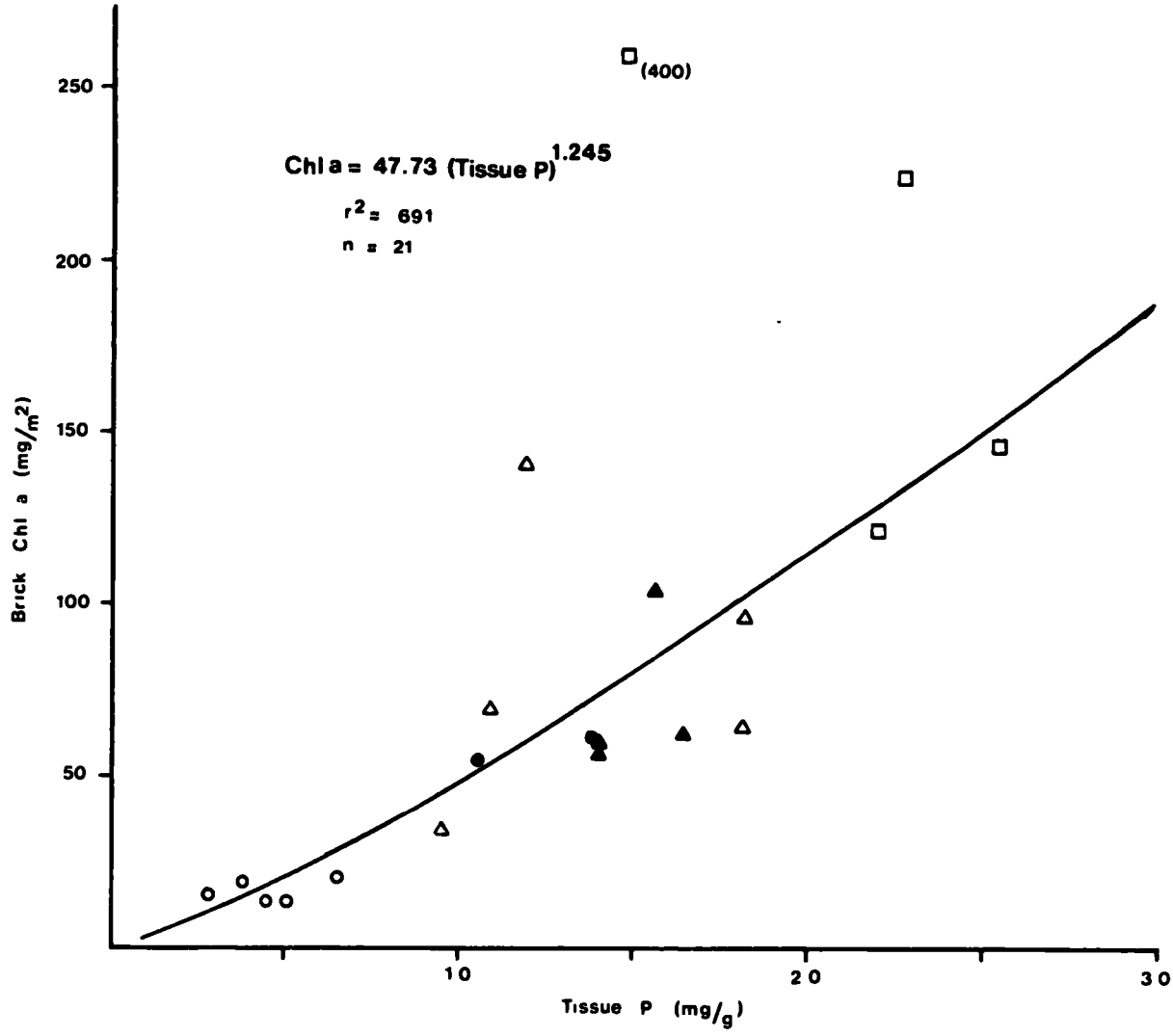


Figure 9

Regression line of monthly mean brick periphyton tissue phosphorus concentrations (mg/gm) on monthly mean stream PO_4P concentrations (mg/l)



MODEL VI

BRICK CHL-A = 47.731 (BRICK TISS-P)$^{1.245}$

where: BRICK CHL-A is in mg/m^2
BRICK TISS-P is in mgP/g dry weight

This model has an R-square value of 0.691. Mean residual error is 49.8%, and ranges from 5.2% to 406% of the predicted values. The mean upper and lower 95% confidence limits of the predicted value are 325% and 11.4% respectively.

Other Aspects of Periphyton Growth and Measurement

The ratio of ash-free weight (AFW gm/m2) to uncorrected chlorophyll-a (UCCHLA, mg/m2) is known as the Autotrophic Index (AI). Autotrophic index values are generally interpreted as indicators of the trophic level of periphyton communities. High AI values (>200) are found in heterotroph-dominated communities and lower values are found where autotrophic organisms are dominant (APHA, et al. 1981).

The periphytometer-derived AI values were negatively correlated with the natural logs of several nutrient parameters (e.g. lnPO4P $r=-.55$, $p<.01$ and lnINORN, $r=-.48$, $p<.01$). This indicates that periphyton communities become more autotrophic as nutrient levels increase. The brick-derived AI values were negatively correlated with PTOT ($r=-.46$, $p<.01$) and PO4P ($r=-.46$, $p<.01$), but were insignificantly correlated with in stream nitrogen concentrations (INORN, TOTN, and NH3N).

The brick-derived AI values were substantially larger than those derived from periphytometers (Table 17). The reason for this is uncertain, but may be related to colonization time (two weeks for the periphytometers, four weeks for the bricks) or other factors such as depth, velocity or substrate carrying capacity.

A study comparing the periphyton on artificial and natural substrates was conducted (Babros 1981). The study was carried out on the Kohlsville River (periphyton-dominated stream), and concluded that periphytometers estimate chlorophyll-a acceptably, but underestimate the AFW of natural substrates. The AI values for the natural substrate samples were much higher than any found on periphytometers. The mean of three AI estimates (using pheophytin-corrected chlorophylls) was 950. The use of uncorrected chlorophyll values would decrease this value to approximately 500, which is still much higher than the corresponding periphytometer estimates of 125-228.

Table 17

Mean Values for Autotrophic Index Values, as Estimated from the Bricks (BAI) and Periphytometers (PAI) (The values given are the means of all samples available)

Site	BAI	PAI
Bark-Wolf	388	261
Bark-Lurvey	231	85
Bark-Masonic	123	59
Kohlsville	368	173
Sugar Creek	382	137

Velocities were determined at periphytometers when the slides were placed and collected. Correlation of ash-free weight and chlorophyll values with velocity (using all available data) did not yield significant results. On two dates, however, the periphytometers did show a negative correlation of ash-free weight with velocity. R-squared values for the regression of velocity against ash-free weight values were 0.95 and 0.49, $n = 6$ in both cases. Each of these regressions used periphytometers which were exposed to identical nutrient and light conditions.

Some concern has been expressed as to whether or not ambient phosphate concentrations in small streams are naturally high enough to "saturate" periphyton growth. The positive correlations of ortho and total phosphate concentration with both brick and periphytometer estimates of chlorophyll-a would seem to indicate otherwise. Other studies and measurements designed to estimate a "saturation level" offer further evidence that increased phosphorus levels will lead to increased periphyton growth in small streams (Auer and Canale 1982; Rosemarin 1982; Lehman, et al. 1976).

Summary

Horner and Welch (1981) have shown that equations can be developed to predict chlorophyll-a from temperature, velocity and phosphorus concentrations. A total of six equations were developed, each of which was only applicable to a particular sample period and velocity range. The coefficients of the parameters (temperature, velocity and phosphorus concentration) exhibited substantial differences between colonization periods and velocity ranges.

Intensive studies designed to describe the growth of a single species of algae (*Cladophora glomerulata*) in the littoral region of the Great Lakes further demonstrate the complexity of periphyton growth (Auer, et al. 1982).

The results of this study and others (Auer, et al. 1982; Horner and Welch 1981)

strongly suggest that increased nutrient levels, particularly phosphorus, will stimulate the growth of periphyton in small streams. The increase in biomass will be modified by many other factors, including temperature, light, velocity and inherent characteristics of the dominant species or community (e.g. resistance to sloughing). Because of the variability of the importance of these factors, it does not appear feasible to develop a "universal model" capable of accurately predicting areal periphyton biomass in small streams. Site specific models, however, appear to be relatively accurate and easily obtainable.

DIEL DISSOLVED OXYGEN STUDIES

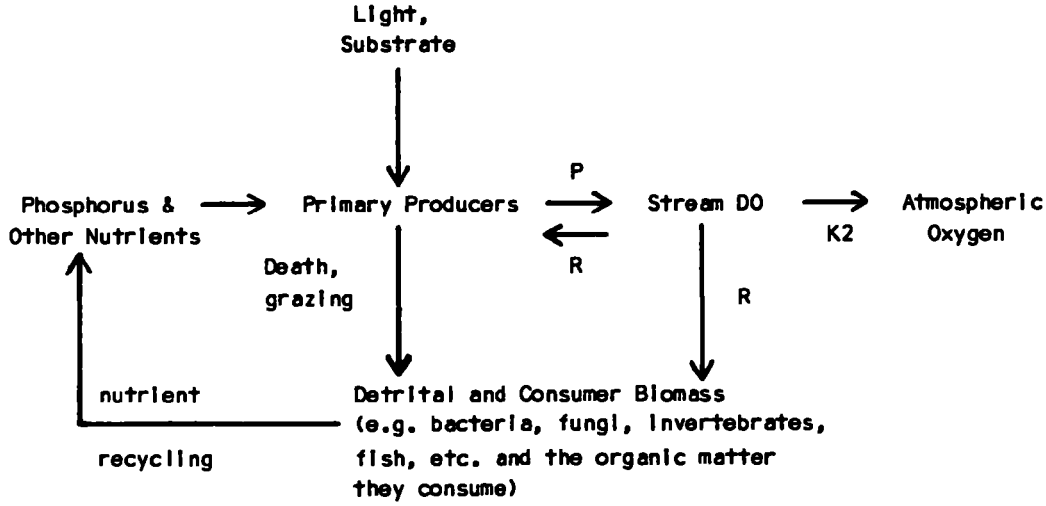
INTRODUCTION

It is convenient to divide the factors responsible for diel stream dissolved oxygen (DO) fluctuations into two general categories; metabolic and physical. The metabolic category consists of plant and animal metabolic activity. The net effects can be positive or negative (DO production or consumption) depending on time of day and biological community composition. The physical category includes the effects of reaeration, loading from tributaries and groundwater inflow. Loadings from tributaries and groundwater inflow were not obvious at the study sites, and were not considered in this study. The instantaneous contribution of reaeration to stream DO content is positive when the water is undersaturated and negative when the water is oversaturated with respect to DO. The magnitude of the deficit or surfeit will determine the magnitude of the instantaneous rate.

Figure 10 represents a simple path by which the impact of phosphorus on stream dissolved oxygen levels can be examined. The substances which are dissolved in the water are generally expressed as concentrations (e.g., mg/l dissolved oxygen). The biomass quantities are usually expressed as "weight per unit area" (e.g. grams of macrophyte dry weight per square meter). The rates of photosynthesis (P) and respiration (R) can be measured in light and dark enclosures (bottles or boxes), or approximated from diel surveys. These rates can be expressed as units of oxygen produced or consumed per unit biomass per unit time (grams oxygen/kg dry weight/hr). For the measurement of photosynthesis, a light level must be defined. The primary producer biomass includes both macrophytes and periphyton,

Figure 10

Path diagram of factors which regulate stream dissolved oxygen concentration.



and the consumer biomass includes a rather diverse group of organisms, such as bacteria, invertebrates and fish. The respiration or photosynthesis rates per unit biomass can be expected to vary for different organisms, as well as with life stage, light, temperature, etc.

Over the course of a diel survey (about 24 hrs), much of this variation can be ignored if the community composition is assumed to remain constant. Community respiration is the sum of all types of respiration, and community photosynthesis is the sum of all types of photosynthesis. The measured rate of change of stream dissolved oxygen is due to community photosynthesis and respiration and therefore diel curve analysis yields community rates.

The other major process which impacts stream dissolved oxygen is the reaeration rate. K2 is a constant which expresses the proportion of a deficit which will be satisfied per unit time. In order to specify a rate (quantity per unit time) due to reaeration, this constant must be multiplied by a deficit (Cs-Co). Cs represents the saturation concentration of dissolved oxygen, which is calculated from temperature data assuming normal atmospheric pressure (760 mm Hg). Co is the measured concentration of dissolved oxygen. When $Cs > Co$, a deficit exists, and the product $K2(Cs-Co)$ should be positive, indicating that oxygen is being gained by the stream. When $Co > Cs$, the product $K2(Cs-Co)$ should be negative, indicating that oxygen is being lost to the atmosphere.

The most notable aspect of Figure 10 is that phosphorus only directly impacts the primary producers (macrophytes and periphyton). This relationship has been explored in detail and has resulted in various models (Canale and Auer 1982, Dillon and Rigler 1974, Jones and Bachman 1976) as well as the phosphorus and macrophyte biomass model presented in this report. Figure 10 then represents a simple path by which the impact of phosphorus on stream dissolved oxygen levels can be examined.

Primary producers are capable of removing oxygen from the water as well as adding it. Other factors, such as reaeration and non-photosynthetic organisms, also influence stream dissolved oxygen content. The relative importance of each of these factors is site specific, but some generalizations are possible.

Several authors have remarked that small streams tend to be net consumers of dissolved oxygen (e.g. Hynes 1970, Vannote et al. 1980). On a daily basis, respiration tends to exceed production. This suggests that respiration in small streams is not strictly a function of primary producer biomass, since a positive net production is required for the accumulation of plant biomass. If the primary producer biomass is getting larger (as demonstrated by growth on periphytometers and seasonal increases in the harvested macrophyte biomass) and community respiration is larger than community photosynthesis, a significant part of the community respiration must be due to non-photosynthetic (consumer) organisms. The relationship between phosphorus and community respiration thus seems likely to be naturally variable due to differing amounts of respiration attributable to consumer organisms (invertebrates, fish, bacteria, etc.) which are not likely to be phosphorus limited.

Photosynthesis, since it is only a function of primary producers, would seem more likely to correlate well with phosphorus. However, due to seasonal changes in photosynthetic efficiency, variable amounts of biomass, self shading, daylength and community composition, this relationship is likely to be difficult to define.

Reaeration, the third major factor responsible for stream dissolved oxygen fluctuations, has no direct relationship with phosphorus. The only possible impact results from ponding of the stream due to macrophyte growth. Ponding (increased depth and decreased velocity) could be expected to decrease the reaeration rate.

If the quantities and rates in Figure 10 can be approximated from field studies and modeling efforts, a good approximation of stream dissolved oxygen content should be possible.

METHODS OF ESTIMATING P, R, AND K2

Box Studies

Light and dark box studies were conducted on the Bark River in late July and early August. The average values of net photosynthesis (Pnet) and respiration (R) for three days are presented in Table 18. The light values (photosynthetically active radiation [PAR, 400-700 nM]) for the incubation periods were around 60-91 x 10^{15} quanta/sec/cm^2, well in excess of the "typical" light saturation levels of 25-30 x 10^{15} quanta/sec/cm^2 reported by Westlake (1966). The Pnet values are far below the 10 gO2/kgDW/hr^1 reported by Westlake (1966). Self shading could account for at least part of the difference. The plant densities under the boxes were at or above Westlakes' calculated plant densities for optimum daily net production. Other possible explanations include internal storage of oxygen, decreased productivity due to senescence, bubble formation and possibly leakage of the boxes.

The dark box estimates of respiration were in better agreement with Westlakes' estimate of 1.5 gO2/kgDW/hour. The values from the box studies were slightly higher, which is not surprising since the boxes would also include sediment oxygen demand (SOD), as well as periphyton and invertebrate respiration. For example, on 29 July, the DO concentration in the box decreased 1.18 mg/l in the first hour. Volume of the box was 80 liters, so total consumption of DO was 80 x 1.18 = 94.4 mg O2 in one hour. The macrophyte biomass enclosed in the box was 51.94 gDW. Straight division (94.4/51.94) gives 1.82 mgO2/gDW/hr. If the "true rate" was 1.5 mgO2/gDW/hr, the consumption due to macrophytes would be 1.5 x 51.94 = 77.91 mgO2. This leaves 16.49 mgO2 (94.4 - 77.91) unaccounted for. If this remainder is entirely attributed to SOD, the SOD rate would be 16.49/0.28 m^2 = .059 gO2/m^2/hr, which is within the range of .0125 -.125 gO2/m^2/hr reported by Edberg and Hofsten (1973). The results also seem comparable to those of Owens and Edwards (1962).

Table 18

Box Study Summary

DATE	NET PHOTOSYNTHESIS	RESPIRATION
29 JULY, 82	3.00	1.82
	2.68	1.51
10 AUGUST,82	3.25	1.76
12 AUGUST,82	2.31	2.42

Pnet and R values in gO2/kgDW/hour.

"Daily totals" of P and R from the box studies can be calculated if photosynthesis is presumed directly proportional to light and respiration is assumed constant. At 4 gO2/kg DW/hr (@ 80 x 10^{15} Quanta/sec/cm^2), an average day in July (3 x 10^{21} Q/sec/cm^2) would give a gross production value of 42 gO2/kg DW/day. Respiration would be 1.5 g/kgDW/hour, or 36 gO2/kg DW/day. Net production then, would be about 6 gO2/kg DW/day. On a very overcast July day, total light could be as low as 1 x 10^{21} Q/sec/cm^2, reducing gross photosynthesis to 14 gO2/kg DW/day, and Pnet would be -22 gO2/kg DW/day.

Each biomass sample harvested in the box studies was also analyzed for nutrient content. Nutrient levels were low, and typical of the Bark River-Wolf Road site. One box study was conducted at the Masonic Home site, but it failed to show a substantially higher net photosynthesis rate.

Bottle Studies

Bottle studies were conducted as another method of approximating productivity. The results are listed Table 19. Net and gross productivities from the bottle studies are

Table 19

Bottle Study Results. In gO2/kg Dry Weight/hr

Date	Site	Pnet	R	Replicates	Species
28 July	BM	20.0		3	Potamogeton spp.
		15.7		3	Heteranthera dubia
15 Aug	B	10.3		9	Vallisneria americana
			1.0	4	" "
		7.3		6	Heteranthera dubia
			1.5	2	" "
15 Aug	BL	11.2		5	V. americana
			2.7	2	" "
		9.6		8	H. dubia
			2.3	2	" "
20 Aug			0.47	5	Myriophyllium spp.
			0.62	5	H. dubia
			0.86	5	Potamogeton spp.
1 Sept	BM		1.3	12	V. americana
			1.7	12	H. dubia
3 Sept	BL		2.3	4	H. dubia
			8.1	4	"epiphytes" (periphyton dislodged from macrophytes)
			2.4	4	H. dubia and epiphytes

substantially higher than those of the box studies. Respiration values are approximately the same.

There are at least three possible explanations for the discrepancy between the box and bottle Pnet estimates. The first is that the bottles included primarily leaf and/or stem tissue, whereas the boxes contained entire plants in situ. Some whole plants were included in the bottle studies, but due to individual bottle variability, small sample numbers and obviously artificial conditions, no conclusions are warranted. The second possible explanation is internal storage of oxygen (Wetzel 1975). This effect could be pronounced in intact plants, such as in the boxes. Leaf and stem fragments in the bottles could be expected to store less oxygen. The third explanation is that self shading could be more important in the boxes than the bottles. Field observations pertaining to the box studies do not support the self shading hypothesis, but further, more carefully constructed experiments would be needed before self shading could be discounted.

