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Bioavailable Phosphorus in the Bear River System

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BIOAVAILABLE PHOSPHORUS IN THE

BEAR RIVER SYSTEM, UTAH

by

Kenneth Warren Barker

A thesis submitted in partial fulfillment
of the requirements for the degree

of

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in

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Kenneth Warren Barker

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ABSTRACT

Bioavailable Phosphorus in the

Bear River System, Utah

by

Kenneth W. Barker, Master of Science

Utah State University, 1988

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Department: Civil and Environmental Engineering

The bioavailable fraction of phosphorus (BAP) in the lower Bear River system waters was investigated. BAP plays a critical role as the limiting nutrient for algal production and eutrophication in proposed reservoirs in the Bear River system. The Bear River system has a hardness ranging between 180-240 mg/L as $CaCO_3$ which significantly affects BAP.

BAP estimation was done by a modified *Selenastrum capricornutum* Printz Algal Assay Bottle Test. The algal bioassay is considered the best estimator of BAP because no chemical tests or indicator parameters are available. Autoclaving and UV radiation were found to be unacceptable means for sterilization because of phosphorus precipitation and inability to kill all the protozoa, respectively. Whole water samples were sterilized by gamma radiation. Hydrogen peroxide formed by gamma radiation was minimized by sparging with nitrogen gas, and adding peroxidase to remove low hydrogen peroxide concentrations. Soluble reactive phosphorus concentrations changed during radiation.

The algal photosynthetic consumption of CO_2 in the assay procedure raised the pH from 8 to as high as 10, which resulted in significant quantities of phosphorus precipitating with calcium and becoming unavailable. To minimize the effects of precipitation, the following recommendations are made: (1) bubble the bioassay flask with a CO_2/air gas mixture to minimize pH increase; and (2) use a high inoculum (10^5 cells/ml) of *S. capricornutum* that have been phosphorus starved for several days to maximize luxury uptake.

Bioavailable phosphorus was estimated for each of the sources in Cache County. There are three major point sources (Logan, Hyrum, and Preston wastewater treatment plants) that contribute significant quantities of phosphorus. There are approximately 200 feedlots in the Cache Valley, and approximately 744,000 acres of land in Cache County which contribute runoff to the Bear River system. In Cache County, point sources contribute 28,200 (46%) kg BAP/yr, livestock runoff contributes 2,500 (4%) kg BAP/yr, and land runoff contributes 28,600 to 33,600 (50%) kg BAP/yr. Bioavailable phosphorus from land runoff was calculated by using export coefficients, which are usually accurate within a factor of two.

A comprehensive phosphorus management plan is required to reduce available phosphorus from all sources to minimize algal blooms in the receiving waters.

(109 pages)

INTRODUCTION

Overview

Excessive algal blooms associated with eutrophication cause water quality problems in many reservoirs that supply water for domestic, agricultural and industrial uses. Drinking water treatment costs will be increased due to the removal of the odors and tastes associated with algae, the production of trihalomethane precursor material, clarification problems and higher chlorine demand. Algal blooms will also degrade recreational aspects of reservoirs due to odors, aesthetics, hindrance of swimmers and clogging of boat motors. Certain species of cyanobacteria (blue-green algae) are especially harmful because they are undesirable as food to grazing zooplankton species, they cause reduced light penetration, and they reduce dissolved oxygen, resulting in anaerobic conditions, which may mobilize iron and manganese thereby causing additional potable water use problems (Miller et al., 1983).

Algal growth can be limited by the surrounding environment or by a specific nutrient. The Law of Tolerance states that environmental factors such as light, pH and temperature outside of certain ranges can limit growth (Odum, 1954). When excessive nutrients are present in a lake and cause eutrophic conditions, plant growth probably becomes limited by environmental (extrinsic) factors (e.g., light, temperature or pH) rather than by nutrients (Porcella and Bishop, 1975).

Liebigs Law states that growth of a plant is dependent on the amount of food-stuff which is presented to it in minimum quantity. In other words, the essential material available in amounts most closely approaching the critical minimum needed will tend to be the limiting

one (Odum, 1954). This law implies that a single nutrient limits growth at any one time (Miller et al., 1978).

Algae require carbon, nitrogen, phosphorus and sulfur in relatively large quantities for growth. Sulfur is essential in the formation of proteins and promotes the formation of chlorophyll. Sulfur is usually quite abundant in natural waters (Coker, 1954). Other nutrients that are required in trace amounts are potassium, magnesium, manganese, boron, and iron as part of the chlorophyll molecule or in its formation (Coker, 1954). These nutrients are usually not limiting because they are only needed in minute quantities and come into the water system by airborne particulate matter, and runoff from surrounding areas. Usually carbon, in the form of CO_2, nitrogen or phosphorus is the limiting nutrient, depending upon their relative abundance in the water (O'Kelly, 1973). For algal production to be reduced, the limiting nutrient concentration must be at the level where algal productivity is proportional to, or limited by that nutrient (Porcella and Bishop, 1975).

Weiss (1976) suggested that if the ratio of soluble inorganic nitrogen to total soluble phosphorus was greater than 13 the waters were phosphorus limited, when this ratio was in the range of 9-12, both nutrients were limiting, and when the ratio was below 8 nitrogen was limiting. Phosphorus is usually the limiting nutrient in lakes and reservoirs due to the additional nitrogen often made available through nitrogen fixation by cyanobacteria (Porcella and Bishop, 1975; Maki et al., 1984). Carbon is usually the limiting nutrient only if the algal production of the water body is so high that not enough CO_2 can be absorbed by the water from the atmosphere. Carbon is usually in such excess, that it would be impractical to control it as a limiting nutrient. For this reason, the present study focuses on phosphorus management.

Phosphorus can be in four forms: (1) orthophosphate (PO_4^{3-}) which is available for biological metabolism without further

breakdown; (2) polyphosphates which are mainly unavailable, but can undergo hydrolysis in aqueous solutions and revert to orthophosphate (within a period as short as a few hours); (3) organic phosphorus which can become available with the breakdown of organic matter by bacterial action and dissolution; and (4) particulate inorganic phosphates (either precipitated as mineral phosphates or sorbed to clay minerals) which require dissolution, usually by a pH decrease, to become readily available to algae as dissolved orthophosphate (Tchobanoglous, 1979; Porcella and Bishop, 1975; Van Wazer, 1973).

Bioavailable Phosphorus

Another important phosphorus category is bioavailable phosphorus (BAP) which is defined as the phosphorus which can be readily utilized by algae. Bioavailable phosphorus is usually only a fraction of total phosphorus (TP), while soluble reactive phosphorus (SRP) determined by the ascorbic acid method can be greater or less than BAP. Generally phosphorus is available to algae only as orthophosphate (Porcella and Bishop 1975).

The types of phosphorus are continually changing between available and unavailable forms. Lee et al. (1978) stated that phosphorus may become unavailable by sorption on sediments; coprecipitation with iron, aluminum oxides and calcium; and uptake by aquatic organisms. For the Bear River, Sorensen et al. (1987) stated that important phosphorus removal mechanisms may be chemical precipitation (especially with calcium) in the stream, sedimentation of phosphorus bearing solids, and biological immobilization. Lee et al. (1978) also stated that, the longer the transit time of phosphorus, the greater the primary productivity in the river; or the greater the suspended sediment load in the river, the greater will be the conversion of initially available P to unavailable forms. Edmundson (1972) found that the total phosphorus concentration during the summertime algal

bloom in eutrophic lakes, when most of the phosphorus is in the algal cells, appears to be closely related to the winter-dissolved orthophosphate concentrations, so phosphorus is cycling from the cells to the water column. The phosphorus is released back to the water column by the death and subsequent decomposition of the cells.

Cycling also occurs for particulate phosphorus. Organic particulate phosphorus can be made available by microbial mineralization in a relatively short time (Golterman, 1973). Some inorganic particulate phosphorus may be made available to algae through solubilization reactions (Golterman, 1973).

Phosphorus also cycles in lakes. Hooper (1973) stated that phosphorus compounds may become available in lakes by (1) in situ decomposition of the phytoplankton and zooplankton organisms themselves, (2) excretion by the plankton organisms, (3) regeneration from bottom sediments and transport to the photosynthetic areas, and (4) *in-situ* release of dissolved organic compounds by algae and bacteria and subsequent breakdown of the organic compounds into soluble phosphate. These processes may operate simultaneously or separately.

Phosphorus can also be cycled from sediments that contain iron. Insoluble ferric phosphate can be reduced to a soluble ferrous phosphate under anaerobic conditions (Holt et al., 1970). Sulfide has a stronger affinity to ferrous iron than phosphate, so phosphate will then become available in the ortho-phosphate form (Messer and Ihnat, 1983). This process will be reversed when aerobic conditions occur and phosphorus will become unavailable (Holt et al., 1970). Phosphorus continually cycles between available and unavailable forms.

Bioavailable Phosphorus Estimation

Algal bioassays (Miller et al., 1978) can be used to estimate BAP. One of the major disadvantages of BAP estimation is that it is a relatively expensive and time consuming test (Bradford and Peters, 1987). Peters (1981) felt that BAP should be approximated by total phosphorus because current assays for available phosphorus are at, best, cumbersome and at worst inadequate. U.S. Environmental Protection Agency (1980a) shared this opinion stating that BAP should be viewed with a high degree of uncertainty and as only a "ball park" approximation of algal productivity. Present practice in nutrient-response regressions and loading models implies that TP is an adequate estimator of bioavailable phosphorus, so BAP may not even need to be estimated (Bradford and Peters, 1987).

Rast and Lee (1982) found that acceptable estimates of eutrophication could be determined by inputting various total phosphorus concentrations into the U. S. Organization for Economic Cooperation and Development (OECD) phosphorus loading models. The United States portion of the North American study included 34 water bodies located primarily in the north central and northeastern United States (Rast and Lee, 1980). However, these empirical models are based on data with considerable variance and very broad confidence intervals. Many eutrophication and phosphorus management applications require more accurate and precise estimations of potential algal production potential (Bradford and Peters, 1987).

Studies conducted in New Jersey by Trama and McIntosh (1985) showed that BAP could be estimated by soluble reactive phosphorus (SRP) and total phosphorus (TP). They found for one watershed in New Jersey that ratios of BAP to TP ranged from 0.03 to 1.00 with a mean of 0.40 and the ratios of BAP to SRP ranged from 0.04 to 4.00 with a mean of 1.20.

Rigler (1973) stated that SRP analysis could have two errors when predicting orthophosphate, which is often considered 100% BAP: (1) filtration might damage delicate algal cells and release phosphate phosphorus or readily hydrolyzed phosphate esters into the filtrate; (2) H_2SO_4 could hydrolyze free phosphate esters and release ortho-phosphorus from fulvic acid-metal phosphates or from colloidal iron phosphate. Rigler (1973) also felt that the discrepancy between SRP and ortho-phosphorus will probably prove to be significant only in unpolluted lakes in which phosphorus is in short demand. Twinch and Breen (1982) found that SRP underestimated BAP in the soluble fraction at low concentrations and overestimated BAP at SRP concentrations ≥ 20 μg P/L.

Another chemical method to estimate BAP is with sodium hydroxide extractions which have been used to estimate BAP for sediment core samples containing iron. Sodium hydroxide phosphorus represents primarily phosphorus loosely bound to the surface of ferrous-ferric hydrous oxide gels, and which would be readily released upon reduction of the iron to Fe(II) when the sediment interface becomes anaerobic (Messer and Ihnat, 1983).

It is difficult to estimate BAP because phosphorus is continually changing forms. BAP estimation is only applicable for one instant in time and one set of conditions at the instant that it is measured. BAP is a very dynamic parameter with organic phosphorus becoming available over time with biochemical attack and dissolution, condensed phosphates becoming available by hydrolysis, and the effects of a pH change upon the precipitation and solubilization of particulate phosphate.

The algal bioassay is probably still the best way to estimate BAP since no chemical method has produced satisfactory estimations of BAP for all the various watersheds and their respective water chemistries throughout the world. In the bioassay procedure the water sample

needs to be sterilized to eliminate all life so that the test algae will neither be grazed upon nor have competition for the limiting nutrient. Filtration may underestimate BAP, since particulate BAP will not be accounted for. Therefore a whole water sample should be used. Phosphorus may precipitate with hardness in the water when autoclaved, resulting in a low BAP. Ultraviolet radiation may not destroy all the protozoa which can graze on the algae and result in a low BAP estimate. Two hundred thousand mWS of ultraviolet radiation is needed to destroy *Paramecium*, a type of protozoa (Hoffman, 1974).

Algal Bioassays

Algal bioassays are useful for determining algal productivity and nutrient limitation because they integrate the effects of all intrinsic factors such as the chemical composition of the water (Porcella and Bishop, 1975). The algal bioassay can estimate toxicity by using dilution techniques, and estimate the specific limiting nutrient via spiking with one or more nutrients and observing the response of the algae (Porcella and Bishop, 1975). The algal bioassay will measure the nutrient forms that are available for algal growth versus the total nutrient concentration as measured by chemical analysis (Miller et al., 1978). According to Miller et al. (1978) the U.S. Environmental Protection Agency's standard algal bioassay is meant for:

   1. Assessment of a receiving water to determine its nutrient status and sensitivity to changes in N and P loading. 2. Evaluation of materials and products to determine their potential stimulatory or inhibitory effects on algal growth in receiving waters. 3. Assessment of effects of complex wastes originating from industrial, municipal and agricultural point or non-point sources to define their impact upon receiving waters. (p. 5)

The standard algal assay has been modified slightly by various investigators to allow it to be used to estimate BAP (Bradford and Peters, 1987; Dorich et al., 1984).

Sources of Phosphorus

Cycling of phosphorus from sediments, degradation of organic phosphates and the hydrolysis of polyphosphates to orthophosphates may serve as phosphorus sources to algae, but the primary, though not always the most immediately important sources of phosphorus are those external to a lake (Porcella and Bishop, 1975). The external sources include domestic and industrial wastewater, dairies, feedlots, agricultural runoff and erosion which can all contribute phosphorus in varying quantities depending on their proximity to a water body and the relative proportions of each in the watershed. For example, Loehr (1974) estimated that 73% of the total phosphorus load to Lake Erie came from wastewater treatment plants. For a different watershed, agricultural runoff, contributed 52% of the total phosphorus load to Lake Canadarago, N. Y. Total phosphorus in runoff from forest land, range land and cropland was 0.03-0.9, 0.08, and 0.06-2.9 kg/yr/ha, respectively (Loehr, 1974).

Phosphorus from land runoff

In some watersheds land runoff is the biggest contributor of total phosphorus. Crop lands can be the major contributor of soluble phosphorus depending upon the fertilizer application rate and method (Porcella and Bishop, 1975). Often times range lands can have substantial quantities of decaying vegetation upon the surface which can be high in organic phosphorus and is highly susceptible to transport because of the low density of organic matter (Porcella and Bishop, 1975).

Grobler and Silberbauer (1985) felt that land use has been mistakenly identified in many studies as the cause of non-point pollution problems, whereas the controlling factor actually is land form (soil composition). Runoff from soils high in clay and organic matter usually have higher concentrations of TP than the original soil (Green, et al., 1978; Overcash and Davidson, 1980). For example runoff sediment in one situation contained 60 percent clay while the soil only contained 18 percent. The runoff had a higher TP content than the original soil due to the higher clay content (Sharpley, 1980a). Runoff will contain enriched concentrations of suspended clay because of its smaller particle size and dispersive properties; and enriched concentrations of organic matter because of its lower density, and smaller size. The particle density of most silicate minerals in soils varies between 2.60 and 2.75 g/cm^3 while organic matter falls in the range of 1.2 to 1.5 g/cm^3 (Brady, 1974). The density of clay particles is between 2 and 3 g/cm^3 (Weast, 1976). Clays are colloids with large surface areas and a negative charge (van Olphen, 1963). The negative charge keeps the clay particles from agglomerating, which enhances their dispersive action and slow settling characteristics (van Olphen, 1963). Even though clay soils have a high TP content, Mancini et al. (1983) found that most of this phosphorus was unavailable to algae.

The rocks in the soil can be important as well as the clay and organic matter content in predicting the quantity of phosphorus in the runoff. Grobler and Silberbauer (1985) showed that sedimentary rocks in South Africa contain more phosphorus than igneous rocks. The U.S. Environmental Protection Agency (1980a) and Tisdale and Nelson (1975) have given the following soil descriptions in regards to their erosivity and phosphorus characteristics for croplands: Sandy/gravel soils (1) do not erode easily, (2) have a low cation content, and (3) cause a general downward flow of water to the groundwater (high infiltration capacity). Thus phosphorus export via runoff is low.

Clay soils (clay loams, silt loams, etc.) can remove phosphorus by two mechanisms: (1) replacement of a hydroxyl group or (2) formation of a clay-cation-phosphate linkage. Clay soils have a high erodibility, and a low infiltration capacity. Therefore phosphorus export via runoff is high. Organic soils have a high nutrient content. As this soil is used for cultivation, it decomposes rapidly and organic phosphorus is mineralized. Therefore phosphorus export via runoff is high.

The relationship between TP and BAP in runoff is not completely understood. Schaffner and Oglesby (1978), stated that the consequence of using BAP instead of TP is to de-emphasize the importance of agriculture relative to the other sources since potentially large quantities of phosphorus can be tied up with particulate matter and not be available.

The time of year as well as soil composition are significant influences on the amount of phosphorus transported off the land. Spring snowmelt runoff has been shown to carry greater mass loads of phosphorus than runoff during other times of the year (Hanson and Fenster, 1969). Most snowmelt runoff phosphorus originates in plant residues that accumulate during winter on the frozen soil. The phosphate released by these residues does not have sufficient time to interact with the semi-frozen soil during the spring runoff period and the residues are easily removed with overland flowing water (Porcella and Bishop, 1975).

Not only is the soil composition and time of year in runoff important, but the length of time that this phosphorus associated with particulate matter will remain in solution is also important. A study done in Indiana determined that 90 percent of the suspended sediment was less than 20 μm in size (Dorich et al., 1984). Armstrong et al. (1979) determined that particles less than 2 μm will stay in suspension longer than 76 days and particles between 2 to 20 μm will

stay in suspension from 0.76 to 76 days. This means that phosphorus associated with clay (< 2μm) is likely to remain suspended.

Land runoff can also contribute phosphorus to irrigation return water which would be expected to contain relatively low concentrations of phosphorus. This is demonstrated in a study on the Snake river where the river had phosphorus concentrations of 21 μg/L, but drainage irrigation water, originally diverted from the river, had 4 μg/L. This is a 70% reduction of phosphorus in irrigation return water (Carter et al., 1971).

Another form of runoff can come from feedlots. Feedlot runoff is extremely high in phosphorus (10-620 kg TP/yr/ha) and bacteria which will result in water quality degradation of any nearby reservoirs or streams (Filip and Middlebrooks, 1976; Loehr, 1974). A common practice during early agricultural development was to locate feedlots where natural drainage aided the transport of runoff wastes to the nearest water body (Kreis and Shuyler, 1972).

Phosphorus from wastewater treatment plants

Wastewater treatment plants (WWTP's) along with land runoff are usually considered the major contributors of phosphorus to water bodies. Phosphorus in domestic wastewater comes from human wastes, food wastes and condensed inorganic phosphate compounds used in various household detergents. Raw domestic wastewater has typically 4 to 15 mg/L as total phosphorus which is composed of 1 to 5 mg/L as organic phosphorus and 3 to 10 mg/L as inorganic phosphorus. Approximately 10% of insoluble phosphorus is removed in primary treatment. The secondary treatment transforms polyphosphates and organic phosphates to ortho-phosphorus, but removes very little phosphorus (Tchobanoglous, 1979). This means that secondary treatment increases bioavailability of the phosphorus. Young et al. (1982) presents data that confirms the transformation of influent phosphorus

to BAP in the WWTP process by showing that influent wastewater is 60 to 80% BAP while the effluent total phosphorus was 83% BAP. Wastewater treatment plants can be designed and operated to remove phosphorus by precipitation with chemical addition, biological uptake (activated sludge) or land application. These processes usually follow secondary treatment.

Phosphorus Management

The relative contribution of phosphorus for each of the sources discussed needs to be determined before an effective phosphorus management plan can be designed and implemented. It is important to know BAP when ranking the importance of various sources in regards to eutrophication. For example, land runoff can contribute large total phosphorus loads with much of it unavailable to algae while, total phosphorus loads from WWTP's are almost 100% available.

