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Bear River Water Quality: Bioavailable Phosphorus Measurement, Sources, and Control

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Final Report

to the
Division of Water Resources
Utah Department of Natural Resources

Utah Water Research Laboratory
Utah State University
Logan, Utah

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SUMMARY OF BEAR RIVER WATER QUALITY STUDIES

Background

Recent proposals have been made to develop reservoirs in the Bear River Basin below Bear Lake (Utah Division of Water Resources, 1983; Utah Association of Conservation Districts, 1986). Water in the proposed reservoirs would provide municipal, industrial, irrigation, recreation, and hydropower generation water for northern Utah. Questions have been raised about the suitability of water quality in the Bear River and its tributaries for these uses. In response to these questions a series of water quality studies were begun in 1984 by the Utah Water Research Laboratory under the sponsorship of the Utah Department of Natural Resources, Division of Water Resources. The principal findings of these studies and research published prior to these studies are summarized below.

Bear River Water Quality Studies Before 1984

Sorensen et al. (1986) discussed most of the studies dealing with water quality in the Bear River and its tributaries that were published prior to 1984. Only a dozen or so studies of water quality in the Bear River Basin have been published. Water quality data have been collected at various sites on the Bear River since 1947 (Waddell, 1970). The U. S. Geological Survey (USGS) and the Utah Bureau of Water Pollution Control (BWPC) have entered their water quality monitoring data for the Bear River in the U. S. Environmental Protection Agency's STORET data storage and retrieval system since 1964. More recent monitoring data collected by the BWPC are also available directly from the BWPC.

Previous studies of Bear River water quality have identified high concentrations of fecal indicator bacteria (coliforms), biochemical oxygen demand (BOD_5), salinity, and phosphorus concentrations to be occasionally in excess of beneficial use standards. Concerns over possible high concentrations of nitrate have also been expressed.

Current Water Quality Concerns in the Bear River System

Salinity

Data tabulated by Waddell (1970) covering the period of March 1952 to July 1968 show concentrations of total dissolved solids (TDS) below Cutler Reservoir, near Collinston, Utah, of 900 to 1010 mg/L in samples collected on various summer and fall dates in 1962 and 1964 when discharge rates were less than 25 cfs. However, TDS was as low as 257 mg/L on June 14, 1965 when 1600 cfs were being discharged. An analysis of Bear River STORET data for 1961 through 1969 (UWRL, 1976) found that 11 of 17 (~65%) of the samples taken between 16 May and 31 December (i. e., during the irrigation return and base flow season) near Collinston contained more than 500 mg TDS/L. Sorensen et al. (1986) reported an average concentration of 542 mg TDS/L with a range of from 252 to 1272 mg TDS/L in a review of STORET data from January 1977 to December 1983. The Utah Department of Health (1986) regulates salinity in public drinking water supplies by requiring specific approval of source exceeding 1000 mg TDS/L by the Utah Safe Drinking Water Committee. The U. S. Environmental Protection Agency (1986) has established a criterion of 250 mg/L for sulfate and chloride in domestic water supplies to avoid taste (primarily from chloride) and laxative effects (primarily from sulfate).

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The review of STORET data from January 1977 to December 1983 by Sorensen et al. (1986) found that concentrations of chloride averaged 136 mg/L and ranged from 22 to 530 mg/L immediately below Cutler Reservoir. From 1952 to 1968, at the same location, chloride concentrations ranged from 31 to 410 mg/L (Waddell, 1970). Sulfate concentrations have consistently been below 100 mg/L. These data suggest that salinity concentrations in the Bear River below Cutler Reservoir will probably not be a major concern for municipal and industrial supply but management to decrease salinity concentrations may be necessary during periods of low flow to avoid chloride taste.

A sample of water from the Barrens Marsh had high salinity (EC≈8000 μmhos/cm). The potential impact of this salinity on the water quality of the proposed Barrens portion of the Amalga Reservoir needs to be investigated (Sorensen et al., 1986)

Nitrate

Cundy and Conant (1982) reported nitrate-nitrogen concentrations in excess of Utah standards (Utah Department of Health, 1988) for the protection of wildlife (i. e., 4 mg NO_3-N/L) in Robinson Spring, Cub River drainage. Spring water used for municipal supply by the town of Cornish has been found to have concentrations of NO_3-N in excess of the Utah standard for municipal supply (i. e., 10 mg NO_3-N/L; unpublished data). With these exceptions, a review of historical and current water quality data showed that concentrations of nitrate-nitrogen in the surface waters of the lower Bear River Basin have consistently been below the wildlife protection and the drinking water standards.

Coliforms

Coliforms are enteric bacteria that are used to indicate the presence of fecal pollution and a risk of enteric disease from ingesting contaminated water. In the past, the Bear River has carried relatively high concentrations of coliforms. The UWRL (1976) found that the average total coliform concentration in STORET data from the Bear River at Amalga, Utah, prior to 1974 was 15,000/100 mL but they projected, on the basis of more recent data, that future concentrations would average approximately 1,000/100 mL. This decreasing trend was ascribed to the elimination of industrial effluents from the river.

The review of STORET data from 1977 to 1983 by Sorensen et al. (1986) found an average total coliform concentration in the Bear River below the confluence of the Cub River (near Amalga) to be 800/100 mL with a maximum measured concentration of 2,400/100 mL. Below Cutler Reservoir the total coliform concentration averaged 280/100 mL with a maximum concentration of 240,000/100 mL. Coliform data collected in 1984 and 1985 showed a similar pattern. Relatively high concentrations of fecal coliforms were observed in several samples from the Cub and Little Bear Rivers (Sorensen et al., 1986). Utah coliform standards for raw culinary water supplies are 5,000 total coliforms and 2,000 fecal coliforms per 100 mL (Utah Department of Health, 1988). Concentrations of coliforms in the Bear River and its tributaries are occasionally in excess of these standards. The anticipated die-away of coliforms (and pathogens) in an impoundment (e. g., the proposed Honeyville Reservoir) prior to diversion for treatment and use in municipal supplies will probably reduce the concentrations of coliforms to consistently acceptable values.

Biochemical Oxygen Demand (BOD₅)

In their review of Bear River STORET data through 1973, the UWRL (1976) found no trends in BOD_5 concentrations in the Bear River at Amalga and the Bear River at the Idaho-

Utah state line and estimated that the 1985 mean concentration at these sites would remain approximately 4 and 2 mg/L, respectively. The mean BOD_5 concentrations at the Bear River Idaho-Utah state line and the Bear River below the Cub River confluence stations in STORET data from 1977 through 1983 (Sorensen et al., 1986) were 2.4 and 3.6 mg/L, respectively. Maximum observed concentrations were 7 and 6 mg/L, respectively. In this same STORET data set, BOD_5 concentrations in the Bear River below Cutler Reservoir averaged 4.0 mg/L with a maximum of 17 mg/L.

The Utah BOD_5 standard for the protection of recreation, esthetic, agricultural, and wildlife uses is 5 mg/L (Utah Department of Health, 1988). Samples taken from the Bear River and its tributaries in 1984 and 1985 were generally below 5 mg BOD_5/L with the exception of the Little Bear River above the confluence with the Logan River where 54 percent of the samples had BOD_5 concentrations greater than or equal to 5 mg/L. A sample collected April 2, 1985 at this location contained 10 mg BOD_5/L (Sorensen et al., 1986). Obviously, BOD_5 concentrations in the lower Bear River system occasionally exceed the standard of 5 mg/L, but problems with reservoir water quality due to biochemical oxygen demand carried by river water would not be expected.

<u>Phosphorus</u>

In anticipation of constructing reservoirs on the Bear River system, high concentrations of phosphorus carried by the Bear River and its tributaries are major concerns for water quality. Nutrients, especially nitrogen and phosphorus, are most likely to limit the biological productivity of the proposed reservoirs. Because nitrogen may be supplied through biological nitrogen fixation, phosphorus is most likely to limit plant (e. g., algae) production.

Based on total phosphorus data, empirical and computer modeling estimates of the algal production (eutrophication) in the proposed reservoirs indicated that the reservoirs would experience eutrophic conditions during the spring, summer, and/or fall (Sorensen et al., 1986). More than 90 percent of the suspended sediments in the incoming streams to the reservoirs would be removed by settling in the reservoirs leaving a translucent water column to conduct light for algal photosynthesis.

Phosphorus loads to the Mill Creek and Avon Reservoirs will be primarily associated with spring runoff erosion and, depending on the availability of this phosphorus to algae, may stimulate a spring bloom of algae. Since these reservoirs are expected to undergo thermal stratification during the summer, it is likely that sediment phosphorus will be distributed in the water column again in the fall when the water column becomes isothermal and mixes throughout the depth of the reservoir. This may stimulate a second algae bloom. The intensity of algae production during the spring and fall blooms may result in mesotrophic-to-eutrophic conditions (Sorensen et al., 1986).