On August 15, with one exception, plants from the high nutrient (Lurvey) site showed significantly higher (statistically) metabolic rates than plants taken from the low nutrient (Wolf Road) site. The Heteranthera dubia Pnet values were significantly higher (t-test, $p < .05$) for the high nutrient plants. Respiration rates were also significantly higher at the high nutrient site (t-test, $p < .001$, all values). Due to the small numbers of each species involved, species specific respiration rates were not tested.

Epiphytic algae were present on essentially all of the macrophytes, but were particularly dense on those from the high nutrient (Lurvey) site. Gentle rinsing removes most of the (loose) epiphytes, but some (tight) inevitably remain. Cattaneo and Kalff (1980) studied the relative productivity of epiphytic algae ("loose" and "tight") and macrophytes in lakes. They concluded that the relative production depended on season and nutrient levels. Epiphyte production was found to exceed macrophyte production during spring and fall in mesotrophic portions of the lake, but exceeded macrophyte production all year in the eutrophic portions of the lake. This suggests that periphyton can significantly affect community photosynthesis and respiration terms, despite their relatively small biomass, and that high nutrient concentrations may enlarge the contribution of periphyton to total community rates.

The dark bottle experiments of August 20 showed much lower respiration rates than the other dark bottle studies. This is undoubtedly due to the low initial DO concentration (4.85 mg/l), and the fact that the ending concentrations averaged 0.62 mg/l. Owens and Maris (1964) used a series of short incubations to demonstrate that the respiration rate varied with the dissolved oxygen concentration (Figure 11).

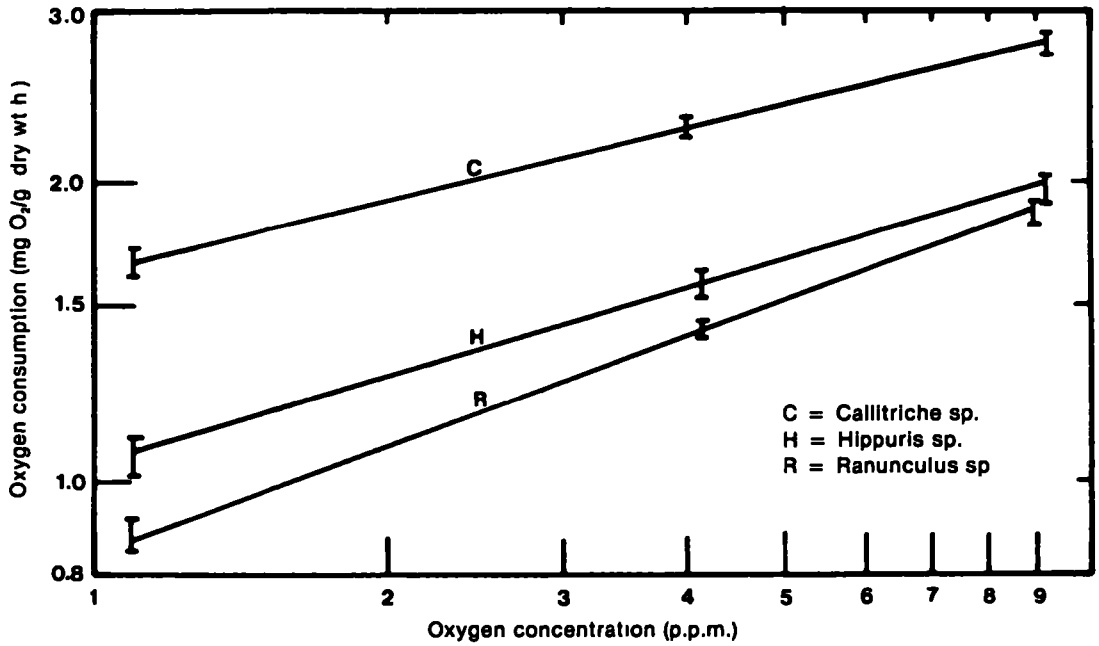
McDonnell and Weeter (1971) also found a decrease in respiration with decreasing DO levels. Unlike Owens and Maris, however, the relationship they found was linear ($R = \alpha + \beta [DO]$), where R = Respiration (mgO2/gDW/hr), and α and β are constants.

Summary of Box and Bottle Study Results

Both the box and bottle studies indicate that macrophyte respiration should be about 1.5 gO2/kg DW/hr. This is in good agreement with the literature estimates of Westlake (1966) and Owens and Maris (1964). The box-derived estimates of photosynthesis seemed lower, and the bottle

Figure 11

The effect of oxygen concentration on plant respiration rates (from Owens and Maris 1964).



estimates higher, than the estimate given by Westlake (1966) of 10 gO2/kg DW/hr at light saturation.

The box study estimates were not numerous enough to statistically test for differences in productivity between low-nutrient and high-nutrient conditions. Bottle studies indicated higher net photosynthetic rates for Heteranthera dubia under high nutrient conditions. Respiration rates, overall, were higher for plants grown under high nutrient concentrations.

DIEL CURVE ANALYSIS

This study also used two modeling approaches to estimate community photosynthesis, community respiration and reaeration. These are referred to as single-station and double-station analyses. When the DO concentration at a single station is monitored over the course of a day, the results of diel changes in the magnitude of both metabolic and physical factors is observable. Analysis of the diel curve for a given station can give estimates of photosynthesis, respiration and reaeration which represent upstream averages. The area for which these averages apply is not clearly defined in the literature, and will be dealt with in a later discussion. A single station method is currently used for stream modeling of WDNR wasteload allocation surveys.

The double-station method analyzes the change in DO between two stations to derive estimates for P, R, and K2. The values for photosynthesis, respiration and reaeration which result from double-station analysis are applicable to the area between stations, but may or may not be representative of the stream as a whole. Both the single-station and double-station methods were used to analyze the data in an attempt to approximate the impact of phosphorus on stream dissolved oxygen levels.

Modeling Assumptions and Parameters

A few assumptions will be made to simplify the modeling process. These are as follows:

- Photosynthesis is directly proportional to light intensity. Some constant (P) when multiplied by light intensity (α) should equal the rate of gross photosynthesis.

- The respiration rate (R) and reaeration coefficient (K2) are constant throughout the day, i.e. that DO fluctuations are not so wide that they significantly influence R and that temperature does not change enough to significantly influence either R or K2 for a given reach.

- The area which is responsible for DO variation at a single-station sample point is homogeneous, and all areas contribute equally, or the area between the two double-station sample points is homogeneous, and causes a linear response in the DO concentration of a mass of water as it moves through the reach.

For modeling purposes, the major terms responsible for stream dissolved oxygen fluctuations ($\Delta DO/\Delta t$) can be characterized as:

- GROSS COMMUNITY PHOTOSYNTHESIS (α P), due to both macrophytes and periphyton.

- COMMUNITY RESPIRATION (R), due to all forms of bacteria, fungi, algae, macrophytes, invertebrates, etc., as well as chemical oxygen demand (COD) (anything which removes oxygen from the water).

- REAERATION, usually represented as K2(Cs-Co), in which K2 is a physical constant which indicates a proportion of the deficit (Cs-Co) which is satisfied per unit time. Cs represents the saturation concentration of dissolved oxygen, which is calculated from temperature data assuming normal atmospheric pressure (760 mm Hg). Co is the measured concentration of dissolved oxygen.

The sum of these three rates [P, R and K2(Cs-Co)] should account for the rate at which the stream concentration of dissolved oxygen is changing, $\Delta DO/\Delta t$. These terms can be combined to approximate a differential equation:

$$\Delta DO/\Delta t = \alpha P + R + K2\ (Cs-Co)$$

It should be noted that the resulting $\Delta DO/\Delta t$ remains constant only as long as all terms [α, P, R, and K2(Cs-Co)] remain constant. If the $\Delta DO/\Delta t$ term is expressed in terms of the deficit ($\Delta(Cs-Co)/\Delta t$) and the differential equation is integrated, the result is:

$$(C_s-C_o)_{t+\Delta t} = (C_s-C_o)_t\, e^{-K_2\Delta t} + [(\alpha P + R)/K_2](1-e^{-K_2\Delta t})$$

This last equation was derived under the assumption that the sum (αP+R) remains constant, and this must be considered when applying the result. The differential form of the equation was used in the double-station analyses, and the integrated form was used in the single-station analyses.

If all of the foregoing assumptions and restrictions are satisfied, the differential and integrated equations should return the same coefficients.

A few simple observations can help clarify these equations, and hopefully represent a simple set of guidelines which define the behavior of diel curves.

First, the $\Delta DO/\Delta t$ term will be positive only when the sum of αP, R and K2(Cs-Co) is positive. In a strictly mathematical sense, all of these terms are independent, and any one of the terms could be the major factor determining the magnitude and sign of the $\Delta DO/\Delta t$ term. More realistically, however, we expect P, K2, Cs and Co to be positive, and R to be negative. Over the course of a day, the αP term should increase with rising light (α) levels, and when the increase is sufficient, the $\Delta DO/\Delta t$ term will become positive, which means that the DO concentration will rise. Towards evening, when the product αP is becoming smaller, the $\Delta DO/\Delta t$ term falls through zero (the highest DO concentration for the day is reached at this point), and then becomes negative which means that the DO concentration is falling. As the DO concentration falls below saturation, the deficit (Cs-Co) begins to increase, which increases the product K2(Cs-Co). Since R is assumed constant and negative, and K_2(Cs-Co) is positive and becoming larger, the two will balance each other and the system will be at "equilibrium" (i.e. $\Delta DO/\Delta t$ will become zero). If a diel curve attains its minimum DO concentration (maximum deficit) prior to dawn ($\Delta DO/\Delta t$ = 0, and α = 0, so αP = 0) we can write:

$$\Delta DO/\Delta t = \alpha P + R + K2(Cs-Co)$$

$$0 = 0 + R + K2(Cs-Co)$$

$$(A) \qquad -R = K2(Cs-Co)$$

$$(B) \qquad -R/K2 = (Cs-Co)$$

Equation (A) indicates that respiration and reaeration are equal. The simple rearrangement in equation (B) defines the maximum deficit that will be achieved. This "maximum deficit" represents an equilibrium concentration of DO. The deficit (Cs-Co) can increase (i.e. Co can decrease) only if R becomes larger or K_2 becomes smaller.

Equilibrium will be attained only if the deficit becomes large enough. The amount of time which must elapse before (Cs-Co) becomes large enough is not obvious from the differential equation. Through examination of the integrated equation, however, we can gain some insight into how long this time period is likely to be.

According to the integrated equation, as the time variable (Δt) increases, the importance of the incoming deficit (Cs-Co) decreases, and the calculated deficit at time "t+Δt" approaches the quotient "(αP + R)/K2". Thus, after sunset, the deficit approaches "R/K2" (because αP = 0). The speed at which the equilibrium level is approached is dependent solely on the magnitude of the K2 term. This is demonstrated in Table 22. As an example, if we specify a K2 value, and allow the 90 percent level to represent a reasonable approximation of the maximum deficit, then we can calculate the amount of time necessary for the maximum deficit (in this case, 90 percent of the maximum deficit) to be achieved (Table 20).

Table 20

Time Required to Satisfy 90 Percent of the Deficit

K_2	Time Required (hours)
1	55.2
2	27.6
5	11.0
10	5.5
15	3.7

At this point, two observations deserve emphasis:

1. The speed at which the maximum deficit is approached is determined solely by K2.

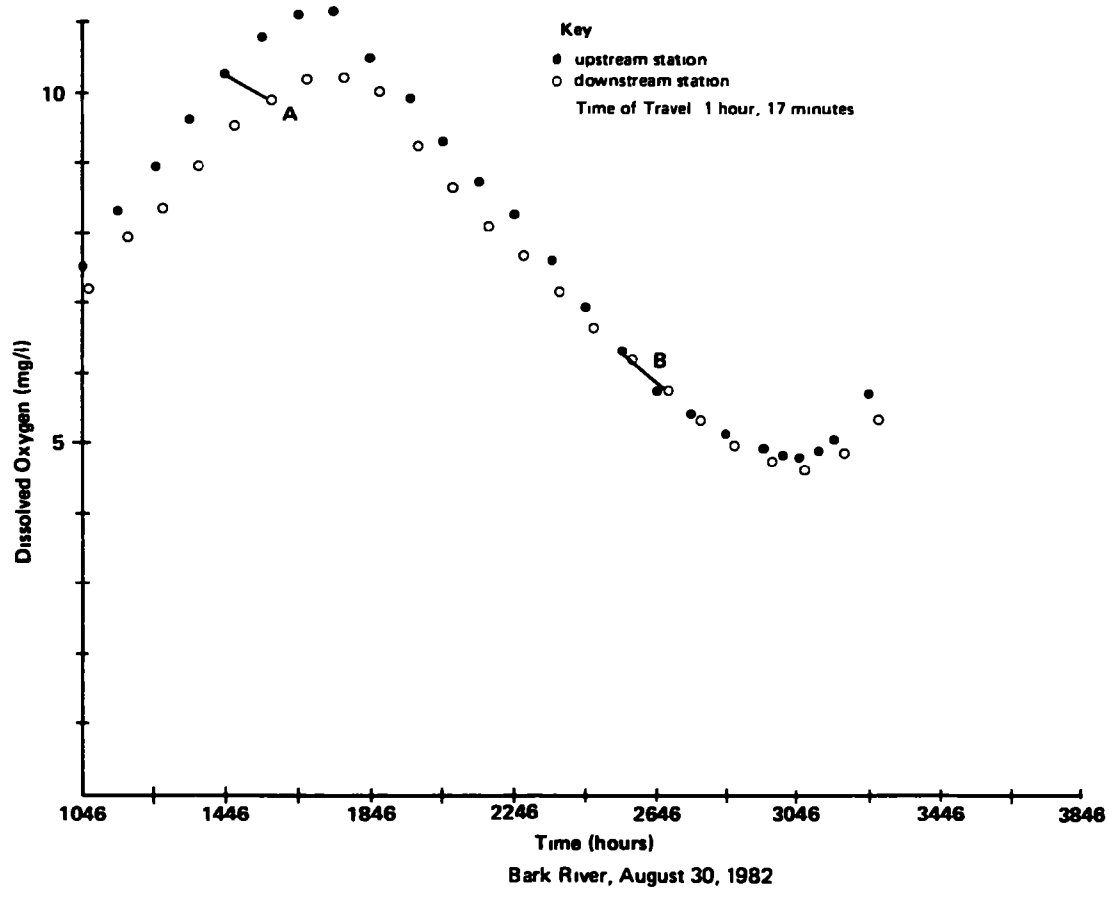
2. The absolute magnitude of the maximum deficit depends on the R/K2 ratio.

Single and Double Station Analyses: Differences in Methodology and Purpose

The major difference between the single and double-station methods is best illustrated by their different interpretations of the $\Delta DO/\Delta t$ term of the differential equation. The single-station method uses the slope of a single diel curve as an estimator of $\Delta DO/\Delta t$. The double-station method uses the change in the DO concentration of a mass of water as it flows from one station to another, divided by the time of travel (TOT) between stations (Figure 12). In case "A" (Fig. 12) the double-station value for $\Delta DO/\Delta t$ would be -0.5 mg/l/hr. The single-station values would be 0.64 and 0.57 mg/l/hr for the upstream and downstream stations, respectively. In case "B", the single and double-station values are essentially equal.

Figure 12

Comparison of single and double-station diel dissolved oxygen methods and purpose



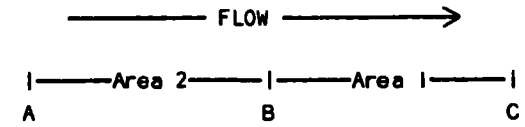
The goal of double-station analysis is to produce a set of coefficients which will accurately predict the DO concentration at a downstream station, if the upstream DO concentration is given, and the temperature and light conditions which prevail for the duration of the time of travel between stations are known. The values of P, R and K2 from double-station calculations are a function of processes within the area between the stations, and may or may not be characteristic of the stream as a whole. If the double-station coefficients are used to generate a single curve (in the same manner that single-station coefficients are used to generate a single curve), the resultant curve may differ from both the upstream and downstream curves from which the double-station estimates were derived. The double-station coefficients will accurately predict the downstream curve only if the upstream curve is given. If the upstream and downstream curves are identical, the curve and coefficients produced by double-station analysis will be identical to the results of the single-station analysis.

If a very good fit is obtained in single-station analysis, the coefficients (P, R, and K2) will reproduce the curve which was obtained for that station. The end result of single-station analysis is a set of coefficients which allows prediction of a DO concentration at any time for a particular station.

Stream Areas Represented by Diel Analyses

Within a homogeneous reach, the area immediately upstream of the sample point will have a greater impact on the dissolved oxygen fluctuations at the sample point than an equivalent area farther upstream. The relative importance of each area, according to the integrated equation, is determined by the magnitude of the K2 value for each upstream area. The larger the K2 value, the smaller the area represented by the single-station method, and the larger the difference between the effect of two upstream areas.

Diagrammatically...



C = single-station sample point
A, B, C = double-station sampling points

For the purposes of this discussion;

I. Area 1 will have a greater impact than Area 2 at point C.

II. The magnitude of this difference is dependent on the K2 value of the entire reach (A - C). A larger K2 value will result in a greater difference between the relative contributions of Area 1 and Area 2 at point C.

The above discussion assumes that Areas 1 and 2 are similar. If there is a great discrepancy between conditions (e.g. biomass, K2, etc.), and the time of travel between the points is short (again relative to the K2 value), the single-station calculations could be more representative of Area 2 than of Area 1. The exact contributions of each area are dependent on the product $(K_2 \times TOT)$. If no differences exist between areas (i.e. the diel curves at each point A, B, and C are identical) the calculated rates could be correctly applied to both areas and the single-station and double-station analyses would be expected to produce the same coefficients (P, R, and K2).

Since the double-station method calculates P, R and K2 values for the area between sample points, the correlation of measured plant biomass (from harvesting and mapping procedures) is potentially straight-forward.

The difference between the single and double-station techniques becomes important when we try to model what will happen to the diel curve at point "C" if a sewage treatment plant discharges at point "B". The only way to predict what will happen to the diel curve at point "C" is by

quantifying the impact of each area. In order to apply coefficients obtained from single-station analysis then, it is necessary to know what area the coefficients were derived from. This concept also has a bearing on the choice of sampling locations (distance or time between sample points) and the question of "how far downstream" the impact of the discharge will reach.

Limit of Reach Length for the Double Station Method

The double-station differential method, as presented above, makes certain assumptions which limit the length of the reach (or the amount of travel time between sample points) to which the method can be applied. The primary assumption is that none of the measured parameters ($\Delta DO/\Delta t$, α or (Cs-Co)) changes significantly during the time interval over which $\Delta DO/\Delta t$ is measured, or that the variation is such that approximation by an average value is justifiable. For example, if photosynthesis is directly proportional to light intensity, and the light intensity varies from 10 to 20 to 30 over a two hour time interval (the "20" value occurring after exactly one hour has elapsed) the total amount of oxygen produced should be correctly predicted by using a single average value of "20" over a two hour period.