Phosphorus management needs to control all of the sources now and in the future to assure a long term water quality improvement. Reduction in algal blooms may require some additional time after implementation of the phosphorus management plan. This time lag might be due to sediment storage of phosphorus and later release to the overlying waters by organic matter decay or redox reactions, it has not been satisfactorily demonstrated that a reduction in phosphorus concentration alone in a lake will cause an immediate reduction in algal blooms. Furthermore, if a phosphorus load reduction program incorporates the removal of mostly particulate phosphorus (e.g. from land runoff), a smaller improvement in phytoplankton-related water quality would be expected than if a greater portion of the available phosphorus load (e.g. wastewater) were removed (Rast and Lee, 1982).

One of the first phosphorus management decisions made by many states was to ban phosphates in detergents thinking that this reduction in phosphorus load would improve the water quality. Maki et

al. (1984) found that even though detergents comprise 20 to 30% of the phosphate load in wastewater; the ban of phosphate in detergents did not measurably improve the water quality in the Great Lakes area. Maki et al. (1984) showed that it can require up to a 92% reduction of phosphorus in wastewater treatment plant effluents before a measurable improvement of water quality can be measured in a water body. Usually, a comprehensive phosphorus management program needs to incorporate control of non-point sources and point sources. In 1976, sixteen states had phosphorus limits between 0.1 to 2 mg/L for wastewater treatment plant effluents (U.S. Environmental Protection Agency, 1976). The State of Utah regulates its water bodies by stating that streams cannot exceed 0.05 mg/L PO_4-P and that lakes and reservoirs cannot exceed 0.025 mg/L PO_4-P (Utah Deptartment of Health, 1978). The present study investigated the Bear River System in Utah to determine the BAP contributions from the various sources.

The Phosphorus Situation for the Bear
River System in Utah

The Utah Water Development Plan

The "State Water Plan for the Lower Bear River Basin" proposes Bear River water be developed for irrigation, municipal, industrial, wildlife/waterfowl, recreation, hydro-power, and flood control. A number of water related studies have been conducted on the Bear River since 1970. The development plans began in earnest in 1980 when the Bear River compact was signed between the states of Idaho, Wyoming and Utah (N.E. Stauffer, Jr., Utah Division of Water Resources, personal communication, June 6, 1988).

The Colorado River will supply the Salt Lake Valley 130,000 to 136,000 acre-feet of water for municipal and industrial use upon completion of the canal. In 1986 it was decided that the Salt Lake

Valley might need an additional 100,000 acre-feet from the Bear River system by the year 2005 for municipal and industrial use (N.E. Stauffer, Jr., Utah Division of Water Resources, personal communication, June 6, 1988).

The Utah Association of Conservation Districts (1986) stated the following facts regarding Bear River water usage: in Box Elder and Cache Counties it is estimated that 42,000 acres of a potential 119,000 acres of irrigable land could be brought under irrigation in the next 30-40 years which would require a diversion of about 111,000 acre-feet of new water; a minimum addition of 124,000 acre-feet of storage water is needed at the Bear River Migratory Bird Refuge to provide adequate management in existing areas and 13,000 to 15,000 acre-feet for additional habitat development.

There are over 17 reservoirs with a capacity over 50 acre-feet in Box Elder and Cache Counties, but with the exception of Willard Bay, each has a capacity under 20,000 feet. Thirty five potential reservoir sites have been selected. The nine best sites are at Washakie, Lampo, Honeyville, West Bay, and East Promontory in Box Elder County, and at Amalga, Cutler, Millcreek, and Avon in Cache County. It has not yet been decided which reservoirs will be built (Utah Association of Conservation Districts, 1986).

Potential eutrophication problems in the proposed reservoirs

Water quality data has been collected on the Bear River system by the Utah Water Research Laboratory (UWRL). This data was used in computer simulation modeling of the eutrophication potential of the proposed reservoirs at Amalga, Honeyville, Oneida, Mill Creek and Avon. It was determined that all of these proposed reservoirs could have at least short periods of eutrophication, and that phosphorus was the limiting nutrient. These models were run using 45 and 85 percent of the total phosphorus as bioavailable phosphorus, but the actual BAP

load of the Bear River and its tributaries was not known (Sorensen et al., 1986).

Methods of eutrophication control

If these reservoirs were to be built, there could be two strategies for eutrophication control. The limiting nutrient can be controlled at the source and/or the effects of eutrophication can be minimized in the water body. Of course, the effects of eutrophication may be negligible or even desirable and no control may be needed at all. This "do nothing" option is seldom acceptable, however. Eutrophication prevention management options are generally the most desirable and are usually directed toward controlling the limiting nutrient.

Since phosphorus was determined to be the limiting nutrient in the Bear River system, the control strategies should be based on its control. Wastewater treatment plants are one of the major sources of phosphorus. Phosphorus can be removed from WWTP effluents by physical treatment which can include sedimentation, flotation and filtration; chemical treatment which can include precipitation with lime, aluminum or iron, chemical-biological precipitation, ion exchange; and biological treatment which can include activated sludge or oxidation ponds. Removal efficiencies range from 70 % for chemical addition followed by sedimentation to 99 % for chemical addition followed by sedimentation and filtration (Nesbitt, 1973).

For a comprehensive management plan to be effective in controlling the limiting nutrient, phosphorus from land runoff should be minimized by using agricultural practices such as minimal tillage that minimize surface disturbance and by placing highly erodible lands into a permanent cover situation (Sorensen et al., 1987).

The second control strategy of minimizing eutrophication in the water body can be accomplished by dredging, sediment treatment for phosphorus inactivation, aeration and/or weed harvesting. Dredging

will: (1) reduce the frequency of summer overturns in very shallow lakes by increasing the volume of the hypolimnion layer; (2) result in a larger volume of hypolimnetic water which in turn contains a larger quantity of oxygen; and (3) reduce the water temperature to increase oxygen solubility, and decrease biological kinetic rates (Stefan and Hanson, 1980). The general concept is that shallow eutrophic lakes can be dredged to such a depth that phosphorus released from the sediments into the hypolimnion is not recycled to the photic zone by lake overturn. This will reduce the standing crop of algae (Stefan and Hanson, 1980). Dredging has been used in several locations, including Wisconsin, where varying improvements of water quality have been seen (Dunst, 1980). *In situ* sediment treatment involves adding a material such as alum to increase the phosphorus binding capacity of the sediment (Barroin, 1980). This process needs to be further evaluated to determine its effectiveness. Aeration increases the dissolved oxygen in bottom waters which minimizes the redox reactions of phosphorus with Fe, Al, and Mn and the cycling of phosphorus from bottom sediments (Pastorok et al., 1980). Harvesting will remove weeds and algae and thus increase the aesthetics of the water body and remove the phosphorus that is contained within the plants (King and Burton, 1980). Harvesting is a short term remedy because the algae will grow back since the phosphorus concentration is usually not significantly reduced.

Water quality managers must decide whether to control eutrophication or pay the increased costs associated with using eutrophied water as a water supply.

OBJECTIVES

Bioavailable phosphorus is the limiting nutrient for algal growth in the Bear River system, Utah. This study involved two processes to better understand BAP. These were: (1) to develop a BAP estimation procedure; and (2) to measure BAP from the various sources in the Bear River system.

The following objectives were addressed in this study:

1) To develop a procedure for estimation of bioavailable phosphorus for the Bear River.

2) To determine sources of bioavailable phosphorus in the Bear River system.

3) To determine the relative contribution of bioavailable phosphorus from the various sources.

4) To determine if bioavailable phosphorus estimation is site specific.

MATERIALS AND METHODS

Bear River Site Description

This study will investigate the Bear River system, shown schematically in Figure 1, which arises from 3048 meters elevation on the north slope of the Uintah mountains in northeastern Utah. It travels back and forth for 676 kilometers through Wyoming, Idaho and Utah (Utah Water Research Laboratory, 1974). The Bear River basin area is approximately 241 kilometers long from north to south and 161 kilometers wide from east to west and encompasses 1.9×10^{10} square meters of primarily agricultural land. The area receives precipitation of 20 to more than 102 cm/year (Utah Water Research Laboratory, 1976).

The rocks in the mountains which feed the Bear River are largely sedimentary while the valleys contain alluvial material from Lake Bonneville. The area is highly calcareous resulting in a hard water river (Utah Water Research Laboratory, 1974).

This study will investigate the water quality of the Bear River from Bear Lake to the Great Salt Lake with emphasis on the portion of the Bear River system contained in Cache Valley which includes Franklin County, Idaho, and Cache County, Utah. Cache Valley contains most of the the proposed reservoirs and sources of phosphorus that will impact these reservoirs. Franklin County only includes the northern portion of Cache Valley and is relatively unimportant in respects to its contribution of phosphorus to the Bear River system.

Cache County encompasses approximately 753,500 acres of which 40% is federally or state owned and used for forest and grazing land. Fifty-seven percent is privately owned and primarily used for agriculture (Cundy and Conant, 1982). The rest is mainly urban with
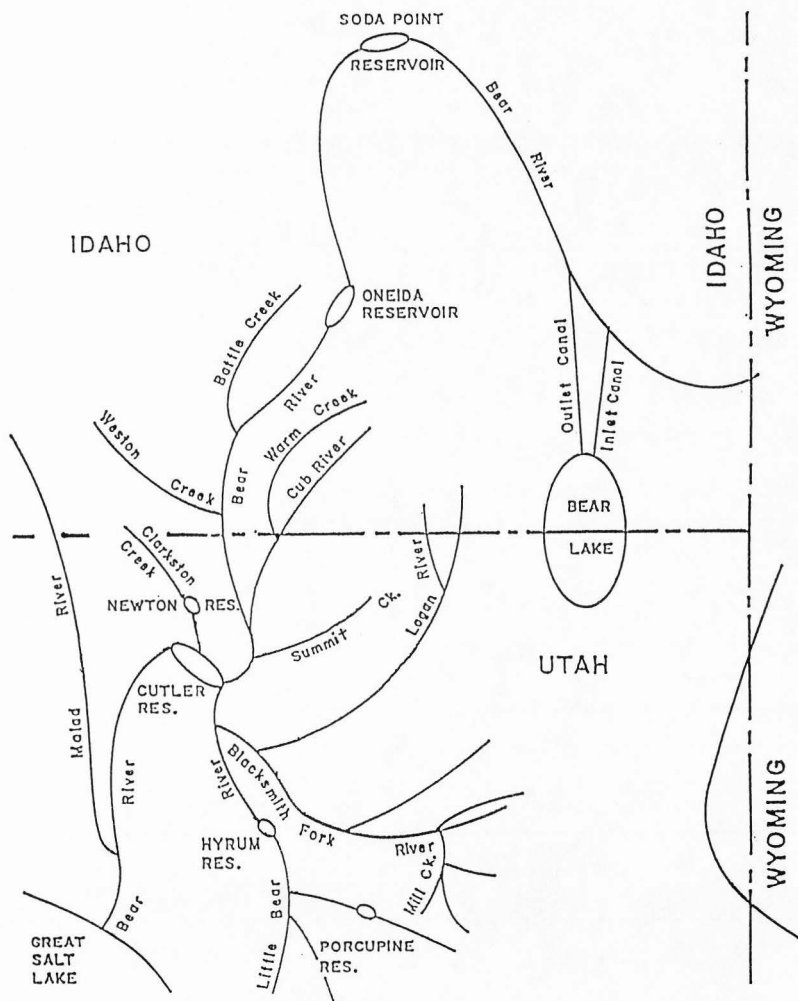
Figure 1.   Map of the major rivers and reservoirs in the Bear River Basin.

Logan being the largest town (population 30,600 in 1985, Cache Valley Data, 1988). The Bear River system supplies water primarily for irrigation purposes, but may in the future be used for municipal and industrial uses.

Approximately one-half of the surface water in Cache Valley enters via the Bear River. The rest of the flow comes from the Cub, Logan, Blacksmith Fork, and Little Bear rivers, minor tributaries, and groundwater (Cundy and Conant, 1982). The surface water leaves Cache Valley through Cutler reservoir. Reservoirs in the valley include Cutler, Hyrum, Porcupine, Newton, and three small reservoirs on the Logan River.

Table 1 summarizes 7 years of record of the chemistry of the Bear River at the USGS gage (# 10118000) below Cutler Reservoir, 64 kilometers upstream from the Great Salt Lake. The Bear River at this point had a pH between 7.4 and 10.5 with a hardness range of 194 to 360 mg/L as $CaCO_3$ (Table 1).

There are several wastewater sources listed in Table 2 that impact the water chemistry, including the phosphorus load of the Bear River and its tributaries. Cache Valley has three wastewater treatment plants (WWTP's) that discharge greater than 1 MGD to surface water. The City of Logan lagoons discharge, on the average, 10 MGD while both Preston and Hyrum WWTP's discharge approximately 1 MGD. There are several smaller lagoon systems that discharge less than 0.25 MGD. The non-domestic wastewater sources include trout farms, slaughter houses, and vegetable canning. E. A. Miller (beef slaughter house) and Whites Trout Farm are the largest industrial sources discharging 0.75 and 8.6 MGD respectively. Approximately 200 dairy and cattle feedlots are in Cache Valley (Wieneke et al., 1980).

Table 1. Water chemistry data for the Bear River below Cutler Reservoir
          (1/77 to 12/83)

Parameter	Mean	Std. Dev.	Minimum	Maximum
Temperature (°C)	11.4	8.7	0.0	26.0
pH	8.2	0.6	7.4	10.5
Total Suspended Solids (mg/L)	52.2	48.3	0.5	175.0
Conductivity, 25 °C (μmhos)	924.4	426.8	450.0	2140.0
Total Dissolved Solids (mg/L)	541.6	241.4	252.0	1272.0
Total Hardness (mg/L as CaCO3)	290.6	39.9	194.0	360.0
Total Alkalinity (mg/L as CaCO3)	253.6	32.8	179.0	315.0
Dissolved Calcium (mg/L)	60.3	9.5	42.0	81.0
Total Iron (mg/L)	0.518	0.419	0.080	1.919
Total Kjeldahl Nitrogen (mg/L)	0.863	0.951	0.200	5.000
Total Phosphorus (mg/L)	0.117	0.047	0.050	0.200
Ortho-Phosphorus (mg/L)	0.056	0.027	0.020	0.100

Data from Sorensen et al. 1986

Table 2. Description of wastewater point sources in the Cache Valley which discharge to the Bear River system

Source	Discharge Type	Flow MGD	Receiving Water
Preston City	Domestic	0.5-1.0	Worm Cr. to Cub R. to Bear R.
Del Monte	Canning	0.54 (July-August)	Cub R. to Bear R.
Richmond City	Domestic	0.02	Cub R. to Bear R.
Logan City	Domestic	12.4	Logan R. to L. Bear R. to Bear R.
White's Trout Farm	Fish Waste	8.6	L. Bear R. to Bear R.
E. A. Miller	Slaughterhouse	0.75	Spring Cr. to L. Bear R. to Bear R.
Hyrum City	Domestic	0.6-1.2	Spring Cr. to L. Bear R. to Bear R.
Wellsville City	Domestic	0.07	L. Bear R. to Bear R.

Erosion and Runoff Study Site Description

An erosion and runoff study was done in the Cache Valley to determine the phosphorus contribution of these sources to the Bear River. This study investigated four streambank locations that had recent erosion (slippage) and four different soil surfaces in a runoff study that included a variety of land uses. The streambank samples were taken at Battle Creek (N 42° 08' 48"; W 111° 54' 52"), Weston Creek (N 42° 01' 38"; W 111° 57' 21"), Bear River at Amalga (N 41° 53' 27"; W 111° 52' 26"), and Little Bear River below Hyrum reservoir (N 41° 38' 01"; W 111° 53' 19"). Table 3 shows that the streambank soils studied were clay loam, loam, and sandy loam.

The runoff study selected four sites for their proximity (within 0.5 mi.) to a stream and having been identified by the Soil Conservation Service (USDA, Soil Conservation Service, Logan, UT Field Office) maps as having high erosion potential. The surface runoff samples were taken at sites above Cutler Reservoir (N 41° 51' 03"; W 112° 02' 25"), adjacent to Weston Creek (N 42° 01' 50"; W 111° 58' 22"), near the Cub River (N 41° 54' 46"; W 111° 51' 52") and near the Blacksmith Fork River near Anderson Ranch (N 41° 35' 56"; W 111° 37' 05"). The site above Cutler Reservoir soil was classified as a Wheelon silt loam with 30-50% eroded slopes, and was a grazing land with small tufts of vegetation and a gravelly soil. The site adjacent to Weston Creek was classified a Bingham gravelly loam with 0-60% slopes, and was composed of loose soil that had been recently planted with no rocks or vegetation on the surface. The site near the Cub River, was classified as a Trenton silty clay loam with 8-20% eroded slopes, and had wheat stubble on the soil. The Blacksmith River Fork site, was classified as a Yeates Hollow extremely rocky silt loam with 30-70% slopes, and was grazing land with sage brush and grasses that had approximately 20% open space. Table 3 shows that the runoff site soils are predominantly silty.

Table 3. Classification of soils by percent clay, silt and sand
for the streambank and runoff sites in Cache Valley

Sample Type	Site	USDA % clay <0.002 mm	USDA % silt <0.04 mm	USDA % sand <2.0 mm	Soil Type
Streambank	Battle Cr.	32.5	35.0	32.5	clay loam
	Weston Cr.	25.0	46.0	29.0	loam
	Bear R. (Amalga)	14.5	21.0	64.5	sandy loam
	Little Bear R.	12.0	19.0	69.0	sandy loam
Runoff	Blacksmith Fork	22.3	60.5	17.3	silt loam
	Weston Cr.	18.0	37.5	44.5	loam
	Cub R.	32.5	51.5	16.0	silty clay loam
	Abv. Cutler Res.	22.3	50.0	27.8	silt loam

Sampling

Sampling collected BAP data on the WWTP's, for the land runoff and streambank studies, and the Bear River and its tributaries. River sampling stations were selected to "isolate" reaches of the river that might be important in terms of phosphorus loading. Tributary locations, adjoining land uses, streambank characteristics, point source discharges, U.S.G.S. gage stations and accessibility were all evaluated when determining sampling locations.

Sample containers were cleaned by washing with a sodium bicarbonate solution and rinsing with 6 N HCl followed by several rinses with double-deionized water.

Some samples were collected by submerging a clean 1 gallon polyethylene container in the stream to a depth of 2 to 4 inches. Other samples were collected from the bridge by filling a well-rinsed polyethylene 2 gal. bucket suspended by a rope into the stream and using the water thus collected to rinse and fill a 0.5 gal. polyethylene bottle. Water samples collected in the bucket were transferred to the bottle quickly to minimize settling of suspended material in the water.

Bottles containing the samples were placed in ice chests with ice and transported to the UWRL, within 12 hours. Samples were stored under refrigeration (5° C) until analyses were complete. Samples for algal assays were then sterilized by autoclaving, filtration, UV or gamma radiation. Aliquots of sterilized and unsterilized water were analyzed for various phosphorus fractions. All samples were either analyzed or appropriately filtered and preserved within 72 hours. All analyses were completed within 7 days (Sorensen et al., 1987).

A special study used streambank samples to determine the possible contribution that streambank erosion may have upon the phosphorus load of the Bear River system. The streambank samples were taken at one

foot intervals along three transects which encompassed the entire vertical height of the eroded streambank. Samples of approximately 20 cm^3 each, were collected by pushing a 60 cc monoject disposable syringe with its end cut off into the streambank. Lengths of the three vertical transects were 38, 26 and 20 feet at Battle Creek; 12, 12 and 13 feet at Weston Creek; 26, 33 and 30 feet at Bear River, Amalga; and 12, 10.5 and 11 feet at Little Bear River below Hyrum Reservoir Samples from all transects were composited in the field for each site.

Another special study was conducted to determine the possible contribution of phosphorus to the Bear River system from land runoff. The soil runoff samples were collected by setting up a rain simulator that was 1.5 feet square and positioned on a platform 2.5 feet above the ground. Rainfall intensity varied from 0.22 to 0.56 in/min due to varying storage tank heights above the rain simulator at individual sites. Runoff collection times ranged from 6 to 15 minutes. The slope of the plots varied from 14 to 40 percent (Table 4). Rainfall continued until one gallon of runoff was collected down slope of the rain simulator by a flat piece of metal with sides which funneled the water into the container. Rainfall was simulated at two plots for each site.