Empirical and computer modeling of eutrophication potential, based on total phosphorous loads to the proposed Amalga Reservoir predicted that it would become very eutrophic near the dam and in the Cub River branch. Similarly, eutrophic conditions are likely to develop near the dam in the Honeyville Reservoir. If high populations of grazing zooplankton develop, the conditions may improve to mesotrophic-to-eutrophic status (Sorensen et al., 1986).

Sorensen et al. (1987) investigated sources of phosphorus to the Bear River system and found that much of the total phosphorus carried by the streams that will feed the proposed

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reservoirs was derived from runoff and land erosion in the watershed. Major contributions of total phosphorus were associated with spring snowmelt runoff and rainfall. Soil and streambank erosion in the Battle Creek, Deep Creek, Fivemile Creek, and Weston Creek areas of Franklin County, Idaho, were capable of contributing substantially to the phosphorus load of the Bear River during runoff events. Land sliding in some areas was thought to enhance bank erosion and hence sediment phosphorus loads in streams. Soil erosion control practices commonly used in agriculture hold considerable promise for substantial reductions in phosphorus loads to the streams that will feed the proposed reservoirs. Stream bank erosion control and landslide management may also help reduce total phosphorus loads but the cost effectiveness of these controls needs to be carefully examined.

For algae to grow, nutrient elements and compounds in their environment must be in a form that the algae can assimilate. Phosphorus contained in the mineral lattice of sediment materials cannot be readily assimilated by suspended (planktonic) algae. The growth of planktonic algae depends on dissolved PO_4-P and other phosphorus made available through extracellular enzymatic activity.

Estimates were made of the algal available (bioavailable) phosphorus transported by the Bear River and its tributaries (Current report). Wastewater treatment facility effluents contributed 20 percent of the total phosphorus and 46 percent of the bioavailable phosphorus entering the Bear River system below Bear Lake. Nearly 10 percent of the total phosphorus may come from the Logan City Lagoon effluent alone. Land runoff contributed up to 80 percent of the total phosphorus and approximately 48 percent of the bioavailable phosphorus. Feed lot and dairy farm lot runoff probably contributed less than 1 percent of the total phosphorus and a maximum of 5 percent of the bioavailable phosphorus passing Cutler Dam in 1985.

The most cost effective control of bioavailable phosphorus would probably be the control of wastewater treatment plant (WWTP) effluent phosphorus. If the phosphorus in the Logan City wastewater lagoon effluent were reduced by 90 percent, the bioavailable phosphorus load to the lower Bear River Basin would be reduced by approximately 32 percent. Wet lands application and/or ferric chloride treatment may be the most cost effective method for phosphorus removal from the Logan City wastewater lagoon effluent.

A rough mass balance of bioavailable phosphorus transport through Cutler Reservoir indicated that bioavailable phosphorus was removed (possibly transformed to unavailable forms and/or deposited in the sediments) in the reservoir. This may greatly reduce the impact of bioavailable phosphorus on algae production in the proposed Honeyville Reservoir. Laboratory algal assays may have underestimated bioavailable phosphorus due to precipitation of phosphorus with calcium and other hardness metals prior to assimilation of the phosphorus by the test organism. Similar precipitation reactions may happen naturally in streams and reservoirs with the hard waters of the Bear River system and help to limit bioavailability of phosphorus in this system.

Water Quality Effects of Exchanging Bear River
Water in Willard Reservoir

An alternative plan for extending the use of Bear River water involves exchanging Weber River water, currently stored in or passed through Willard Reservoir, for Bear River water, thus making Weber River basin water available for other use. Sorensen et al. (1987) investigated the water quality impacts on Willard Reservoir of such an exchange.

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Willard Reservoir was found to be a frequently mixed (polymictic), eutrophic water body with spring and early summer algal blooms that were dominated by *Apanizominon* sp. Chlorophyll *a* concentrations in the surface waters exceeded 66 μg/L (eutrophic range) at one location in March 1987 but were less than 2 μg/L (oligotrophic) at three locations in late August. Empirical models predicted that exchanging Bear River water for Weber River water would not produce a noticeable change in the algal production and trophic state of the reservoir.

On May 28, 1986 the salinity in the surface waters of Willard Reservoir averaged 227 mg TDS/L. By August 20, 1986, the average salinity had increased to 308 mg TDS/L. The historical maximum salinity of the reservoir is approximately 920 mg TDS/L. A mass balance technique was used to calculate the salinity that is likely to develop in Willard Reservoir as a result of exchanging Bear River water for Weber River water. Substituting Bear River water for Weber River water resulted in a calculated increase of 190 mg TDS/L. A salinity increase of 100 to 200 mg/L is not anticipated to interfere with any projected uses of Willard Reservoir water.

<u>Recommendations and information needs</u>

Just because this study indicates that the bioavailable phosphorus load to the proposed reservoirs, especially the Honeyville reservoir, may be low does not necessarily mean that algal production in the reservoirs will be low. This interpretation would be in opposition to the empirical trophic state model predictions of eutrophic conditions based on total phosphorus loading. Eutrophic conditions in Hyrum Reservoir (Sorensen et al., 1986), a reservoir which is somewhat similar to the proposed Honeyville Reservoir also suggest that the Honeyville Reservoir will be eutrophic. This apparent inconsistency in results calls for cautious planning in reservoir development until a more complete understanding of the factors that affect primary production and their impact upon water quality is obtained.

Cutler Reservoir includes considerable amounts of marsh area in the southern portion through which the waters of the Little Bear River flow. There is evidence that bioavailable phosphorus is removed or transformed into chemical forms not readily available to algae through Cutler Reservoir. More information is needed on this "removal" of BAP. Information about the ultimate fate of the phosphorus entering Cutler Reservoir and the potential for release of BAP from the reservoir during fall and winter months when vegetation decays may be especially important to the water quality management of the proposed Honeyville Reservoir.

Similarly, a laboratory assessment of the potential for phosphorus release from the sediments of the proposed reservoirs is needed under aerobic and anaerobic conditions. Microcosm studies may help elucidate the sediment phosphorus cycling potential of reservoir sediments in these hard water reservoirs.

The chemistry of Weber River water diverted into Willard Reservoir is not very different from Bear River water and yet Willard Reservoir supports dense algal blooms, indicating that phosphorus availability is high. Careful comparison of the Hyrum and Willard Reservoirs (and other similar reservoirs) with the proposed Honeyville Reservoir in terms of eutrophication potential would help predict the quality of the proposed reservoir. A more complete socio-economic analysis of the downstream benefits of BAP and eutrophication control is needed before confident planning for recreational and other beneficial uses of the water proceeds.

BIOAVAILABLE PHOSPHORUS IN THE BEAR RIVER SYSTEM

Introduction

Overview

Excessive algal blooms, associated with eutrophication, cause water quality problems in many reservoirs that supply water for domestic, recreational, agricultural, and industrial uses. Algal blooms will degrade recreational aspects of reservoirs due to odors, esthetics, hindrance to swimmers, and clogging of boat motors. Drinking water treatment costs will be increased because of the need to remove odors and tastes associated with algae, the production of trihalomethane precursor material, clarification problems, and higher chlorine demand. 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 produce taste and odor compounds. When algae settle to the bottom of the reservoir and begin to decompose, dissolved oxygen is consumed resulting in anaerobic conditions which may mobilize iron and manganese and cause additional potable water use problems (Miller et al., 1983). High concentrations of algae may deplete dissolved oxygen at night when photosynthesis ceases and oxygen continues to be used for respiration. Fish and invertebrate animal life may be threatened by insufficient dissolved oxygen concentrations.

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 factor (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 relatively 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 were 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

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due to the additional nitrogen often made available through nitrogen fixation by cyanobacteria (blue-green algae) (Porcella and Bishop, 1975; Maki et al., 1984). Carbon is the limiting nutrient only if the algal production of the water body is so high that the rate of CO_2 absorption by the water from the atmosphere is too slow to meet the demands of the algae. However, carbon is usually in excess and it is impractical to control it in most natural aquatic environments. For these reasons, the present study focused on phosphorus management.

Phosphorus can occur in four forms in water: (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 microbial 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) before they can become readily available to algae as dissolved orthophosphate (Tchobanoglous, 1979; Porcella and Bishop, 1975; Van Wazer, 1973).

Sources of Phosphorus to Rivers, Lakes and Reservoirs

Cycling of phosphorus from sediments, degradation of organic phosphates, and the hydrolysis of polyphosphates to orthophosphates may serve as phosphorus sources to algae. However, the primary, though perhaps not the most 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 percent of the total phosphorus (TP) load to Lake Erie came from WWTP's, but agricultural runoff contributed 52 percent of the TP load to Lake Canadarago, New York. 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.