If the integrated equation is used, a different set of restrictions on travel time is appropriate. The integrated equation removes the requirement that $\Delta DO/\Delta t$ remains constant, but the requirements regarding constancy of α, P, R and K2 remain. If the integrated equation is used in the double-station technique, the time of travel and stream character within the reach must remain short enough to assure constancy of these terms.

Maximum Attainable Dissolved Oxygen Deficit

According to both the integrated and differential equations, the maximum attainable deficit is R/K2. If the length of the night is multiplied by an approximated K2 value, we can use the integrated equation to calculate what proportion of the difference between the observed deficit at sunset (Cs-Co) and the calculated maximum attainable deficit at sunrise (R/K2) will be satisfied. The actual concentration of DO, of course, depends on the stream temperature and the magnitude of the deficit at sunset. If night-length is 12 hours, and K2 is greater than 5, more than 90% of the maximum attainable deficit will be satisfied.

The concept of a "half-life" for a deficit is pertinent at this point. For those familiar with the fundamental decay equation, K_2 is a decay constant, and is the only factor that controls the rate at which the maximum deficit is approached. After the elapse of one half-life ($K_2 \times t = .693$), 50% of the maximum deficit has been achieved, after two half-lives ($K_2 \times t = 1.386$), 75% of the maximum deficit has been achieved, after three half-lives, 87.5% of the maximum deficit has been achieved, etc.

Examples which illustrate the importance of K2 in controlling the rate at which the maximum deficit is approached follow.

Case 1:

If, for example, the DO deficit at sunset $(Cs-Co)_{ss}$, is 2 mg/l, R/K2= 4, the length of night is 12 hours and K2 is 5/day, then the deficit after 12 hours (i.e. the deficit at sunrise $(Cs-Co)_{sr}$) can be calculated as follows:

$$(Cs-Co)_{sr} = (Cs-Co)_{ss}^{e^{-K_2\Delta t}} + R/K_2\,(1-e^{-K_2\Delta t}) =$$

$$= 2\,(.08) + 4\,(.92)$$
$$= .16 + 3.68$$
$$= 3.84 \text{ mg/l or } 96\% \text{ of } R/K_2$$

Case 2:

If the initial DO was 2 mg/l higher (deficit at sunset = 0), $(Cs-Co)_{sr}$ would be 0(.08) + 4(.92) = 3.67 (92% of R/K_2).

Case 3:

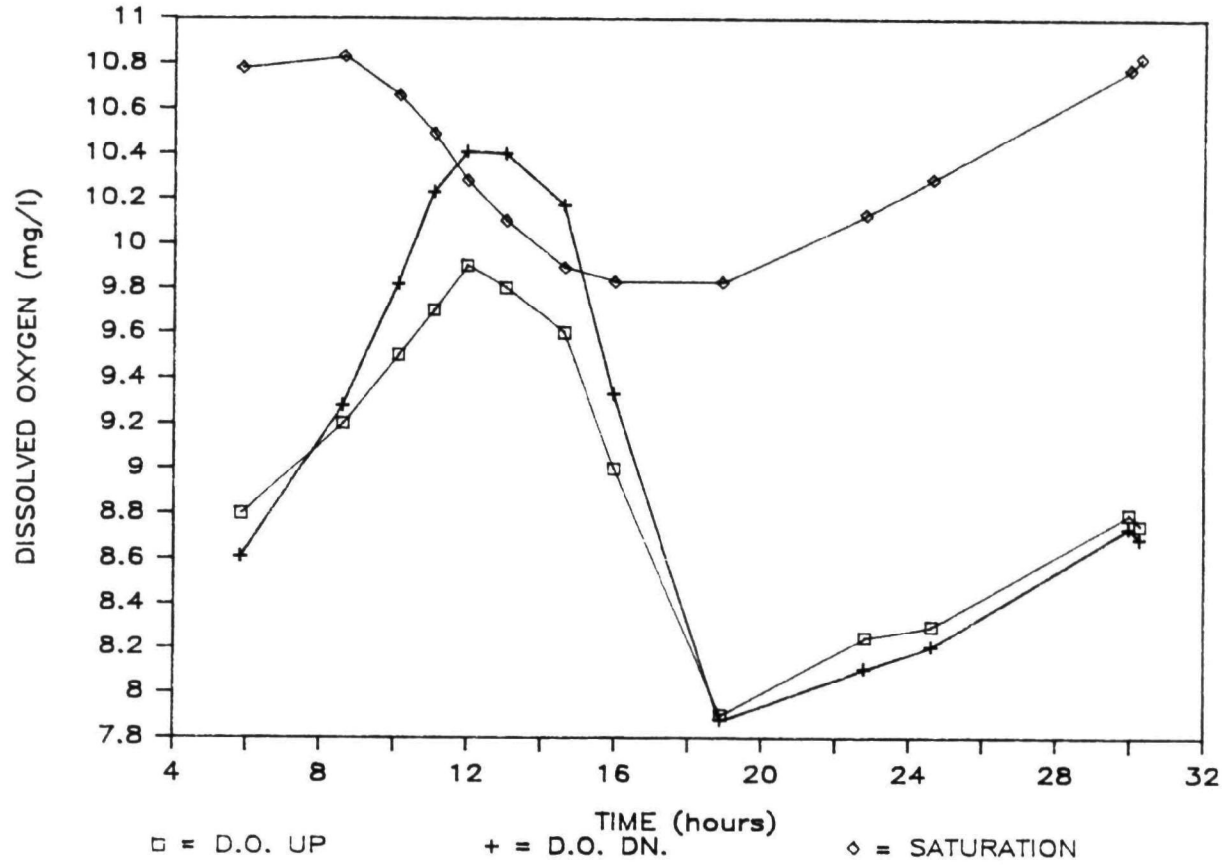
If the R/K2 ratio was 5, (all other parameters as in the initial case) $(Cs-Co)_{sr}$ = 2(.08) + 5(.92) = .16 + 4.6 = 4.76 mg/l (95% of R/K_2).

If the temperature at sunrise was 20°C, Cs would be 9.07 mg/l, and of the above examples, only the third would result in a violation of the 5 mg/l criteria. It is clear from these examples that "R/K2" is very important in determining the maximum deficit, and that according to the model, where K2 is large enough (i.e. $K_2 > 5$), R/K2 essentially specifies the maximum deficit, regardless of the concentration at sunset.

Since a drop in temperature will increase the saturation oxygen level, Cs, normal nighttime cooling will increase the rate at which the maximum deficit R/K2 is approached. Once a deficit of R/K2 is reached, any further drop in stream temperature will cause the dissolved oxygen level, Co, to rise as the stream maintains the equilibrium deficit of R/K2. This is illustrated in Figure 13 which shows the diel curves for 18 August 1981 in the Kohlsville River. Due to a very high reaeration rate (15/day), the equilibrium level is reached almost immediately after sunset. The DO concentration rises throughout the night because reaeration and respiration maintain their "equilibrium" (i.e. (Cs-Co)= R/K2), and temperature decreases cause the saturation concentration to rise. (A decrease of 3°C will increase Cs approximately 0.5 mg/l. If this change is "added" to the above examples (case 1 - case 3), all of them would have reached

Figure 13

Attainment of equilibrium DO deficit (R/K2) and the effects of changing DO saturation in the Kohlsville River (August 18 1981).



"equilibrium". There is some question as to when (i.e. under what conditions) this "adding" treatment is valid, however.

MACROPHYTES AND REAERATION

A potential for change in K2 because of increased macrophyte growth must also be dealt with. Increases in macrophyte density will cause an increase in the drag felt by the water as it flows over the stream bed. Such an increase in drag will cause greater depths (i.e. ponding) of water for equal flow when weeds are present. Mathematically this can be expressed in the Manning flow formula as an increase in the roughness coefficient, "n".

The Manning formula is...

$$Q = 1.486/n \times A \times (R^{2/3}) \times (S^{1/2})$$

By dividing both sides by cross-sectional area (A), we can obtain...

$$V = 1.486/n \times (R^{2/3}) \times (S^{1/2})$$

Where... Q = discharge, in cubic feet per second (cfs)
A = cross sectional area, in square feet
R = hydraulic radius (A/wetted perimeter)
S = slope of the water surface
n = Mannings "roughness coefficient"
V = mean velocity (time of travel)

In the first equation, we can see that if Q is held constant, and "n" is increased, the product [A x ($R^{2/3}$) x ($S^{1/2}$)] must also increase. At least part of this increase could be expected to translate into an increase in depth.

Similarly, in the second equation, an increase in "n" would result in a proportional decrease in mean velocity if "R" and "S" remain relatively constant.

Table 21 gives values of Mannings' "n" for various substrate types (from Corbett 1945). It is important to note that the highest values are associated with "very weedy reaches".

Table 22 shows values of stream discharge (Q), mean velocity (V), mean depth (D), and macrophyte biomass for the Lurvey site (impacted) and the Ashippun site (non-impacted) for the months of June, July and August.

The mean velocity and mean depth change substantially at the Lurvey site (impacted), but not the Ashippun site. If changes in mean velocity and depth due to macrophyte biomass: "n" relationships are to be quantified, factors such as condition (shape?) of the stream bank, character of the stream bed, and slope of the channel would also have to be taken into account.

In the present example, however, we can approximate the effect of ponding due to macrophytes by examining a few equations which were developed to predict K2 from mean depth (D) and velocity (V). The following three equations ranked highest among those equations which used mean depth and velocity to predict K2 (Grant and Skavroneck 1980). All three equations predict K2/day at 25°C.

Padden-Gloyna (1971)*
$$K_2 = 7.73\ (V^{.703}) \times (D^{-1.054})$$

Bansal (1973)*
$$K_2 = 5.26\ (V^{.6}) \times (D^{-1.4})$$

Negulescu-Rojanski (1969)*
$$K_2 = 12.29\ (V/D)^{.85}$$

*from Grant & Skavroneck, 1980

Table 23 shows the K_2 values calculated from the above equations for each month and each stream. The decrease in K_2 from July to August at the Lurvey site is apparently due to macrophyte growth.

Values of Mannings' "n" taken from tables are not exact, and the impact of weed growth upon mean depth and velocity would be even less exact. The various equations

Table 21

Approximate Values of Manning's Roughness Coefficient, "n" (from Corbett 1945).

Channel Description	Channel Conditions			
	Perfect	Good	Fair	Poor
1. Clean, straight bank, full stage, no rifts or deep pools	0.025	0.030	0.035	0.040
2. Same as (1), but with some weeds and stones	0.030	0.033	0.035	0.040
3. Winding, some pools and shoals, clean	0.035	0.040	0.045	0.050
4. Same as (3), lower stages, more ineffective slopes and sections	0.040	0.045	0.050	0.055
5. Same as (3), some weeds and stones	0.033	0.035	0.040	0.045
6. Same as (4), stony sections	0.045	0.050	0.055	0.060
7. Sluggish river reaches, rather weedy or with very deep pools	0.050	0.060	0.070	0.080
8. Very weedy reaches	0.075	0.100	0.125	0.150

Table 22

Discharge (Q in cfs), Velocity (V in ft/sec), Depth (D in ft) and Macrophyte Biomass (in gm/m^2 DWT) for POTW-Impacted (Lurvey) and Non-Impacted (Ashippun) Sites

	Lurvey				Ashippun			
	Q	V	D	Biomass	Q	V	D	Biomass
June, 1982	36	1.2	1.2	75	16.5	.86	1.1	30
July, 1982	28	.71	1.3	151	11.7	.72	1.2	109
August, 1982	29	.38	2.03	289	11.2	.72	1.3	125

Table 23

K_2 Values (/day) for June-August at POTW-Impacted (Lurvey) and Non-Impacted (Ashippun) Sites

	Q (cfs)	Biomass (gm/m^2)	Padden Gloyna	Bansal	Negulescu-Rojanski
LURVEY					
June, 1982	36	75	7.25	4.54	12.29
July, 1982	28	151	4.61	2.97	7.34
August, 1982	29	289	1.86	1.09	2.95
ASHIPPUN					
June, 1982	16	30	6.28	4.20	9.97
July, 1982	12	109	5.06	3.35	7.96
August, 1982	11	125	4.65	2.99	7.43

which predict K2 from mean depth and velocity do not show good agreement in many instances. All of this seems to indicate that an attempt to model this process at this time would be frustrating. Although a study specifically designed to relate K_2 and biomass at a given location could yield quantitative results, the results would be applicable only to the study site, and prove no more useful than the examples presented above. From the present study, the only conclusion to be drawn is that for a given stream, a large increase in biomass could lead to a substantial decrease in K2.

The change in depth could also be expected to lead to a decrease in the rates of respiration and photosynthesis (mg/l/hr). If the depth doubles, the volume of water over a square meter would also be expected to double. This would lead to a sharp decrease in the volumetric (mgO2/l/hr) metabolic rates ascribable to the biomass on the square meter.

The decrease in respiration and the decrease in reaeration in this scenario could be off-setting. The actual increase in R/K2 predicted by assuming that all R is allocatable on an areal basis (no significant BOD or plankton populations), and the change in reaeration is entirely due to the change in depth, varies for the equation used to predict K2. For the Padden-Gloyna equation, the "new" R/K_2 would be four percent higher (new R = .5 times old R, and new K2 = old K2 times $2^{-1.054}$). Similar calculations with the Bansal equation would indicate a 32% increase. Decreases in mean velocity would be certain to accompany increases in depth, so these estimates must be considered conservative.

DEVIATIONS FROM MODELING ASSUMPTIONS

There are cases where K2 doesn't seem large enough to explain the apparent attainment of equilibrium. Among examples are cases where the K2 approximated from the diel data is significantly larger than the K2 predicted by various equations. One possible explanation for this situation lies in the failure of the assumption that respiration remains constant over the course of the day. In all likelihood, respiration will at some point become limited by the availability of oxygen, or even (potentially) be proportional to oxygen availability throughout the day (see Box/Bottle study section).

The question of how much the respiration rate can vary is at least partially answered by the dark bottle experiments. In the range of 5 to 9 mg/l DO, a value of 1.5 mg O2/g DW/hr seems valid. Below 5 mg/l, the rate appears lower. Since we are concerned with keeping the DO levels above 5 mg/l in the stream, it seems prudent to choose the value applicable to DO levels above 5 mg/l.

The variation of respiration with temperature has been explored in some detail (Lassiter 1975, Canale et al. 1982). The general relationship has been expressed as...

$$R_2 = R_1 \times \theta^{T_2-T_1}$$

Where... R_2 = respiration at T2°C
R_1 = respiration at T1°C
θ = a constant which is specific for a specified process (in this case respiration) over a given temperature range.

If θ is assumed to be 1.07, a three degree drop in temperature would lead to a nineteen percent decrease in the R value. This would lead to a nineteen percent decrease in the R/K2 ratio.

For those who prefer to use Q_{10}, a Q_{10} of "2.0" corresponds to θ = 1.07) where

$$R_2 = R_1 \times Q_{10}^{(T_2 - T_1)/10}$$

The temperature variation coefficient (θ or Q10) is not constant for all organisms or metabolic rates (P or R). Different organisms may have optimum rates at different temperatures. For this reason,

"community R", the term which is actually approximated in diel surveys, is likely to vary less than a species specific θ would predict (Odum 1973, McDonnell 1982).

If the R-value from the diel curve analysis is an "average" for a period where a substantial amount of time is spent in the 5-10 mg/l DO range, the use of 1.5 g O2/kg DW/hr rate seems reasonable. If the average night-time DO is substantially below 5 mg/l, the application of a lower rate would be defensible. The ultimate solution of course, would be to alter the diel equations so that they would produce temperature and DO sensitive coefficients. Construction and validation of such a model, however, would be a major project.

The reaeration coefficient K2 is presumed constant as well, but has been shown to vary with temperature according to the equation:

$$K2_{(25°C)} = K2_{(T)}\ 1.024^{(25-T)}$$

According to the equation, a temperature decrease of 3°C (e.g. 23°C to 20°C) would lead to a seven percent decrease in the K2 value, as follows:

$K2_{(25°)} = 5$

$K2_{(23°)} = (5)\ 1.024^{(-2)} = 4.77$

$K2_{(22°)} = (5)\ 1.024^{(-3)} = 4.44$

% change = .33/4.77 = .07 x 100% = 7%

This would cause a slight (+7%) increase in the R/K2 ratio. In the night period, the decrease in R/K2 caused by decreased respiration and the increase in R/K2 caused by decreased K2 are thus opposed and potentially compensatory. The decrease in R/K2 due to decreased respiration is likely to be more important. (See Box/Bottle Study section.)

FURTHER CONSIDERATIONS IMPORTANT TO THE MODELING PROCESS

The question of whether or not respiration and/or K2 remain constant with decreasing temperature and DO levels does not seriously alter the discussion regarding R/K2 and the maximum deficit for at least two reasons. First, a change in the biomass should provide an incremental change in respiration, and second, the respiration rate which results from the diel curve analysis is likely to be an "average" for the site, i.e. it will over-estimate the R value applicable to the period during which DO is lowest, and because of this be more typical of higher DO levels. Thus, although the magnitude of projected changes is not likely to be predicted with absolute accuracy, there is a good degree of certainty that changes within a certain range will occur.

The area (or time of travel) downstream of a point source to which the modeling process should be applied is another consideration. Attempts to document a "phosphorus decay curve" in this study met with limited success. A decrease in phosphorus concentration over the study area was obvious on some days, and lacking on others.

When attempting to specify a region of impact, an important distinction must be made between the true decay observed for non-conservative pollutants, and the decrease in concentration ("decay") observed for nutrients such as phosphorus. CBOD may be considered a non-conservative pollutant. It is oxidized to water and carbon dioxide, and lost from the system. Phosphorus may show an initial "decay" due to uptake by plants and physical adsorption, but it is not eliminated from the system. It may be released at the end of the growing season, re-dissolved through grazing, resuspended during storm flows, etc. While the present study does not rigorously define the area impacted, it does suggest that the area is well in excess of several stream miles.

The actual area included in a model of diel oxygen curves should include the area which is impacted, which must be decided on a case-by-case basis. If phosphorus levels at the lower end of the modeled area approached those encountered upstream of the source, the model could be considered to have accounted for most of the impact.

This will not always be practical, but should be a desired goal. If no decay is obvious, or other sources or factors (tributaries, impoundments, etc.) become important enough to mask the impact, the modeled area will have to be defined and judged on the basis of preliminary studies.