Physical Analyses

The river flow was determined at each of the sampling sites at each sampling time, so that mass transport of phosphorus could be determined. River discharge was determined by using USGS gage stations or was estimated by multiplying the average river velocity by the cross-sectional area (Dunne and Leopold, 1978). The cross-section profile was determined by using surveying techniques to determine water surface elevation in terms of a reference point and river bed profile which was determined by using a sonar depth finder (Sorensen

Table 4. Runoff plot characteristics for four sites (2 plots/site)
where runoff was collected from simulated rainfall

Site	Plot #	Plot size (in x in)	Slope	Rainfall intensity (in/min)	Rainfall depth (in)
Abv. Cutler Res.	1	27 x 19	0.1	0.3	4.0
	2	31 x 19	0.2	0.6	5.0
Weston Creek	1	21 x 18	0.2	0.4	2.5
	2	24 x 24	0.2	0.2	2.8
Cub River	1	26 x 17	0.2	0.3	7.0
	2	23 x 21	0.2	0.2	8.0
Blacksmith Fork	1	27 x 24	0.4	0.3	3.0
	2	22 x 21	0.4	0.2	2.5

et al., 1987). The average velocity was determined by multiplying the
velocity of an orange passing between two fixed points near the known
cross section by 0.85 (Hynes, 1970). Stream flows were determined by
setting up spreadsheets (Microsoft[TM] Excel, Microsoft Corp.) with the
flow profiles and using the stream depth and velocity for flow
determination.

At the same time as samples were collected and flow determined,
water temperature was taken with a glass mercury centigrade
thermometer. Samples were then brought back to the lab and electrical
conductivity was determined using a YSI model 33 S-C-T conductivity
meter.

A special study investigated the relationship of particle size
range to BAP. Several samples had suspended solids and size fractions
determined upon them by initially passing the sample through a 250 μm
brass sieve to remove debris, sand, and relatively large aggregates of
particles. Approximately one liter of the sieved sample was placed in
an ultrasonic bath for two minutes to break up the aggregates.
Following sonication the sample was vacuum-filtered through a 30 μm

opening nylon mesh (Spectra/mesh) fabric. To separate the 30 to 10 μm range particles, the filtrate from the 30 μm filter was passed through a 10 μm filter (Nucleopore polycarbonate membrane). For the 30 to 0.5 μm range, the filtrate from the 30 μm filter was passed through a 0.5 μm Whatman 934AH glass fiber filter. For the 10 to 0.5 μm range, the sample filtrate from the 10 μm filter was then passed through a 0.5 μm Whatman 934AH glass fiber filter. Standard Methods (APHA, 1980) procedures for suspended solids determination were followed for filter preparation and residue determination. The volume of liquid that passed through the filter and the additional weight due to the solids on the 10 and 0.5 μm filters were used to determine the concentration of each size range.

Another special study involved determining phosphorus contributions from streambanks and land runoff to the Bear River. Composite samples of streambank soil were thoroughly mixed and air dried under a laboratory fume hood at room temperature for approximately 24 hours. During drying, large stones and vegetation were picked out and the soil clumps were broken up by hand and sieved through a #10, U.S. standard testing sieve to remove all material larger than 2 mm. After air drying, subsamples were weighed and then oven dried for 24 hours at 103° C for percent moisture determination. A suspension of soil from each site was made by mixing 18 grams of air dried sample and approximately 1 liter of doubly deionized water (DDW). These samples were then sonicated (Bransonic 12 ultrasonic bath) for 5 minutes. The suspension was then passed through a 325 mesh brass sieve (~45μ); a 30 μ Spectramesh® nylon cloth and then diluted to 3600 mL with DDW. These samples were then tumbled at 30 rpm for one hour. A portion of each sample was centrifuged, the supernatant was decanted and soluble reactive phosphorus (SRP) and total phosphorus were determined on the supernatant. The remaining suspension of each sample was sparged with nitrogen for 90 minutes,

capped, sealed with paraffin and gamma irradiated (2 to 3.5 Mrad in 20 hrs.) at a commercial facility (Isomedix (Utah), Inc., Sandy, UT).

Samples from the runoff study were sonicated with a Biosonic IV sonicator (high setting, mid scale) and mixed with magnetic stirrers for 5 minutes. The sample was allowed to stand for one minute so that course materials would settle out; the supernatant suspension was then passed through a 250 μm brass sieve. The sediment that had settled was discarded. Approximately one liter of the sample was passed through a 45 μm brass sieve into a clean half-gallon (polyethylene) bottle and stored overnight at 5° C. The sample was then re-sonicated and shaken for five minutes to re-suspend the sample. Four hundred milliliters (minimal) of the sample was passed through a 30 μm mesh size Spectramesh® nylon filter which had been silicone-glued onto a 3.5 mesh brass sieve. The filter had to be washed several times with tap water to remove the sediment from the clogged filter. After the filter was rinsed following the last sample, a 400 mL aliquot of tap water was passed through the 30 μm cloth and analyzed for TP and SRP as a blank. Tap water was used instead of DDW because tap water was used in the rain simulator. A composite sample was made for each site by combining 180 mL from each plot with 3140 mL DDW for an approximate 1:17 dilution. Diluted samples were placed in clean one gallon polyethylene bottles and tumbled at 30 rpm for 1 hour. Duplicate 30 mL aliquots from each site were removed immediately and centrifuged at 2600 rpm for 10 min., the supernatant was then passed through a pre-rinsed 0.45 μm Gelman Sciences GN-6 filter and analyzed for SRP. Separate 30 mL aliquots in duplicate, were placed in polypropylene bottles and digested for total phosphorus. The remaining portion of the samples were poured into glass bottles, sparged with nitrogen gas, capped, sealed with paraffin, and sent, together with the streambank soil suspensions, for gamma radiation sterilization.

Most samples from both the streambank and runoff studies were very turbid. To ensure adequate light penetration for algal growth in the determination of BAP, the intensity of light passing through an empty 500 mL bioassay flask was measured. The samples were diluted so at least 50% of the light passing through the empty flask would pass through 100 mL of sample in the same flask. One streambank site (Battle Creek) and two runoff sites (Blacksmith Fork River and Weston Creek) required dilution.

Water soluble phosphorus was determined on the streambank and runoff site soil samples by the method of Olsen and Sommers (1982). This method involves shaking a soil sample with distilled water, filtering the suspension and then measuring phosphorus by the ascorbic acid method. The filtrate contained some suspended clay which required turbidity corrections to be made. The turbidity corrections were made in addition to the Olsen and Sommers (1982) procedures, by subtracting the absorbance of samples from which the combined reagent had been withheld from the absorbance of samples after full color development had occurred. The corrected absorbance readings were then used to determine water soluble phosphorus concentrations based on a standard curve prepared with the sample.

Total phosphorus analysis of soil samples was performed by the Soil, Plant and Water Analysis Laboratory at Utah State University, using the method by Olsen and Sommers (1982). Percent calcium carbonate in the soil was determined using a pressure-calcimeter method (Nelson, 1982) except a 1000 mL flask was used and the pressure reading was taken on a mercury manometer once the reading had stabilized. Percent calcium carbonate is based on the principle that carbonates release CO_2 when an acid is added. This release of CO_2 will increase the pressure in a closed system (measured on the manometer) and is related to percent of $CaCO_3$ by using standards. The determination of pH upon the soils was done by a saturated soil paste

method (Richards et al., 1969). The texture was determined by the hydrometer method (Gee and Bauder, 1986).

Chemical Analysis

The pH of the water samples was determined using a Corning model 130 pH meter. The glass pH electrode was calibrated using a phosphate buffer of pH 7 and a borate buffer of pH 10 (VWR Scientific).

A special study involved determining total organic carbon (TOC) on several water samples to determine if TOC could be an indicator of BAP. Total organic carbon was determined by passing the sample through a 250 μm nominal opening brass sieve (U.S.A. Standard Testing Sieve No. 60) and then determining the organic carbon on an Oceanography International Carbon Analyzer (model 05248) using persulfate oxidation with infrared absorption detection of CO_2 (Oceanography International, Inc., 1977).

All of the water samples had SRP and TP analysis done upon them to compare with the BAP determinations. SRP samples and digested TP samples were analyzed by the manual ascorbic acid method (Strickland and Parsons, 1972). Total phosphorus digestions were carried out according to the American Public Health Association (1980) persulfate digestion protocol.

Sterilization for Bioavailable

Phosphorus Estimation

Sterilization of the water sample is the first step required in BAP estimation. Initially UV and gamma irradiation were investigated as two alternative methods of sterilization. Ultraviolet radiation was used initially because no known by-products are formed and a Model SPF teflon-tube ultraviolet device manufactured by Ultraviolet Technology Inc. (El Toro, California) was available (Harris, 1986).

Three experiments were performed to investigate the use of UV radiation for bioassay sample sterilization. A sample collected from the Bear River at Amalga, UT was treated by placing it in a pyrex dish to a depth of about 1 inch and exposing it to UV radiation from a germicidal lamp for 1 hour. Another aliquot was sterilized by autoclaving at 121° C for 30 minutes. Since the UV dose was not sufficient to kill the native algae and protozoans in this sample, other methods were tried. In the second experiment, the model SPF UV reactor, consisting of several 3 ft sections of teflon tubing, each surrounded by four G30T8 (General Electric Co., Schenectady, NY) low pressure mercury germicidal lamps, were used. The water sample (collected from the Bear River at Honeyville, UT) was pumped through 1/4" inner diameter teflon tubes at a rate of approximately 0.12 gal/min and collected in a sterile polypropylene container. Growth of native algae was evident in this sample as well. A third sample was collected from the Bear R. at Benson, UT. This sample was pumped through the large teflon tubing of the reactor described above. Five to ten gallons of water were pumped through the reactor and discarded before turning on the germicidal lamps. After the lamps were turned on, the first ten gallons were discarded and the next three gallons collected in sterile polypropylene bottles at a rate of 5 gal/min.

In summary, the experiments showed that ultraviolet radiation was able to kill all the endogenous algae, but was unable to kill all the protozoa. Protozoa can graze on the algae and result in an underestimation of BAP since it is based on algal growth, so UV radiation was judged unacceptable and a new sterilization procedure had to be found.

Gamma radiation had been used successfully by Dorich et al. (1984) for sterilizing concentrated sediment suspensions prior to BAP estimations. It was decided to evaluate gamma radiation for sterilization of the whole water samples from the Bear River system.

Whole water samples, and in several instances water samples which were filter-sterilized with 0.2 μm membrane filters, were placed in glass bottles and transported to a commercial facility (Isomedix (Utah) Inc., Sandy, UT) for gamma radiation sterilization. Samples received a minimum dose of 2.5 to 3.5 Mrad (cobalt-60 source) during an exposure period of approximately 20 hr.

The first algal bioassay performed with gamma-sterilized water and filter-sterilized, non-irradiated water produced no growth in any of the gamma-irradiated samples. The Microtox™ test (Microbics Corp., Carlsbad, CA) was used to evaluate possible toxicity in those samples. It was suspected that low concentrations of hydrogen peroxide, produced by ionization of oxygen dissolved in the water during the gamma radiation treatment, may have persisted in the samples after irradiation resulting in toxicity to the algae. Despite subsequent efforts to strip oxygen from samples by sparging with N_2 gas prior to irradiation and thus preventing the formation of hydrogen peroxide, toxicity problems were frequent. The use of the enzyme peroxidase (Type IV and VI, Sigma Chemical Co., St. Louis, MO) to break down peroxide in irradiated samples and eliminate toxicity was evaluated. Type VI was initially used because of its higher purity. The less expensive type IV was later evaluated and found to be phosphorus free and capable of detoxifying the sample water. After initial success an experiment was conducted to determine the optimum time and enzyme concentration to eliminate the toxicity.

Algal Bioassays and Calculation
of Bioavailable Phosphorus

Algal bioassays were performed following a modified version of the Environmental Protection Agency (EPA) Algal Assay Procedure (AAP) protocol (Miller et al., 1978). The method used to determine BAP was to measure _in vivo_ fluorescence of algal chlorophyll with known

additions of phosphorus and comparing these with the fluorescence of
the algae developing in the unamended river sample.

Prior to analysis, approximately three liters of sample were
sparged with nitrogen gas for 90 minutes to remove oxygen so as to
minimize formation of hydrogen peroxide during irradiation. The
sample was then gamma-irradiated with a dose of 2.5 to 3.5 Mrad (^{60}Co
source). Untreated samples and gamma-irradiated samples were analyzed
for both TP and SRP to determine any differences caused by the
radiation. Radiated samples were treated with 2000 units/L of
peroxidase (Type IV or VI, Sigma Chemical Co., St. Louis, MO) for 48
hours to eliminate any toxicity due to hydrogen peroxide formation
during irradiation. A Microtox™ test (Microbics Corp., Carlsbad, CA)
was then done on the sample to confirm toxicity removal. Samples
showing less than 5 units of light intensity lost relative to the
control were found not to be toxic to the algae.

Aliquots of treated and toxicity free river water were introduced
into triplicate bioassay flasks, enriched with N and P, and inoculated
with *Selenastrum capricornutum* (10^3 cells/mL in the test flasks).
After it was discovered that samples spiked with 1 mg nitrate-N per
liter often exhibited evidence of nitrogen limitation, nitrogen spikes
were adjusted to 20 times the TP concentration in the sample. Two
levels of phosphorus additions and a nonenriched control were used in
order to verify linearity of algal growth response to the P additions.

In vivo fluorescence measurements were made daily after the third
day of incubation of the test flasks and continued until the peak in
growth occurred. Linear regression analysis of maximum fluorescence
versus added P was performed. The negative value of the intercept of
the regression equation represents bioavailable phosphorus in the
sample. This standard additions method gave precise BAP estimates
(i.e., high correlations between fluorescence and added P) for many
samples. However, as the data set increased in size, the occurrence

of data scatter and poor fit from regression analysis in a significant number of samples became obvious. A second method for estimating BAP was, therefore, investigated. A standard curve was prepared with AAP medium in which the P concentration was adjusted over a range of 10 to 250 µg/l. Triplicate flasks for each concentration were inoculated with 10^3 cells of *S. capricornutum*/mL (final concentration). A second set of flasks was inoculated with 10^5 cells/mL Gamma-irradiated whole water samples amended with nitrogen were also inoculated with *S. capricornutum* at both high and low levels. Bioavailable P was determined by comparing maximum fluorescence of the sample with the calibration curve prepared by linear regression of maximum fluorescence versus P in the standards. Seven to ten day old cultures of *S. capricornutum* maintained according to the AAP protocol of Miller et al. (1978) were used in all assays.

pH Study

During the algal bioassay test the sample pH can increase by as much as two units. Two studies were done to investigate the effects of a pH increase upon the algal bioassay. The first study was to determine the effect of pH change on soluble reactive phosphorus. A surface water sample was collected from the Bear River below Cutler Reservoir for the first study. A three liter portion of the sample was filtered through a 0.45 µm Gelman membrane filter prior to use in pH experiments. Triplicate 100 mL aliquots of filtered sample were introduced into 500 mL erlenmeyer flasks for each pH level. Tris/HCl buffers (2M) were prepared for pH control in the 8.00 to 9.00 range. Carbonate/bicarbonate buffers (2M) were used for pH values between 9.20 to 10.25. Each was prepared so that the desired sample pH would be attained by adding 1 mL of the buffer to 100 mL of river water resulting in a 0.02 M buffer. Appropriate buffers were added and pH was measured using a Corning model 130 pH meter. SRP and pH were

measured after 30 min. and again after 2 hours had elapsed since buffer addition.

The second pH study involved monitoring and controlling the pH while determining BAP during an algal assay test. Compressed carbon dioxide and air from an aquarium air pump were used to control the pH for the algal assay test. The mixture of CO_2 and air was adjusted by variable area flowmeters until the desired pH was obtained. The gas mixture passed through water to saturate the gas. It then entered the algal assay 500 mL erlenmeyer flasks through Kimble 1 mL disposable sterile pipettes. Yellow eppendorf pipette tips were attached to the ends of the 1 mL pipettes to produce a fine stream of gas bubbles. Sterile gauze-wrapped cotton was used as stoppers for the flasks.

Luxury uptake

Luxury uptake was investigated to determine if this could minimize the effects of phosphorus precipitation due to the increase in pH of the medium during the assay. The normal assay procedures were followed except *Selenstrum capricornitum* inoculum sizes of 6.8 X 10^4 cells/mL and 1.3 X 10^5 cells/mL were done along with the usual inoculum size of 10^3 cells/mL Replicates of three to four were done for each inoculum size. In a study conducted on March 22, 1988 the algae from a stationary phase culture was harvested by centrifugation and put into a phosphorus free nutrient medium for a phosphorus starvation period of two days before the bioassay.

Data Reduction and Statistics

Data reduction and statistics is important in evaluating and understanding the significance of the sampling data obtained by the previously described methods. The sampling data was compiled by using the Microsoft™ Excel program on the Apple Macintosh computer.

Ninety five percent confidence intervals were determined for the BAP estimation methods of standard additions and standard curve by procedures in Kleinbaum and Kupper (1978) and the Statview statistical computer program (Feldman and Gagnon, 1986) for the Apple Macintosh computer. Procedures in Kleinbaum and Kupper (1978) were also used to determine whether lines were significantly different by comparison of the slopes and y-intercepts. Analysis of Variance (ANOVA) of the Statview program was used to determine the significance of the affects of various sterilization treatment scenarios upon BAP. Where ANOVA indicated significant ($P \leq 0.05$) differences among the treatments, the Least Significant Difference was used to identify specific significant differences between means. The t-test, also of the Statview program was used to test the significance ($P \leq 0.05$) of the difference between the means of two data sets. The Cricket graph program (Rafferty and Norling, 1986) for the Apple Macintosh computer was used to determine regression lines and R-values for the data. A correlation was determined to be statistically significant if at the 95 % confidence level ($P \leq 0.05$), the slope was different from 0 and when the correlation coefficient, R was greater than 0.71. The value of r^2 would then be greater than 0.5, suggesting that more than 50% of the data variance was explained by the regression line.

Power Peaking

A special study was done on power peaking to determine its effects upon phosphorus transport. Hydraulic output can be increased by power peaking which is accomplished by increasing the flow through the turbines. The water from the turbines is discharged to the river resulting in a higher flow. Both Cutler and Oneida reservoirs are operated by Utah Power and Light Company for hydroelectric power generation in a power peaking mode. Sorensen et al. (1987) describes the power peaking study in the following paragraphs:

In cooperation with Utah Power and Light Company and the U.S. Geological Survey we sampled for ortho and total phosphorus over a power peaking cycle for both Cutler and Oneida Reservoirs. On December 3, 1986, at 7:00 a.m., Cutler Reservoir was discharging minimal flow (approximately 20 cfs). The flow was increased hourly in increments of 1000 cfs until a maximum of 4000 cfs was reached. Water flow was then decreased back to minimum flow at the same rate. Surface water samples were collected at 30 minute intervals at the USGS gauging station 800 yards downstream from the power plant tail race during the time that the flow rate was changing. After the flow had stabilized at each increment of change, samples were collected at approximately 1 foot intervals from the river bottom using a sampler at points approximately 1/3 and 2/3 the distance across the river.