In some watersheds land runoff is the biggest contributor of TP. Crop lands can be the major contributor of soluble phosphorus depending upon the amount and method of fertilizer application. Range lands can have substantial quantities of decaying vegetation upon their surfaces which may be high in organic phosphorus and 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 nonpoint pollution problems. When the controlling factor is actually land form (soil composition). Runoff from soils high in clay and organic matter are usually enriched in TP relative to 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 also had a higher TP content than the original soil (Sharpley, 1980a). Runoff will contain enriched concentrations of suspended clay (because of clay's smaller particle size and dispersive properties) and enriched concentrations of organic matter (because of organic matter's 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

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large surface areas and a negative charge. 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 the phosphorus was unavailable to algae.

The rocks in the soil can also be important for 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 descriptions in regards to the erodibility and phosphorus characteristics of soil for croplands: sandy/gravel soils do not erode easily, have a low cation content, and 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), observed that the consequence of accounting bioavailable phosphorus (BAP) pollution instead of total phosphorus 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 be unavailable.

The time of year as well as soil composition are significant influences on the amount of phosphorus transported off of the land. Spring snow-melt runoff has been shown to carry greater mass loads of phosphorus than runoff during other times of the year (Hanson and Fenster, 1969). Most phosphorus from snowmelt runoff 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 spring runoff and the residues are easily removed with overland flowing water (Porcella and Bishop, 1975).

The length of time phosphorus that is associated with particulate matter remains in suspension 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.

Irrigation return water may contain relatively low phosphorus concentrations since many soils immobilize phosphorus. This was demonstrated in a study on the Snake River where the river had phosphorus concentrations of 21 μg/L; however, drainage irrigation water, originally diverted from the river, had 4 μg/L. This is a 70 percent 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) (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).

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WWTP's are often major contributors of phosphorus to water bodies (Loehr, 1974). 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 percent of the insoluble phosphorus is removed during the primary treatment of wastewater. Secondary wastewater treatment transforms polyphosphates and organic phosphates to orthophosphorus, but removes very little phosphorus (Tchobanoglous, 1979). This means that secondary treatment increases bioavailability of the phosphorus.

Young et al. (1982) present data that confirms the transformation of influent phosphorus to bioavailable phosphorus (BAP) in the WWTP process by showing that influent wastewater total phosphorus is 60 to 80 percent BAP while the effluent total phosphorus was 83 percent BAP. WWTP's 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 Transformations and Cycling

Phosphorus in the aquatic environment is continually changing between biologically 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) suggested 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) concluded 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 the conversion of initially available phosphorus to unavailable forms.

Phosphorus also cycles in lakes. 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. Phosphorus was cycled from the cells to the water column (i. e., the phosphorus was released back to the water column by the death and subsequent decomposition of the cells). Hooper (1973) stated that phosphorus compounds may become available in lakes by in-situ decomposition of the phytoplankton and zooplankton organisms themselves, excretion by the plankton organisms, regeneration from bottom sediments and transport to the photosynthetic zone, and 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.

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

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 for ferrous iron than phosphate, so phosphate will be replaced and become available in the orthophosphate form (Messer and Ihnat,

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1983). This process will be reversed when aerobic conditions occur and phosphorus will become unavailable (Holt et al., 1970). Phosphorus continually cycles between biologically available and unavailable forms.

Bioavailable Phosphorus

Bioavailable phosphorus is defined as the phosphorus which can be readily used by algae. Bioavailable phosphorus is usually only a fraction of total phosphorus, while soluble reactive phosphorus (SRP), determined by the ascorbic acid method (APHA, 1980), can be greater or less than BAP. Generally, a large fraction of BAP in natural waters is orthophosphate (Porcella and Bishop, 1975).

Bioavailable Phosphorus Estimation

Algal bioassays (Miller et al., 1978) can be used to estimate BAP. Major disadvantages of BAP estimation is its expense and time (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. The U.S. Environmental Protection Agency (1980a) shared this opinion stating that BAP estimates 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 BAP, so BAP may not 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 phosphorus loading models for the U. S. Organization of Economic Cooperation and Development (OECD). The U. S. portion of the North American OECD 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 estimates of potential algal production (Bradford and Peters, 1987).

From studies conducted in New Jersey, Trama and McIntosh (1985) suggested that BAP could be estimated by soluble reactive phosphorus (SRP) and total phosphorus. They found for one watershed 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.

Soluble reactive phosphorus is often considered 100 percent BAP. Rigler (1973) stated that SRP analysis could have two errors when used to predict orthophosphate: filtration might damage delicate algal cells and release phosphate phosphorus or readily hydrolyzed phosphate esters into the filtrate; or H_2SO_4, used in the analysis, could hydrolyze free phosphate esters and release orthophosphorus from fulvic acid-metal phosphates or from colloidal iron phosphate. Rigler (1973) felt that the discrepancy between SRP and orthophosphorus will probably prove to be significant to algal production only in unpolluted lakes in which phosphorus is in short supply. 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.

Sodium hydroxide extractions have been used to estimate BAP for sediment core samples containing iron. Sodium hydroxide extractable phosphorus primarily represents phosphorus loosely bound to the surface of ferrous-ferric hydrous oxide gels which would

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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 measurements are only applicable for one moment in time and for the conditions under which it is measured. BAP is a very dynamic parameter; organic phosphorus becomes available over time with biochemical attack and dissolution; condensed phosphates become available by hydrolysis, and a change in pH may affect 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.

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). Algal bioassays can estimate toxicity by using dilution techniques and estimate the specific limiting nutrient via spiking with one or more nutrients by observing the response of the algae (Porcella and Bishop, 1975). Bioassays 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 nonpoint sources to define their impact upon receiving waters. (p. 5)

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

Phosphorus Management

The relative contribution of phosphorus for each source 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, much of which is unavailable to algae. Total phosphorus loads from WWTP's, however, are almost 100 percent available.

Phosphorus management needs to control current and future sources to assure long term control of algal production. After a phosphorus management plan is implemented, more time may be required to reduce algal blooms. This time lag might be caused by sediment storage of phosphorus and its later release to the overlying waters by organic

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matter decay or redox reactions. It has not been satisfactorily demonstrated that a reduction in phosphorus concentration alone will cause an immediate reduction in algal blooms. Generally, in-lake phosphorus concentrations must be reduced below some threshold level before algal production decreases substantially. 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 percent 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 a 92 percent reduction of phosphorus in WWTP effluents may be required before achieving a measurable improvement in water quality. How much reduction in phosphorus is required to actually begin improving water quality depends upon the threshold level. Usually, a comprehensive phosphorus management program needs to incorporate control of both nonpoint and point sources. In 1976, sixteen states had phosphorus limits between 0.1 to 2 mg/L for WWTP effluents (U.S. Environmental Protection Agency, 1976). The State of Utah has regulated water quality by setting in-stream standards of 0.05 mg/L PO_4-P and a standard for lakes and reservoirs of 0.025 mg/L PO_4-P (Utah Department of Health, 1988).

Phosphorus in the Bear River System

The Bear River System

The Bear River originates from 10,000 ft elevation on the north slope of the Uintah mountains in northeastern Utah and travels back and forth for 420 miles through Wyoming, Idaho, and Utah, terminating in the Great Salt Lake (Utah Water Research Laboratory, 1974). The Bear River Basin area is approximately 150 miles long from north to south and 100 miles wide from east to west; it encompasses 7340 square miles of primarily agricultural land (Figure 1). The area receives precipitation of 8 to more than 40 in/yr (Utah Water Research Laboratory, 1976).

The rocks in the mountains which feed the Bear River are largely sedimentary, and 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 focused on bioavailable phosphorus in the Bear River Basin from Bear Lake to Honeyville, Utah (the lower Bear River Basin). Emphasis was placed on the portion of the Bear River system contained in Cache Valley which includes Franklin County, Idaho and Cache County, Utah (Figure 1). Most of the proposed reservoirs are to be located in Cache Valley; important sources of phosphorus that may impact these reservoirs are found in Cache Valley.