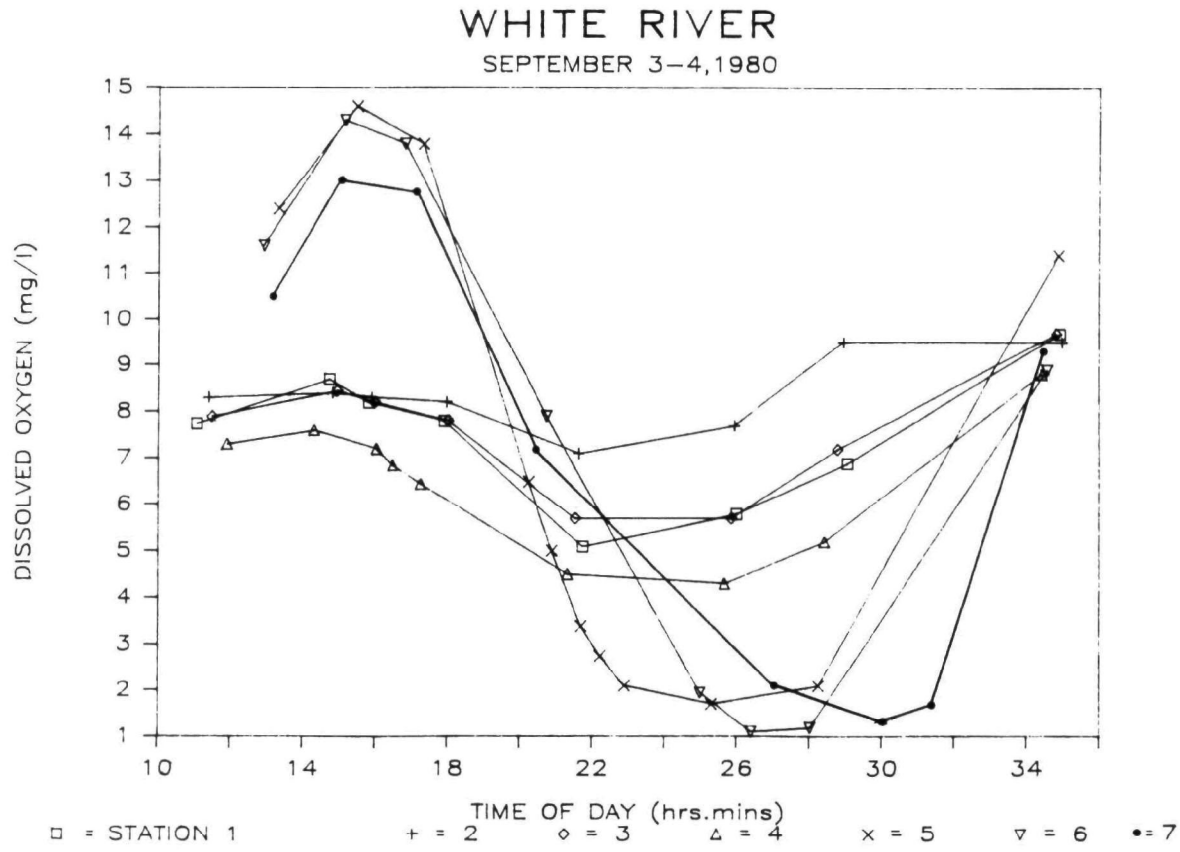
The impact of BOD and the impact of phosphorus are separate entities. BOD may be responsible for depleted oxygen levels at one point, and phosphorus at another. A good example of this occurs on the White River below the Lake Geneva POTW. The BOD sag occurs within the modeled area, but oxygen depletion due to macrophytes is obvious at the last station (Figure 14).

RESULTS AND IMPLICATIONS OF DIEL ANALYSES

The results of the double-station and single-station analyses are supportive of the dissolved oxygen model and the assumptions under which it was developed. Plots of R/K2 versus the maximum deficit (Figures 15 and 16) show good agreement with theoretical considerations. It should be pointed out that the values returned from the diel curve analyses are "first

Figure 14

Illustration of BOD oxygen "sag" and "sag" attributable to plant growth in the White River



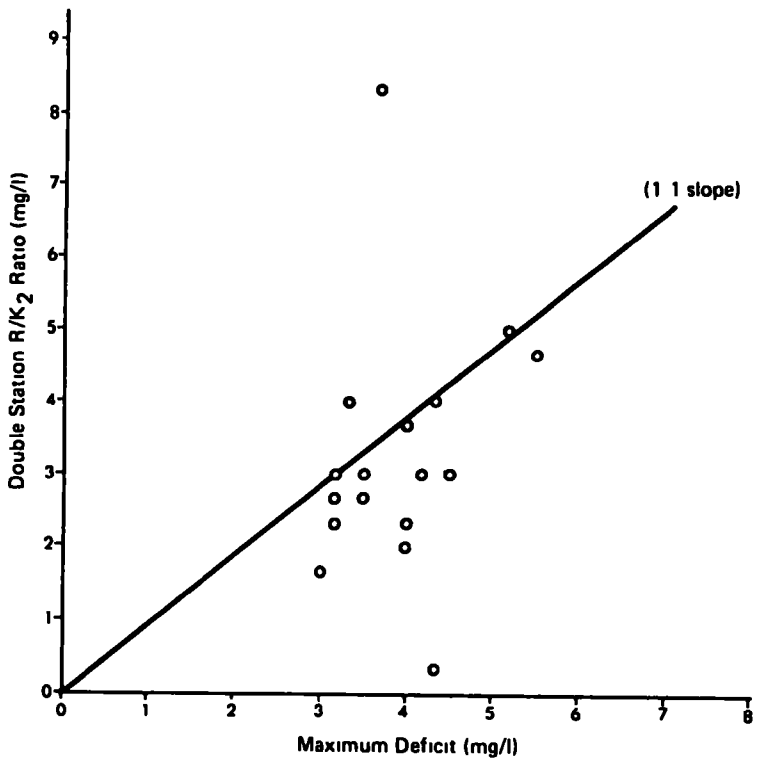


Figure 15

Theoretical maximum deficit (R/K2) and observed maximum deficit for the double-station method.

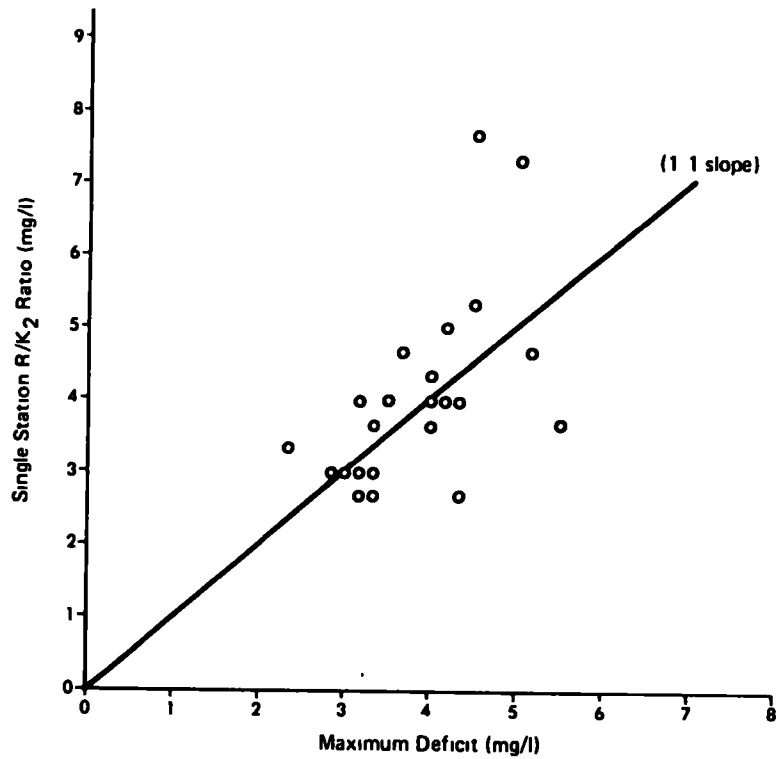


Figure 16

Theoretical maximum deficit (R/K2) and observed maximum deficit for the single-station method.

approximations". In a true modeling setting, the coefficients are subject to adjustment in validation and verification procedures. Such adjustments would be expected to improve the relationship between R/K2 and the maximum deficit.

If the estimates of respiration from diel curve analysis and the respiration expected on the basis of 1.5 g O2/kg DW/hr are compared (Figures 17 and 18), it is obvious that the diel estimates are much higher. This suggests that other factors (BOD, SOD, invertebrates, periphyton, etc.) are important when calculating the total community respiration estimate.

Graphs of gross photosynthesis versus biomass are presented in Figures 19 and 20. Double-station estimates are generally much higher than single-station estimates. The 1982 double-station estimates, which were derived from larger reaches with longer travel times, are generally lower than the 1981 estimates when corrected for volume (Figure 20). The statistical fits of the 1982 data sets were also improved. The 1982 double-station data and both year's single-station data are below the estimates of photosynthesis derived from Westlake's (1966) biomass approximations. The photosynthesis approximations from the box studies are in better agreement with the diel estimates, but the scatter of the diel estimates still precludes serious modeling effort.

Table 24

Diel Modeling Coefficients for Single and Double-Station Analyses

SINGLE STATION (INTEGRAL) COEFFICIENTS

Data Set	K2 (1/day)	P (mg/l/day)	R (mg/l/day)
1	1.0±0.5	10.9	13.3
7	0.9±0.8	9.2	13.3
9	0.8±0.7	7.0	11.3
13	1.4±1.0	10.7	19.5
16	2.2±1.5	19.4	29.7
18	3.9±2.5	1.3	10.3

Table 24 (con't)

DOUBLE STATION INTEGRAL COEFFICIENTS

Data Set	K2 (1/day)	P (mg/l/day)	R (mg/l/day)
1	2.6±0.4	9.9	14.9
2	2.8±0.3	6.7	17.8
3	2.7	8.2	16.5
4	5.7	12.0	24.4
5	9.3±3.8	19.4	35.9
6	**	**	**
7	3.2	6.9	20.6
8	2.6±0.5	8.5	17.0
9	1.7±0.8	7.5	10.8
10	3.7±0.7	10.3	17.1
11	6.3±1.3	13.3	24.7
12	7.6±1.1	12.2	26.7
13	4.3±1.0	10.7	19.5
14	0.6±1.2	8.8	10.1
15	6.3±1.0	12.4	23.9
16	5.9±2.9	15.7	22.1
17	7.2±2.7	13.9	24.1
18	7.3	10.7	20.8

DOUBLE STATION DIFFERENTIAL COEFFICIENTS

Data Set	K2 (1/day)	P (mg/l/day)	R (mg/l/day)
1	2.7±0.5	10.6	15.0
2	2.8±0.3	9.4	17.9
3	2.7±0.2	10.2	16.5
4	4.8±0.7	11.2	21.3
5	6.0±1.1	13.0	24.1
6	6.5±2.4	12.7	22.7
7	3.2±0.4	8.3	21.0
8	2.1±0.4	8.7	14.5
9	1.5±0.7	9.6	10.6
10	3.4±0.7	10.7	16.8
11	5.4±1.0	13.9	22.7
12	6.2±0.7	12.6	23.2
13	4.1±1.0	11.4	19.4
14	4.8±1.2	15.0	21.9
15	5.7±0.7	13.9	22.5
16	4.4±1.7	16.4	21.1
17	5.7±1.0	14.4	22.0
18	6.6±1.4	11.3	21.3

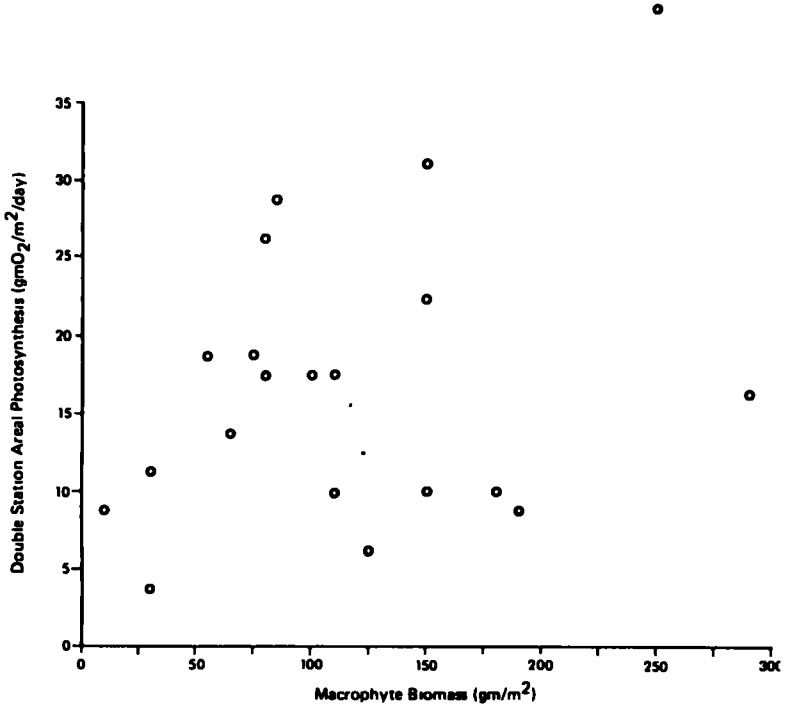


Figure 17

Double-station respiration estimates from diel and plant biomass data.

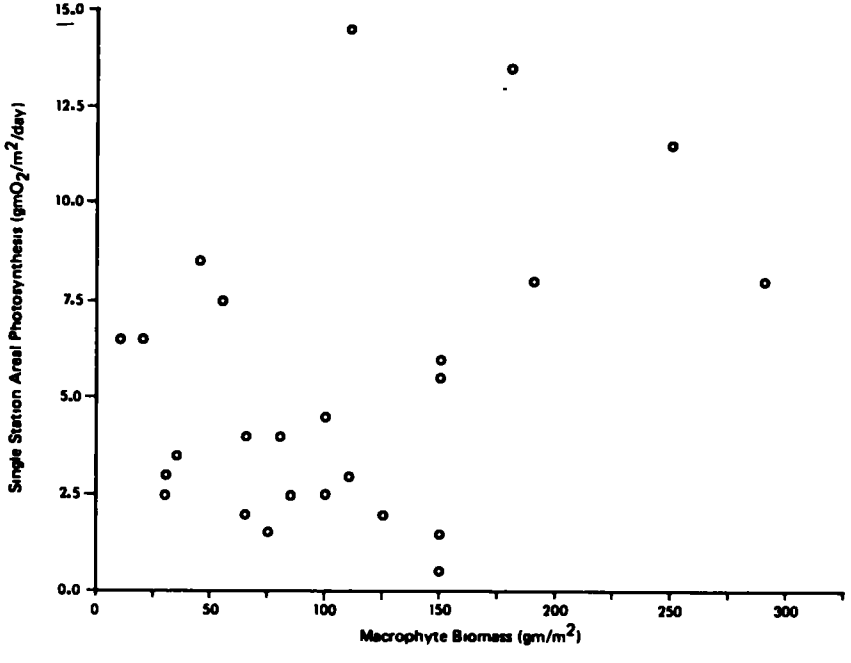


Figure 18

Single-station respiration estimates from diel and plant biomass data.

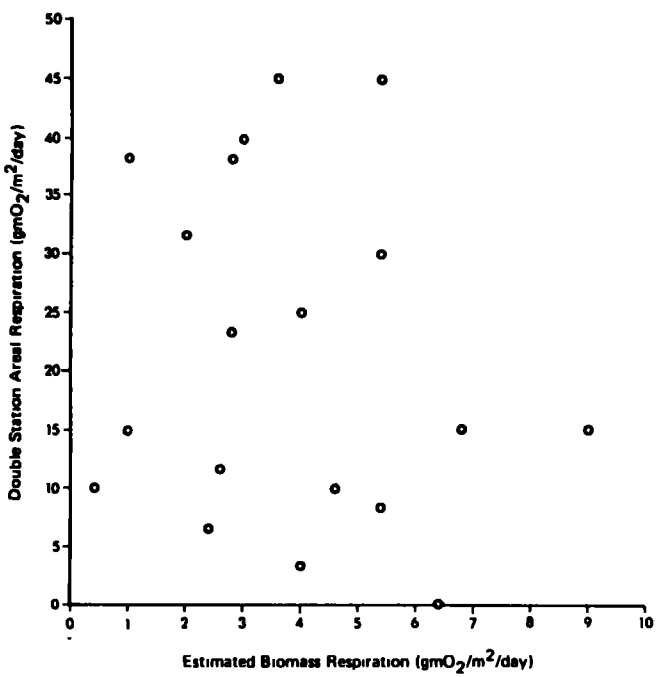


Figure 19

Double-station areal photosynthesis estimates from diel and plant biomass data.

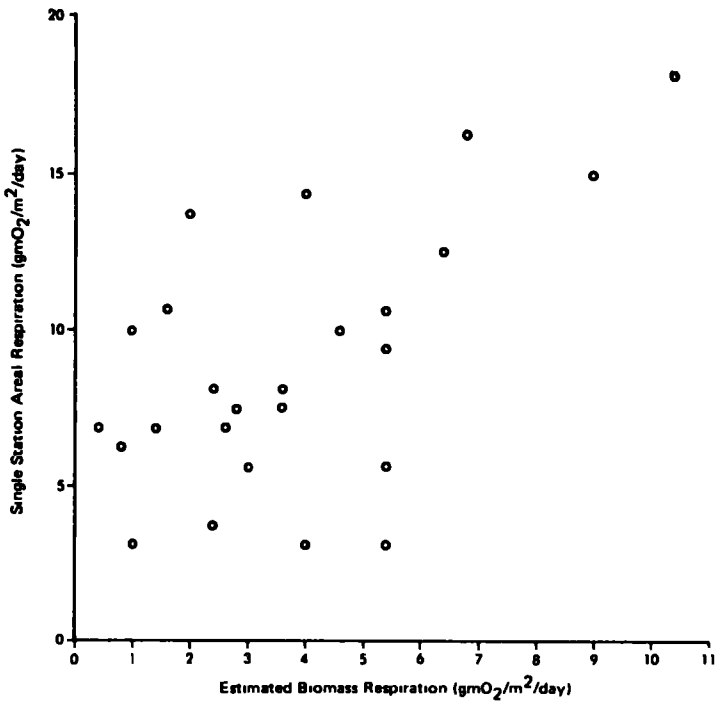


Figure 20

Single-station areal photosynthesis estimates from diel and plant biomass data.

Comparison of Double and Single Station Data

A demonstration wasteload allocation survey was conducted on the Bark River on 30 August-2 September, 1982. The intent was to gather data which would allow comparison of the single and double-station techniques, and hopefully corroborate some of the assumptions of the model.

Seven diel stations were established on the Bark River, downstream of the Dela-Hart POTW outfall. The locations of these stations, in miles downstream of the outfall, were 0.0, 0.3, 0.6, 0.85, 1.15, 1.42 and 1.75. Dissolved oxygen (DO) was monitored at all stations for about 48 hours. Light was monitored at the 0.85 mile station. Temperature was monitored at the 0.3, 1.15, 1.42 and 1.75 mile stations. The last three stations (1.15, 1.42 and 1.75) were sampled for dissolved oxygen and temperature at two to three hour intervals. All other data were continuously recorded. An interpolation routine (AISPIN/AISPEV, available at the Madison Academic Computing Center) was used to generate "continuous" data for the last three stations.

Three separate analyses of the data set were performed (Table 24):

1. The single-station analysis was performed on each of the seven diel curves, using the integrated equation.

2. Double-station analysis was performed on all possible pairs of diel curves (0.0-0.3, 0.0-0.6, 0.0-0.85, etc.), using the differential equation.

3. Double-station analysis was repeated, using the integrated equation.

The resultant coefficients, and the "fit" of the individual determinations, can be used to test some of the assumptions of diel curve analysis, and answer some questions which pertain to the choice of a modeling method. The following section shall examine the following questions:

- What upstream area does the single-station analysis represent?

- Do the differential and integrated equations return the same coefficients when "short" reaches are analyzed?

- How long must a reach be before the differential equation fails, and when is it necessary to employ the integrated equation?

- How does increasing reach length impact the accuracy of the double-station technique?