At 9:00 a.m. on December 4, 1986, a power peaking cycle was begun at Oneida Reservoir. Flow was increased hourly in 1000 cfs increments until a maximum of 2800 cfs was reached and then decreased back to minimum flow (40 cfs) in 1000 cfs increments. River sampling was begun at 6:30 p.m. at the USGS Gage Station at the Utah-Idaho border, approximately 32 river miles downstream from the reservoir. A permanently installed sampler, used for suspended sediment monitoring by the USGS, was used to collect water samples at 30 minute intervals from approximately one foot above the river bottom. Surface water samples were collected hourly for the duration of the cycle. On the following evening we intended to collect samples at the Benson Bridge (66.6 miles downstream) but were unable to detect any increased flows. (pp.29-30)

RESULTS AND DISCUSSION

Indicators of Bioavailable Phosphorus

Total and soluble reactive phosphorus

Bioavailable phosphorus estimation is time consuming and expensive, so indicator parameters were investigated to determine if they could provide adequate estimations of BAP. Initially total and soluble reactive phosphorus were investigated. Table 5 and Figures 2, 3, 4, 5, 6, and 7 present the phosphorus data obtained for the Bear River System. Figure 2 is a plot of TP plotted versus BAP for river water and the data has a statistically significant (P $\leq$ 0.05) correlation (R = 0.81). This correlation is questionable, however, because 6 of the 10 BAP concentrations are from sites that have WWTP's discharging within 1 to 7 miles upstream of them; 58 % of the sites have less than 8 μg/L; and the two data points on the far right may control the correlation more than is justified. The data in Figures 3 and 4 do not show a statistically significant (P > 0.05) correlation of SRP with BAP (R = 0.59) and SRP with TP (R = 0.52) respectively for river water. Sample data from wastewater effluents shows that TP plotted against BAP (Figure 5) and SRP plotted against BAP (Figure 6) have R-values of 0.97 and 0.96 respectively which suggest a good correlation, but the slopes of the regression lines are not significantly (P > 0.05) different from zero. The data plotted in Figure 7 shows a statistically significant (P $\leq$ 0.05) correlation (R $\approx$ 1.00) between SRP and TP for wastewater samples.

In summary, there does not appear to be a reliable correlation between BAP and SRP, and BAP and TP for the Bear River system. In some areas of the U.S., total phosphorus is a good estimator of BAP, but for the Bear River this is not the case (Figure 2).

Table 5.   Soluble reactive, total and bioavailable phosphorus data
           for the Bear River system

River	Sample Location	Date	SRP μg/L	TP μg/L	BAP μg/L
Bear	Weston Cr.	5/27/87	52.	224.	<8.
	West of Fairview, UT-ID gage	5/11/87	0.	388.	<8.
	West of Fairview, UT-ID gage	7/12/87	20.	100.	<8.
	West of Fairview, UT-ID gage	8/18/87	19.	67.	<8.
	Abv Cutler Res., W. of Benson	5/11/87	0.	138.	<8.
	Abv Cutler Res., W. of Benson	5/27/87	37.	224.	<8.
	Abv Cutler Res., W. of Benson	7/12/87	18.	130.	<8.
	Blw Cutler Res., UPL gage	5/11/87	14.	125.	13.
	Blw Cutler Res., UPL gage	5/27/87	34.	208.	<8.
	Blw Cutler Res., UPL gage	7/12/87	14.	131.	<8.
	West Side Canal	8/18/87	12.	163.	<8.
	W. of Honeyville	5/11/87	0.	142.	41.
Cub	South of Richmond	8/18/87	17.	128.	18.
	Worm Cr.	6/21/87	129.	274.	156.
L. Bear	S. Fork Blw Davenport Cr.	5/11/87	13.	24.	<8.
	S. Fork Blw Davenport Cr.	5/27/87	37.	181.	12.
	Blw Hyrum Res., Hwy 101	7/12/87	68.	106.	<8.
	Abv Logan R. confluence, 6th S.	7/12/87	66.	137.	<8.
	Spring Creek	8/18/87	289.	332.	141.
	Benson Marina	5/27/87	74.	227.	52.
	Benson Marina	7/12/87	211.	458.	274.
	Benson Marina	8/18/87	130.	336.	139.
Blacks. Fork	Blw Anderson Ranch, gage	5/11/87	7.	17.	<8.
Logan	Blw Logan lagoon outfall	7/12/87	14.	490.	302.
Point Source	Richmond Lagoon effluent	6/21/87		2800.	2450.
	Hyrum WWTP effluent	6/21/87	4890.	5190.	5830.
	Wellsville Lagoon effluent	6/21/87	862.	1130.	860.
	Logan Lagoon effluent	6/21/87	1700.	1960.	1518.
	Preston WWTP effluent	8/18/87	2300.	2520.	1303.
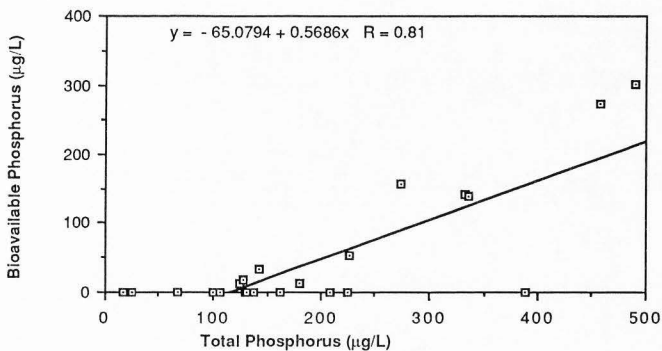
Figure 2.   Total phosphorus data plotted against bioavailable phosphorus data in river water (Table 5).
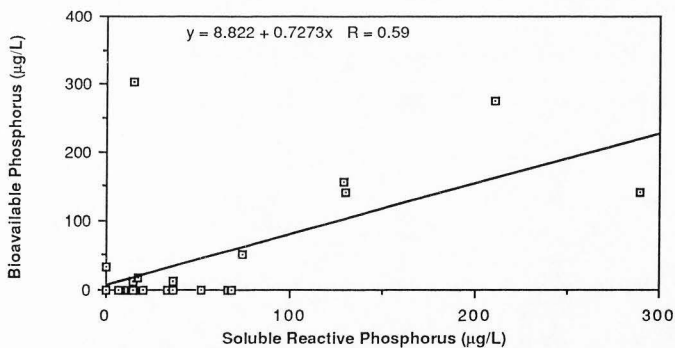
Figure 3.   Soluble reactive phosphorus data plotted against bioavailable phosphorus data in river water (Table 5).
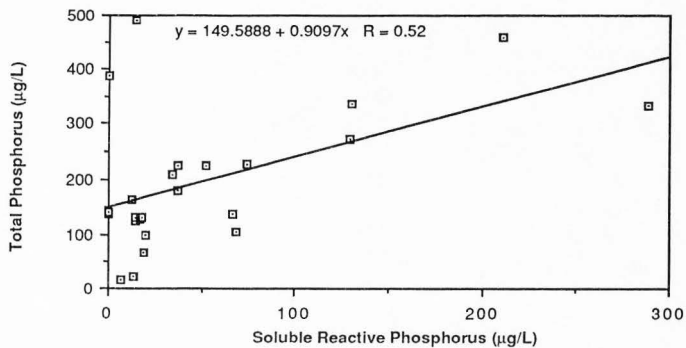
Figure 4. Total phosphorus data plotted against soluble reactive phosphorus data in river water (Table 5).
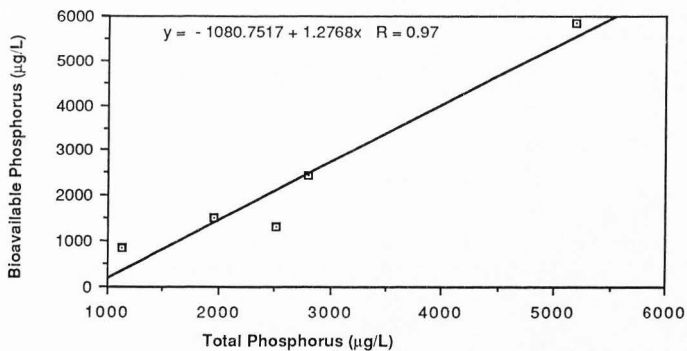
Figure 5. Total phosphorus data plotted against bioavailable phosphorus data for wastewater point sources (Table 5).
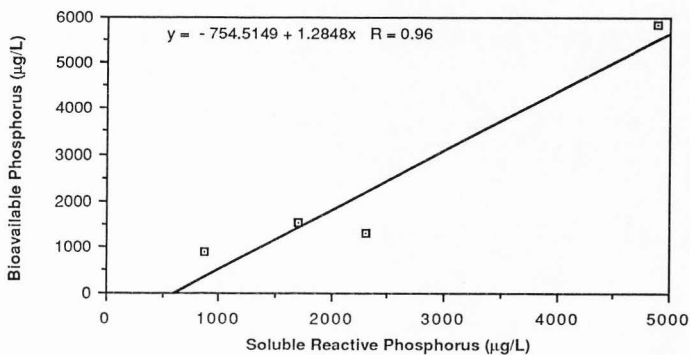
Figure 6.   Soluble reactive phosphorus data plotted against bioavailable phosphorus data for wastewater point sources (Table 5).
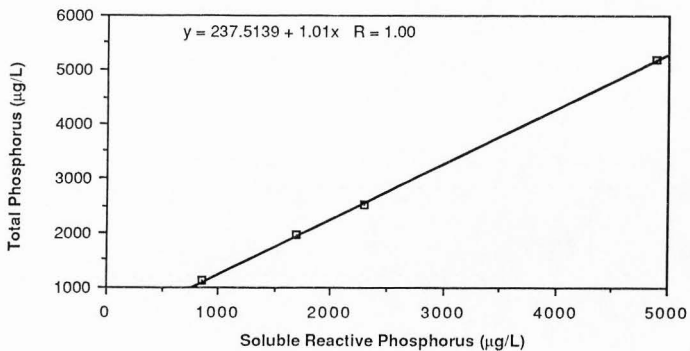
Figure 7.   Soluble reactive phosphorus data plotted against total phosphorus data for wastewater point sources (Table 5).

Comparison of the fractions of BAP, SRP and TP between river water and wastewater effluent samples is shown in Table 5. BAP was approximately 88% of TP and 98% of SRP for wastewater samples. While for river samples with BAP greater than 8, BAP was approximately 41% of TP and 120% of SRP. The WWTP's effluent is highly available due to the complex waste in the influent being converted to available forms by the treatment process. Also, a low particle load in the WWTP's effluent relative to river water in the Bear River basin, could explain that SRP is 72% of TP for WWTP's, while for Bear River water SRP is only 34% of TP.

Total organic carbon

Total organic carbon (TOC) was also investigated as a possible indicator for BAP. Walker (1983) used data collected from lakes and reservoirs in the United States to determine a positive correlation between TP and TOC measurements. This appears reasonable because land and livestock runoff can contain large concentrations of organic matter.

Table 6 presents the TOC and BAP data collected for the Bear River system. The observed relationship between TOC and BAP in river water is shown in Figure 8. The regression analysis of these data suggests that there is a statistically significant ($P \leq 0.05$) relationship ($R = 0.82$) between these measurements. However, all but one of the data points with BAP > 8 $\mu g/L$ are from stream or reservoir sample sites that have wastewater discharged 1 to 7 miles upstream of them. This suggests that the relationship cannot be applied to river waters in general. It is noteworthy that the correlation between TOC and BAP in wastewater effluents is not statistically significant ($P \leq 0.05$; $R = 0.05$; Figure 9). The lack of correlation between TOC and BAP in WWTP effluents is not surprising because one of the main objectives of a WWTP is to remove organic carbon, but other nutrients such as phosphorus are not necessarily removed in the same proportion. It was

Table 6.   Total organic carbon and bioavailable phosphorus data
           for the Bear River system

River	Sample Location	Date	TOC (mg/L)	BAP (μg/L)
Bear	Weston Cr.	5/27/87	5.	<8.
	West of Fairview, UT-ID gage	7/12/87	6.	<8.
	Worm Cr.	6/21/87	8.	156.
	Richmond Lagoon effluent	6/21/87	27.	2450.
	Abv Cutler Res., W. of Benson	5/27/87	5.	<8.
	Abv Cutler Res., W. of Benson	7/12/87	7.	<8.
	Blw Cutler Res., UPL gage	5/27/87	7.	<8.
	Blw Cutler Res., UPL gage	7/12/87	7.	<8.
L. Bear	S. Fork Blw Davenport Cr.	5/27/87	9.	12.
	Whites Trout Farm Effluent	6/21/87	4.	<8.
	Hyrum WWTP effluent	6/21/87	4.	5830.
	Blw Hyrum Res., Hwy 101	7/12/87	5.	<8.
	Wellsville Lagoon effluent	6/21/87	10.	860.
	Abv. Logan R. confluence, 6th S.	7/12/87	5.	<8.
	Benson Marina	5/27/87	6.	52.
	Benson Marina	7/12/87	10.	274.
Logan	Logan Lagoon effluent	6/21/87	10.	1518.
	Blw Logan Lagoon outfall	7/12/87	11.	302.
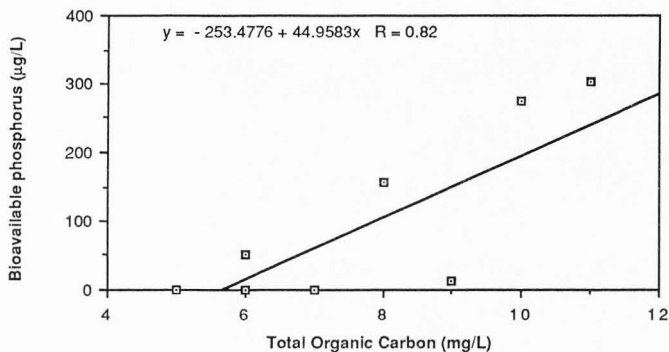
Figure 8.   Total organic carbon data plotted against bioavailable phosphorus data in river water (Table 6).
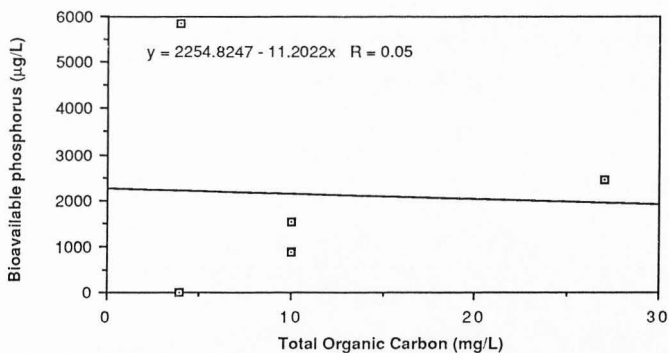
Figure 9.   Total organic carbon data plotted against bioavailable phosphorus data for wastewater point sources (Table 6).

decided that the TOC and BAP relationship would not be further investigated since there does not appear to be a reliable correlation and the precision of our TOC analysis ($\pm$ 1 mg C/L) may not be adequate to allow resolution of changes in TOC relative to BAP changes in river water.

Particle size range

The last potential indicator investigated was the correlation between BAP and particle size ranges. Several researchers have proposed that BAP may be approximated by adding 20% of particulate phosphorus to ortho-phosphorus (Dorich et al., 1980; Rast and Lee, 1982; U.S. Environmental Protection Agency, 1980a). The high surface area of clays, fine silt, and colloidal size organic matter may have appreciable amounts of sorbed phosphate that may be available to algae through ion exchange or solubilization reactions. Dorich et al. (1984) found that aggregates that contained the least amount of clay also contained the least amount of total phosphorus. These solids associated with available phosphorus can easily be transported along a river system.

Particle size ranges of 30 to 0.5 μm, 30 to 10 μm, and 10 to 0.5 μm were investigated to determine if there was a significant correlation with BAP for the Bear River system. These size ranges were separated by filtration after sonication to breakup the aggregates (Table 7). The data in Figure 10 shows that BAP concentrations are not correlated with suspended solids in any of the size ranges composed of individual particles. Dorich et al. (1984) found that soil aggregate size (groups of particles) did not correlate with phosphorus concentration because of the similarity of the primary particle size distribution within the aggregates. Even with no measurable BAP (Figure 10), the suspended solids concentrations in all of the size fractions were quite variable. There also appears to be

Table 7. Ranges of suspended solids and bioavailable phosphorus data for the Bear River system

River	Sample Location	Date	Suspended Solids (mg/L) for the ranges			BAP µg/L	TP µg/L	SRP µg/L
			30 to 0.5 µm	30 to 10 µm	10 to 0.5µm			
Bear	West of Fairview, UT–ID gage	7/12/87	31.8	14.0	19.3	<8.	100.	20.
	West of Fairview, UT–ID gage	8/18/87	24.7	8.7	13.0	<8.	67.	19.
	Abv Cutler Res., W. of Benson	7/12/87	63.2	29.0	38.6	<8.	130.	18.
	Blw Cutler Res., UPL gage	7/12/87	35.2	11.8	13.6	<8.	131.	14.
	West side canal	8/18/87	50.4	5.8	33.3	<8.	163.	12.
Cub	South of Richmond	8/18/87	26.6	7.7	12.9	18.	128.	17.
L. Bear	Blw Hyrum Res., Hwy 101	7/12/87	2.4	0.9	1.9	<8.	106.	68.
	Abv Logan R. confluence, 6th S.	7/12/87	30.3	14.0	17.0	<8.	137.	66.
	Spring Creek	8/18/87	7.8	4.0	3.7	141.	332.	289.
	Benson Marina	7/12/87	51.2	4.0	39.0	274.	458.	211.
	Benson Marina	8/18/87	38.4	5.0	35.0	139.	336.	130.

no correlation between total phosphorus or, SRP and suspended solids as shown by the data in Figures 11 and 12.

In summary for all of the indicators investigated, there appears to be no reliable correlation between TP, SRP, TOC, particle size range, and BAP. Although the data analyzed to date is limited, it seems unlikely that a useable index of BAP in Bear River system waters can be derived by considering TOC and/or particle size distributions along with SRP and total phosphorus data.
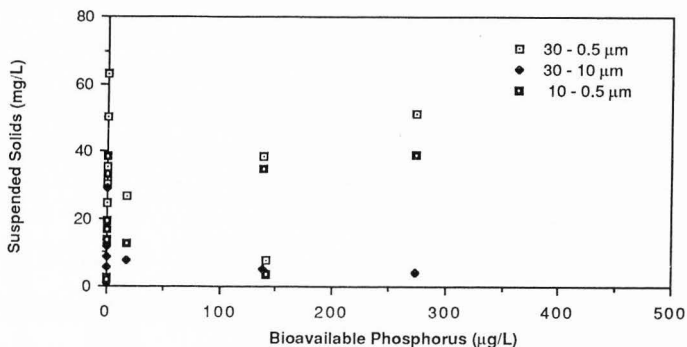
Figure 10.   Suspended solid ranges plotted against bioavailable phosphorus data in river water (Table 7).
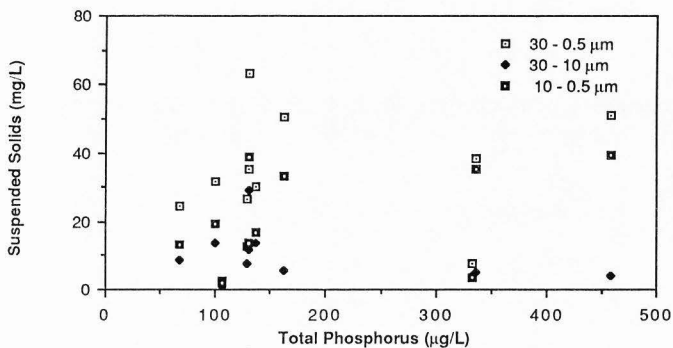
Figure 11. Suspended solid ranges plotted against total phosphorus data in river water (Table 7).
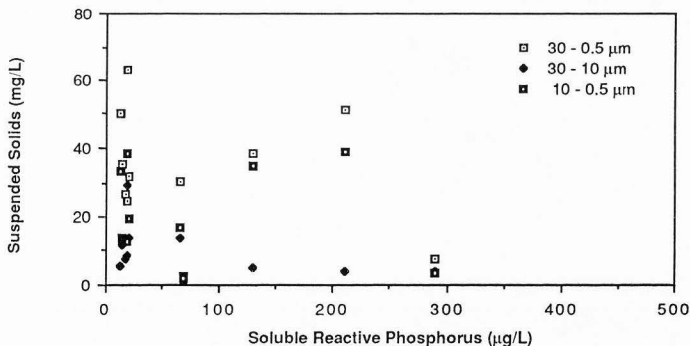
Figure 12. Suspended solid ranges plotted against soluble reactive phosphorus data in river water (Table 7).

Bioavailable Phosphorus Estimation

Water hardness affects upon BAP

As discussed earlier, one of the disadvantages of BAP estimations was that they are time consuming, requiring up to two weeks, and over this time period, a portion of the initially available phosphorus can become unavailable due to precipitation, adsorption or settling. It is also difficult to relate a BAP estimation back to where the sample was taken because during the bioassay, the sample pH can increase by as much as two units. This increase in pH can cause phosphorus to precipitate out, therefore resulting in a low BAP estimate for the river.