Cache County encompasses approximately 753,500 acres; 40 percent of the land is federally or state owned and is used for forest and grazing land. Fifty-seven percent is privately owned and primarily used for agriculture (Cundy and Conant, 1982). The remaining three percent is mainly urban (Logan, Utah, is the largest town--population

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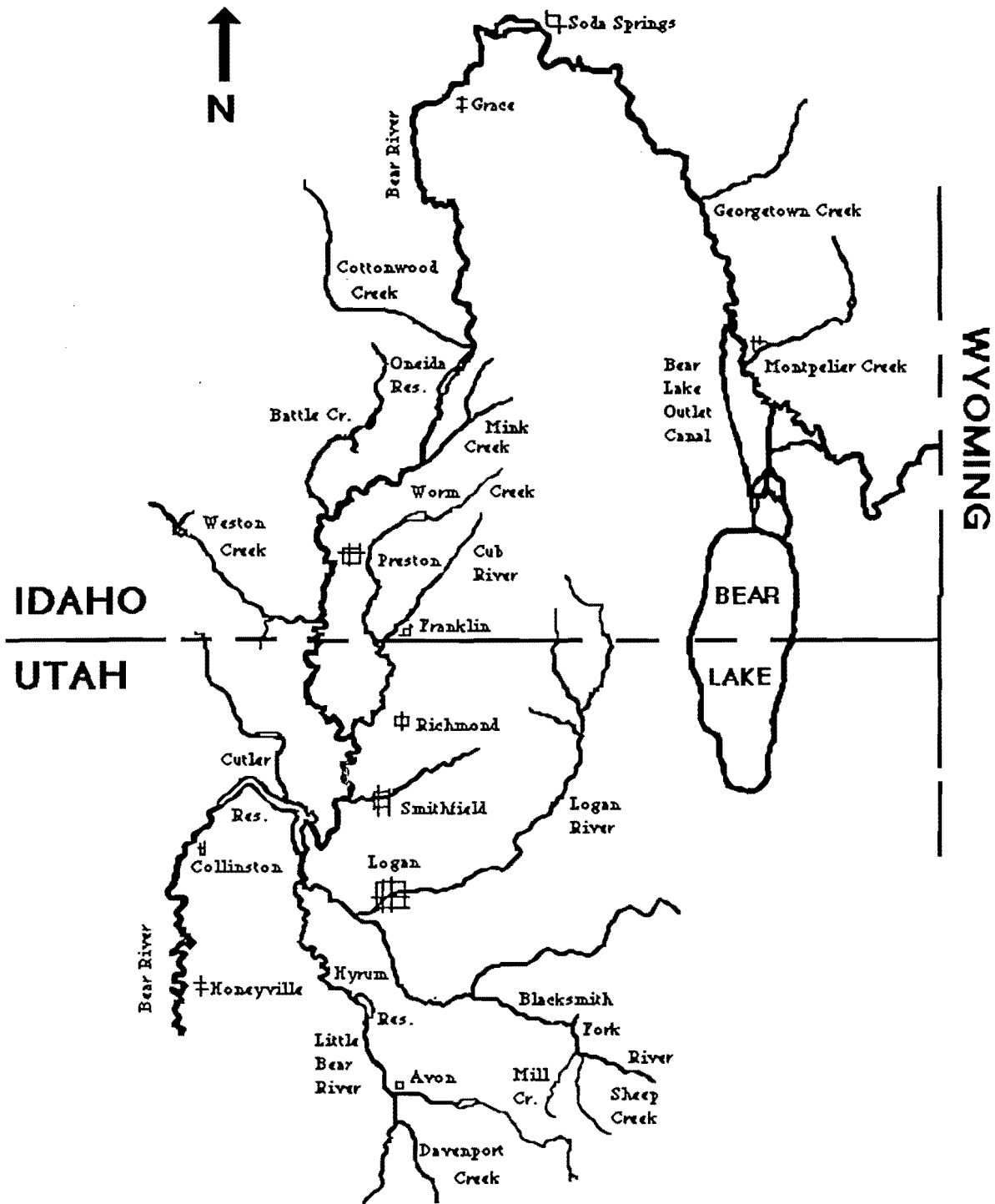
Figure 1. Schematic map of the major rivers and reservoirs in the Bear River Basin.

30,600 in 1985; Cache Economic Development, 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 other half 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.

Bear River Water Chemistry

Table 1 summarizes seven recorded years of Bear River chemistry at the Utah Power and Light Company gauge below Cutler Reservoir (USGS station 10118000 and Utah Bureau of Water Pollution Control station 490198), 40 mi (64 km) upstream from the Great Salt Lake. Table 2 summarizes water quality data at this station from May 1984 through April 1985. It is clear from these data that the Bear River at this point is a turbid, warm, alkaline, hard water river.

Table 3 lists several wastewater sources that impact water quality, including the phosphorus load of the Bear River and its tributaries. Cache Valley has three WWTP's that discharge greater than one million gallons per day (MGD) to surface water. The City of Logan lagoons discharge, on the average, 10 MGD while both Preston and Hyrum's WWTP's discharge approximately one MGD. There are several smaller lagoon systems that discharge less than 0.25 MGD. The industrial wastewater sources include trout farms, slaughter houses, and vegetable canning. E. A. Miller Inc. (beef packing plant) 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).

Bear River Phosphorus Dynamics

Earlier research has investigated phosphorus sources and transport in the lower Bear River system (Sorensen et al., 1986; Sorensen et al., 1987). Based upon the flow weighted average concentration of total and soluble reactive phosphorus (Table 2) and the total flow passing Cutler Dam from May 1, 1984 through April 30, 1985 (3,161,213 acre-feet; ReMillard et al., 1984, 1985), approximately 470 ton (430,000 kg) of total phosphorus and 170 ton (160,000 kg) of soluble reactive phosphorus passed Cutler Dam (Sorensen et al., 1986).

Sorensen et al. (1987) monitored Bear River total phosphorus and soluble reactive phosphorus concentrations at various locations from the Bear Lake Outlet canal, Idaho, to Honeyville, Utah in mid-April, May, June, August, and early November, 1986. They found that total phosphorus concentrations were relatively high in April and June and relatively low in May, August, and November. Total phosphorus concentrations tended to increase downstream at sustained rates, ranging from approximately 1.5 to 0.05 μg/L·mi. Soluble reactive phosphorus concentrations were highest in May and June (~50 μg/L) and lowest in August and November (~10 μg/L). Soluble reactive phosphorus concentrations were surprisingly uniform over the monitored 200 miles of the river. The highest sustained rate of soluble reactive phosphorus concentration increase was approximately 0.13 μg/L·mi on June 10. On August 13, the soluble reactive phosphorus concentration at Honeyville was only 6 μg/L higher than in the Bear Lake Outlet Canal, and concentrations actually decreased slightly from below Oneida Reservoir to Honeyville.

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Table 1. Summary of selected Bear River water quality measurements below Cutler Reservoir (STORET station 490198) from January 1977 through February 1983.

Variable	Units	Mean	Std. Dev.	Minimum	Maximum	No. Observations
Temperature	°C	11.4	8.7	0	26	36
Dissolved Oxygen	mg/L	9.9	3.2	5.2	17.9	34
Conductivity	μmhos/cm @ 25 °C	924.4	426.8	450	2140	42
Total Dissolved Solids	mg/L	541.6	241.4	252	1272	45
pH	units	8.2	0.6	7.4	10.5	34
Sodium	mg/L	94	90.7	18	360	37
Potassium	mg/L	8.5	4.1	1	18	37
Calcium	mg/L	60.3	9.5	42	81	36
Bicarbonate	mg/L	304.7	41.6	196	384	37
Carbonate	mg/L	3.5	9.6	0	54	36
Chloride	mg/L	135.9	140.8	22	530	36
Sulfate	mg/L	50.7	15.2	21	75	37
Fluoride	mg/L	0.29	0.13	0.03	0.6	18
Total Alkalinity*	mg/L	253.6	32.8	179	315	37
Carbon Dioxide	mg/L	3.3	1.7	1	8	37
Total Hardness*	mg/L	290.6	39.9	194	360	37
Silica (dissolved)	mg/L	13.3	2.8	7	17	18
Total Iron	mg/L	0.52	0.42	0.08	1.92	37
Total Arsenic	μg/L	3.3	1.5	1.5	7	31
Total Cadmium	μg/L	1	0	1	1	29
Total Copper	μg/L	10.5	2.4	5	15	30
Total Lead	μg/L	5.8	1.9	3	10	29
Total Mercury	μg/L	0.15	0.23	0	1.3	35
Total Selenium	μg/L	0.73	0.24	0.5	1	30
Total Zinc	μg/L	22	21	5	110	31
Suspended Solids	mg/L	52.2	48.3	0.5	175	39
Turbiditiy	FTU	28.6	25.2	1.9	100	36
Total Kjeldahl Nitrogen	mg/L	0.86	0.95	0.2	5	41
Ammonium Nitrogen	mg/L	0.22	0.3	0	1	43
Nitrate + Nitrite Nitrogen	mg/L	0.53	0.3	0.1	1.15	35
Nitrate Nitrogen	mg/L	0.53	0.4	0	1.4	20
Total Phosphorus	mg/L	0.12	0.05	0.05	0.2	40
Orthophosphorus	mg/L	0.06	0.03	0.02	0.1	18
Total Organic Carbon	mg/L	10.2	9	1	38.8	34
Chemical Oxygen Demand	mg/L	20	13.6	4	85	38
Biochem. Oxygen Demand (5-day)	mg/L	3.95	3.69	1.2	17	18
Total Coliforms	MPN/100 mL	280	14	23	240000	17
Fecal Coliforms	MPN/100 mL	210	12	4	23000	17

*As Calcium Carbonate.