Area Represented By Single Station Coefficients

The relative positions of the DO stations for a wasteload allocation survey on the Bark R. (in miles downstream of the outfall), the coefficients returned by the single and double-station analyses, and the time of travel between stations are shown in Table 25. The P, R and K2 values are arranged to correspond with the area or position from which they were derived. The double-station values listed between the 0.0 and 0.3 mile stations were derived for the reach between these two points. Single-station values are adjacent to the station from which they were derived, and represent averages applicable to some area upstream of the station for which they were derived.

As might be expected, the double-station and single-station values do not seem to closely agree. The determination of P, R and K2 for longer reaches (e.g. the 0.0 to 0.85 or 0.0 to 1.15 reaches) failed to yield sufficient data with which to determine and validate the area represented by the single-station analyses.

If we assume an "average K2" for the 0.0 to 1.85 mile reach of 2/day, and use the observed travel time of 7.2 hours (0.30 days), we can see that the incoming deficit at the 0.0 mile station still has a substantial impact on the deficit that arrives (7.2 hours later) at the 1.85 mile station.

Table 25

Single and Double-Station Coefficients for the Bark River Wasteload Allocation Survey

Single-Station K_2	P	R	Stream Miles (TOT in hours)	Double-Station K_2	P	R
1.8	11.3	14.5	0.0			
			(1.65)	2.7	10.6	15.0
1.6	10.0	14.8	0.3			
			(1.27)	3.2	8.3	21.0
1.8	7.2	12.9	0.6			
			(1.10)	1.5	9.6	10.6
2.7	8.6	12.7	0.85			
			(1.16)	4.1	11.4	19.4
2.7	7.7	11.4	1.15			
			(1.07)	4.4	16.4	21.1
4.5	9.2	15.4	1.42			
			(0.97)	6.6	11.3	21.3
4.8	11.1	17.4	1.85			

The calculations are as follows:

$$(Cs - Co)_{1.85} = (Cs - Co)_{0.0}\, e^{-K2\,\Delta t} + ((\alpha P + R)_{/K2})(1 - e^{-K2\,\Delta t})$$
$$= (Cs - Co)_{0.0}\, e^{-0.6} + ((\alpha P + R)_{/2})(1 - e^{-0.6})$$
$$= (Cs - Cs)_{0.0}\,(.55) + ((\alpha P + R)_{/2})(.45)$$

P and R values are in mgO2/liter/day. K_2 values in 1/day.

If we assume a larger K2 value (4/day), the importance of the incoming deficit diminishes, but is still significant.

$$(Cs - Co)_{1.85} = (Cs - Co)_{0.0}\,(.30) + ((\alpha P + R)_{/4})(.70)$$

From this perspective it would appear that the area represented by the 1.85 mile single-station analysis is in excess of 1.85 miles. These calculations are similar to assuming that an "average P, R and K2" can be uniformly applied to the area upstream of the 0.0 mile station and would produce an "average (Cs-Co) ", and that another set of "average P, R and K2" values uniformly applies to the 0.0-1.85 reach, and should produce an "average (Cs-Co)".

This is a "steady-state" viewpoint. Under these conditions, little or no variation in DO occurs over time at a given station. Under these conditions, the time of travel (or distance) which must elapse before the "new" P, R and K2 values are fully expressed is primarily a function of the new K2 value. In reality, the incoming deficit is not constant, and in this instance fluctuates around "0". It is clear that the magnitude of both the incoming deficit and the $((\alpha P + R)/K2)$ term will be important. When the incoming deficit is large, its impact may mask the effects of the area immediately upstream. If the incoming deficit is small (<1?), however, it is likely to have a small (relative to the impact of the $((\alpha P + R)/K2)$ term) impact on the deficit at a downstream station.

The term $(\alpha P + R)/K2$ is also presumed constant in this discussion. Like (Cs-Co), however, the value of this term could be expected to fluctuate around zero, and exert a variable impact upon actual deficit.

In a real stream, natural variations in the magnitude of P, R or K2 can be expected to play equally important roles, and complicate matters even more.

Comparison of Mean Single and Double Station Values for the Entire Survey Area

The single and double-station techniques generated two separate estimates of the coefficients which are characteristic of the study reach. A t-test can be used to determine whether the "average" single-station estimate for each of the coefficients is significantly different from the "average" double-station estimate. The probabilities that the "true means" are equivalent are 0.42, 0.11 and 0.09 for K2, P and R respectively. Only the six double-station reaches shown in Table 26 were included in this test.

Casual inspection of Table 25 seems to indicate that both estimates of K2 increase in the downstream direction (with the exception of the 0.6-0.85 mile double-station estimate), but there are no apparent patterns in the variations of P or R.

Table 26

Mean Modeling Coefficients and their Standard Deviations for the Entire Survey Area

Coefficient	Single-Station Values			Double-Station Values		
	K2	P	R	K2	P	R
Mean	2.83	9.3	14.2	3.75	11.3	18.1
Standard Deviation	1.30	1.6	2.0	1.74	2.8	4.4
Number of Samples	7	7	7	6	6	6

The above data and discussion suggest that the area represented by the single-station method is smaller than the area (time of travel) required to reduce the incoming deficit to a small quantity. If a study of this nature was conducted where the reaeration rate was higher, a better definition of the area represented by the single-station method might be possible. A computer simulation of the effects of changing values of P, R and K2 would also be useful.

Double Station Coefficients and "Average Double Station Coefficients"

Figures 21 through 23 represent the comparison of calculated coefficients and coefficients determined by averaging the coefficients returned for the shortest reaches. This exercise was carried out to demonstrate the failure of the "double station differential" method when reach length, or the product "K2TOT" becomes too large. The plots demonstrate that as TOT increases, the estimates of R and K2 fall further from the theoretical 1:1 line which would be expected if no failure occurred. The estimates of P seem relatively unaffected by reach length. R and K2 coefficients returned from the longest reaches are much higher than would be expected from the averages of the short reaches. Even the shortest "averages", i.e. those returned from the average of two one hour reaches, lie above the 1:1 line that would be expected if only random errors in the determinations were the cause. This suggests that a systematic error is involved, and that the error increases with increasing reach length.

A slightly anomalous point is that the "statistical fit" of the determinations generally improves with increases in travel time. The above diagrams and figures indicate that coefficients derived from the long travel times are incorrect. This suggests that "statistical fit" (e.g. large r-square values and small confidence limits) is not necessarily a good indicator of "correctness".

Integrated and Differential Coefficients for the Double Station Technique

The integrated equation was used to calculate a second set of double-station coefficients for the WLA data set. The resultant coefficients are compared with those returned by the differential equation in Figures 24 through 26. A t-test was performed to test whether or not the two methods (differential and integrated) returned mean values for P, R, and K2 which were significantly different. The mean values, their standard deviations and the probability that the mean values are statistically equal are presented in Table 27. (A value less than 0.05 is commonly interpreted to indicate statistical inequality.)

The results of these tests indicated that the "whole-reach" mean values returned by the two analytical methods (differential and integral) were not significantly different at the five percent level.

A paired t-test was also performed to determine whether the reach-by reach differences between analytical methods were significant. The results indicated that P,

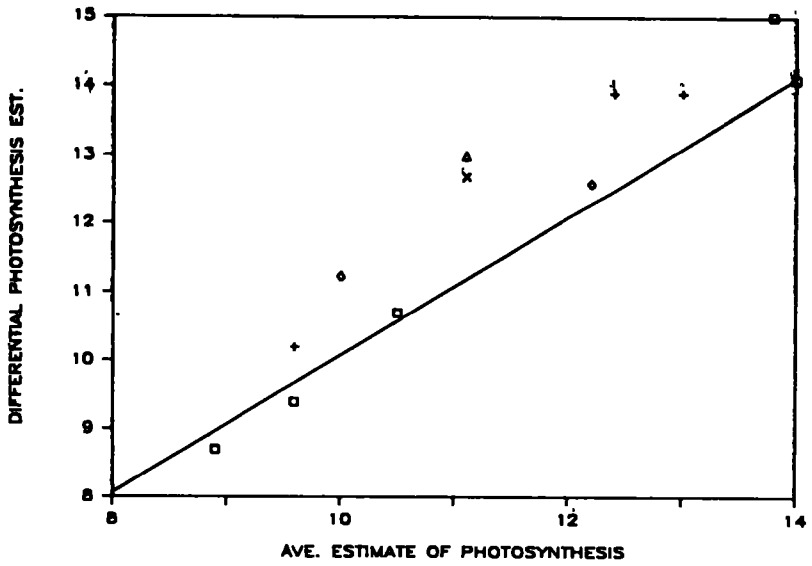


Figure 21

Average differential photosynthesis coefficients versus measured differential photosynthesis coefficients (gm $O_2/m^2/day$). The line drawn represents a "perfect" relationship (slope = 1, intercept = 0).

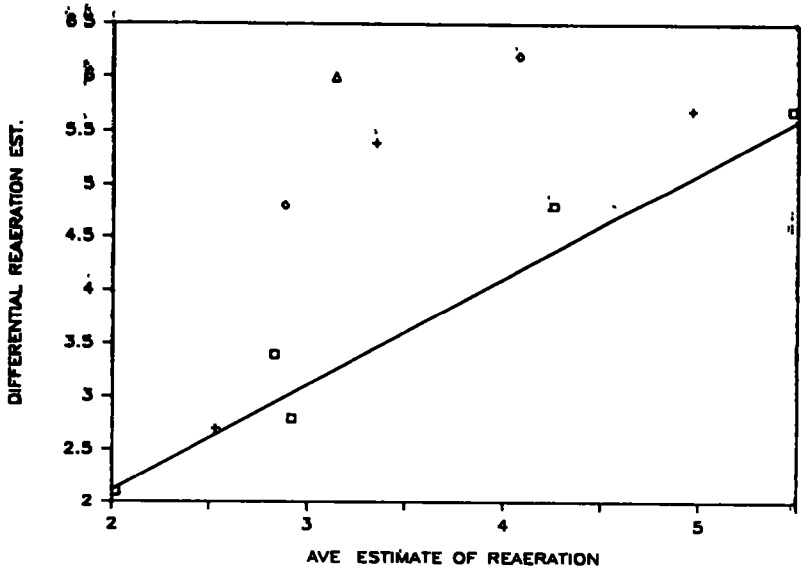


Figure 22

Average differential K2 versus measured differential K2 coefficients (gm $O_2/m^2/day-1$). The line drawn represents a "perfect" relationship.

Note: Symbols represent different travel times:
□ = 2-3 hours, + = 3-4 hours, ◇ = 4.5 hours, △ = 6 hours, x = 7.2 hours

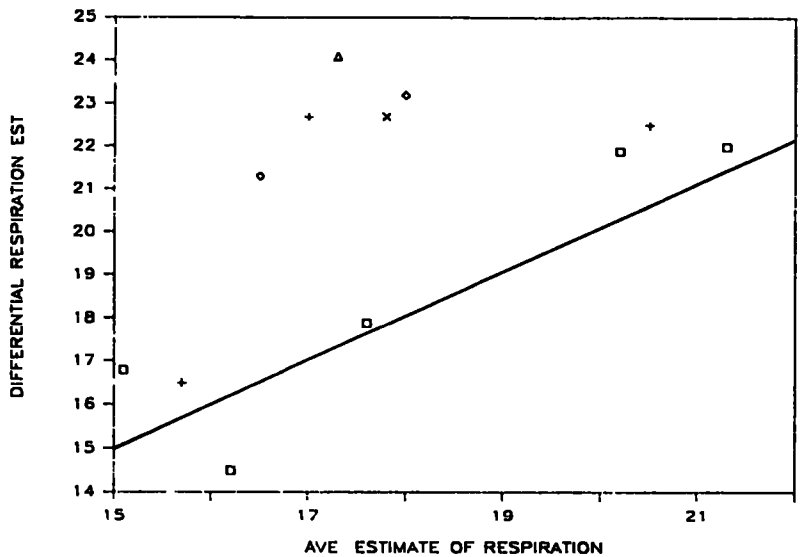


Figure 23

Average differential respiration versus measured differential respiration coefficients (gm O_2/m^2/day). The line represents a "perfect" relationship.

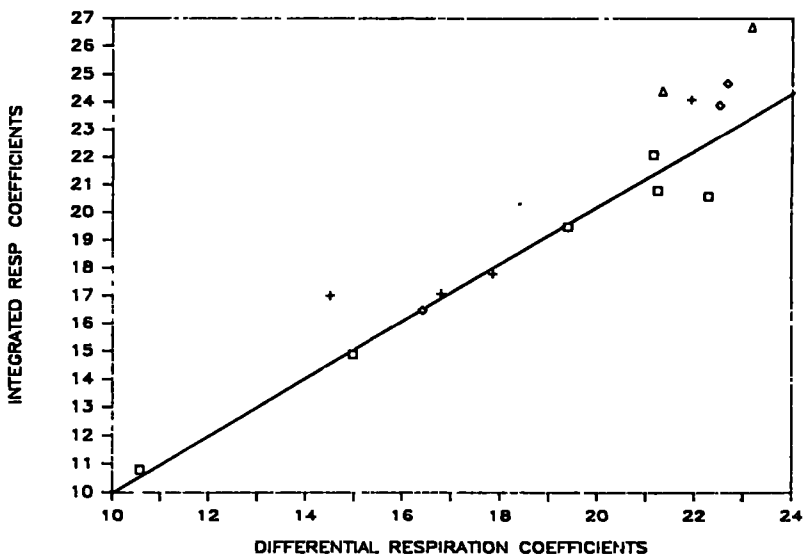


Figure 24

Comparison of differential and integrated double-station coefficients for respiration (gm O_2/m^2/day). The line represents a "perfect" relationship.

Note: Symbols represent different travel times:
□ = 2-3 hours, + = 3-4 hours, ◇ = 4.5 hours, △ = 6 hours, x = 7.2 hours

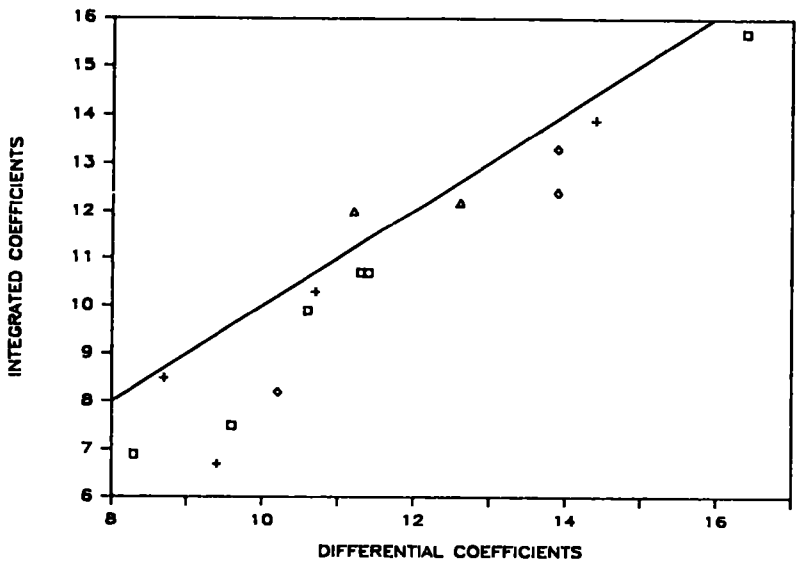


Figure 25

Comparison of differential and integrated double-station coefficients for photosynthesis (gm O_2/m^2/day). The line represents a "perfect" relationship.

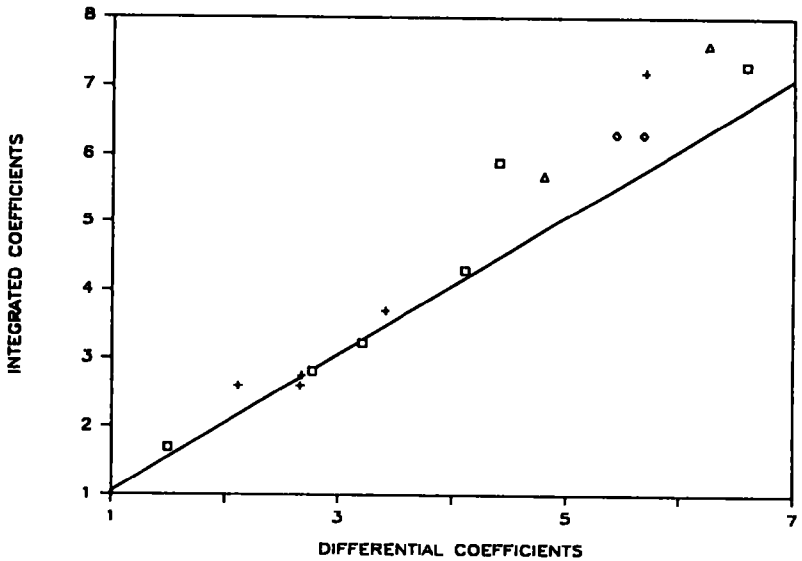


Figure 26

Comparison of differential and integrated double-station coefficients for reaeration (1/day). The line represents a "perfect" relationship.

Note: Symbols represent different travel times:
□ = 2-3 hours, + = 3-4 hours, ◇ = 4.5 hours, △ = 6 hours, x = 7.2 hours

Table 27

Comparison of Mean Double-Station Coefficients

Coefficient	Double-Station Coefficients: Integrated			Differential			
	Mean	S.D.	N	Mean	S.D.	N	$P(\mu_1=\mu_2)$
P (mg/l/day)	10.59	2.69	15	11.85	2.28	18	0.39
R (mg/l/day)	20.06	4.38	15	19.75	3.72	18	0.83
K2 (mg/l/day)	4.67	2.02	15	4.37	1.62	18	0.65

R and K2 estimates from the differential method were significantly different from the integral estimates of P, R and K2 (p=.0013,.0010 and .0273 respectively).

Regression analyses were performed to further characterize the relationship between the differential and integral coefficients. Integrated coefficients are designated by the prefix "I" and differential coefficients by the prefix "D". The equations which resulted are presented below. The values in parantheses are the standard deviations of the coefficients.

$$IP = 1.11(DP) - 2.14 \quad n=15$$
$$(\pm 0.10) \qquad (\pm 1.19)$$

$$IR = 1.10(DR) - 1.07 \quad n=15$$
$$(\pm 0.10) \qquad (\pm 2.02)$$

$$IK2 = 1.24(DK_2) = 0.41 \quad n=15$$
$$(\pm 0.07) \qquad (\pm 0.29)$$

If the differential and integral results are truly equivalent, the slope (coefficient of the differential term) should not be significantly different from "1.0". The slopes of the lines relating the two estimates of P and R are not significantly (± 2 standard deviations) different from "1.0". The slope relating IK2 and DK2 appears significantly different from "1.0".

DISCUSSION

The results of the single-station: double-station comparison are in general agreement with theoretical considerations. The single-station and double-station coefficients are comparable when reach length (study area) is large enough, despite relatively poor agreement for short reaches. The P, R, and K2 values which result from the last single-station sample point also appear close to the average double-station values. Despite these findings, it is still difficult to define the area represented by the single-station analysis. The same experimental design, executed where reaeration is higher or reach length is longer (and more DO stations are involved) could reveal the controlling factors. For the present, we can only cite "general agreement" between methods, for long (K2 x TOT $>$ .9) reaches.