Currently BAP is determined by a modified *Selenastrum capricornutum* Printz Algal Assay Bottle Test. This assay can raise the pH from natural conditions of approximately 8 up to 10 because of the algal photosynthetic consumption of carbon dioxide. REDEQL-EPAK (Ingle et al., 1980), an aqueous chemical equilibrium computer model, was used to determine the effects that a pH increase would have upon phosphorus. Bear River data was used in the model as the pH was increased from 8 to 10. The model calculated that 99.9% of the phosphorus would be precipitated with calcium at equilibrium conditions with a pH of 10. Equilibrium conditions represent limits, which the system is moving towards, but rarely reaches, so probably less phosphorus is precipitated with calcium in the Bear River than predicted by REDEQL. Snoeyink and Jenkins (1980) state that if, in natural waters, phosphate levels were controlled by equilibrium with calcium hydroxyapatite ($Ca_5OH(PO_4)_3$), the phosphate levels would be so low that phosphate would not be a concern as a limiting nutrient for algal growth. The rate of formation of hydroxyapatite is very slow with intermediate calcium-phosphate containing species forming first allowing phosphorus concentrations to far exceed predicted equilibrium concentrations (Snoeyink and Jenkins, 1980). Phosphorus precipitation

during the bioassay probably makes BAP estimates low. However, river water pH is also frequently higher than 8, and phosphorus precipitation is likely to occur in the stream, so bioavailability could be low naturally.

In using the REDEQL model it was seen that both pH and calcium were important factors in the precipitation of phosphorus and its resultant unavailability. To determine if there was a correlation between calcium and pH a stepwise regression was done on data collected by Greene et al. (1975) for the Snake River in Idaho, Oregon, and Washington. The Snake River is located in the same geographical area, and the data had a relatively broad range of pH, BAP and TH. More data was available from the Snake River study than has been collected for the Bear River. Greene et al. (1975) used the algal bioassay test to collect data for the water quality study on the Snake River Basin to determine the limiting nutrients. Bioavailable phosphorus numbers were estimated by using the algal yields given by Greene et al. (1975) in the following ratio:

$$BAP = \frac{50* \ (Y_N/ \ (Y_{N+P}))}{1- \ (Y_N/ \ (Y_{N+P}))} \tag{1}$$

Where: Y_N = algal yield with 1 mg/L of nitrogen added

Y_{N+P} = algal yield with 1 mg/L of nitrogen and 50 μg/L of phosphorus added

50 = 50 μg/L of phosphorus added

This ratio was used on only 14 of the 18 sample results reported because these were the only ones with either nitrogen or phosphorus limiting for algal growth. Total phosphorus, SRP, total hardness, hydrogen ion concentration and iron concentration were also given for the sample sites.

These parameters and their interactions for 14 sites were used in a stepwise multiple regression analysis for BAP using the Statview computer program (Feldman and Gagnon, 1986). The variables and interactions evaluated were: BAP, TP, SRP, TH, H^+, Fe, SRP/TP, $TH*H^+$, $SRP*TH*H^+$, $TP*SRP$, $TH*H^+*(SRP/TP)$, $TP*TH*H^+$, $TP*TH$, $SRP*TH$. The regression procedure identified two important variables: (1) the product (interaction) of SRP, total hardness and hydrogen ion concentration ($SRP*TP*H^+$) and, (2) total hardness. The r^2 value of the equation with both of these variables was 0.941.

$$BAP = a(SRP*TH*H^+) + b(TH) \qquad (2)$$

These statistics indicate that the interaction of SRP, total hardness and pH are major factors in the availability of phosphorus in the Snake River.

A multiple regression analysis was then done using the same parameters for the Bear River data as was done on the Snake River. Six data points were evaluated for the Bear River. The regression procedure identified the interaction of TP, total hardness, and hydrogen ion concentration (pH) as describing the largest fraction of the BAP sample variance ($r^2 = 0.879$).

$$BAP = a(TP*TH*H^+) \qquad (3)$$

This analysis does indicate that total hardness and pH play a major role in regulating BAP for the Bear River system. The total hardness for the Bear River system is between 180-240 mg/L as $CaCO_3$. By comparison, the surface water hardness of the northwestern U.S. is between 0-120, the East coast has less than 60, and the Great Lake states are between 60 to 180 with a few areas getting as high as 240 mg $CaCO_3$ hardness/L (Geraghty, et al., 1973). The difference in surface water hardnesses across the U.S. could provide one reason why

in some areas of the country TP correlates closely with BAP while in other parts there are no correlations. In hardwater areas the TP would be mainly in unavailable forms because of phosphorus being precipitated with the hardness contributing ions. The Bear River system has very hard water in comparison to most of the areas in which algal assays for estimation of BAP have been conducted.

Selection of sterilization procedure

The algal bioassay requires the sample to be sterilized, so that no protozoa can graze upon the algae, resulting in a low BAP estimation. Water hardness can be an important factor in picking a sterilization procedure due to the potential precipitation of phosphorus.

Filtration is a common way to sterilize a sample in preparation for an algal assay. Filter sterilization may underestimate BAP by eliminating phosphorus associated with particulate matter. Several studies have estimated that 20% of particulate phosphorus is available (Dorich et al., 1980; Rast and Lee, 1982; U.S. Environmental Protection Agency, 1980a). It was decided that a possible 20% error was unacceptable in the current study. O'Kelly (1973) stated that measurements of phosphate available for growth based on soluble phosphorus (filtered) alone does not take into account the presence of phosphorus in particulate detritus or absorbed on particles of silt or clay, and both are available for algal growth. It was felt whole water samples would provide a better estimation of bioavailable phosphorus for the Bear River system.

Autoclaving is another fairly standard method for sterilization of water samples used in algal assays and has proven satisfactory in many studies, especially those conducted in the Southeastern United States (Raschke and Schultz, 1987). In a 1984 experiment on Bear River water, separate aliquots of the sample were spiked with nitrogen (N)

and phosphorus (P) prior to autoclaving. The effect of autoclaving was very pronounced. Samples spiked with N and/or P before autoclaving produced very little algal growth (*in-vivo* fluorescence), whereas samples spiked after autoclaving supported significantly higher maximum algal populations. Autoclaving resulted in the precipitation of P and a low algal population. In another experiment, soluble reactive phosphorus decreased from 52 to 14 μg/l upon autoclaving in a sample collected from the Bear River in March, 1986.

UV treatment, another sterilization procedure, did not affect SRP concentration in the March, 1986 sample or in a sample collected in April. Algal bioassays were then set up with both autoclaved and UV treated samples in August, 1984 and March, 1986. The effect of autoclaving on BAP was dramatically demonstrated in both experiments: BAP estimates were < 1 and 6 μg/l in autoclaved samples; and 28 and 79 μg/l in UV treated samples. Although, UV treatments did not kill all the native algae (protozoa were not observed, but possibly present), we feel that the comparisons are valid, since unspiked UV treated samples with and without *Selenastrum capricornutum* reached about the same maximum fluorescence.

Gamma radiation sterilization was also investigated. The positive aspects of gamma radiation are that it allows use of a whole water sample without causing precipitation of phosphate such as autoclaving, and does complete sterilization which UV cannot consistently do. Gamma radiation appears to be the best sterilization method for the Bear River system. However, some chemical changes due to gamma irradiation were observed.

Total phosphorus concentration did not change with gamma radiation, but Figure 13 shows that 83 % of the samples showed an increase in SRP after gamma radiation. Ten, fifty and eighty percent of the samples had a SRP increase of 63 μg/L (276%), 17 μg/L (22%), and 6 μg/L (8%) respectively. Figure 14 shows that 17% of the samples

showed a decrease in SRP after gamma radiation. Ten percent of the samples decreased by 7 µg/L (21%).

Table 8 shows the effects of gamma radiation upon samples taken on March 22, 1988. SRP was significantly (P ≤ 0.05) increased after gamma radiation for the Benson Marina site while the SRP's for below Cutler Reservoir were not significantly different (P ≤ 0.05). For the Benson Marina BAP estimation using a low inoculum size, the gamma irradiated sample is significantly higher (P ≤ 0.05) than gamma plus filtration and filtration alone, but filtration alone is not significantly (P ≤ 0.05) different from gamma plus filtration. The BAP for the high inoculum size for Benson Marina shows that the gamma radiated sample is significantly higher (P ≤ 0.05) than gamma plus filtration and filtration alone. Filtration alone for BAP is significantly higher (P ≤ 0.05) then filtration followed by gamma. For the below Cutler Reservoir site the BAP's for the low inoculum were all below the detection level (i.e., 8 µg BAP/L). The BAP for the high inoculum shows that the gamma radiated sample is significantly higher than gamma plus filtration (P ≤ 0.05) and filtration alone (P ≤ 0.05). Filtration alone for BAP is significantly lower (P ≤ 0.05) than gamma plus filtration.

The general trend seen in Table 8 is that gamma radiation alone results in a higher BAP than gamma with filtration and filtration alone. One possible explanation for this is that filtration removes particulate phosphorus which could become available upon gamma radiation for a non-filtered sample. This may overestimate BAP if the gamma radiation causes more particulate phosphorus to become available than would actually occur in the natural system.

Not only does gamma radiation cause a change in SRP, but it can cause the formation of hydrogen peroxide which is toxic to the algae.
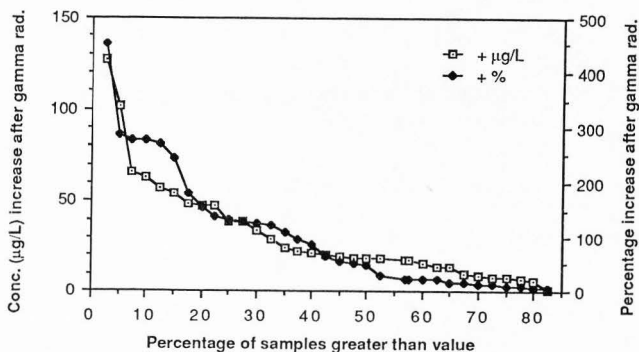
Figure 13.   Soluble reactive phosphorus concentration increase with its accompanying percentage increase due to gamma radiation.
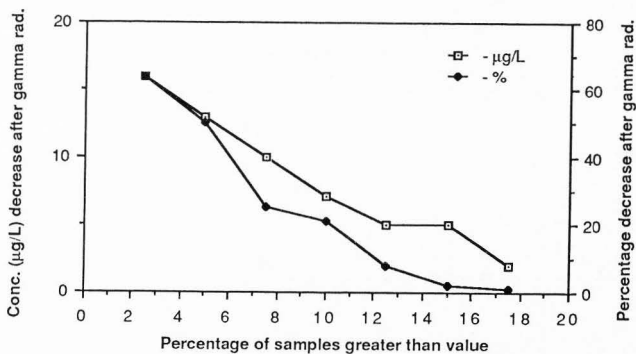
Figure 14.   Soluble reactive phosphorus concentration decrease and its accompanying percentage decrease due to gamma radiation.

Table 8. Effects of gamma radiation upon an average SRP and BAP for three sterilization scenarios on samples taken on March 22, 1988

Location	Inoculum Size	Treatment*	SRP before gamma (µg/L)	SRP after gamma (µg/L)	BAP (µg/L)
Benson Mar.	Low	a			132.7
	Low	b	150.5	173	88.7
	Low	c			95.7
	High	a			246.7
	High	b	150.5	173	148.0
	High	c			176.0
Bear R. Blw.					
Cutler	High	a			49.3
Reservoir	High	b	72	68.5	42.3
	High	c			26.7

* a: is gamma radiated only; b: is filtration followed by gamma radiated; c: is filtration only.

Sorensen et al. (1987) discussed the toxicity induced in samples by gamma radiation sterilization as follows:

> Due to radiation induced toxicity of the water, bioavailable phosphorus estimates were obtained for only filtered, non-irradiated samples for the first two sampling periods. For the April 21 collection, estimates ranged from 3.6 (Little Bear) to 13.3 µg P/L (Bear R., Honeyville). Bioavailable phosphorus estimates for the May 12 samples were 7.6 (Little Bear) and 26.7 µg/L (Honeyville). Microtox tests performed on the filtered samples collected May 12 indicated toxicity (23 - 31 Microtox units) in the irradiated samples. We collected a sample from the Bear River above Oneida Reservoir on June 10 and sparged it with N_2 for 1 hour to strip oxygen from solution hoping to prevent hydrogen peroxide formation through the reactions of singlet oxygen (Foote, 1968). Microtox tests indicated no toxicity and a trial bioassay resulted in good growth of *S. capricornutum* in this sample. The next two set of bioassay samples (23 June and 13 August) were also sparged with N_2 prior to irradiation. Good growth of *S. capricornutum* was

exhibited in most samples in each set, but toxicity was
evident in the Blacksmith Fork samples collected on both
dates and in the Little Bear sample in August. Microtox
tests performed on the 23 June Little Bear and Blacksmith
Fork samples indicated toxicity in both samples.
Apparently, sparging with N_2 did not consistently remove
sufficient oxygen from solution to prevent hydrogen peroxide
formation.

Unfortunately, samples used for phosphorus analyses in
samples collected for bioavailability assays in April
through August were contaminated with phosphate from
membrane filters used in sample preparation and all results
are unreliable. It is, therefore, not possible to evaluate
what fraction of the total phosphorus in the samples was
available to algae. Complete sets of phosphorus data are
available for samples collected in September through
December. Post irradiation toxicity prevented obtaining
bioavailable phosphorus estimates for most of these samples
however. In September, bioavailable phosphorus at the
Honeyville station was 31 μg/L. Orthophosphorus at this
station was 20 μg/L before irradiation and 42 μg/L after
irradiation. One sample from the November set was selected
for an experiment which investigated the use of the enzyme
peroxidase to break down peroxide and possibly eliminate
toxicity. The sample (Blacksmith Fork R.) was treated with
450 units of peroxidase per liter and allowed to stand at
room temperature overnight in the dark. Microtox results
changed from 17.8 units to 0 after the enzyme treatment
indicating the removal of toxicity in this sample. Samples
collected on 1 December were treated with 150 units of
peroxidase per liter and allowed to stand overnight.
Microtox tests indicated that the samples were still toxic.
Additional peroxidase was added to bring the enzyme
concentration up to 450 units activity/L before the algal
bioassay was set up, but some toxicity evidently persisted
in most (perhaps all) of these samples. A bioavailable
phosphorus estimate was obtained at only one site, the Bear
R. above Oneida Reservoir. Orthophosphorus at this site was
< 5 μg/L before radiation treatment and 13 μg/L after
treatment. Peroxidase addition did not affect the
orthophosphorus concentration.

The effects of different concentrations of peroxidase
and reaction time on the toxicity of irradiated samples were
evaluated in an experiment conducted in mid-December. Water
was collected from the Bear R. near Honeyville. One gallon
was sparged with nitrogen for 1 hr. A second sample was not
sparged. Both samples were irradiated and returned to the
laboratory where they were sub-sampled and treated with
peroxidase concentrations of 160, 500, 1000, 2000 units/L.
Microtox tests were performed after 2 hr. had elapsed and
again after 16 hr. The 2000 unit/ L treatment reduced
toxicity to 1 Microtox unit in the N_2-sparged sample and to

6 in the non-sparged sample after a 16-hr reaction period. Relatively high toxicity levels remained in all other treatments. No algal bioassay was set up for this sample. Future bioavailable P research will probably utilize a 2000 unit/L peroxidase treatment for more than 16 hr. following radiation sterilization of the water samples. (pp. 43,49)

Bioavailable phosphorus estimation by standard addition

Once the sample had been sterilized, standard additions were used in the algal bioassay to estimate BAP. In quantitative analysis, standard addition is a common way to correct for interferences from the sample matrix and produces reliable values unless some internal process prevents the addition from being linear. In algal bioassays something, or some reaction in the sample could inhibit growth (e.g., toxicant, precipitation of nutrients, or lack of nutrients) or stimulate the growth (e.g., vitamins, amino acids, temperature, or light).

For example, Figure 15 shows that an unamended Bear River sample collected on July 22, 1987 from below Cutler Reservoir produced a fluorescence of 7.0 indicating algal growth, but the BAP determined, based on the "linear" response to P addition was a negative 5 μg/L (negative y-intercept). A November 22, 1987 sample from the same site had a lower fluorescence of 4.4, but the resulting BAP estimate was higher; 14 μg/L. This discrepancy might be explained by an increase in pH (due to algal photosynthetic consumption of CO_2) which would cause the precipitation of phosphorus with calcium resulting in non-linearity of the availability of phosphorus and misleading BAP values. Calcium-phosphate forms several intermediates before it reaches the stable hydroxyapatite form. The different intermediates have different solubility products and proportions of calcium to phosphate which might explain the curve seen in Figure 16 for the higher pH. Figure 16 shows that a reduction of phosphorus can occur due to a pH
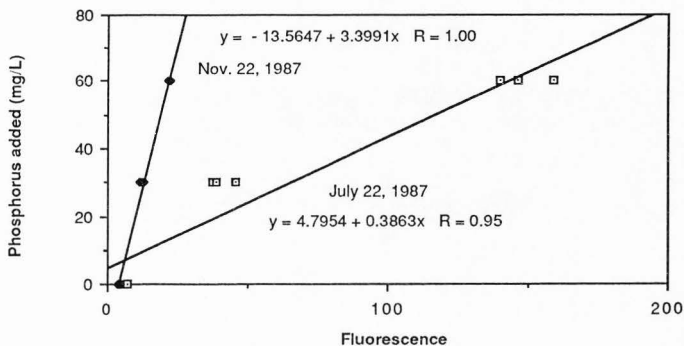
Figure 15. Variable results of standard addition at the same sample site for two different sample dates.
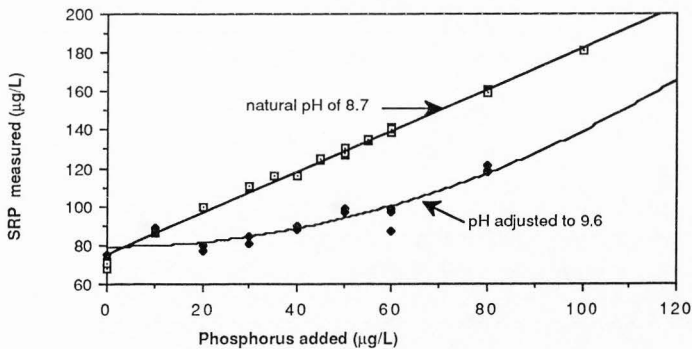
Figure 16. Effects of pH upon standard additions linearity.

occurs rapidly in the first ten minutes and then continues slowly for
increase and result in a non-linear response to phosphorus addition.
A linear relationship is shown in Figure 16 for the natural pH of 8.7,
but a pH of 9.6 results in a non-linear relationship.

This next study also investigated the phenomenon of a non-linear
response of phosphorus addition due to an increase in pH. This was
done by raising the pH incrementally with buffer additions and
measuring the decrease in SRP after 30 minutes. This time period was
provided to allow the system to become evenly mixed and allow
precipitation to begin (Figure 17). After 120 minutes, SRP was
measured again and the change was found not to be statistically
different (P $\leq$ 0.05) between 30 minutes and 120 minutes. This
indicates that most of the phosphorus is precipitated within the first
30 minutes after pH adjustment. When the pH of this sample was
lowered again to a pH of 8 for 20 hours, dissolution of precipitated
phosphorus could not be statistically proven. Griffin and Jurinak
(1974) found that in soils, phosphorus adsorption and precipitation
the next 4 hours. Desorption occurs much slower, requiring 6 to 7
hours in soils (Griffin and Jurinak, 1974). The REDEQL model was run
for the same pH's used in the above experiment and approximately the
same sample composition (ortho-phosphorus equals 50 $\mu g/L$). The model
calculations showed that as pH was increased from 8.70 to 9.48 the
ortho-phosphorus equilibrium concentration went from 0.35 $\mu g/L$ to 0.15
$\mu g/L$ while the balance of phosphorus was in solid form with calcium.
Both the laboratory results and REDEQL calculations agree that as pH
is increased, ortho-phosphorus, and hence BAP concentration would
decrease for Bear River water. Equilibrium concentrations (0.35 to
0.15 $\mu g/L$) calculated by REDEQL are approximately 100 times less than
the measured concentrations (Figure 17) which shows that the river
composition is not at equilibrium. This suggests that phosphorus
inputs replenish that lost to precipitation with time in the river.
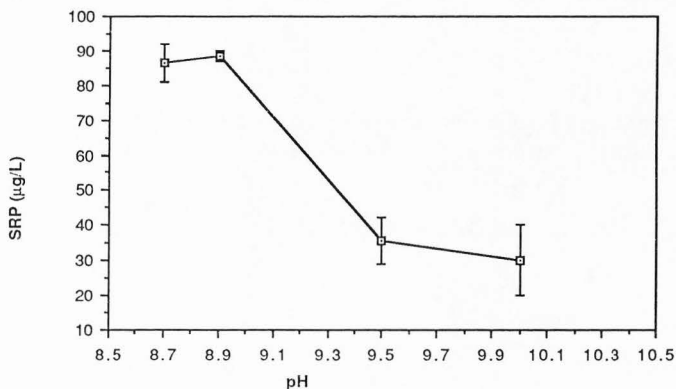
Figure 17.   The change of soluble reactive phosphorus with an increase
             in pH (95 % confidence interval error bars drawn).