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Table 2. Summary of selected Bear River water quality measurements below Cutler Reservoir (STORET station 490198) from May 1, 1984, through April 30, 1985.

Variable	Units	Flow Weighted Average	Simple Mean	Std. Dev.	Minimum	Maximum	No. Observations
Temperature	°C	11.47	10.54	8.19	-0.2	22.7	12
Dissolved Oxygen	mg/L	8.16	8.07	1.41	5.9	10.6	12
Conductivity (Field)	μmohs/cm	603	623	75.09	445	717	12
Total Dissolved Solids	mg/L	366	378	44.18	258	418	12
pH	Units	8.15	8.33	0.46	7.8	9.2	12
Sodium	mg/L	34	36	7.32	16	44	12
Potassium	mg/L	5	5	1.44	3	9	12
Calcium	mg/L	57	59	6.49	51	71	12
Magnesium	mg/L	31	32	8.10	18	41	12
Chloride	mg/L	38	40	8.59	18	51	12
Sulfate	mg/L	41	43	11.01	21	56	12
Total Alkalinity*	mg/L	239	246	23.71	192	277	12
Total Hardness*	mg/L	269	279	45.15	200	347	12
Total Iron	mg/L	0.52	0.46	0.34	0.03	0.94	12
Total Manganese	μg/L	53	50	31.91	10	105	12
Suspended Solids	mg/L	73	67	54.26	7	193	12
Turbidity	NTU	36.58	33.21	20.58	4.9	66	12
Total Kjeldahl Nitrogen	mg/L	0.73	0.73	0.26	0.3	1.2	12
Ammonium Nitrogen	mg/L	0.19	0.20	0.15	0.1	0.5	12
Nitrate Nitrogen	mg/L	0.72	0.74	0.34	0.35	1.61	12
Nitrite Nitrogen	mg/L	0.03	0.03	0.04	0.01	0.14	12
Total Phosphorus	mg/L	0.11	0.12	0.05	0.03	0.2	11
Orthophosphorus	mg/L	0.04	0.04	0.03	0.01	0.12	11
Total Organic Carbon	mg/L	23.21	25.22	27.24	6.2	104.2	12
Chemical Oxygen Demand	mg/L	18	17	9.17	10	35	12
Biochem. Oxygen Demand	mg/L	3	3	0.78	2	4	12
Total Coliforms (MF)	#/100 mL	388	379	276.01	30	800	8
Fecal Coliforms (MF)	#/100 mL	186	201	416.12	0	1300	9
Fecal Streptococci (MF)	#/100 mL	546	516	332.93	50	1100	9
Flow Passing Cutler Dam	cfs		4188.50	1452.99	2640	8258	12

*As calcium carbonate.

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Table 3. 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.
Del Monte	(July-August discharge only)		
	Canning	0.54	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.
White's Trout Farm	Fish Waste	8.6	L. Bear R. to Bear R.
E. A. Miller Inc.	Slaughterhouse	0.75	Spring Cr. to L. Bear R.
Hyrum City	Domestic	0.6-1.2	Spring Cr. to L. Bear R.
Wellsville City	Domestic	0.07	L. Bear R. to Bear R.

Total phosphorus concentrations in the Little Bear River and Davenport Creek, which will feed the proposed Avon Reservoir, were highest during spring runoff when sediment loads were high and decreased to relatively low conentrations during the remainder of the year. Soluble reactive phosphorus concentrations tended to be higher (≤ 65 µg/L) in April and May but decreased to less than 20 µg/L at several locations in the summer and fall samples.

Similarly, phosphorus concentrations in the Blacksmith Fork River and Mill Creek, which will feed the proposed Mill Creek Reservoir, were highest during spring run off and decreased during the summer and fall. Upland erosion during runoff affected total phosphorus concentrations most. Concentrations of both total and soluble reactive phosphorus were consistently higher in Mill Creek than in the Blacksmith Fork. Soluble reactive phosphorus concentrations were consistently below 20 µg/L in the Blacksmith Fork at Anderson Ranch while Mill Creek soluble reactive phosphorus was consistently above 25 µg/L.

In summary, lower Bear River system phosphorus dynamics reflect the influence of surface runoff events with total phosphorus being the more dynamic phosphorus measure. Soluble reactive phosphorus concentrations tend to be more uniform. Summer and fall, 1986, soluble reactive phosphorus data on the Bear River are so uniform over a river distance of approximately 200 miles that sources and sinks of soluble reactive phosphorus appear to be nearly balanced. If soluble reactive phosphorus reflects biological availability of phosphorus, BAP concentrations may be relatively low in the Bear River and its tributaries below Bear Lake.

Study Objectives

Bioavailable phosphorus is the limiting nutrient for algal growth in the Bear River system. This study was designed to better understand BAP and to perform an economic evaluation of BAP control from wastewater treatment operations.

The following specific objectives were addressed in this study:

1. Development of a procedure for estimation of bioavailable phosphorus for the Bear River.

2. Identification of the sources of bioavailable phosphorus in the lower Bear River basin.

3. Determination of the relative contribution of bioavailable phosphorus from the various sources.

4. Economic assessment of effective methods for bioavailable phosphorus control.

Materials and Methods

Water Sampling and Sample Preparation

Samples from wastewater treatment facility effluents, land runoff, streambank soils, and the Bear River and its tributaries were collected during this study. 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. gauge 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 gallon bucket suspended by a rope into the stream and using the water thus collected to rinse and fill a 0.5 gallon 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, and 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 investigated the relationship of suspended particle size range to BAP. Suspended solids size fractions were determined for several samples by initially passing the sample through a 250 μm brass sieve to remove debris, sand, and relatively large aggregates of particles. Then approximately one liter of the sieved sample was placed in an ultrasonic bath (Bransonic 12)for two minutes to disperse aggregated soil particles. Following sonication, the sample was vacuum-filtered through a 30 μm opening nylon mesh (Spectramesh®) fabric. To separate the 30 to 10 μm 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

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10 μm filter was also 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.

Soil and Runoff Sampling and
Sample Preparation

A special study used streambank soil samples to determine the possible contribution that streambank erosion may have upon the phosphorus load of the Bear River system. This study investigated four locations that had recent erosion. Table 4 lists the location of the sampling sites and the textural classification of their soils. The streambank samples were taken at 1 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 cm^3 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. 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 for five minutes. The suspension was passed through a 325 mesh brass sieve (~45μ), a 30 μ Spectramesh® nylon fabric, and then diluted to 3600 mL with DDW. These samples were then tumbled at 30 rpm for one hour. A portion of each suspension was centrifuged, the supernatant was decanted and SRP and TP 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 in preparation for BAP determination.

Another special study was conducted to determine the possible contribution of phosphorus to the Bear River system from land runoff. Four sites were selected for their proximity (within 0.5 mi.) to a stream and for their high erosion potential (USDA, 1974). Table 4 lists the location and textural classification of the soil of each of these sites. The soil at the site above Cutler Reservoir was classified as a Wheelon silt loam with 30-50 percent eroded slopes and was a grazing area with small tufts of vegetation and a gravelly soil. The site adjacent to Weston Creek was classified as a Bingham gravelly loam with 0-60 percent 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 percent eroded slopes and had wheat stubble on the soil. The Blacksmith Fork River site was classified as a Yeates Hollow extremely rocky silt loam with 30-70 percent slopes and was grazing land with sage brush and grasses with approximately 20 percent open space. 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

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Table 4. Sampling locations and textural classification of the soils at the streambank and runoff study sites in Cache Valley.

Sample Type	Site	Location (North Latitude/ West Longitude)	Soil % clay	% silt	% sand	Texture Classification
Streambank	Battle Creek	42°08'48"/111°54'52"	32.5	35.0	32.5	clay loam
	Weston Creek	42°01'38"/111°57'21"	25.0	46.0	29.0	loam
	Bear R. @ Amalga	41°53'27"/111°52'26"	14.5	21.0	64.5	sandy loam
	Little Bear R.	41°38'01"/111°53'19"	12.0	19.0	69.0	sandy loam
Runoff	Blacksmith Fork	41°35'56"/111°37'05"	22.3	60.5	17.3	silt loam
	Weston Creek	42°01'50"/111°58'22"	18.0	37.5	44.5	loam
	Cub River	41°54'46"/111°51'52"	32.5	51.5	16.0	silty clay loam
	Abv. Cutler Res.	41°51'03"/112°02'25"	22.3	50.0	27.8	silt loam

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 5). 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.

Runoff samples were sonicated with a Biosonic IV probe sonicator (high setting, mid scale) and mixed with magnetic stirrers for five minutes. The sample was allowed to stand for one minute so that coarse 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 resonicated and shaken for five minutes to resuspend the sample. Four hundred milliliters (minimal) of the sample were 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, 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 one hour. Duplicate 30 mL aliquots from each site were removed immediately and centrifuged at 800 X g for 10 min., the supernatant was then passed through a prerinsed 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 was poured into glass bottles, sparged with nitrogen gas, capped, sealed with paraffin, and sent with the streambank soil suspensions for gamma radiation sterilization in preparation for BAP determination.