The results of the "differential" : "average differential" comparison were mixed. The differential equation failed to return "average coefficients" (for R and K2) for long reaches, as expected. The fact that the differential estimates of P for the longest reaches were not appreciably different from the expected average values indicates that linear processes can be accurately determined in very long reaches. The failure for K2 (and therefore R) is possibly related to inappropriate determination of the "average deficit" to be applied over the time of travel. The results suggest that time of travel for the double-station differential method be limited to one to two hours. It is uncertain as to whether the time limit should be shorter in streams where reaeration is higher.

The integrated and differential comparison showed unexpectedly good agreement between the two methods. The results were similar to the "differential" : "average differential" comparison, with best agreement occurring between differential and integral estimates of P. The estimates of R and K2 from the long reaches seemed to substantially overestimate the true (as indicated from weighted averages of short reaches) values. The integral equation, as implemented, did not alleviate problems associated with long travel times. The reason for this is unclear. Further investigation in this area is warranted.

SUMMARY OF DIEL STUDIES

There are many possible reasons for the scattered diel estimates of P, R and K2. Inaccuracy of data collection techniques could be a major source of error. The double-station technique suffers more from slight inaccuracies in DO measurement than does the single-station technique. Seasonal changes in the photosynthetic efficiency of the plants, self-shading, and the presence or absence of periphyton as other photosynthetic agents could all lead to the observed results.

In addition, it appears that the assumptions upon which the model is based are sometimes violated. Figures 27 and 28 show several examples where light and photosynthesis are apparently not linearly related. These graphs are from the double-station analysis, and represent the change in DO across a reach, corrected for reaeration, at different light levels. It is worth noting that the curves "flatten out" around $30x10^{15}$ Q/cm2/sec, which is near the range of $25-30x10^{15}$ Q/cm2/sec quoted in the literature as being a "typical light saturation level". This non-linearity may or may not lead to an error in the estimate of "P" and "R". (In most cases it does not appear to be a serious error.) It is an inaccuracy in the technique, and deserves attention.

The assumption that "R" remains constant throughout the night is probably also in error, according to the bottle and box studies. Further studies would be needed to define the magnitude of the error, and if indeed it would be possible to correct this error on a routine basis.

Finally, each of these diel surveys covered approximately twenty-four hours. Day to day variations in light levels are likely to add variability to the determinations of "P".

From the results of the simulated wasteload study, it appears that continuously recorded data (light, temperature and DO) collected over a period of about 48 hours provided the best (in a statistical sense) data.

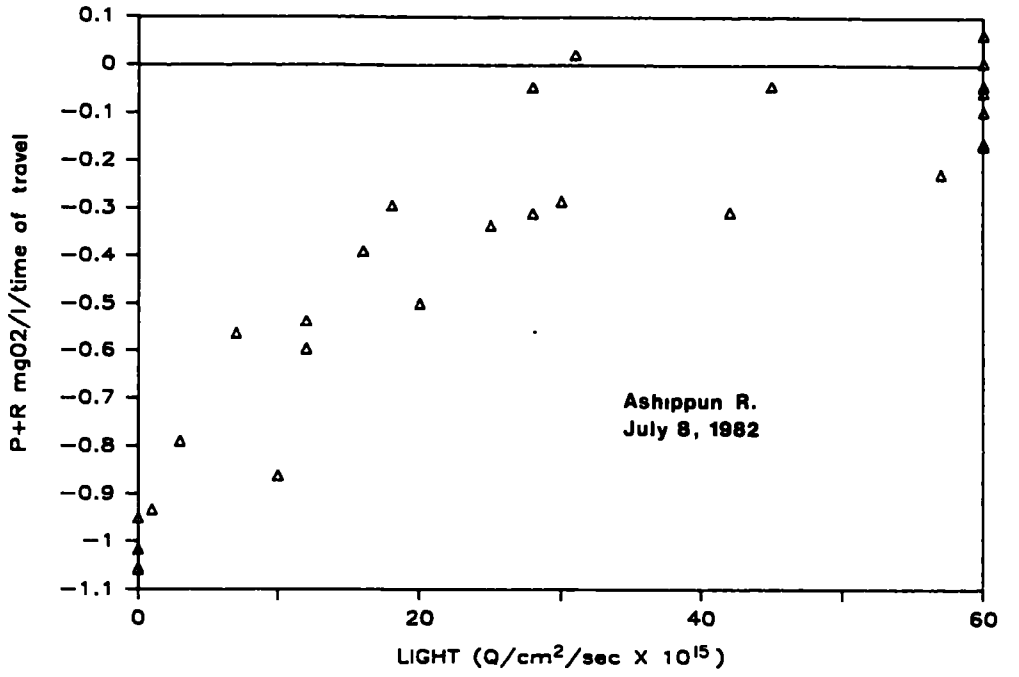
The choice of the modeling method used to determine the K2 and R value for a given stream is complicated by the fact that the exact relation between single and double-station coefficients has not been adequately explored. It is clear, however, that the results of the two methods will differ in most cases, and that they do so because of their different determinations of the $\Delta DO/\Delta t$ term.

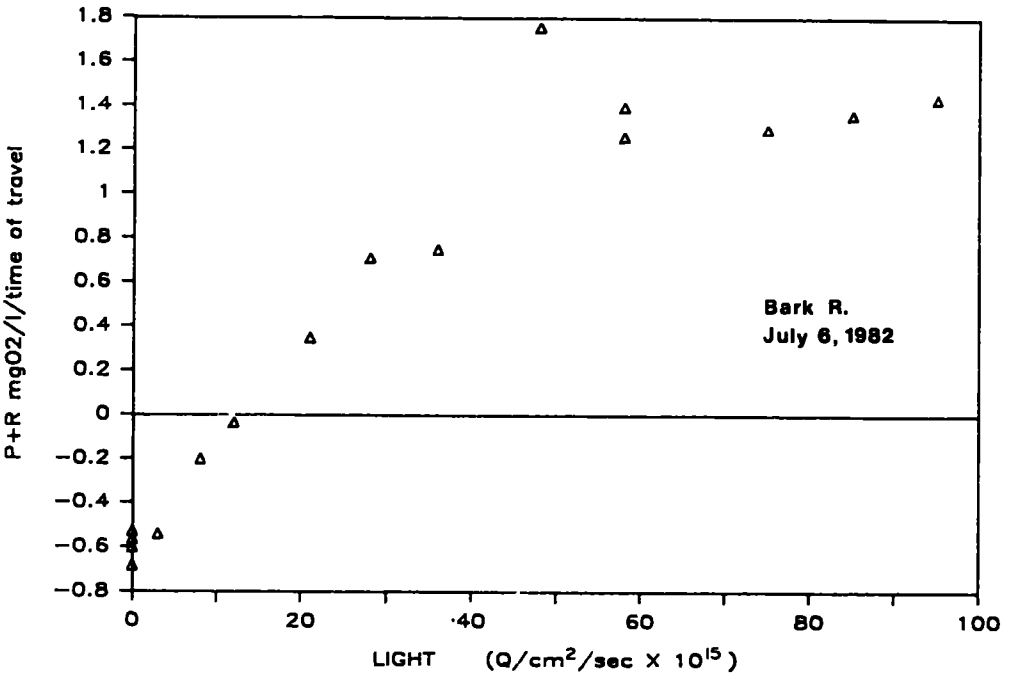
It seems prudent to conduct waste-loads with a design conducive to double-station analysis wherever possible. The small amount of extra effort and expense will provide additional (and possibly better) data. If problems are encountered with the double-station analysis, the data are still available for single-station analyses.

If the differential double-station method is used, care must be taken to choose stations which are relatively close, with less than two hours travel time between them. Travel times shorter than two hours may be advisable where K2 is expected to exceed 5/day. (Further research is needed to determine more exact time of travel restrictions.) When travel time is less than one half hour, very accurate determinations of the $\Delta DO/\Delta t$ term are necessary. Poor resolution of the $\Delta DO/\Delta t$ term may make analysis very difficult and prone to inaccuracy.

Figures 27 and 28

Examples from the Ashippun and Bark rivers demonstrating non-linearity of the relationship between light and photosynthesis (apparent light saturation).





The integrated double-station method, as implemented in this study, remains restricted by the assumption that sum ((∝P+R)/K2) remains constant. It thus seems reasonable to restrict the travel time to less than two hours in this method.

POTENTIAL METHOD OF ALLOCATING PHOSPHORUS

The foregoing data and discussion suggest a simple way in which phosphorus may be incorporated into the wasteload allocation process. Field surveys or existent data can provide estimates of reaeration and respiration. If a change in the amount of phosphorus which will be discharged is estimated, and the upstream phosphorus concentration is known, the macrophyte biomass can be estimated from the models presented earlier in this paper. The change in biomass projected by the model (i.e. "Biomass predicted at present phosphorus level" minus "Biomass predicted at the new (projected) phosphorus level") can be used to estimate a change in the respiration rate.

Experiments presented elsewhere in this paper, as well as in the literature, suggest that macrophytes respire at the rate of 1.5 g O2/kg DW/hr, which is equivalent to 36 g O2/kg DW/day. If the biomass is projected to increase by 100 g/m2, this would increase the areal respiration rate by 3.6 g O2/m2/day, which can be translated into a mg/l/day estimate by multiplying by average depth and dividing by the number of liters present over a square meter area at a depth of one foot (if the average depth was one foot, 3.6 g $O2/^2$/day x 1 ft x 1 m^2/304.8 l = 11.8 mg O2/l/day). The new respiration rate, the "measured respiration rate + projected change", can be divided by the "estimated or measured K2 rate" to estimate what the new maximum deficit will be if K2 is greater than 5/day. If K2 is less than 5/day, the projected change can be added to the respiration term in the model to estimate the maximum deficit.

The advantage of the "R/K2" methodology lies in its ability to predict the maximum deficit regardless of daytime DO fluctuations. Application of the integrated model should yield approximately the same result, yet is much more cumbersome.

In streams where K2 exceeds 5/day, the equilibrium concept will also allow the impact of non-point sources to be quantified, if an increase in phosphorus or sediment oxygen demand can be linked to non-point sources. Difficulties inherent in projecting a new photosynthesis rate make application of the integrated model difficult in situations where K2 is less than 5/day.

This procedure would augment the present BOD allocation process. In conjunction with CBOD and NBOD, some level of phosphorus with attendant increases in plant biomass and respiration would result in violation of the 5 mg/l criteria.

CONCLUSIONS

The dissolved oxygen concentration of small streams has been modeled as a function of three basic terms: P, R, and K2.

Under high reaeration conditions (K2 >5/day), the maximum deficit is essentially specified by the quotient R/K2, and phosphorus induced increases in biomass (see Macrophyte section) can be expected to increase the maximum deficit (decrease the minimum DO level).

Under lower reaeration conditions (K2 <5/day), the result of increasing biomass is less clear, but should still be modelable through the present "BOD allocation process". If the modeled area includes less than twelve hours of travel time, then analysis of the night time changes can give the "expected deficit at sunrise" (the incoming deficit at sunset (upstream of discharge) would be expected to remain unchanged). Where more than 12 hours of travel time are included in the modeled area, some approximation of the deficit at sunset is necessary. This approximation would have to be done on a site specific basis.

The allocation of phosphorus under these guidelines is very similar to, yet separate from the present BOD allocation procedures. Some increase in phosphorus levels could be expected to result in violation of the 5 mg/l stream DO standard, even if BOD levels are reduced to negligibly small quantities.

SUMMARY AND CONCLUSIONS

The purpose of the Phosphorus and High-Flow Assessment study was to define methods of dealing with phosphorus in setting appropriate water quality goals or standards. The overall study initially evaluated the feasibility of both point and non-point source control of phosphorus. The field study reported here addressed the nutrient control objective, and was aimed at defining low-flow or sustained stream phosphorus contributions rather than high-flow, or event-related phosphorus loadings.

In addition to investigating the impacts of phosphorus in small stream systems, the field study also evaluates methods of documenting phosphorus impacts in streams and recommends monitoring strategies.

Stream and sediment nutrients were compared to rooted plant and attached algae growth in selected southeastern Wisconsin stream reaches in 1981 and 1982. The impacts of in-stream nutrients and plant growth on stream diel dissolved oxygen (DO) characteristics were also investigated. These reaches provided a variety of physical and biological characteristics as well as a wide range of water and sediment nutrient conditions. Streams receiving wastewater treatment plant effluents were also included in the study.

STREAM MACROPHYTES

Based on frequency of occurrence of macrophytes on specific substrate types, sediment interstitial water/stream phosphorus concentration ratios and macrophyte tissue nutrient concentration data, the study reaches were categorized into two groups, Type I and Type II. Various investigators have reported rapid exchange between sediment pore water and overlying stream water in larger substrate sizes. Macrophytes in Type I streams, characterized as growing over larger substrate sizes, are suspected of obtaining growth nutrients from the overlying water. Significant relationships were described between stream phopsphorus concentrations and macrophyte biomass in Type I streams. Based on these relationships, a predictive equation was developed which predicts maximum summer biomass from mean summer (June-August) PO4P concentrations (Model I).

In these stream types, the model predicting maximum plant biomass from in-stream PO4P is most applicable and may provide a good predictive tool for assessing phosphorus impacts in streams. Macrophytes in Type II streams, occuring over primarily silt substrates, are suspected of deriving growth nutrients from the sediments.

Macrophyte tissue nutrients were also significantly related to in-stream nutrient concentrations. A predictive equation was developed which describes macrophyte tissue phosphorus concentrations as a function of mean summer in-stream PO4P concentrations (Model II). As with the macrophyte biomass/stream PO4P model, tissue nutrient concentrations of macrophytes in those stream reaches identified as Type II, were higher than that which could be obtained from the ambient water alone. These Type II streams (where sediments are suspected of being the primary nutrient source) were clearly identified as not belonging to the the Type I stream relationship. This model (Model II) indicates that the maximum tissue phosphorus concentration is dependent on the in-stream mean summer PO4P concentration in Type I streams.

A third equation (Model III) was developed to describe the relationship between macrophyte tissue phosphorus concentration and maximum stable summer plant biomass. This model is somewhat sensitive to timing of the harvesting as tissue nutrients are rapidly lost from senescing plants.

The three least squares regression models presented may provide an alternative method for determining the primary macrophyte nutritive source, and specify the proper model to assess stream macrophyte production and phosphorus inputs in different stream Types. This provides a basic tool

with which to determine existing levels of macrophyte biomass and project changes in stream macrophyte populations due to changing phosphorus inputs. It is suggested, however that the results and the macrophyte models be further tested to improve their applicability to a larger number of situations.

The macrophyte mapping and harvesting methods developed and refined during the study appear to adequately describe stream macrophyte communities. These methods are similar to those employed in present Waste-load Allocation Surveys. Based on the results of this study, a monitoring protocol with recommendations for its use in P-assessment surveys is appended to the study report.

STREAM PERIPHYTON

Stream periphyton were harvested from glass-slide samplers and bricks, exposed for two and four weeks, respectively. The two-week exposure of glass-slide periphytometers was employed to test what is usually considered an optimum or "standard" exposure period. The longer brick exposure periods were designed to approximate a naturally occurring periphyton population's response to nutrients.

Chlorophyll-a was positively correlated with in-stream phosphorus and inorganic nitrogen concentrations. Brick chlorophyll, however, was most strongly related to in-stream PO4P. Brick values were also more strongly correlated with stream nutrients than periphytometer values. Based on these relationships, a least squares regression model was calculated describing the Brick chlorophyll and in-stream PO4P relationship (Model IV). Ash-free weight biomass measurements did not appear directly related to stream nutrient concentrations.

Periphyton tissue nutrients were also highly correlated with in-stream nutrients. Similar to the chlorophyll-a: in-stream PO4P relationship, brick collections were more strongly correlated with nutrients than glass-slide collections. A least squares regression model was calculated describing brick periphyton tissue nutrients as a function of in-stream PO4P (Model V).

Although periphytometer chlorophyll did correlate with in-stream phosphorus, in-stream sampling variability and the influence of physical factors (e.g. current velocity, shading, temperature) precluded serious modeling effort.

Brick periphyton chlorophyll was also correlated with tissue phosphorus and a model calculated to express this relationship (Model VI). The periphyton results suggest that bricks, placed for four-week exposure periods, more closely reflect nutrient impacts than glass-slide collections exposed for two weeks. The correlations of brick nutrients with water nutrient concentrations, similar to macrophyte tissue and in-stream nutrients suggest that brick periphyton collections, representing a naturally occurring periphytic community, support the marophyte study results.

SEDIMENT

Sediment interstitial water nutrients and bulk sediment nutrient samples were collected within macrophyte-populated areas and outside of macrophyte areas.

Macrophyte biomass was positively correlated with sediment interstitial PO4P. No correlation was apparent between interstitial nitrogen and macrophyte parameters. Sediment interstitial PO4P was also correlated with in-stream PO4P concentrations, inorganic nitrogens less so. Statistical T-tests showed no significant differences between nutrients in macrophyte beds and concentrations outside of these areas within stream reaches. Bulk sediment nutrient content was also not clearly related to interstitial nutrient concentrations.

These results substantiate the relationships described by the stream macrophyte

results, suggesting that macrophyte nutritional needs in Type I streams are satisfied primarily through shoot absorption or indirectly through stream water exchange through the substrate.

DIEL STUDIES

Single-station and double-station diel dissolved oxygen analyses were conducted monthly in 1981 and 1982. The purpose of the modeling was to determine photosynthesis, respiration and reaeration values for the date each stream was monitored. *In situ* light and dark bottle and box studies were also conducted to independently measure photosynthesis and respiration.

Estimates of photosynthesis from the modeling results generally agreed with *in situ* measurements. Light saturation was also demonstrated in many of the diel curves. Measured respiration rates from both the bottle and box studies show good agreement with values reported in the literature. Modeling estimates of respiration, however, were usually higher than could be accounted for by measured plant biomass alone. This is attributed to other forms of biological respiration and sediment oxygen demand.

Results from the study and theoretical developments indicate that by increasing primary producer populations, phosphorus in streams will impact photosynthesis, respiration and reaeration capacity. Plant growth in streams will result in incremental increases in community photosynthesis (oxygen production) and respiration (oxygen consumption) and decreased stream reaeration. The effect which substantial macrophyte growth can have on stream reaeration capability is potentially severe. This is due to ponding of stream water by macrophytes, decreasing surface area to volume ratios, and increasing channel roughness.

Using the macrophyte biomass/PO4P model presented in this report, the maximum stream dissolved oxygen (DO) deficit (minimum night-time DO) at sunrise appears modelable as a function of stream PO4P. This should be workable as long as the area modeled has a time of travel which is less than the night length. If time of travel is greater than night length, the "deficit at sunset" must be specified.

Stream reaeration will also influence the ability to specify or predict the night-time maximum DO deficit. Where reaeration is low (less than 5/day) the small stream wasteload modeling process can be used to project the DO deficit at sunrise. Where reaeration is high (greater than 5/day), the small stream model predicts the deficit will equal respiration divided by the reaeration rate (R/K2). Plots of R/K2 against the maximum observed deficit showed that the predicted equilibrium deficit level (R/K2) is commonly achieved.

SITE-SPECIFIC PHOSPHORUS ALLOCATIONS

The overall results of the Phosphorus Assessment field studies indicate a need for phosphorus control in small stream systems. Inability to define phosphorus as a pollutant in a traditional sense has hindered establishment of phosphorus water quality standards in Wisconsin. Relationships between stream phosphorus, macrophyte biomass and stream DO characteristics appear to provide a method to approach phosphorus control.