The next study investigated the effects of a pH increase upon BAP
estimation.   These effects were monitored during an algal assay test
by controlling pH near neutrality for some of the samples and having
no pH control on the remaining samples.   A CO_2 and air gas mixture was
bubbled through the culture to control pH.   The results of this study
are shown in Table 9.

The Benson Marina site shows a significantly higher (P $\leq$ 0.05) BAP
with pH kept between 6.7 and 7.2 versus a much lower BAP with the pH
as high as 9.7.   The estimated BAP concentrations for the site below
Cutler Reservoir shows a significantly higher (P $\leq$ 0.05) BAP with pH
kept below 6.9 versus a much lower BAP with a pH high around 8.9.   No
BAP could be measured in the Blacksmith Fork sample at either high or
neutral pH.   It appears that an increase of pH for Benson Marina and
below Cutler Reservoir samples resulted in an apparent reduction of
BAP.   This generally agrees with the REDEQL program results which
predict a high (> 99 %) precipitation of PO_4-P with calcium in Bear

Table 9.   The effects on BAP estimation while controlling the algal
           culture pH near 7.0

Site	pH range Without CO_2	BAP Without CO_2 ($\mu g/L$)	pH range CO_2 with Air	BAP CO_2 with Air ($\mu g/L$)
Benson Mar.	7.5-9.7	60.5	6.7-7.2	119
Blw. Cutler Res	8.1-8.9	0	6.9	16.5
Blacksmith Fork	8.5-8.9	0	6.6-7.0	0

River water.   Scherfig et al. (1973) found that CO_2 addition to
maintain pH between 7 and 8 resulted in 45 to 74 % increases of algal
growth (g of cells/L). The Benson Marina sample showed a 97% increase
in estimated BAP.

Precipitation of phosphorus due to an increase of pH might be
minimized by using a large inoculum size to encourage rapid luxury
uptake of BAP by the inoculum.   Luxury uptake refers to the uptake and
storage of phosphorus by the algae beyond those levels required for
immediate growth (Keenan and Auer, 1974).   Keenen and Auer (1974) made
the following findings: (1)   one of the problems associated with
luxury uptake is that it could result in a larger ortho-phosphorus
concentration in solution due to the release of stored intracellular
phosphorus which will act as an extraneous source of phosphorus in
situations where phosphorus concentration in the sample is low;   (2)
the detrimental effects of luxury uptake by introducing additional
phosphorus into the sample can be minimized by a starvation period;
(3)   the influence of luxury uptake of phosphorus on algal bioassays
is a function of the phosphorus concentration in the original (stock
culture) growth medium and of the length of time during which the
cells are starved of phosphorus;   (4)   *Selenastrum capricornutum*

culture) growth medium and of the length of time during which the cells are starved of phosphorus; (4) *Selenastrum capricornutum* exhibits luxury uptake of phosphorus.

On March 22, 1988 a *S. capricornutum* culture in stationary growth phase was put into a phosphorus free medium for two days before being used as an inoculum in a culture. The results were compared with results using an inoculum from a culture at stationary phase, but without holding the cells in phosphorus free medium (February 22, 1988). Both of these sample dates used an inoculum of 10^3 cells/mL in an AAP culture. No significant ($P \leq 0.05$) increase of fluorescence was detected between the two cultures for given phosphorus additions (Figure 18). Since these results are from one culture, the effects of phosphorus starvation needs to be studied further before any conclusions about the effect on luxury uptake can be drawn.

It was hypothesized that luxury uptake could be used advantageously for estimating BAP in Bear River system samples. Cells added in the bioassay inoculum would utilize luxury uptake and store the available phosphorus within the cell. The phosphorus then would not be susceptible to precipitation due to a pH increase within the culture. The quicker the available phosphorus could be removed from the system by cell incorporation, the smaller the effect of phosphorus precipitation would be on BAP estimation.

A low inoculum had been used for the previous study. A high inoculum had also been used for the same sample, so that the effects of luxury uptake could be investigated for two inoculum sizes. Data in Figures 19 and 20 are from the high (1 X 10^5 cells/mL) and low inoculum (1 X 10^3 cells/mL) sizes for data taken February 22 and March 22, 1988. Statistical comparison of the slopes and y-intercepts for both the low and high inoculums on both dates found that they were significantly different ($P \leq 0.05$). Both of these figures show that the higher inoculum size produces more growth (fluorescence) than the low inoculum.
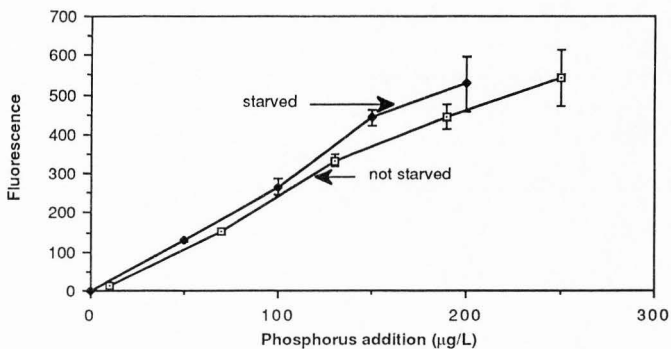
Figure 18.   Effect of algal starvation upon fluorescence with phosphorus addition (95% confidence interval error bars drawn).
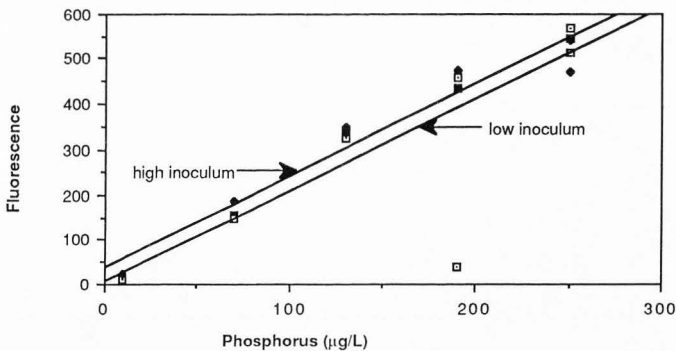
Figure 19.   Fluorescence response of a high and low inoculum size of *S. capricornutum* to phosphorus additions on February 22, 1988.
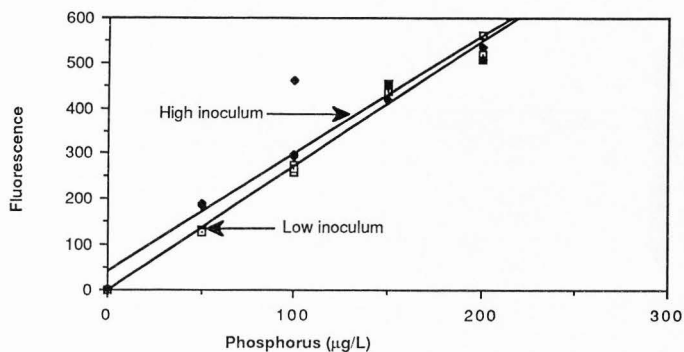
Figure 20. Fluorescence response of a high and low inoculum size of *S. capricornutum* to phosphorus additions on March 22, 1988.
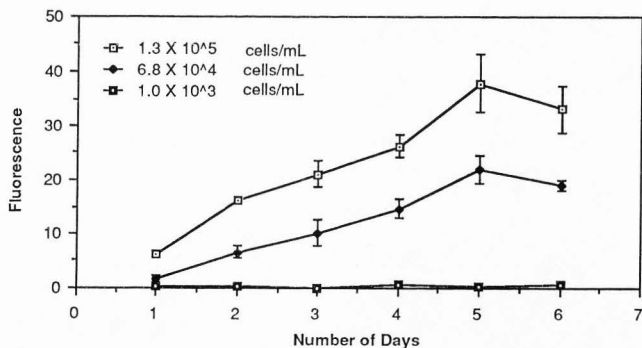
Figure 21. Fluorescence over time for three inoculum sizes (95% confidence interval error bars drawn).

cells/mL exceeds the maximum fluorescence of the 1 X 10^3 cells/mL inoculum within the first day. The inoculum size of 1.3 X 10^5 cells/mL exceeds the maximum fluorescence of the 6.8 X 10^4 cells/mL within the first 2 days. This faster response can be important in counteracting the effects of precipitation due to a gradual pH increase over a one to two week period. This suggests that a high inoculum could be used to minimize the effects of phosphorus precipitation due to a pH increase.

<u>External standard curve</u>

The previously discussed data showed that use of an internal standard curve resulted in a non-linear response to phosphorus as pH was increased. It was then decided to try an external standard curve which might provide more consistent BAP estimates. An external standard curve for <i>S. capricornutum</i> growth in response to P was done in March, 1988 by growing <i>S. capricornutum</i> in algal assay medium (Miller et al. 1978) with varying concentrations of PO_4-P.

There was 72 $\mu g/L$ (1.8 $\mu molar$) of calcium in the algal assay medium. This calcium could have precipitated 6 and 23 percent of the added 200 and 50 $\mu g/L$ of PO_4-P, respectively, if solid $CaHPO_4$ (pK_{so} = 6.66) formed as an intermediate to hydroxyapatite (Snoeyink and Jenkins, 1980). Less than 6 to 20 percent of the phosphorus could precipitate from any phosphorus addition since a portion of the calcium would be utilized by the algae and would be unavailable for precipitation. The linearity of the standard curve seen in Figure 22 indicates that PO_4-P precipitation was not a factor in algal response to PO_4-P addition because the line does not curve to suggest any precipitation.

The minimal detectable change in fluorescence when using the external standard curve was determined to be 8 μg of BAP/L. This was determined by the 95% prediction confidence interval of the
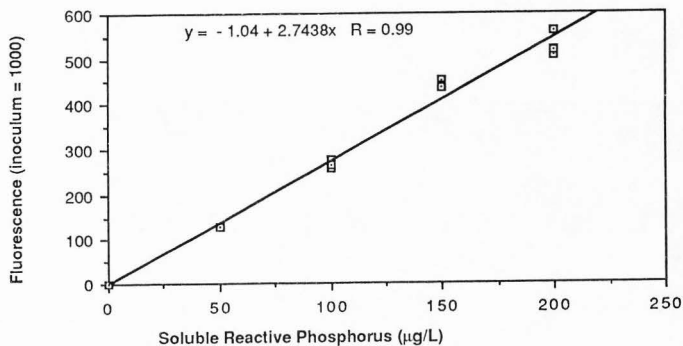
Figure 22. Linearity of external standard curve conducted on March 22, 1988.

fluorescence at 0 phosphorus based on the regression of fluorescence with SRP. The SRP corresponding to this fluorescence was determined using the regression equation.

The equation for calculating BAP from the fluorescence of the sample culture without phosphorus addition was:

$$BAP = (Fluorescence + 1.04) / 2.7438 \qquad (4)$$

A disadvantage of using an external standard curve is that all natural constituents are not present in the artificial medium and their inhibitory and stimulatory effects on the algae are not accounted for. Bear River water quality data and ability of samples to support algal growth suggests that toxic or inhibitory constituents are either not present or their concentrations are insignificant in regards to affecting the growth of the algae. Copper, one of the most toxic elements to algae and one of the most likely to be present, has usually been below 30 μg/L of total copper in Bear River samples.

usually been below 30 µg/L of total copper in Bear River samples. Soluble copper inhibits *S. capricornutum* growth at 50 µg/L (U.S. Environmental Protection Agency, 1980b). A maximum concentration of 105 µg/L of total copper was observed in samples collected between January 1977 and December 1983 in the Bear River (Sorensen et al., 1986). Soluble copper ranges from 10 to 1 % of total copper for water ranging between pH's of 7.5 to 8.0 (Lindsay, 1979). A pH of 7.5 to 8.5 is typical of Bear River water. Therefore, the maximum soluble copper in a Bear River sample would be 10.5 µg/L which would be below the toxic level for algae. Similar computations for other heavy metals indicate that toxic concentrations are unlikely in this hard, alkaline water.

Recommended BAP estimation procedures

In summary, the standard Algal Assay Bottle Test (Miller, et al., 1978) was primarily developed to determine the limiting nutrient and the presence of any toxicants. This procedure has been modified to allow BAP estimation by using standards. Miller et al. (1978) suggests that the pH of the algal cultures be maintained below 8.5 to insure the availability of carbon dioxide. A pH increase for the Bear River samples resulted in phosphorus precipitating with calcium.

The apparent affects of precipitation of phosphorus with calcium on BAP estimation can be minimized by the use of an external standard curve, luxury uptake by an initial starvation period of the inoculum and large inoculum size, and pH control by bubbling with carbon dioxide. Luxury uptake would maximize the phosphorus uptake within the cells during the initial phase of algal growth and minimize the phosphorus concentration in solution which would be available for precipitation. Precipitation of phosphorus with calcium is pH dependent, so control of pH will minimize precipitation. These procedures would probably produce the maximum BAP estimate for the Bear River system. This does not mean this would be the most

accurate, but it would give the largest BAP estimate and a margin of safety in making management decisions. A more accurate procedure might include maintaining the sample pH at the natural river pH. The BAP estimations for this study used the March 22, 1988 external standard curve with an inoculum of 10^3 cells/mL and no pH control.

Sources of Bioavailable Phosphorus

Phosphorus contributed by point sources

The Cache Valley has five wastewater treatment plants that discharge into the Bear River system, including: the City of Preston (trickling filter), City of Richmond lagoons, City of Logan lagoons, City of Wellsville lagoons and City of Hyrum (oxidation ditch). The City of Franklin has a total containment lagoon, while Richmond and Wellsville were designed to be total containment facilities, but currently discharge less than 0.1 MGD. White's Trout Farm and E. A. Miller are the only significant industrial waste dischargers to surface water in the Cache Valley.

Table 10 shows that, based on the average BAP concentration and average flow, the wastewater treatment plants in Cache Valley contribute approximately 30,000 kg of BAP/year to the Bear River system. The Logan lagoons contribute 73% of the total WWTP effluent BAP while Preston WWTP and Hyrum WWTP contribute 6 and 21%, respectively. This means that if the Logan Lagoons effluent phosphorus was completely eliminated from entering the Bear River system, the BAP load from wastewater treatment plants would be reduced to approximately 8,200 kg/yr. If Logan lagoons, Preston WWTP and Hyrum WWTP effluents were all removed from the Bear River system, the point source BAP load would be reduced by 99.5% to only 160 kg/yr. Realistically these effluents could not be completely removed from the Bear River system, but treatments are available that would remove 90%

Table 10. Cache Valley point sources that discharge into the Bear River system

River	Effluent Samples	Date	Flow cu. ft/s	Concentrations			Average mass discharge		Percentage of Total BAP
				TP µg/l	SRP µg/l	BAP µg/l	TP kg/yr	BAP kg/yr	
Bear	Preston WWTP	6/21/87	1.0	2210.	1470.	1560.	2697.	1800.	6.0
		2/22/88		3830.	3290.	2500.			
L. Bear	White's Trout Farm	6/21/87	31.0	*	*	< 8	1272.	< 179	0.0
		2/22/88	25.0	57.	23.	< 8			
	Hyrum WWTP	6/21/87	1.4	5190.	4890.	5830.	7571.	6300.	21.0
		2/22/88		6920.	7000.	4180.			
	Wellsville Lagoon	6/21/87	0.1	1130.	862.	860.	101.	90.	0.3
Logan	Logan Lagoon	2/22/88	14.4	3170.	2530.	1710.	40768.	22000.	73.3
						Total =	52409.	30000.	

*= sample was contaminated

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of the PO_4-P; the form which is highly available. If a 90% reduction of PO_4-P from the three major plants was instituted then PO_4-P, and hence BAP from the point sources, would be reduced by 89.5% to 3,200 kg/yr. A 90% reduction in phosphorus for only the Logan lagoons could reduce the BAP contribution from WWTP's by 65.5% to 10,400 kg/yr to the Bear River system.

Phosphorus values from the literature were reviewed to determine if the concentrations measured for the point sources in Cache Valley were within the same range as the literature values. Typical values from the literature show that total phosphorus ranges from 4 to 15 mg/L for untreated wastewater, but since secondary treatment removes a small percentage (<10%) of total phosphorus, this range would be typical for effluents (Tchobanoglous, 1979). The TP concentration from the Cache Valley wastewater treatment plants ranges from 1.1 to 5.2 mg/L (Table 5). These phosphorus concentrations reflect the relatively dilute wastewaters treated by these plants. Excessive infiltration and inflow problems in the sewer systems probably explain these low concentrations. Rast and Lee (1982) recommend a phosphorus loading value of 1.1 kg P/cap.-yr. Using this value for the City of Preston, City of Hyrum and City of Logan results in phosphorus loads of 4,200 kg/yr, 9,300 kg/yr and 33,000 kg/yr, respectively (Table 11).

Table 11. Phosphorus loading calculation for the three major wastewater plants in Cache Valley using the factor of 1.1 kg P/cap.-yr

City WWTP	Population using system	Pop. eq. from Industries	Total Population	Calculated Discharge Kg P/yr
Preston	3800	0	3800	4200
Hyrum	4600	3850	8450	9300
Logan	30000	0	30000	33000

The calculated loading comes within a factor of two of the measured values (Table 10);   a reasonable variance.

Phosphorus contributed by livestock runoff

Another source of phosphorus comes from dairy and feedlot waste carried into streams by runoff.   Based on the modeling results of Wieneke et al. (1980), Sorensen et al. (1987) estimated that 2.5 Mg/y of total phosphorus was contributed to the Bear River system from feedlots.   This was equivalent to approximately 0.6% of the total phosphorus (440 Mg/y) that passed by Cutler Reservoir in 1985.   The contribution from feedlots in regards to BAP was estimated.   The average BAP concentration is 22.5 µg/L below Cutler Reservoir and the water year flow in 1985 was 2.6 X 10^{12} L.   The BAP which then passed by Cutler Reservoir was approximately 59 Mg/y.   If all the total phosphorus (2.5 Mg/y) contributed from feedlots was bioavailable then it would compose only 4.3 percent of all the BAP that passes below Cutler Reservoir.   Feedlot phosphorus could be highly available since 5 to 25% of the total phosphorus is ortho-phosphate and much of the remaining phosphorus fraction is decomposable organic material (Wieneke et al., 1980).

Wieneke et al. (1980) found that there was no linear correlation between TP and SRP in feedlot runoff.   This might be due to phosphorus being in the organic form which would not be measured as SRP.   The organic matter would break down over time and become available, so BAP would probably be high.   According to Wieneke et al. (1980) all of these feedlots are within 24 hours river time of Cutler Reservoir. More studies need to be conducted to determine BAP transport along the river and reservoirs to determine the potential impact upon downstream reservoirs from livestock runoff.

Phosphorus contributed by land
runoff and streambank erosion

The transport of phosphorus in runoff from agricultural land is commonly regarded as one of the major factors controlling the eutrophication of natural waters (Sharpley, 1980a). In contrast, Raschke and Schultz (1987) found, in limited work, that very little nonpoint source phosphorus was bioavailable.

Sagher (1976) found that most of the inorganic phosphorus was not available in calcareous soils. Cache Valley has mainly sedimentary geology, so it would be expected that the soils would be high in total phosphorus, but would be low in BAP due to its calcareous soils.