Table 5. Runoff plot characteristics for four sites (2 plots/site)where runoff was collected from simulated rainfall.

Site	Plot #	Plot size	Slope	Rainfall intensity	Rainfall depth
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

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 with DDW so at least 50 percent 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.

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 gauge stations or was estimated by multiplying the average river velocity by the cross-sectional area (Dunne and Leopold, 1978). The cross-section profile was obtained using surveying techniques to determine water surface elevation relative to a reference point and the river bed profile (the profile was determined by using a sonar depth finder) (Sorensen 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™ Excel) with the stream cross-section areas and using the stream depth and velocity for flow determination.

While 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 measured using a YSI model 33 S-C-T conductivity meter.

Soil texture was determined by the hydrometer method (Gee and Bauder, 1986).

Chemical Analyses

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

A special study involved determining total organic carbon (TOC) on several water samples to see if TOC could be an indicator of BAP. Total organic carbon was measured 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 by using an Oceanography International Carbon Analyzer (model 05248) which uses persulfate oxidation with infrared absorption detection of evolved CO_2 (Oceanography International, Inc., 1977).

All of the water samples were analyzed for SRP and TP. Persulfate digestion (APHA, 1980) was used for TP analysis. SRP samples and digested TP samples were analyzed by the manual ascorbic acid method (Strickland and Parsons, 1972).

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

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determine water soluble phosphorus concentrations based on a standard curve prepared with the sample.

By using the Olsen and Sommers method (1982) the Soil, Plant, and Water Analysis Laboratory at Utah State University analyzed the total phosphorus in the soil samples.

Percent calcium carbonate in the soil was determined using a pressure-calcimeter method (Nelson, 1982). A 1000 mL flask was used and the pressure reading was taken on a mercury manometer once the reading had stabilized.

Soil pH was determined by a saturated soil paste method (Richards et al., 1969).

Algal Assay Procedures

To determine BAP, algal bioassays were performed following a modified version of the Environmental Protection Agency (EPA) Algal Assay Procedure (AAP) protocol (Miller et al., 1978). Algal growth in nutrient (nitrogen or nitrogen plus phosphate) amended, sterilized cultures was monitored as *in vivo* fluorescence of chlorophyll. Maximum standing crop of the algae (i. e., maximum fluorescence) in the culture was used to indicate the available phosphorus in the culture.

Since not all available phosphorus may be in solution, filter sterilization of the samples was not acceptable. Autoclave sterilization caused precipitation of a large fraction of the SRP in Bear River system samples (Sorensen et al., 1987) and could not be used. Therefore, sterilization with electromagnetic radiation was used.

Sample sterilization by gamma radiation

Ultraviolet (UV) radiation sterilization was tried initially because no known chemically ractive or toxic by-products are formed during radiation and a Model SPF teflon-tube ultraviolet device (Ultraviolet Technology Inc., El Toro, CA) was available (Harris, 1986). Three experiments investigated the use of UV radiation for bioassay sample sterilization. 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 test algae and result in an underestimation of BAP since it is based on algal growth, so UV radiation was judged unacceptable (Barker, 1988).

Gamma radiation had been used successfully by Dorich et al. (1984) for sterilizing concentrated sediment suspensions prior to BAP estimations. This was evaluated 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 dose of 2.5 to 3.5 Mrad (cobalt-60 source) during an exposure period of approximately 20 hr.

The first algal bioassay performed using 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. Low concentrations of hydrogen peroxide, produced by ions and free radicals in the water during the gamma radiation treatment (Turner, 1986), may have persisted in the samples after irradiation resulting in toxicity to the algae. Despite

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subsequent efforts to strip oxygen from samples by sparging with N_2 gas prior to irradiation and thus minimizing the formation of hydrogen peroxide, toxicity problems were frequent. Peroxidase enzyme (Type IV and VI, Sigma Chemical Co., St. Louis, MO) was evaluated to see if it could break down peroxide in irradiated samples and eliminate toxicity. 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. Through an experiment the optimum time and enzyme concentration needed to eliminate toxicity was determined.

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 peroxide formation during irradiation. A Microtox™ test (Microbics Corp., Carlsbad, CA) was used 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.

Assay procedure

Aliquots of sterilized and toxicity free river water were introduced into three 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 to verify the linearity of algal growth response to the P additions.

After the third day of incubation of the test flasks, *in vivo* fluorescence measurements were made daily 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 BAP 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 growth response curve was prepared with AAP medium in which the P concentration was adjusted over a range of 10 to 250 µg/l. Three 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 on the algal bioassay.

During the first study the effect of pH change on soluble reactive phosphorus was determined. 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. Three 100 mL aliquots of filtered sample were placed in 500 mL erlenmeyer flasks for each pH level. Tris/HCl buffers (2M) were prepared for pH control in the 8.0 to 9.0 range. Carbonate/bicarbonate buffers (2M) were used for pH values between 9.2 to 10.3. 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 and a glass electrode. SRP and pH were measured after 30 min. and again two hours after 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 flasks through Kimble 1 mL disposable sterile pipettes. Small 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 of phosphorus by algae was investigated as a process to minimize the effects of phosphorus precipitation due to the increase in pH of the medium during the algal assay. The normal assay procedures were followed except *Selenastrum capricornutum* inoculum sizes of 6.8 X 10^4 cells/mL and 1.3 X 10^5 cells/mL were used along with the usual inoculum size of 10^3 cells/mL. Three or four replicates were set up for each inoculum size. In one study, the algae from a stationary phase culture was harvested by centrifugation and put into a phosphorus free nutrient medium for two days of phosphorus starvation before it was used to inoculate the bioassay.

Data Reduction and Statistics

Sampling data were compiled on computer spread sheets (Microsoft™ Excel) using the Apple Macintosh computer. The Statview computer program (Feldman and Gagnon, 1986) for the Apple Macintosh computer was used to calculate confidence intervals for BAP estimates, analyses of variance (ANOVA) and Least Significant Differences for sterilization treatment affects on BAP, and t-tests for determining the significance of diffences between various data sets. The Cricket Graph program (Rafferty and Norling, 1986) was used to calculate regression equations and r-values. A correlation was determined to be statistically significant if at the 95 percent 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 percent of the data variance was explained by the regression line (Kleinbaum and Kupper, 1978).

Results and Discussion

Sampling Results

Measurements of flow, temperature, conductivity, TP, SRP, and BAP are tabulated in Appendix A for water samples collected between April 19, 1987 and May 21, 1988. Portions of these data are used, as needed, to support the discussion which follows.

BAP Methods Development

Selection of a Sterilization Procedure

The algal bioassay requires the sample to be sterilized to eliminate indigenous algae and protozoa.

Filtration is a common way to sterilize a sample in preparation for an algal assay. Filter sterilization may lead to an underestimate of BAP by eliminating phosphorus associated with particulate matter. Several studies have estimated that 20 percent 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 percent error was unacceptable in the current study. O'Kelly (1973) stated that measurements of phosphate available for growth based on soluble (filterable) phosphorus 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. Whole water samples would provide a better estimation of bioavailable phosphorus for the Bear River system.

Autoclaving is a standard method for sterilization of whole 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) where hardness is likely to be be less than 60 mg/L as $CaCO_3$ (Geraghty, et al., 1973). In a 1984 experiment on Bear River water, separate aliquots of the sample were spiked with nitrogen (N) and phosphorus (P) prior to autoclaving. After autoclaving, these amended samples produced very little algal growth (*in-vivo* fluorescence), whereas samples spiked after autoclaving supported significantly higher maximum algal populations. 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.

Gamma radiation sterilization allows the use of a whole water sample without causing precipitation of a large fraction of the phosphate like autoclaving does. In contrast to UV irradiation, gamma radiation can consistently kill all of the organisms in the water. Gamma radiation was chosen as the sterilization method for the current investigations. However, some chemical changes due to gamma irradiation were observed.

Total phosphorus concentration did not change with gamma radiation, but Figure 2 shows that 83 percent of the samples showed an increase in SRP after gamma radiation. Ten percent of the samples increased by ≥ 276 percent of their preradiation SRP concentration, 50 percent of the samples increased by ≥ 22 percent of their preradiation SRP concentration, and 80 percent of the samples increased by ≥ 8 percent of their preradiation SRP concentration. Seventeen percent of the samples decreased in SRP concentration. Decreases in SRP concentration were relatively small with only 6 percent of the samples decreasing by more than 10 percent of their preradiation SRP concentration (Figure 3).