The results of the primary producer and diel studies suggest that phosphorus could be allocated to streams, on a site-specific basis. This would be done in a manner similar to current methods of allocating Biochemical Oxygen Demand (BOD). Given the macrophyte/phosphorus relationships developed in this study, it appears possible, at least in streams where plants obtain phosphorus from the water, to project changes in macrophyte biomass based on projected changes in in-stream PO4P concentrations.

If minimum night-time dissolved oxygen can be described by R/K2, then the additional

respiratory oxygen demand of the projected increase due to macrophyte biomass can be added as an additional form of BOD. In addition, theory predicts that reaeration (K2) will also decrease as macrophytes become more abundant in the channel. This effect would serve to drive the minimum night-time DO concentration even farther downward.

Other areas of key concern, attributable to phosphorus, include stream ponding, alteration of channel characteristics, changes in the ability of streams to maintain "healthy" night-time dissolved oxygen concentrations, alteration of natural stream habitat and production of undesirable aesthetic conditions due to macrophyte growth. Other agencies in the United States have adopted phosphorus standards based on the above concerns. Appropriate criteria were generally applied through water use classifications (US EPA 1980). In addition to the above incorporation of "Macrophyte BOD" into the current WDNR Wasteload Allocation process, considering these other concepts in development of phosphorus control strategies for Wisconsin is recommended.

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APPENDIX I

STUDY REACH DESCRIPTIONS

INTENSIVE SURVEY REACHES

SUGAR CREEK, WALWORTH CO., T4N-R17E, S15

Sugar Creek (SC), a medium-gradient stream, drains predominantly agricultural and muck-farmed land. The headwaters are ditched with extensive agricultural tile drainage. Sugar Creek is the most heavily nonpoint source impacted stream in the study.

The area upstream of the study reach is predominantly wetlands and fresh-meadow, providing a good buffer along the stream length. Other than overhanging terrestrial grasses and brush, there is virtually no shading of the reach. Increased turbidity during summer low-flow was attributed to rough fish activity.

The Sugar Creek study area was about 60 m (195 ft) in length with an average width of 6.7 m (22 ft). Sample transects were numbered consecutively from downstream to upstream with approximately 11 m (40 ft) between transects. The mean instream depth of this reach was 0.2 m (0.8 ft), with a mean annual flow of 15.4 cfs. Substrate was predominantly sand and gravel with silt overlay once macrophytes became stablished. During the growing season (June-September), submerged macrophytes were restricted to the shallower water and gravel/rubble substrate of the left side of the channel.

ASHIPPUN RIVER, WAUKESHA CO., T8N-R17E-S32

The Ashippun River is a low-gradient stream draining agricultural land. There is little shading of this section of the stream, with good buffer along the length immediately above the study area. Cattle pasturing adjacent to the river in the upper watershed appeared the most common nonpoint source problem. This reach is also heavily impacted by agricultural NPS pollution. The watershed area is roughly one-half the size of Sugar Creek's.

The Ashippun study area is divided into three reaches, delineated by an old berm which at one time served a mill (Ashippun-Mainstem), and a small island which divides the downstream flow into the North and South Branch reaches. The water serving the reaches is essentially of the same quality.

Ashippun-Mainstem

The mainsteam reach (AM) in 1981 was approximately 40 m (130 ft) long with an average width of 5 m (17 ft), and divided into 5 transects, 9 m (30 ft) apart. Submersed macrophyte growth was restricted to the deeper, center channel from June-September. Emergent vegetation (Sparganium eurycarpum) occurred along the right bank. In 1982, mean depth of this reach was 0.4 m (1.2 ft) with a mean annual flow of 15.5 cfs. Bottom materials were predominantly sand and gravel except in the area associated with the Sparganium, which is a thick silt bed. There is little direct shading of the reach.

This reach was lengthened in 1982 for a total reach length of 235 m (780 ft), with a mean width of 6 m (20 ft). Mean depth of the Ashippun reach in 1982 was .5 m (1.3 ft). Predominant substrate was sand and gravel.

Ashippun-North Branch

The Ashippun-North Branch reach (AN) was approximately 26 m (87 ft) long and averaged 4 m (13 ft) wide. Submersed macrophyte growth occurred throughout the reach in June-September, 1981. Substrate in this reach was primarily rubble/gravel with little silt. This area received shade from bank willows for a brief period in the morning. This reach averaged 0.2 m (0.7 ft) in depth with a mean annual flow of 9.3 cfs, or 60% of the Mainsteam flow. This reach was discontinued in 1982.

Ashippun-South Branch

The South Branch reach (AS) was roughly 32 m (104 ft) long and 3 m (10 ft) wide. Bottom materials varied from sand and gravel to gravel and rubble. Macrophytes occurred primarily in the gravel and rubble substrate. Mean depth of this reach was 0.2 m (0.6 ft) with a mean annual flow of 6.0 cfs, with represents 40% of the Mainsteam discharge.

Both the North and South Branch reaches were divided into 4 transects, separated by 9 m and 11 m (30 and 35 ft) respectively.

KOHLSVILLE RIVER, WASHINGTON CO., T12N-R18E-S35

The Kohlsville River (KR) is a high gradient low order stream. The study reach was located at CTH "D", upstream of the impoundment at Kohlsville. For most of its length above this point, Kohlsville River is shallow, limiting the fishery to forage fish. The water upstream of the study reach is predominantly agricultural, mostly in hay and grain crop production. The Kohlsville study reach is one of the smallest watersheds of the study.

The Kohlsville study reach was 56 m (185 ft) long and averaged 2 m (7.5 ft) in width. The reach is generally shallow, with rubble and gravel substrate. Mean depth during the 1981 study period was 0.2 m (0.7 ft) with a mean annual flow of 4.2 cfs. The reach was divided into 9 transects, approximately 6 m (20 ft) apart. Periphyton and mosses are the dominant primary producers, sometimes growing in a thick, felt-like mat on the larger substrate classes; no macrophytes were observed.

This reach is shaded in the early morning by an Oak lot. The area immediately upstream of the study area, however, is almost totally shaded.

This reach was discontinued in 1982.

BARK RIVER, WAUKESHA CO.

There were two study reaches on the Bark River in 1981 and three in 1982, one upstream and two downstream of the Dela-Hart POTW outfall.

Bark-Wolf, T7N-R17E-S26

This study reach (BW), upstream of the Dela-Hart POTW outfall, is roughly 1.1 km (0.7 mi) downstream of Crooked Lake. Land use upstream and adjacent to this study site is primarily agricultural, however there is also a large percentage of recreational/open space land. Because of the short distance between the study reach and the Crooked Lake outlet, there is very little land area contributing directly to the stream at this point, and there is a good buffer area along the stream.

Substrate within the Bark-Wolf reach is predominantly sand and gravel. There is little macrophyte growth in the study area and for a distance up and downstream. There are, however, areas upstream of the reach which support moderate periphyton and macrophyte growth.

In 1981, the Bark-Wolf study reach was 68 m (226 ft) long and averaged 9 m (30 ft) in width. Mean depth over the 1981 study period was 0.4 m (1.3 ft) with a mean annual flow of 26.0 cfs. Bark-Wolf had the lowest nutrient concentrations of all the study sites, reflecting the influence of the upstream lake.

This reach was lengthened to about 300 m (1000 ft) in 1982.

Bark-Lurvey, T7N-R17E-S35

The Bark-Lurvey study reach (BL) is located 1.4 km (0.85 mi) downstream of the Dela-Hart outfall. The Dela-Hart treatment plant, which went on-line in August 1980, has a design capacity of 2.2 mgd and summer effluent limits of 10 mg/l for BOD_5 and suspended solids, 2 mg/l ammonia, 6 mg/l DO and pH of 7.6; there are no phosphorus limits. Incorporated into the treatment process are rotating biological contractors (bio-discs), sand filtration and cascade-type final effluent aeration. Low ammonia concentrations, high NO_2-NO_3N and P04P concentrations are discharged. Mean discharge at the outfall was estimated at 2 cfs in 1981 and 1.6 cfs in 1982.

As with Bark-Wolf, a very small watershed area contributes to the Bark between the upstream (Wolf) and the Bark-Lurvey reaches, agriculture being the dominant land use. The section of stream below the outfall is characterized by numerous gravel riffles and runs. There is little shading of this area and there is a good buffer along the stream length.

In 1981, the Bark-Lurvey study reach itself was 67 m (220 ft) long with an average width of 12 m (40 ft). There were 12 transects, separated by 6 m (20 ft). Mean reach depth during the 1981 study period was 0.4 m (1.3 ft). The mean annual flow was 28.4 cfs. During the growing season (June-September), reach depth increased without a corresponding increasing in flow due to the ponding effect of macrophytes.

A farm bridge splits the reach into two sections. The downstream section is characterized by shallow water depth and predominantly rubble/gravel substrate; the upstream portion by deeper water and gravel/sand substrate. The entire reach is dominated by submerged macrophytes, with emergent vegetation (Sparganium eurycarpum) along the right bank. The study reach is mostly unshaded.

This reach length was expanded to about 480 m (1600 ft) in 1982.

Bark-Masonic, T6N-R17E-S3.

This study reach, 3 km (1.85 mi) downstream of the Dela-Hart POTW outfall (1.6 km or 1 mi below the Lurvey site) was added in 1982. Predominantly agriculture and residential land drain to the reach between this and the Lurvey site. The reach was added to Bark-Wolf and Bark-Lurvey as permanent stations on the Bark River.

Substrate within the Bark-Masonic reach is predominantly sand and gravel. There is little direct shading of this reach. Adjacent land supports Tamarack and shrub growth.

The Masonic reach is about 242 m (800 ft) long with an average width of 13 m (44 ft). Nine transects were established for mapping and macrophyte harvesting.

SYNOPTIC SURVEY REACHES

Bark-Wahlschlaeger, T8N-R18E-S23.

This site, located upstream of Nagawicka Lake, is approximately 74 m (245 ft) long with a mean width of 9 m (30 ft). Mean depth at the time of harvesting was .4 m (1.2 ft). Substrate was predominantly sand and gravel. There is little direct shading of this reach.

Land use upstream and adjacent to the site is predominantly wetlands and low-density residential. This reach is not impacted by a POTW discharge.

Mukwonago River, T5N-R18E-S25.

This site is approximately 1 km (0.5 mi) downstream of the Lower Phantom Lake dam. The reach sampled was approximately 300 m (1,000 ft) long with a mean width of 16 m (53 ft). Substrate was predominantly gravel. Mean depth at the August 11, 1982 macrophyte harvest was .4 m (1.3 ft).

This reach is similar to the Bark-Wolf reach both in water chemistry and physical characteristics. There is no direct shading of the reach. Adjacent land use is primarily open space/sedge meadow.

Milwaukee River - Main Branch, T13N-R19E-S18.

The Milwaukee River-Main Branch reach was sampled September 2, 1982, approximately 1 km (.8 mi) downstream of the Campbellsport POTW discharge. The sample reach was 211 m (695 ft) long with ten transects selected for macrophyte harvests. At the time of the survey, mean reach width was 9 m (31 ft) and mean depth was .3 m (1.1 ft). Land use adjacent to the reach was predominantly meadow/open space with little direct shading of the reach.

Present WPDES permit effluent limits are 30 mg/l for BOD_5 and suspended solids.

Milwaukee River - East Branch, T12N-R19E-S2.

The study site of the East Branch of the Milwaukee River is approximately 91 m (300 ft) long with a mean width of 12 m (38 ft). Twelve transects within this reach were selected for mapping and macrophyte harvesting. The reach was sampled on August 16, 1982. At that time, mean depth was .3 m (1.1 ft). Substrate was predominantly gravel and sand with silt along the banks.

Land use adjacent to the sampling site is sedge-meadow. There is a mill-pond approximately 3 km (.9 mi) upstream at New Fane.

White River, T2N-R18E-S17.

The White River reach is located approximately 2.6 km (1.6 mi) downstream of the city of Lake Geneva POTW. Present WPDES interim permit limits are 45 mg/l BOD_5 and suspended solids and 1.0 mg/l total phosphorus. The existing POTW has trouble meeting the phosphorus limit. Dam manipulation and seiches in Lake Geneva, which frequently cause water to flow over the spillway, cause fluctuations in stream depth and velocity.

Macrophytes were harvested August 25, 1982. At this time, mean depth was .5 m (1.7 ft). Reach length was 288 m (950 ft) with a mean width of 9 m (31 ft). Substrate in the reach was predominantly gravel and sand.

Land use above the site is primarily grass meadow floodplain. The stream is generally unshaded, however portions of the reach sampled receive shade during part of the day.

Scuppernong River, T5N-R17E-S19.

This site was approximately 91 m (300 ft) long, with a mean width of 5 m (17 ft). Mean depth at the time of harvesting was .3 m (1.1 ft) with sand and gravel substrate. Harvesting was conducted September 7, 1982.

Land use upstream and adjacent to the study reach is predominantly agriculture. Portions of the upper watershed are extensively ditched.

Pewaukee River, T7N-R19E-S26.

This reach is approximately 6.5 km (4 mi) downstream of Pewaukee Lake and the City of Pewaukee. Land use adjacent to the reach is primarily agriculture and low-density residential. The City of Pewaukee, however, contributes substantial storm drainage to the river during wet weather. The Pewaukee POTW, which wend off-line in October, 1981, also discharged to the Pewaukee River.

The study reach was approximately 91 m (300 ft) long with a mean width of 6 m (19 ft). Ten transects were selected for mapping and macrophyte harvesting, August 24, 1982. Mean depth at the time of harvest was .2m (.5 ft). Substrate was predominantly gravel with sand.

Cedar Creek, T10N-R19E-S13.

The Cedar Creek reach was located approximately 6.5 km (4 mi) downstream of Little Cedar Lake. This reach was 86 m (290 ft) long with a mean width of 5 m (16 ft). Mapping and harvesting were conducted at 11 transects within the reach on September 8, 1982. At that time, mean depth was .2 m (.5 ft). Substrate was predominantly gravel with rubble.

Land use adjacent to the reach is primarily agriculture with some wetlands contribution.

Mt. Vernon Creek, T5N-R7E-S2.

Mount Vernon Creek is a groundwater-fed stream, supporting an excellent trout fishery. The study reach was 120 m (400 ft) long with a mean width of 6 m (18 ft). Substrate was predominantly gravel with sand. Macrophyte harvests and stream mapping were conducted September 10, 1982. At the time of harvest, mean reach depth was .4 m (1.2 ft). Eleven transects within the reach were sampled.

Land use upstream and adjacent to the reach was predominantly agriculture and pasture.

Black Earth Creek, T8N-R6E-S36.

Black Earth Creek is a groundwater-fed stream which also supports an excellent trout fishery. The Black Earth study reach was approximately 112 m (370 ft) long with a mean width of 11 m (37 ft). Stream mapping and macrophyte harvesting were conducted at ten transects within the reach on September 10, 1982. At that time mean depth of the reach was .4 m (1.4 ft).

Black Earth Creek receives effluent from the Cross Plains POTW, approximately 11 km (7 mi) upstream of the study reach. Present permit limits are 30 mg/l BOD and suspended solids and 2/7 mg/l ammonia (summer/winter). Land use adjacent to and upstream of the reach was primarily pasture and wetlands.

Fox River (Upper Fox), T12N-R9E-S4.

The Fox River study reach was sampled approximately 1 km (.6 mi) downstream of the Portage POTW discharge. The reach was 136 m (450 ft) long with a mean width of 17 m (57 ft). Mapping and macrophyte harvesting was conducted at ten transects within this reach. The Fox River study reach was sampled August 26, 1982. At that time, mean reach depth was .5 m (1.5 ft).

APPENDIX 2

STREAM REACH SPECIES LIST

A species list is presented for each stream reach at the time maximum biomass was harvested. Species are given in order of percent occurrence. Percent occurrence was calculated from the mapping data which was collected at the time of harvesting. The value given for percent occurrence is the number of times a species occurred divided by the total number of sample points mapped in a reach. Mean relative abundance is given for each species. Species abundance ratings were assigned to each species for each occurrence of a given species according to the criteria in Table 4. Mean relative abundance is the mean of the species abundance ratings for each species.

Emergent species were not included in the harvesting surveys. For this reason, some species may occur on the species list but which were not harvested.

Ashippun River

August 1981

	Percent Occurrence	Relative Abundance
Potamogeton pectinatus	28.7	1.63
Sagittaria rigida	16.3	1.62
P. zosteriformis	14.0	1.80
P. amplifolius	10.1	1.56
Sparganium eurycarpum	7.9	1.71
Ceratophyllum demersum	3.9	1.00
Lemna minor	1.7	1.00

August 1982

S. rigida	40.7	2.73
S. eurycarpum	14.8	1.88
P. zosteriformis	8.3	1.89
P. amplifolius	6.5	1.71
L. minor	1.9	2.00
P. pectinatus	0.9	2.00
Heteranthera dubia	0.9	2.00
C. demersum	0.9	1.00

Bark River-Lurvey Farms

August

	Percent Occurrence	Relative Abundance
H. dubia	64.9	1.82
L. minor	43.2	1.85
Vallisneria americana	40.5	2.10
P. nodosus	20.7	1.17
C. demersum	20.7	1.00
Nymphaea sp.	9.9	1.09
P. pectinatus	7.2	1.25
Anacharis canadensis	5.4	1.00
Myriophyllum sp.	4.5	1.00
Scirpus sp.	3.6	1.00
P. crispus	0.9	1.00

August 1982

H. dubia	67.4	3.17
V. americana	28.3	2.97
P. nodosus	10.5	1.41
P. pectinatus	9.7	1.36
C. demersum	1.9	1.80
L. minor	1.9	2.80
S. eurycarpum	1.6	1.25
Myriophyllum sp.	1.6	1.25
Scirpus sp.	1.2	1.66
A. canadensis	0.8	1.00
P. zosteriformis	0.4	1.00
P. crispus	0.4	2.00

Bark River-Wolf Road

August 1981

	Percent Occurrence	Relative Abundance
Potamogeton sp.	24.6	1.00
C. demersum	18.8	1.31
P. pectinatus	17.4	1.08
Myriophyllum sp.	15.9	1.09
V. americana	14.5	1.10
H. dubia	13.0	1.00
P. zosteriformis	4.3	1.00
Scirpus sp.	2.9	1.00

Bark River-Wolf Road (con't)

	Percent Occurrence	Relative Abundance
Ausust 1982		
P. pectinatus	34.8	1.43
Myriophyllum sp.	28.0	1.11
V. americana	26.1	1.10
Potamogeton sp.	24.8	1.43
H. dubia	6.2	3.40
Najas flexilis	3.1	1.00
C. demersum	1.2	1.00

Sugar Creek

August 1981

	Percent Occurrence	Relative Abundance
P. americanus	58.1	2.90
S. rigida	9.3	2.38
August 1982		
P. americanus	64.9	3.81
S. rigida	17.6	3.15

Black Earth Creek

August 1982

	Percent Occurrence	Relative Abundance
P. zosteriformis	63.4	2.86
A. canadensis	17.2	2.70
Ranunculus sp.	13.4	1.83
P. pectinatus	5.2	1.57
L. minor	3.7	2.80
Scirpus sp.	1.5	1.00
P. crispus	0.7	1.00