Rainfall and snowmelt are the major causes of land runoff. The phosphorus concentration in soil runoff decreases in each successive runoff event during rainy periods (Sharpley, 1980b; Wendt and Corey, 1980). Phosphorus is replenished during dry periods due to atmospheric deposition and organic matter buildup. A Bear River sample taken at the UT-ID state border in February 1986 during a high runoff event indicated phosphorus loads of 89 g TP/sec and 37 g SRP/sec. This high runoff event was a very rare ($\approx 1/100$ years) occurrence. The magnitude of phosphorus transport during this event can be put in perspective when compared to the average 1986 load of 9 g TP/s and 3 g SRP/s. These loads convert to annual loads of 270,000 kg TP/yr and 95,000 kg SRP/yr without including the February data. The February event contributed 30,700 kg TP/4 days and 12,800 kg SRP/4 days which is 11 and 14 %, respectively, of the annual loads. Runoff events can contribute large amounts of phosphorus during a short time period, but over an entire year the relative magnitude is reduced.

To obtain an annual estimate of the total phosphorus contributed by land runoff in Cache County, export coefficients for entire watersheds from Rast and Lee (1982) were used along with land use acreages (Cundy and Conant, 1982) of Cache County (Table 12). It was estimated that 55,000 to 65,000 kg of TP/year was contributed to the

Table 12. Phosphorus contribution from land runoff for various land uses in Cache County, UT

Land use	Acres	sq. meters	Total Phosphorus Rast & Lee Export Coeff. g/m2/yr	Designation	Low Total Phosphorus kg/yr	High Total Phosphorus kg/yr
Non-irrigated Cropland	60,000.	2.4E+08	0.05	rural/agriculture	12,000.	12,000.
Irrigated Pasture *	120,000.	4.9E+08	0.05	rural/agriculture	24,000.	24,000.
Marshland	5,000.	2.0E+07	0	wetlands	0.	0.
Multiple Use	307,000.	1.2E+09	0.005-0.01	forest	6,200.	12,000.
Native Grazing	218,000.	8.8E+08	0.005-0.01	forest	4,400.	8,800.
Recreation	1,000.	4.0E+06	0.005-0.01	forest	20.	40.
Urban	19,000.	7.7E+07	0.1	urban	7,700.	7,700.
Wildlife	19,000.	7.7E+07	0.005-0.01	forest	380.	770.
				Total (kg/yr) =	55,000.	65,000.

* includes all irrigated cropland

Bear River system by land runoff. Rast and Lee (1982) state that the approach of using export coefficients should provide an estimate, usually accurate within a factor of two, of the amounts of nutrients expected to enter a waterbody. Sorensen et al. (1987) estimated that 440 Mg of TP/year passed by Cutler Reservoir dam in 1985. This estimate included the phosphorus contributed from the point and non-point sources from the entire Bear River basin.

A special study was done to try and estimate the percentage of TP that was BAP in runoff. Since rain is the major cause of land runoff, a rain simulation study was done in September, 1987 on four sites with different soils, slopes, and land cover characteristics (Table 13). The sites were: a wheat field, a recently planted barren field, and two range land sites in the Cache Valley. The sample date probably results in a lower estimation of phosphorus loss than would be expected in a spring runoff event since the additional factors of organic matter accumulation over the fall and snowpack would not affect the loss of top soil and plant litter. The rain simulation apparatus gave us the benefit of better experimental control but some error may have occurred due to the small number of sample plots done and their respective small areas. The simulated rain intensity ranged from 0.22 to 0.56 in/min. Typical rainfall in the Cache Valley is 0.02 in/min for a 5 minute duration over a 1 year return period and 0.05 in/min for a 5 minute duration over a 100 year return period (State Climatologist office, USU, Logan, UT). The simulated rain intensity was 10 times greater than actual rainfall. This excess rain intensity may have produced a lower BAP estimate because of its higher erosive power which would wash away larger particles, and lower the fraction of clays and organic matter, and thus result in a lower BAP fraction per unit mass of suspended solids than would be expected from natural runoff.

Table 13. Soil characteristics from runoff and streambank sampling sites; BAP and TP measurements from simulated runoff and streambank suspensions in Cache Valley

Location	Soil (< 2 mm)						Suspension				
	Soluble Phos. µg P/ g soil	TP µg P/ g soil	% clay	% CACO3	pH	% Organic* matter	Sus. solids mg/L	BAP		TP	
								µg/L	µg/ g SS	µg/L	µg/ g SS
Runoff**											
Blacksmith Fork	13.2	850	22.25	0	7.1	6.29	422	2744.	6502.4	2392	5668
Weston Cr.	3.91	130	18	11.6	7.84	1.95	932	735.	788.6	1366	1466
Cub R.	6.88	860	32.5	15.3	7.77	2.26	297	51.	171.7	2444	8229
Abv. Cutler Res.	11.75	130	22.25	1.7	7.25	6.28	426	433.	1016.4	1156	2714
Streambank											
Battle Cr.	7.97	710	32.5	10.2	8.13	0.74	2560	15.	5.9	2529	988
Weston Cr.	2.58	640	25	7.7	7.67	2.52	2485	10.	4.1	932	375
Little Bear R.	3.2	600	12	5.1	7.74	1.91	1010	1.	1.4	2027	2007
Bear R. (Amalga)	2.34	560	14.5	10.2	7.64	1.67	560	6.	9.8	1487	2655

*Organic matter = (1.724) X Organic carbon and is measured on the top 3 in. of soil including duff.
** All runoff parameters are averages from two plots except BAP which was a single measurement

The BAP load from runoff can be estimated by using the ratios of (μg BAP/g SS)/(μg TP/g SS) for the four runoff sites. The ratios are 1.15, 0.54, 0.02 and 0.37 μg BAP/μg TP for the runoff sites at the Blacksmith Fork, Weston Creek, Cub River and Above Cutler Reservoir, respectively. The large range might be due to the Blacksmith Fork site having 0 % $CaCO_3$, and is a range land with an accumulated organic layer on the surface while the Cub River has 15.3 % $CaCO_3$, and is a cropland with the organic matter being tilled into the soil. The average of the ratios is 0.52 which was multiplied by the TP high and low export estimates (Table 12) to obtain an estimate for BAP export. It is felt that the average value would provide the best BAP estimate due to the large range of a limited data set. The estimated BAP loads would range from 28,600 to 33,800 kg BAP/yr using this ratio. These estimated BAP loads could be high because the measured BAP values do not account for any BAP loss as it travels through the watershed and becomes unavailable due to precipitation and plant uptake. Sorensen et al., 1987 and Ahuja et al., 1982 point out that the relative contribution of sediment and phosphorus from any soil to a water body depends on its erosivity, slope, and distance from the water. This runoff total phosphorus estimate needs to be taken with caution since the export coefficients are national averages and may not be representative for a calcareous area such as Cache County.

Phosphorus contributed from streambanks was also investigated because bank erosion and landsliding was obvious in many locations in Cache Valley. The relative phosphorus contribution for streambanks in respect to runoff can be seen by dividing the BAP concentrations by the suspended solids concentrations which allows a comparison on a common basis. This comparison shows that the runoff samples were several orders of magnitude greater in BAP than the streambank samples. Runoff samples come from top soils that have continual phosphorus replenishment from atmospheric deposition, fertilizer

addition and/or decaying vegetation. The streambanks are naturally low in phosphorus and continually lose it by root uptake. In agreement, Taylor and Kunishi (1971) state that streambanks can actually act as phosphorus sinks because they are usually low in phosphorus content.

The final analysis of the data in Table 13 showed that no correlations existed for either the runoff or streambank samples between BAP and % clay, % $CaCO_3$ and pH. The lowest pH, lowest % $CaCO_3$ and average clay percentage gave the highest BAP estimate (Blacksmith Fork). The lowest BAP estimate for runoff samples was obtained for the Cub River sample which had a higher pH, the highest percent $CaCO3$, and the highest clay content.

The relative contribution of phosphorus from each of the sources

Cache County will be used as the common base to compare the relative contributions of phosphorus from the various sources (Table 14). The point sources (not including Preston WWTP) in Cache County contribute 28,200 kg BAP/yr. The majority of the feedlots are in Cache County, so it will be assumed that all of the 2,500 kg BAP/yr is contributed from livestock runoff to the Bear River in Cache County. The BAP contribution from land runoff of 28,600 to 33,600 kg BAP/yr was calculated for Cache County only. Point sources contribute 46%, livestock runoff contributes 4% and land runoff (midpoint of 31,100 kg/year of BAP) contributes 50% of the bioavailable phosphorus load. The majority of phosphorus for the point sources only comes from three WWTP plants, while the livestock runoff comes from approximately 200 feedlot areas and the land runoff is very diffuse encompassing 744,000 acres. It is noteworthy that BAP has been undetectable at the Bear River sampling site above Cutler Reservoir, even though this site would receive most of the land runoff in Cache Valley. This suggests that even though land runoff can contribute 50% of the BAP, an undetectable amount of it reaches Cutler Reservoir. Land runoff may

have insignificant impact upon downstream reservoirs. With this in mind, a comprehensive management plan still needs to be developed which will minimize the phosphorus from all the sources.

Sorensen et al. (1987) stated that not only should the total mass of phosphorus from any one source in a year's time be considered in phosphorus management, but the mass of algal available phosphorus contributed must also be taken into account. Control of the sources which have the greatest impact in terms of contributing the largest BAP to the reservoir should be ranked highest (Sorensen et al., 1987), and a management plan should apply the best management practices (BMP) to those sources which will reduce the BAP load to the reservoirs in the most cost effective manner.

Table 14. Bioavailable phosphorus contributions from various sources in Cache County, UT

	Total Phosphorus kg/yr	Bioavailable Phosphorus kg/yr
3 Wastewater Treatment Plants discharge	28,200	28,200*
Phosphorus from livestock runoff	2,500	2,500
Phosphorus from land runoff	55,000 to 66,000	28,600 to 33,800

*For Wastewater and livestock runoff, BAP/TP ≈ 1.0

The proposed Honeyville reservoir will be immediately below Cutler reservoir and is the main reservoir impacted by the Cache Valley sources. The three wastewater treatment plants (Logan, Hyrum and Wellsville) are the closest sources to the Honeyville reservoir and therefore should be ranked highest since they will contribute the most BAP. E. A. Miller, a slaughter house in Hyrum, needs to be studied to determine if it fits into this category.

Phosphorus control and treatment

Technology is available that can reduce the phosphorus from WWTP's by 90%. The Innovative and Alternative Technology Assessment Manual (Reckhow et al., 1980) discusses the economics of chemical precipitation, biological and land treatment for phosphorus removal. The construction and operation and maintenance costs from the Innovative and Alternative Technology Assessment Manual were updated to June, 1988 by using the ENR construction cost index of 4531.58 (ENR, 1988) (Table 15).

Alum, ferric chloride and lime addition are the three most common precipitation methods. Removal efficiencies ranged from 75% for lime to 97% for ferric chloride addition. These processes produce sludges which, because of their toxicity may require additional treatment depending on the disposal methods. For a 1 MGD WWTP, ferric chloride was the least expensive control strategy for construction with costs of $46,000, while alum addition was the most expensive at $49,000. The annual operation and maintenance costs for a 1 MGD WWTP showed that lime clarification was the cheapest at $19,000, while alum addition was again the most expensive at $55,000. For a 10 MGD WWTP ferric chloride addition was the least expensive with construction costs of $150,000 and lime clarification being the most expensive at $210,000. The annual operation and maintenance for a 10 MGD WWTP showed that lime clarification was the least expensive at $200,000 with alum addition being the most expensive at $400,000 (Table 15). Even though lime clarification is relatively inexpensive compared to the other strategies, it should only be recommended with caution because of its low, (75%) removal efficiency.

Land treatment technology ranges from wetlands to overland flow with removal efficiencies ranging between 0 to 90% depending on the site and design characteristics. Construction costs and annual operation and maintenance costs show that wetlands are the least

Table 15. Economics for phosphorus removal from wastewater effluents by various treatments
(Data from U.S. EPA, 1980a and ENR = 4531 for June, 1988)

Treatment Process	Wastewater Flow (MGD)	Construction Cost ($)	Operation/ Maintenance Cost ($)	Performance T. Phosphorus Removal (%)
Wetlands	1	26,000	8,200	0-94
	0	140,000	37,000	
Rapid Infiltration,	1	420,000	47,000	0-90
Underdrained	10	3,200,000	245,000	
Rapid Infiltration,	1	370,000	37,000	0-90
Not Underdrained	10	2,900,000	220,000	
Land Treatment, Slow Rate,	1	1,200,000	59,000	80-99
Sprinkler (CP), Underdrained	10	9,800,000	440,000	
Land Treatment, Slow Rate,	1	1,200,000	56,000	80-99
Sprinkler (CP), Not Underdrained	10	9,800,000	440,000	
Land Treatment, Slow Rate,	1	86,000	56,000	80-99
Gravity (RF), Not Underdrained	10	370,000	320,000	
Land Treatment, Slow Rate,	1	1,200,000	71,000	80-99
Gravity (RF), Underdrained	10	5,400,000	340,000	
Overland Treatment (OD), Gravity	1	980,000	23,000	90-99.9
	10	9,100,000	150,000	
Phostrip (Activated Sludge)	1	860,000	70,000	>90
	10	1,600,000	100,000	
Alum Addition	1	49,000	55,000	94
	10	180,000	400,000	
Ferric Chloride Addition	1	46,000	42,000	56-97
	10	150,000	310,000	
Lime Clarification of Raw	1	47,000	19,000	75
Wastewater	10	210,000	200,000	

expensive for treating both 1 and 10 MGD while land treatment options are the most expensive (Table 15).

Biological phosphorus removal treatment is usually accomplished by an activated sludge process either in a combined or split treatment system in regards to carbon removal. This process can remove greater than 90% of the phosphorus. Estimated construction costs are $860,000 for 1 MGD and $1,600,000 for 10 MGD. Annual operation and maintenance costs are $70,000 for 1 MGD and $100,000 for 10 MGD.

The selection of treatment technology should be based on existing facilities, operator expertise and cost effectiveness. The beneficial use of land treatment should be accounted for if additional irrigation or wetlands are provided.

Land runoff contributes about 50% of the total BAP and probably should be ranked second because of the difficulty of its control. Land runoff also contributes significant quantities of sediment which are filling in the channels and reservoirs. Hanson and Fenster (1969) stated that we shouldn't ignore erosion into our lakes, but we should not consider the sediment in addition to nutrient enrichment. The cost effectiveness of land runoff control should account for both BAP and sediment removal from the Bear River system. Phosphorus in land runoff can be minimized by using no-till or low-till agriculture, maintaining a crop cover, controlling runoff channels, using green belts adjacent to the water courses, maintaining wetlands, and using proper fertilizer application and dosage. No-till agriculture was developed in the 1960's and is now gaining national acceptance. Cache County agriculture in 1987 only used 0.2% of no-till and 9.9% of low-till (Sorensen et al., 1987). The 1985 Federal Farm Bill (PL 99-198) contains the Conservation Reserve Program (CRP) in which farmers are encouraged, through 10-year contracts with the U.S. Department of Agriculture to stop growing crops on highly erodible cropland and plant it to grass or trees. The Soil Conservation Service, Logan

(Gwen Christiansen, personal communication, June 8, 1988), reported that Cache County had 45,000 acres eligible for CRP out of 200,000 acres of cropland; 17,250 acres (38%) had taken advantage of this program.

Streambank erosion should be controlled to reduce the sediment load, but not necessarily to control BAP. Streambanks can be stabilized by low porosity covers, loose material covers, vegetation or modification of the stream channel. The Corps of Engineers (1981) found that, generally, across the U.S. stream bank stabilization was not cost effective.

Livestock runoff contributes approximately 4% of the BAP. Wieneke et al. (1980) stated that separation of cattle from the receiving stream by approximately 60 m (200 ft.), significantly reduced the impact of the waste on the stream. The Cache County Zoning Ordinance does not establish a separation distance for cattle, but states, "Setback distances: A. The applicant shall demonstrate that his waste management system will minimize any wastes from entering a waterway; canal, drain, or ditch; lake or reservoir; wetland or watertable, consistent with federal, state, and local laws and regulations " (Chapter 13-A, Agriculture Zone, amended November 15, 1983, Section 13-6-3). The Utah Bureau of Water Pollution Control (Brian Elwell, personal communication, June 16, 1988) has a policy that if a county does not have a setback regulation, then the wastewater from operation and runoff cannot be discharged into a water body.

A benefit-cost analysis needs to be done to determine whether it is more cost effective to control the sources or treat the water before use. Sorensen et al. (1986) determined that it would cost $3.62/acre-ft to $10.47/acre-ft for treatment of Bear River water by coagulation/flocculation, taste and odor control and chlorination. The recreational benefits (boating, aesthetics, fishing etc.) of better quality water in the Bear River and reservoirs needs to be

determined and subtracted from the costs of source treatment before a meaningful cost analysis can be determined.

Transportation of Bioavailable Phosphorus

Effects of power peaking upon phosphorus concentration in the Bear River

When discussing transportation of phosphorus it is important to distinguish between concentration and mass. Algae influence the phosphorus concentration surrounding them resulting in a concentration gradient as the phosphorus is taken up by the algal cell. BAP needs to be within this realm of influence for the algae to be utilize it, so BAP concentration is important. Mass transport is related to concentration by flow. The mass transport value can be misleading because even with a large mass transport, the concentration can be low due to large flows.

A power peaking study was done for both Oneida and Cutler reservoirs to determine the effect upon phosphorus transport. Figures 23 and 24 show the effects of power peaking. For below Cutler Reservoir (Figure 23), the stream surface total phosphorus concentration reached a maximum of 50 μg/L, and the depth profile total phosphorus concentration reached a maximum of 48 μg/L. The depth profile was obtained by lowering and raising a USGS continuous sampler throughout the river depth at approximately 1/3 and 2/3 the distance across the river. Flow reached a maximum of 4000 cfs at approximately one hour before the maximum TP concentrations were measured. The SRP measurements remained relatively constant and no outstanding maximum is observed.