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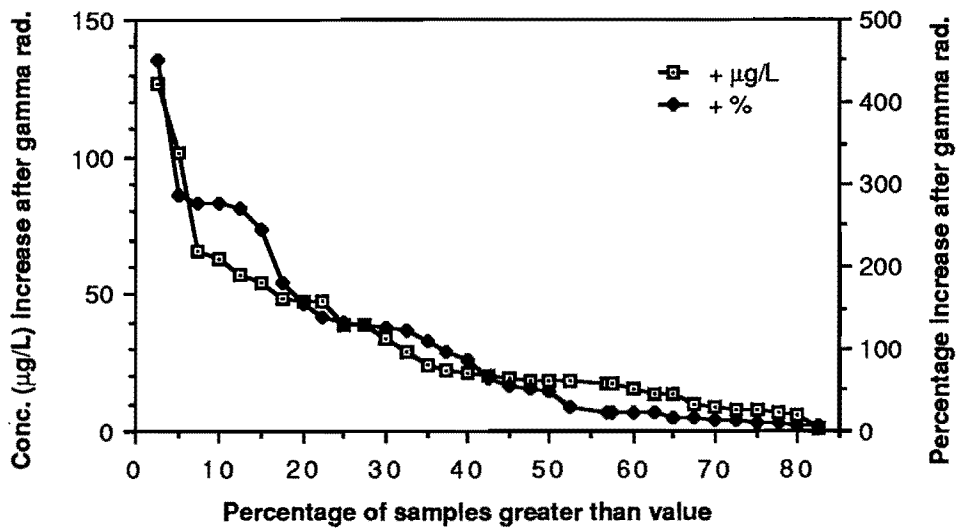
Figure 2. Soluble reactive phosphorus concentration increase with its accompanying percentage increase due to gamma radiation.
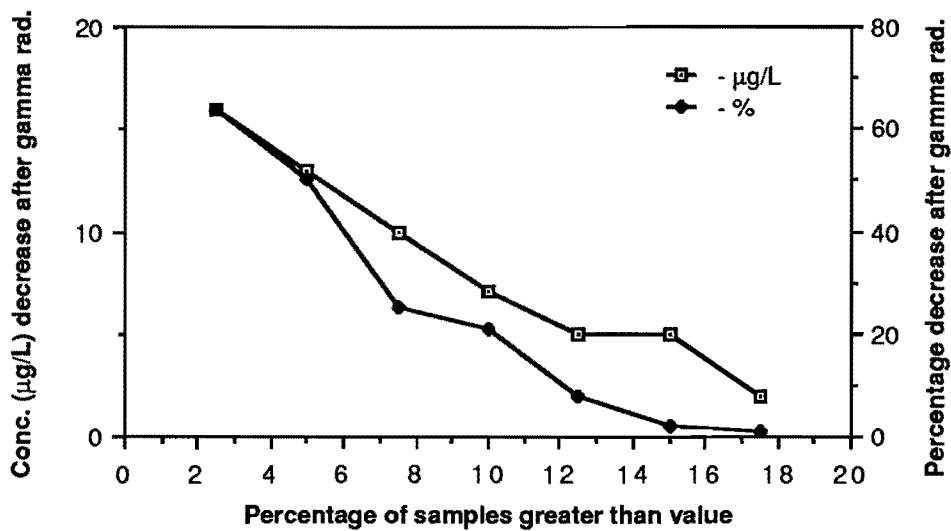
Figure 3. Soluble reactive phosphorus concentration decrease and its accompanying percentage decrease due to gamma radiation.

Table 6 shows the effects of gamma radiation on samples taken on March 22, 1988. SRP was significantly ($p \leq 0.05$) increased after gamma radiation for the Benson Marina site while the SRP's for the site below Cutler Reservoir were not significantly different ($p \leq 0.05$). For the Benson Marina sample, using a low inoculum size, the gamma irradiated BAP estimate was significantly higher ($p \leq 0.05$) than that for gamma plus filtration and filtration alone, but BAP in the filtration alone treatment was not significantly ($p \leq 0.05$) different from the gamma plus filtration treatment. The BAP estimate using the high inoculum size for the Benson Marina gamma radiated sample was significantly higher ($p \leq 0.05$) than for the gamma plus filtration and filtration alone treatments. With the high inoculum, BAP estimated for filtration alone was significantly higher ($p \leq 0.05$) than filtration followed by gamma. For the site below Cutler Reservoir the BAP estimates for the low inoculum were all beneath the detection level (i. e., < 8 μg BAP/L). For the high inoculum, the gamma radiated sample BAP estimate was significantly higher than gamma plus filtration ($p \leq 0.05$) and filtration alone ($p \leq 0.05$). BAP estimated for filtration alone was significantly lower ($p \leq 0.05$) than that for gamma plus filtration. In general, gamma radiation alone resulted 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 result in an overestimate of BAP if the gamma radiation causes more particulate phosphorus to become available than would naturally occur.

Table 6. 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	Low	a	150.5		132.7
Marina	Low	b	150.5	173	88.7
	Low	c	150.5		95.7
	High	a	150.5		246.7
	High	b	150.5	173	148.0
	High	c	150.5		176.0
Bear River					
below	High	a	72		49.3
Cutler	High	b	72	68.5	42.3
Reservoir	High	c	72		26.7

*a = gamma radiated only
b = filtration followed by gamma radiated
c = filtration only.

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Bioavailable Phosphorus Estimation
by Standard Addition

In quantitative chemical analysis, standard addition is a common method used to correct for interferences from the sample matrix. This method produces reliable results unless some internal process prevents it from causing a precise and predictable analytical response. Initially, for "reliability" the standard addition was used in performing algal bioassays for BAP. In algal bioassays precipitation of nutrients, lack of nutrients, toxicity or some other negative factors in the sample could inhibit growth while growth might be stimulated by vitamins, amino acids, light, or other positive factors in a way that is not linearly related to added phosphorus. For example, Figure 4 shows that the unamended Bear River sample collected on July 22, 1987 from below Cutler Reservoir produced a maximum fluorescence of 7.0, indicating measurable algal growth; however, the BAP estimate, based on a 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 maximum fluorescence of 4.4, but the resulting BAP estimate was higher; 14 μg/L. This discrepancy in results might be explained by an increase in pH (due to algal photosynthetic consumption of CO_2) in the assay sample which would cause the precipitation of phosphorus with calcium resulting in a non-linear availability of added phosphorus and misleading BAP estimates. 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 5 for the higher pH. Figure 5 shows that addition of PO_4-P to Bear River water at pH 9.6 may not result in a linear chemical availability as measured by the SRP procedure. A linear relationship between added PO_4-P and SRP is shown in Figure 5 for the natural pH of 8.7.