Bark River-Walschlaeger Road

August 1982

	Percent Occurrence	Relative Abundance
C. demersum	45.7	3.21
P. zosteriformis	39.4	1.90
P. pectinatus	26.8	1.29
S. eurycarpum	7.1	1.67
L. minor	6.3	1.88
Sagittaria sp.	3.9	2.00
Nymphaea sp.	3.1	1.50

Cedar Creek

September 1982

	Percent Occurrence	Relative Abundance
P. pectinatus	92.2	2.58

Fox River

August 1982

	Percent Occurrence	Relative Abundance
H. dubia	63.4	2.92
Potamogeton sp.	53.6	1.99
V. americana	28.1	2.63
L. minor	5.2	1.25
Myriophyllum sp.	2.6	2.25
Zizania aquatica	1.3	1.00
C. demersum	0.7	1.00

Milwaukee River-Campbellsport

September 1982

	Percent Occurrence	Relative Abundance
P. pectinatus	87.8	4.52
C. demersum	5.2	3.00
L. minor	4.3	2.60
S. rigida	1.7	1.00
A. canadensis	0.9	1.00
P. zosteriformis	0.9	1.00

Milwaukee River-East Branch

August 1982

	Percent Occurrence	Relative Abundance
S. rigida	45.8	2.90
P. pectinatus	30.1	1.54
Potamogeton sp.	25.9	1.79
H. dubia	4.2	1.29
S. eurycarpum	2.4	2.25
Iris sp.	1.2	1.50
C. demersum	0.6	1.00

Mount Vernon Creek

September 1982

	Percent Occurrence	Relative Abundance
Ranunculus sp.	55.5	3.75
A. canadensis	24.5	3.48
Hypericum elliptlcum forma aquaticum	17.3	1.68
Zannichellia palustris	16.4	2.72

Mukwonago River

August 1982

	Percent Occurrence	Relative Abundance
Najas flexilis	49.0	1.35
P. pectinatus	48.5	1.15
V. americana	40.2	1.07
Myriophyllum sp.	24.5	1.14
Chara	22.5	1.09
H. dubia	11.8	1.04
Potamogeton sp.	8.3	1.35
A. canadensis	2.9	1.00
Scirpus sp.	2.0	1.50
P. zosterformis	1.0	1.00

Pewaukee River

August 1982

	Percent Occurrence	Relative Abundance
P. pectinatus	93.2	3.95
L. minor	26.5	1.36
Myriophyllum sp.	5.4	2.38
P. crispus	3.4	3.60
C. demersum	0.7	1.00

Scuppernong River

September 1982

	Percent Occurrence	Relative Abundance
S. eurycarpum	33.6	2.40
A. canadensis	32.7	3.70
S. rigida	27.4	1.84
Potamogeton sp.	24.8	2.57
Z. aquatica	5.3	1.67
P. amplifolius	5.3	1.00

APPENDIX 3

METHODS FOR EVALUATING MACROPHYTE POPULATIONS IN SMALL STREAM SYSTEMS - APPLICATION OF PHARTS METHODS TO ROUTINE WATER QUALITY INVESTIGATIONS.

INTRODUCTION

The PHARTS data analyses have indicated that maximum stable summer macrophyte biomass can be accurately predicted in Southeastern Wisconsin streams. Two linear regression models were developed which predict summer macrophyte biomass. These models are based on macrophyte percent coverage estimates and mean summer phosphorus concentrations within streams. Further development and testing of these models for use in other parts of the state requires additional data over a wider range of stream types and conditions.

Collecting data to use in the model involves a limited amount of field work. To use both models, stream reach macrophyte biomass, percent of the streambed with macrophyte cover and mean growing season in-stream phosphorus and nitrogen concentration data need to be collected. Mapping provides the percent coverage estimates for the reach and harvesting provides quantitative plant biomass data to compare with the percent macrophyte coverage and water chemistry data. Substrate class and distribution within the stream reach is used to evaluate macrophyte substrate preference and their potential to supply macrophyte nutrients. This work is conducted over a relatively short stream reach.

There are, then, two separate elements of macrophyte assessment; mapping and harvesting. The harvesting element is designed to provide corroborative data for model development and refinement. The mapping element provides the data to use in the model. It is probable that mapping will be the only element routinely conducted.

Criteria for selecting stream reaches as well as sample collection requirements and the macrophyte mapping and harvesting methodologies are discussed.

SITE SELECTION

The following criteria should be followed in selection of a stream reach which will provide the best obtainable data:

- Reasonably uniform distribution of macrophytes and substrate type within the reach;
- Maximum reach depth of 2-3 ft. The stream must be workable with waders. Greater depth will also increase the potential for light-limited growth which will obscure any macrophyte/nutrient relationships;
- Annual mean flow of less than 60 cfs;
- Maximum stream top width of 60-70 ft;
- Stream should be relatively unshaded and free of constructions (e.g. trees, boulders, pools, logs);
- Stream reach length should be a minimum of 300 ft and maximum of 2,000 ft.

METHODS

Stream macrophyte community assessment involves conducting a three-part survey; water chemistry collections, macrophyte mapping, and macrophyte harvesting. Mapping and harvesting are conducted as close to the time of maximum biomass as possible. Typically, this is in late August or early September in the SE District. Equipment required to conduct the surveys includes; wading rod; flow meter, tag-line(s), Surber bottom sampler, survey flags or pins, mapping and harvesting field data forms (attached), plastic bags, random number tables, and plastic wash tubs with 1/4 inch mesh bottoms.

Stream mapping procedures are similar to those used in collecting top-width and stream cross-section data for wasteload allocation surveys. The primary difference is in more accurate descriptions of substrate types and macrophyte percent coverage. A sample reach is selected which is representative of the portion of stream to be characterized. Transects within the reach are established, equidistant if possible, for mapping and plant harvesting.

The following summarize mapping and harvesting procedures.

Macrophyte Mapping Procedures

Macrophyte and stream channel mapping is conducted at each reach transect prior to selecting harvest quadrats and plant removal. Collection of these data provide percent macrophyte cover and substrate composition of the stream channel. Procedures are similar to stream cross-section or flow measurements (without velocity) and involve the following:

1) Ten to fifteen transects within the reach should be selected with a minimum of 30 feet between each transect. Distance between transects should be uniform, measured parallel to the thalweg. Transects are placed perpendicular to the direction of stream flow. Transects should not be located near major obstructions in the stream (e.g. trees, boulders, deep pools and logs).

2) Ten to twenty observation points should be taken along each transect with a maximum of three feet between each point. The observation points are one square-foot quadrats. Transect widths include open-water areas and do not include zones of emergent bank vegetation (e.g. cattails, bulrush, burreed).

3) The observations recorded at each quadrat (observation point) are; the distance of that point from the left streambank, depth at that point, estimates of the percent composition of each bottom type (substrate type), percent of quadrat covered by macrophyte and percent of each species present. It is convenient to "imagine" a one square-foot area around the observation point to estimate the percent macrophyte cover (or percent open area) and substrate types. It's also convenient and quicker to use macrophyte species codes rather than writing the full species name, and the number rating corresponding to a given percent coverage (Attachment 1). Data forms are provided for recording this information. These data will provide the estimated macrophyte percent coverage values to be used in the model.

Macrophyte Harvesting Procedures

Macrophyte harvesting is accomplished by re-establishing or using the original mapping transects. Transects should be marked when mapped so that the harvesting will be conducted at the same location as the mapping. It is usually more efficient to have two tapes and two crews, one mapping and the other harvesting. This is not always necessary. A minimum of 40-50 samples should be collected per reach, with a minimum of 3-4 samples per transect.

1) A random number table is used to select the sample quadrats within each zone or along the transect. The sample quadrats should be selected no closer than three feet from the tag-line to avoid areas disturbed by mapping activities. The maximum distance from the tape should be 20 feet or 1/2 the distance to the next transect, whichever is the least.

2) A 2-4 digit random number is used to pick each sample quadrat with the transect. The first one or two digits (depending on total transect width) is the distance from the left stream bank. The third and/or fourth digit(s) is the distance up- or downstream of the transect. If the digit is even, the quadrat is placed that distance upstream of the transect. If the digit is odd, the quadrat is placed downstream of the transect. Different random numbers or a different column of numbers are used for each transect. If macrophyte growth occurs in distinct zones within the stream channel, sample locations should be weighted by the size and occurrence of the zones.

3) A Surber sampler (one square-foot) is placed on the stream bed with the random number coordinate at its center. Percent macrophyte coverage of plants rooted in the quadrat, species abundance, depth and water velocity should be recorded at each quadrat prior to plant harvesting. All plants originating within the frame are harvested with the roots. A small hand garden cultivator works best to get the roots. The harvested plants are dumped into plastic washtubs (with the screen mesh bottom) and thoroughly washed with stream water. The sample is sorted in the field to remove stones, sticks, fish and invertebrates. The sorted and rinsed sample is placed in a plastic bag, labeled with the transect and sample number. Samples are transported to the lab on ice and refrigerated (do not freeze) until processed.

4) In the lab, samples are separated by species, placed in numbered, pre-weighed 20# paper bags and dried at 60 degrees C to constant weight in a forced-air oven. Drying approximately 5-10 grams (dry) of plant material (about one handful) per bag should take 24 hrs. Larger portions in each bag will lengthen the drying time.

Water Sampling

Mean growing season phosphorus and nitrogen concentrations provide the best data for predicting summer macrophyte biomass. The object of the collection is to characterize the nutrient concentrations occurring over the greater part of the season. Ideally, grab samples should be collected every three weeks from mid-May through the end of August. Grab samples should also be collected at near-normal flow (not necessarily Q7-10 or Q7-2). This best represents the conditions that plants have experienced during the growing season. For this reason, high-flow samples, if collected, are generally not included in calculation of the mean growing season stream chemistries.

APPENDIX 4

DIEL DATA COLLECTION AND ANALYSIS METHODS

A variety of methods have been developed to approximate the diel fluctuations of dissolved oxygen in streams, lakes and rivers. These have generally been developed in response to a specific need, and the usefulness and applicability of each of these methods is a direct function of the needs of the investigator. In the present study, two specific goals required consideration. First, an attempt to determine stream photosynthesis and respiration rates, and relate these to measured plant biomass quantities, and second, to determine whether or not plant biomass could be expected to significantly impact stream dissolved oxygen (DO) concentrations, especially the minimum concentration.

Since the double station method provides estimates of photosynthesis, respiration and reaeration for a defined area (the area between stations), it was thought to be the best tool for obtaining photosynthesis and respiration estimates, which could then be compared with areal biomass estimates from harvest data.

Assumptions necessary to the modelling process restrict the length of the reach which is modelable through the double station technique. Travel times of one half to two hours gave good results where K2 ranged from 1.0-6.0/day. Natural variations in the model parameters tends to hinder attempts to correlate double and single station estimates of photosynthesis, respiration and reaeration.

The single station method determines coefficients which are "upstream averages" for an area which is determined by the "average upstream" reaeration rate (K2). The larger the value of K2, the smaller the area represented by single station analysis. The coefficients which result from single station analysis should reproduce the curve observed for the site from which the coefficients were derived. The single station method may be preferable where the goal is prediction of dissolved oxygen at a given time and place.

Two different equations were used in the present study. The differential equation (I) was originally proposed by Odum (1956). The integrated equation (II) was advanced in part by Blain and McDonnell (1967) and independently derived in a form that included photosynthesis by WDNR staff. Either equation is suitable for single or double station analysis.

I. $\Delta DO/\Delta t = \alpha P + R + K_2 (Cs-Co)$

II. $(Cs-Co)_{t+\Delta t} = (Cs-Co)_t e^{-K2\,\Delta t} + [(\alpha P + R)/K2](1 - e^{-K2\,\Delta t})$

The major difference between the single and double station methods lies in their determination of the $\Delta DO/\Delta t$ term of the differential equation, or the (Cs-Co) terms of the integrated equation. In the single station method, the data is taken from a single diel curve. $\Delta DO/\Delta t$ is the slope of the diel curve at a given time. The (Cs-Co) terms are the deficit at the specified times ("t" or "t + Δt").

The double station technique requires that two diel curves be obtained, one for an upstream station and one for a downstream station. It also requires that the time of travel between stations is known. In the double station method, $\Delta DO/\Delta t$ is the difference between the upstream DO concentration at time "t" and the downstream concentration and time "t + time of travel (TOT)", divided by the time of travel.

Values for light and temperature are necessary for both equations and both methods. Either can be approximated, but it is best if both are measured. The temperature data is used to calculate what the "saturation concentration" of DO is. Several equations are available for this purpose. One which is commonly used is as follows (J. Sanit. Eng. Div., Am. Soc. Civ. Eng., 1960):

$$Cs = 14.652 - .41022(°C) + .007991(°C)^2 - 0.000077774(°C)^3$$

The concentration which results is that which would occur if the atmospheric pressure is 760 mmHg. Variations in atmospheric pressure are assumed to cause only negligible variations in the saturation concentration.

Light should be measured in a way that closely approximates the light levels which the plants experience. The best technique appears to be measuring light below the water surface, at a depth that is similar to the mean depth of the stream. Although this is not an exact measurement, it does eliminate problems associated with surface reflectance, partially compensates for attenuation of light with depth and shading due to bank vegetation or the horizon. Where plants have grown enough to reach the surface, this method probably underestimates the amount of light the plants actually receive. It should be realized that the main purpose of this measurement is to provide a means by which photosynthesis can be proportioned.

An example is presented below to help clarify these statements, and show how these measurements are used to calculate double station P, R, and K2 coefficients. In the example, DO and temperature were continuously recorded at the upstream station, DO was continuously recorded at the downstream station, and light was continuously recorded at a point near the downstream station. A dye study revealed that the time of travel between stations was 18 minutes. The upstream DO and temperature data were read off the stripchart at one hour intervals. Then the downstream DO data were read off the downstream stripchart at times which corresponded to "upstream times + 18 minutes". Temperature was assumed to be constant within the reach. Light for each sample period (8:00-8:18, 9:00-9:18, etc) was also read from a strip chart. Upstream DO readings could be taken at more frequent intervals if more sample points were desired.

Upstream Station				Downstream Station	
TIME	DO (mg/l)	TEMPERATURE (°C)	LIGHT Q/sq cm/sec x 10^{15})	TIME	DO (mg/l)
0400	5.56	22.6	00.0	0418	5.70
0500	5.58	22.4	00.0	0518	5.70
0600	5.57	22.2	00.0	0618	5.70
0700	5.59	22.1	02.0	0718	5.85
0800	5.80	22.1	13.0	0818	6.10
0900	6.12	22.1	11.0	0918	6.55
1000	6.32	22.4	20.0	1018	6.78
1100	6.70	22.8	42.0	1118	7.20
1200	6.82	23.1	45.0	1218	7.40
1300	7.03	23.5	38.0	1318	7.55
1400	7.04	23.8	19.0	1418	7.40

Some preliminary calculations are necessary before the data is in a form that is amenable to analysis. Specifically, we need to determine Δ DO/Δt and (Cs-Co) for each sample interval. For the double station method, Δ DO/Δt is simply the difference between the upstream and downstream values. (If the single station method was used, a rough approximation of Δ DO/Δt

could be obtained by subtracting consecutive readings in the upstream or downstream columns. The t value would then be the time interval between readings.) For the period 0400-0418, the double station $\Delta DO/\Delta t$ is 5.70 - 5.56 = 0.14 mg/l. The time component is not included at this point. The next step is to calculate (Cs-Co). Cs is calculated from the temperature data and the equation presented earlier. At 22.6°C, Cs would be 8.31 mg/l. To determine the "average deficit" for the period, we need to average the upstream and downstream DO values before determining the deficit. For the 0400-0418 interval, the average Co value is 5.63 mg/l, and the average deficit (Cs-Co) is 8.31-5.63 or 2.68 mg/l. The light value for the period is 0.0. If this process is repeated for each sample interval, we end up with three columns: $\Delta DO/\Delta t$, (Cs-Co) and light (α).

If $\Delta DO/\Delta t$ is specified as the independent variable, and (Cs-Co) and light (α) are specified as dependent variables, a multiple regression may be performed on the data set. The coefficients which are returned from the regression analysis are "P" (the coefficient of the light (α) term), K2 (the coefficient of the (Cs-Co) term) and "R" (the intercept, or constant). Since no time correction was made to the $\Delta DO/\Delta t$ term, all coefficients are in units of "per time of travel". To convert the R and K2 terms to units of "per day", simply multiply by the number of time of travel units which occur in a twenty four hour period. (For the 18 minute time of travel specified, the coefficients would be multiplied by 80.0.) The photosynthesis term is in units of "mg/l/unit light/TOT". As a raw coefficient it specifies the number of mg/l that would be produced at a light intensity of 1.0 for the specified time of travel. To convert this to a per day rate, the coefficient should be multiplied by the total quantity of light received in a day, and divided by the quantity of light received during the time of travel at a light intensity of "1.0". The total quantity of light received in a day can be determined graphically (the area under a light curve), or through mathematical subroutines. An example of converting the raw coefficient to a value proportionate to light follows.

$$P = X \text{ (mg/l)}/(1.0 \times 10^{15} \text{ quanta/sq cm/sec})/18 \text{ minutes}$$

$$= X \text{ (mg/l)}/1080 \times 10^{15} \text{ quanta/sq cm}$$

Where "X" is the coefficient of the light variable (from the regression analysis)

If this value is multiplied by the total quantity of light/sq cm received in a day, the product would represent the amount of oxygen produced in a day. (An average July day would be about 3×10^{21} quanta/sq cm/day at the water surface.) If other light units are used, the appropriate alterations must be made to these calculations.

To convert from "mg/l/day" to mg/sq meter/day estimates, the mg/l/day estimates were multiplied by the average number of liters per square meter (304.8 liters/sq meter/1 foot depth x average reach depth in feet).

Single station analysis could proceed in a similar manner, except that the $\Delta DO/\Delta t$ term would be taken from a single curve. The Δt would be the time between DO readings on a single curve. Calculation of per day and per unit area coefficients could proceed in a similar manner.

If the integrated equation is used, the initial and final deficits must be used in a nonlinear regression routine. [Blain and McDonnell (1967) used the night-time data to calculate R and K2, and then calculated P from the daytime data.] P, R and K2 values can be similarly determined from the nonlinear regression coefficients.

If recorded data is not available, a method of approximating intermediate data points may be necessary. "Approximation and interpolation" subroutines are available through the Madison

Academic Computing Center (MACC). The particular routines used in this study were cubic polynomial spline interpolations (AISPIN/AISPEV).

Regression routines usually provide an estimate of the "goodness of fit" of the data to the proposed equation and resultant coefficients (e.g. r-squared, F-tests, t-ratios of coefficients, etc.) The behavior of the data and the assumptions inherent in the model should be examined before the results are accepted. An example of a situation where statistical fit does not properly indicate erroneous results is presented in the "Differential vs Average Differential Coefficient" discussion.

Violations of the assumptions of the model are common (see "Deviations from Assumptions" section). The effect of these violations may or may not be severe, but they should be examined on a case by case basis, and dealt with if necessary. Plots of $\Delta DO/\Delta t$ (corrected for reaeration by subtracting K2(Cs-Co)) versus light are useful in confirming the linearity of the light:photosynthesis relationship. Similarly, plots of $\Delta DO/\Delta t$ (corrected for reaeration, again) during the night time hours can be used to confirm the constancy of respiration. These two simple checks can add to the information to be gained from diel curve analysis.