On the Bear River at the Idaho/Utah border 32 miles downstream from the reservoir, the effects of the Oneida Reservoir power peaking were measured (Figure 24). The stream surface total phosphorus concentration reached a maximum of 50 μg/L, the near bottom total
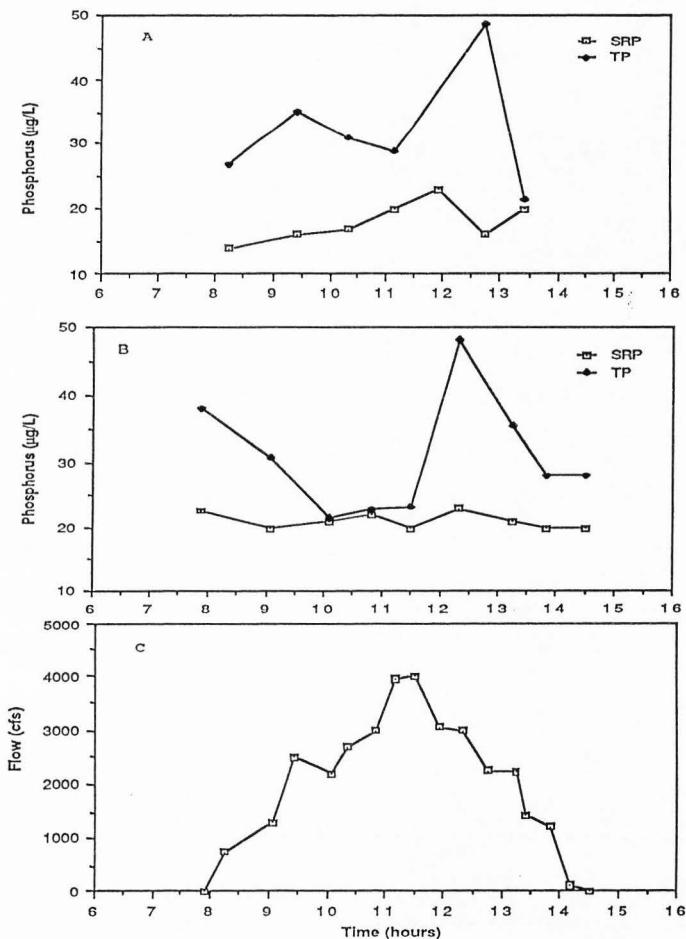
Figure 23. Bear River below Cutler Reservoir during a power peaking operation on December 3, 1986. The concentration of SRP and TP on the surface (A). The concentration of SRP and TP over a composited water profile (B). The flow of the Bear River (C).
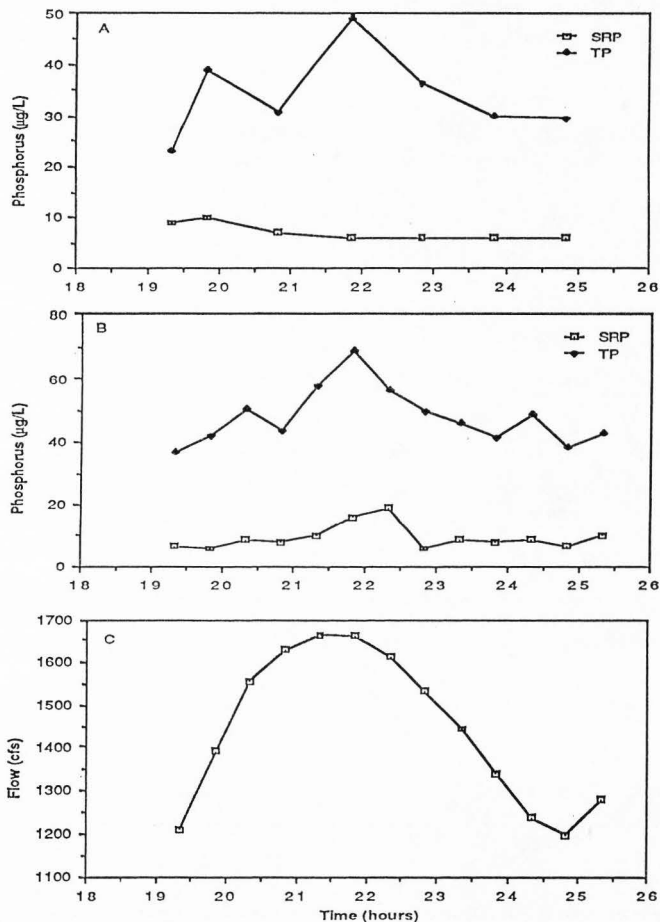
Figure 24. Bear River below Oneida Reservoir at the UT-ID border during a power peaking operation on December 4, 1986. The concentration of SRP and TP at the surface (A). The concentration of SRP and TP at 1 foot off the river bottom (B). The flow of the Bear River (C).

phosphorus concentration reached a maximum of 68 µg/L, and the flow reached a maximum of 1680 cfs, all at approximately the same time. The SRP measurements remained relatively constant with the near bottom phosphorus concentration reaching a peak of 18 µg/L at approximately one hour after the flow reached its peak. The total phosphorus peaks were probably due to suspension of the stream bed material with total phosphorus being comprised mainly of particulate phosphorus and only a small percentage of SRP.

Verhoff et al. (1982) made the following conclusions for rivers in western Ohio over a time period which included several storms: (1) the peak of the total phosphorus concentration almost always leads the flow rate peak of the river at any station, (2) the total phosphorus concentration declines to its low flow value before the river flow becomes low. This discrepancy is probably due to the phosphorus being suspended from the streambed during the power peaking study while the phosphorus came from surface runoff during storms in the study done by Verhoff et al. (1982). The Verhoff et al. (1982) findings agree with the phosphorus flushing action from storms which was previously discussed. This study showed that power peaking is important in the transport of TP, but does not appear to be a major factor in SRP transport. The relationship between BAP and TP and SRP needs to be better understood before the importance of power peaking upon BAP can be determined.

Concentrations of bioavailable phosphorus through Cutler Reservoir

The concentration of BAP through Cutler Reservoir is important in understanding the transport of phosphorus through it and the impacts it will have upon the downstream Honeyville Reservoir. Vollenweider (1968) estimated that, based on the nutrient budget of 15 eutrophic lakes, an average of 49% of the phosphorus entering the lakes was

retained in them, presumably in the sediments. Porcella and Bishop (1975) found that a similar ratio, 54%, of the influent phosphorus remained in Hyrum Reservoir, Utah. Hyrum Reservoir has a detention time of 52 days for high flows and 123 days for low flows (Sorensen et al., 1986). Cutler Reservoir is a river-run reservoir with retention times ranging between 2 to 5 days. An average 5 foot depth was approximated from random samples and an approximate surface area of 5.4×10^7 ft^2 were used to calculate the retention time for Cutler Reservoir. Probably less than 54% of the influent phosphorus remains in Cutler Reservoir because its retention time is shorter than Hyrum. While actual data for Cutler Reservoir shown in Table 16 was used to calculate that 69% (October 20, 1987) of the influent TP can be stored, but on another date in the same month (October 3, 1987) 193% of the influent TP was released. Cutler Reservoir as a river run reservoir does not appear to consistently store or release TP. The proposed Honeyville reservoir will have an average retention time of 58 days (Sorensen et al., 1986). Honeyville Reservoir might store as much as 54% of the influent phosphorus since its expected retention time will be similar to Hyrum.

Logan, Hyrum and Wellsville wastewater treatment plants in Cache Valley discharge to the Little Bear and Logan rivers within 4, 9, and 9 miles, respectively, of Cutler Reservoir. Table 17 presents the October 3, 1987 phosphorus and flow data that is graphed in Figure 25. The outflow from Cutler Reservoir is measured at the site below Cutler Reservoir. Table 16 shows that the Bear River usually has greater flows ranging from 3.3 to 19.3 times as much as the combined Little Bear and Logan rivers except on October 20, 1987 where the flows were approximately the same. The Bear River has lower TP concentrations ranging from equal to as little as 30% of those in the Logan and Little Bear rivers. The Bear River has BAP concentrations so low they are unmeasurable (Table 16). The higher flows, lower TP's and BAP's of the Bear River can dilute the phosphorus concentration in the

Table 16.   Flow, total and bioavailable phosphorus concentration balance around Cutler Reservoir

Date	Ab. Cutler Res. Bear River			Benson , inflow L. Bear and Logan R			Below Cutler Res Calculated value		(1) <8=8	(2) <8=0	Below Cutler Res. measured values		
	Flow cfs	TP µg/L	BAP µg/L	flow cfs	TP µg/L	BAP µg/L	flow cfs	TP* µg/L	BAP* µg/L	BAP* µg/L	Flow cfs	TP µg/L	BAP µg/L
5/27/87	1578	222.	< 8	293	225.	51.	1871	222.	15.	8.	3226	206	< 8
7/12/87	1049	129.	< 8	164	453.	271.	1213	173.	44.	37.	778	130	< 8
8/18/87	808	123.	< 8	243	333.	138.	1051	171.	38.	32.			
10/3/87	1291	114.	< 8	67	229.	66.	1358	120.	11.	3.	2156	146	< 8.
10/20/87	284	79.	< 8	351	240.	33.	635	168.	22.	18.			

(1) setting BAP for Ab. Cutler Res. equal to 8
(2) setting BAP for Ab. Cutler Res. equal to 0
* concentrations calculated by adding mass flows of Bear R. with L. Bear and Logan R.
  and converting this sum back to concentration

Little Bear and Logan rivers and explain most of the decrease in concentration. At mile 12, BAP was reduced to a non-measurable concentration while TP concentration was reduced by 20%. One possible explanation for the lack of a decrease in phosphorus concentration at mile 9, might be that the Bear River water had not completely mixed with the Little Bear and Logan river water at the sampling point. The sample might have been taken in the section only impacted by the Little Bear and Logan river, so no phosphorus concentration change was detected.

Table 17.   Transport of phosphorus through Cutler Reservoir on October 3, 1987

Included Rivers	Distance from Logan lagoons miles	Flow ft^3/s	Total Phosphorus $\mu g/L$	Bioavailable Phosphorus $\mu g/L$
Bear R.		1291	114	< 8
L. Bear, Logan R	6	67	231	67
L. Bear, Logan, Bear R.	9	887	196	60
L. Bear, Logan, Bear R.	12	808	159	< 8
L. Bear, Logan, Bear R	15	515	150	< 8

Table 16 is a mass balance of Cutler reservoir with the influent comprised of the sites Above Cutler Reservoir, Bear River and the Benson Marina which accounts for the Logan and Little Bear rivers. The calculated values below Cutler Reservoir are the summation of the Bear, Little Bear and Logan rivers which should equal the measured flow and concentration below Cutler Reservoir if the parameter is conserved through the reservoir. The flow differs by as much as a factor of 2, while the TP concentrations are within 33% of each other. The differences in flow and TP concentration could be explained by the error incurred in obtaining the measurements, and the variable flow of
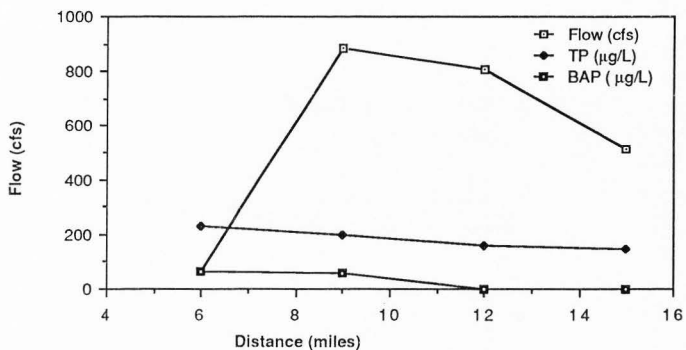
Figure 25.   Changes in flow, total and bioavailable phosphorus with distance through Cutler Reservoir on October 3, 1987.

the reservoir due to power peaking.   Two scenarios were used to calculate the BAP concentrations below Cutler Reservoir: (1) the BAP above Cutler Reservoir is assumed to be 8 or;   (2) the BAP above Cutler Reservoir is assumed to be 0.   These two scenarios show that dilution can account for most of the decrease in concentration in BAP, but both scenarios yield BAP concentrations greater than what was measured below Cutler Reservoir.   This suggests there might be another removal mechanism for BAP.   This removal mechanism might be a combination of plant uptake, precipitation and/or settling.

Figure 26 presents another way to look at the mass balance of Cutler Reservoir.   The percent change is determined by subtracting the flow (ft^3/s), TP (mg/s) and BAP (mg/s) of the inputs (Bear River above Cutler Reservoir plus Little Bear River at Benson Marina) from the output of Cutler Reservoir, the result is then divided by the output and multiplied by 100 to obtain percent.   Flow and TP are well correlated (r^2 = 0.95), while BAP does not correlate with flow ($r^2 \approx$ 0.00), but is consistently lost.   If the percent change is negative,

then flow, TP and BAP is stored within the reservoir. While if the percent change is positive then Cutler reservoir is releasing flow and TP. In summary it appears that TP and flow are generally conserved through Cutler Reservoir, but the data suggests that BAP may not be conserved and is actually removed through the reservoir.
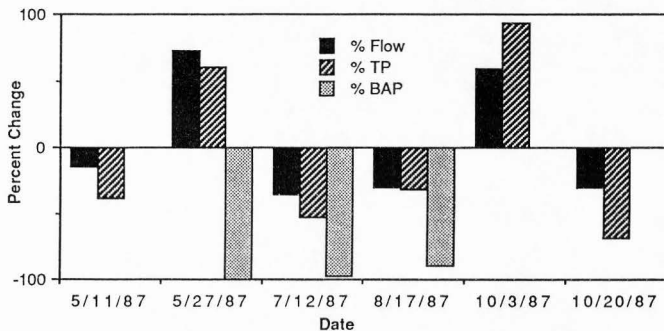

Figure 26.   Percent change of total phosphorus (TP), bioavailable phosphorus (BAP) and flow through Cutler Reservoir [((output - input)/output)*100].

SUMMARY AND CONCLUSIONS

Estimation of Bioavailable Phosphorus

Prior studies had identified that phosphorus was the limiting nutrient for algal productivity in the Bear River system. This study investigates the fraction of total phosphorus (TP) which is available for algal growth, referred to as bioavailable phosphorus (BAP). BAP estimation is best done by using a *Selenastrum capricornutum* Printz algal assay test (algal bioassay test) modified by estimating the equivalent BAP from an algal growth versus PO_4-P concentration standard curve. This test, however, is time-consuming and expensive. There is presently no known chemical method to estimate BAP. Therefore possible BAP indicators were investigated such as total phosphorus (TP), soluble reactive phosphorus (SRP), total organic carbon (TOC), and particle size range (30-0.5 μm, 30-10 μm, 10-0.5 μm). No reliable correlations were found between these parameters and BAP for Bear River water.

Another possible indicator would be the interaction of water hardness, TP, and hydrogen ion concentration (pH). A stepwise regression was done on the BAP data collected from the Bear River. This regression found that the interaction of TP, total hardness, and pH explained 87% of the data variance. Thus it appears this interaction may be a good indicator of BAP for the Bear River. Unfortunately, insufficient time was available during the current research effort to verify the predictive capability of this relationship.

BAP estimation through algal bioassay is site-specific and dependent on the make up of the water sample. If the water has a naturally high pH, aluminum, iron, calcium and/or magnesium

concentration then phosphorus will precipitate out. Studies conducted in the lab on Bear River samples showed that 58% of SRP will be lost within 30 minutes when the pH is increased from 8.7 to 10.0.

An underestimation of BAP will thus result unless precipitation can be minimized. This research has shown that precipitation can be minimized in three ways. First, an external standard curve could be used, so that calcium precipitation would not cause a non-linear standard curve. Second, luxury uptake could be utilized by using a large inoculum that has been phosphorus starved. Available phosphorus would be incorporated into the cell before the pH has increased. An inoculum size of 1.3×10^5 cells/mL resulted in a 17-fold increase of fluorescence over the usual inoculum size (1×10^3 cells/mL) used in the algal assay. Third, the pH during the assay can be controlled by bubbling it with an air/CO_2 gas mixture. For one site, a 2 fold increase in apparent BAP concentration occurred when the pH was maintained by a CO_2/air gas mixture around neutral, as opposed to letting the pH increase to 9.7. These procedures result in a maximum BAP that would give the planner a safety margin in determining the algal productivity potential of the water body. A more accurate estimate of BAP might be obtained by controlling the pH near the natural pH of the water body.

Algal bioassay tests require sterilization for consistent BAP estimations. All the protozoa must be killed to prevent their grazing upon the algae. In order for any available phosphorus associated with particulate matter (up to 20 %) to be accounted for, whole water samples must be used. This eliminated the possibility of sterilization by filtration. Autoclaving resulted in precipitating phosphorus out of solution, while UV radiation, another common sterilization procedure, was unable to kill all of the protozoa. Gamma radiation, the last sterilization process tried, resulted in a relatively small number of samples losing SRP, and was able to kill all of the protozoa. Gamma radiation (2.0-3.5 Mrad in 20 hours, ^{60}Co

source) was used for sample sterilization throughout this study. As the study progressed, it was determined that gamma radiation caused 50% of the samples to increase in SRP by a minimum of 17 μg/L, while only 10% of the samples decreased a minimum of by 7 μg/L. Gamma radiation also resulted in hydrogen peroxide formation, which is toxic to algae. The samples were, therefore, sparged with nitrogen gas for 90 minutes to remove oxygen and prevent formation of hydrogen peroxide. Sparging alone was not enough to eliminate toxicity consistently. Two thousand units/L of peroxidase were added, and the sample was incubated for 48 hours, to remove all of the toxicity. Gamma radiation will need to be studied further to understand its limitations and determine whether it is the best available sterilization procedure for hardwater systems.

BAP was estimated in this study with an inoculum of 10^3 cells/mL, no pH control, and an external standard curve.

The Sources of Bioavailable Phosphorus, Their

Relative Contributions, and Their Control

A prerequisite in developing a phosphorus management plan is to know the relative BAP contributions from each of the sources in the watershed. Several reservoirs are planned for the Bear River system. Previous modeling efforts on the Bear River used fractions of 45% and 85% of TP as available phosphorus, to determine the eutrophication potential of the reservoirs. The studies reported here suggest the available fraction of TP is between 0 to 60% for the Bear River and its tributaries. The models may have over predicted the eutrophication potential due to use of a higher fraction of TP being available.

The BAP contributions from various sources in Cache Valley were estimated, so that a phosphorus management plan could be developed.

BAP was measured on samples from the wastewater treatment plant (WWTP's) effluents. Total phosphorus loadings from livestock runoff were determined by a previous study which had modeled the Cache Valley feedlots. The BAP from land runoff was determined using an average BAP/TP ratio of 0.52 from 4 different rainfall-simulated runoff sites that had been identified as high erosion areas. This ratio was then multiplied by the total phosphorus, calculated by export coefficients, expected from Cache County due to land runoff.

The phosphorus from wastewater treatment plants and livestock runoff is approximately 100% bioavailable while phosphorus from land runoff can have very low bioavailability. In Cache County, point sources (City of Logan lagoons, City of Hyrum WWTP, City of Wellsville lagoons) contribute 28,200 kg BAP/yr, livestock runoff contributes 2500 kg BAP/yr, and land runoff contributes 28,600 to 33,600 kg BAP/yr. The point sources contribute 46%, livestock runoff contributes 4%, and land runoff contributes 50% (the midpoint was used) of the BAP.

The BAP from point sources would probably be easiest to control since three wastewater treatment plants comprise 99% of the total BAP from point sources. Even if only the Logan lagoons had phosphorus treatment (90% reduction), this would reduce the point source BAP contribution by 65% or reduce the overall BAP from all sources by 32%. The livestock runoff comes from approximately 200 sites, and the land runoff comes from 744,000 acres, both of which would be more difficult to control. The wastewater treatment plants appear to be the easiest and most effective place to begin phosphorus reduction.

The majority of the wastewater plants discharge relatively close to the proposed Honeyville reservoir. The data presented here suggest that phosphorus removed from the Logan lagoons effluent may be the most effective in reducing BAP concentrations due to its large contribution of phosphorus. A comprehensive management plan will need

to be developed which will reduce all of the BAP sources before the control of eutrophication is likely to be complete.

RECOMMENDATIONS FOR FURTHER RESEARCH

BAP estimation still has a lot of variables that are not fully understood. BAP estimation is greatly affected by pH changes when excess iron, calcium, magnesium or aluminum are present. Utilization of luxury uptake by using a large inoculum, and controlling pH by bubbling with CO_2, have been briefly investigated and need further research to confirm the initial findings presented here. Indicators for BAP estimation need to be found before the concept of BAP can be widely used because of the time requirement, complexity and expense of the present BAP estimation procedure.

The transport of BAP in a river or reservoir system needs to be better understood. It is known that BAP can be precipitated out, taken up and released by aquatic life, and settled out and suspended again by flow. The significance of these various mechanisms needs to be investigated further.

Once BAP estimation is understood, then computer models need to be developed using BAP. These models will allow a better understanding of BAP transport and allow management decisions on how to control eutrophication.

ENGINEERING SIGNIFICANCE

This study investigated several factors which a phosphorus control manager needs to be aware of when estimating available phosphorus in a hard water system and predicting its accompanying algal growth. Total phosphorus (TP) is not a good predictor of the algal production of a water body because it is not well correlated with the available fraction of phosphorus because much of the phosphorus has precipitated with calcium and is unavailable. This phenomenon probably holds true for any hardwater system that has excess iron, calcium, aluminum or magnesium. There are no indicators or chemical tests that adequately estimate BAP.

Probably the best available method for bioavailable phosphorus estimation is the modified algal assay test described in this study. This test can give misleading BAP estimates in hardwater systems because of phosphorus precipitation. The sterilization procedures of filtration, autoclaving, and gamma radiation used as a preliminary step in the assay, all affected the phosphorus pool. Ultraviolet radiation, another sterilization procedure, was unable to consistently kill all of the protozoa.

For the Cache Valley in northeastern Utah, both the wastewater treatment plants (WWTP's) and land runoff were determined to be significant contributors of BAP. This means that the phosphorus management plan needs to control phosphorus from both the land and WWTP's. The impact of BAP from WWTP effluents was clearly seen in river samples, but BAP from land runoff was often undetectable.

This study found that prediction of algal productivity is very complex, especially for hardwater systems. The manager needs to be aware of the assumptions that were made in estimating BAP, so that proper decisions can be made.

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