The non-linear response to phosphorus addition due to an increase in pH was investigated by raising the pH incrementally with buffer additions and measuring the decrease in SRP after 30 minutes. This period was provided to allow the system to become evenly mixed and allow precipitation to begin (Figure 6). After 120 minutes, SRP was measured again. The change in SRP 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 occurs rapidly in the first ten minutes and then continues slowly for the next four hours. Desorption occurs much slower, requiring six to seven hours in soils (Griffin and Jurinak, 1974). REDEQL-EPAK (Ingle et al., 1980), an aqueous chemical equilibrium computer model, was run for the same pHs used in the above experiment and approximately the same sample composition (50 μg PO_4-P/L). The model calculations showed that as pH was increased from 8.70 to 9.48 the orthophosphorus equilibrium concentration went from 0.35 μg/L to 0.15 μg/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, orthophosphorus and hence BAP concentration would decrease for Bear River water. Equilibrium concentrations (0.35 to 0.15 μg/L) calculated by REDEQL are approximately 100 times less than the measured concentrations (Figure 6), showing that the river composition is not at equilibrium. This suggests that phosphorus inputs replenish that lost to precipitation with time in the river.
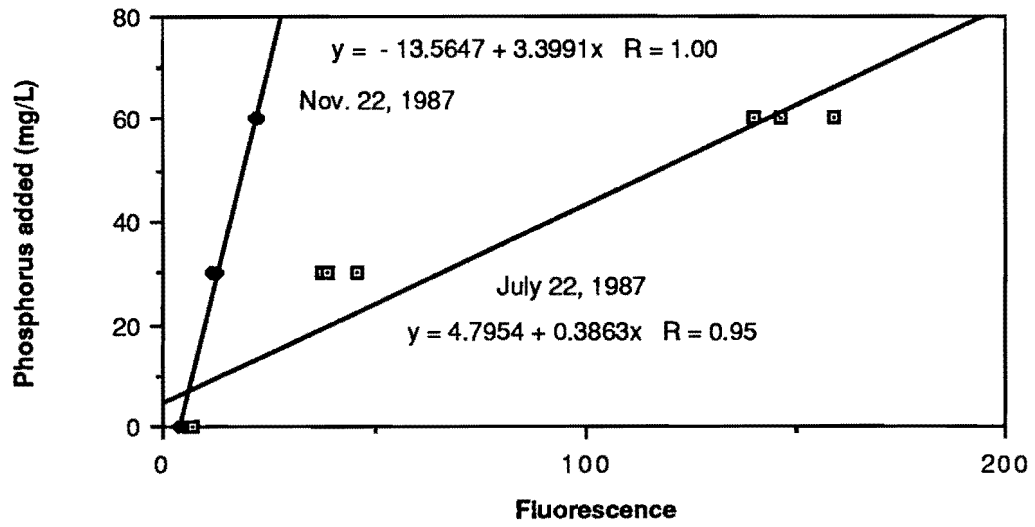
Figure 4. Variable results of standard addition at the same sample site for two different sample dates.
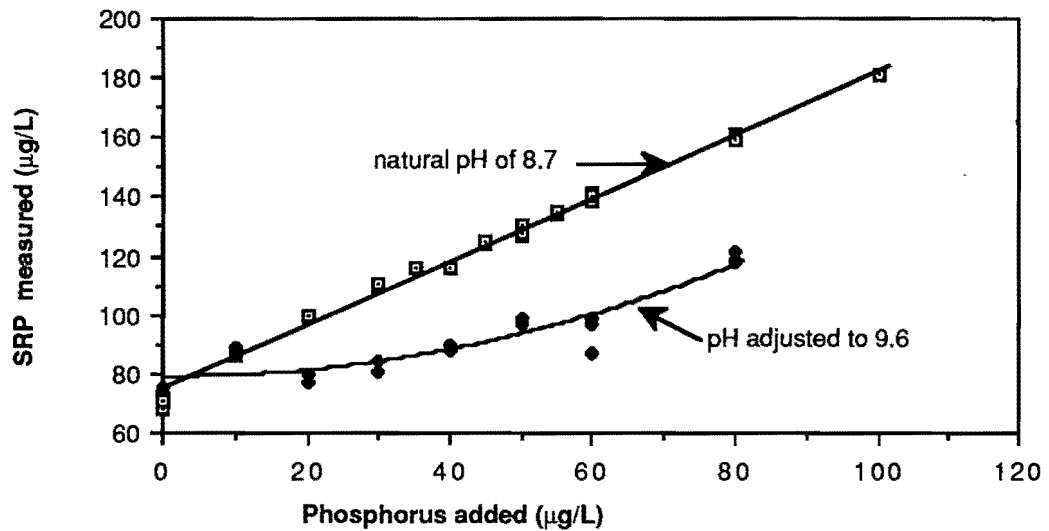
Figure 5. Effects of pH upon standard additions linearity.
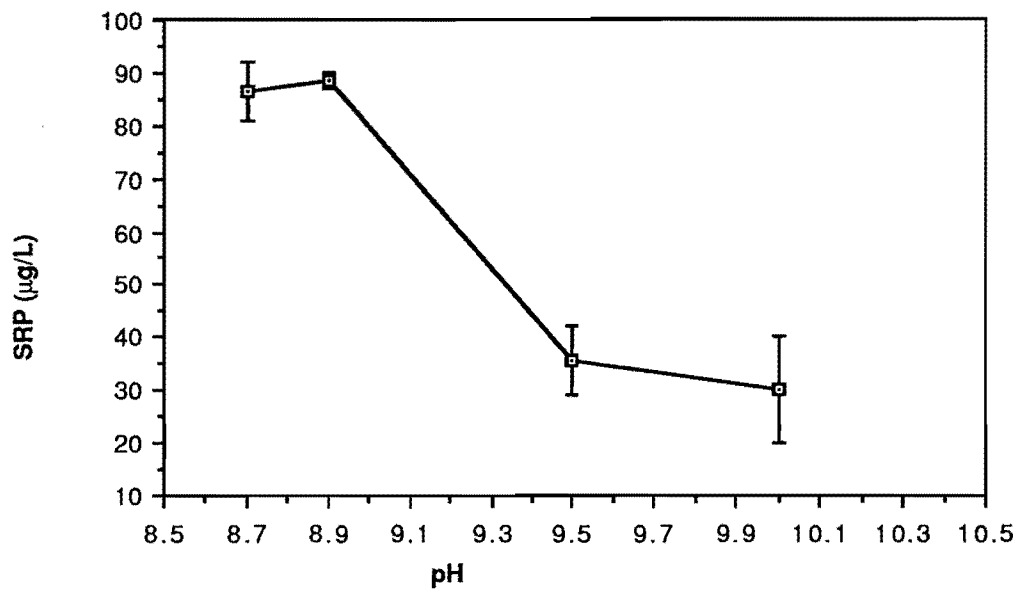
Figure 6. The change of soluble reactive phosphorus with an increase in pH (95% confidence interval error bars drawn).

pH Control Effects

The effects of a pH increase on BAP estimation 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 7.

The Benson Marina sample shows a significantly higher ($p \leq 0.05$) BAP with pH kept between 6.7 and 7.2 than with the pH as high as 9.7. The estimated BAP concentrations for the sample from 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 of around 8.9. No BAP could be measured in the Blacksmith Fork sample at either high or neutral pH. An increase of pH for the Benson Marina and below Cutler Reservoir samples resulted in a decrease in apparent BAP. This generally agrees with the REDEQL program results which predict a high (> 99 percent) precipitation of PO_4-P with calcium in Bear River water. Scherfig et al. (1973) found that CO_2 addition to maintain pH between 7 and 8 resulted in 45 to 74 percent increases of algal growth (g of cells/L). The Benson Marina sample showed a 97 percent increase in estimated BAP under these conditions.

Luxury Uptake of Phosphorus

Precipitation of phosphorus due to an increase of pH during an algal assay 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). Keenan and Auer (1974) made the following findings: (1) *Selenastrum capricornutum* exhibits luxury uptake of phosphorus, (2) luxury uptake could result in a larger orthophosphorus concentration in the assay 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, (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, and (4) the detrimental effects of luxury uptake by introducing additional phosphorus into the sample can be minimized by a starvation period.

Table 7. The effects on BAP estimation while controlling the algal culture pH near 7.0.

Site	pH range Without CO_2	BAP Without CO_2 (μg/L)	pH range CO_2 with Air	BAP CO_2 with Air (μg/L)
Benson Marina	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	< 8

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 use luxury uptake, store the available phosphorus within the cell, and pass it to daughter cells as growth occurred. 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 would be the effect of phosphorus precipitation on BAP estimation.

To test the affects of phosphorus starvation on luxury uptake and BAP results, 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. Both of these samples used an inoculum of 10^3 cells/mL in an AAP medium. No significant ($p \leq 0.05$) difference in fluorescence was detected between the two cultures (Figure 7). More extensive data are needed before any conclusions about starvation affects on luxury uptake of phosphorus can be drawn.

A low inoculum (1 X 10^3 cells/mL) and a high inoculum (1 X 10^5 cells/mL) were used in two split samples so that the effects of luxury uptake could be investigated. Data in Figures 8 and 9 are from the high (1 X 10^5 cells/mL) and low inoculum (1 X 10^3 cells/mL) sizes for Bear River samples, taken February 22 and March 22, 1988, respectively. Statistical comparison of the slopes and y-intercepts for both the low and high inocula on both dates showed that the higher inoculum size produced significantly ($p \leq 0.05$) more growth (fluorescence) in response to PO_4-P addition than did the low inoculum. The effect of using the higher inoculum is more clearly illustrated in Figure 10. These data show that the fluorescence of a culture, without added phosphorus, inoculated with 1.3 X 10^5 cells/mL exceeded the fluorescence of both a 6.8 X 10^4 and a 1.0 X 10^3 cells/mL inoculum after 1 day of incubation, and after two days cultures with both the 1.3 X 10^5 and the 6.8 X 10^4 cells/mL inocula exceeded the fluorescence (growth) of the culture with the 1.0 X 10^3 cells/mL inoculum. The effect, on BAP estimation of using a high inoculum in the March 22, 1988 samples is shown in Table 6. These results suggest that more phosphorus was available for growth to the algae in the high inoculum cultures and that a high inoculum could be used to minimize the effects of phosphorus precipitation due to a pH increase in the assay culture.

External Standard Growth Response Curve

The data show that use of an internal standard curve (standard additions) resulted in a nonlinear response to phosphorus as pH was increased in some samples. An external standard growth response curve was produced to provide BAP estimates that were consistent with observed algal growth. The external standard curve for *S. capricornutum* growth in response to P was produced by growing *S. capricornutum* in algal assay medium (Miller et al., 1978) with varying concentrations of PO_4-P (Figure 11).

There were 72 μg/L (1.8 μmolar) of calcium in the algal assay medium. This calcium could have precipitated 6 and 23 percent of the added 200 and 50 μg/L of PO_4-P (respectively) if solid $CaHPO_4$ ($pK_{so} = 6.66$) formed as an intermediate to hydroxyapatite (Snoeyink and Jenkins, 1980).

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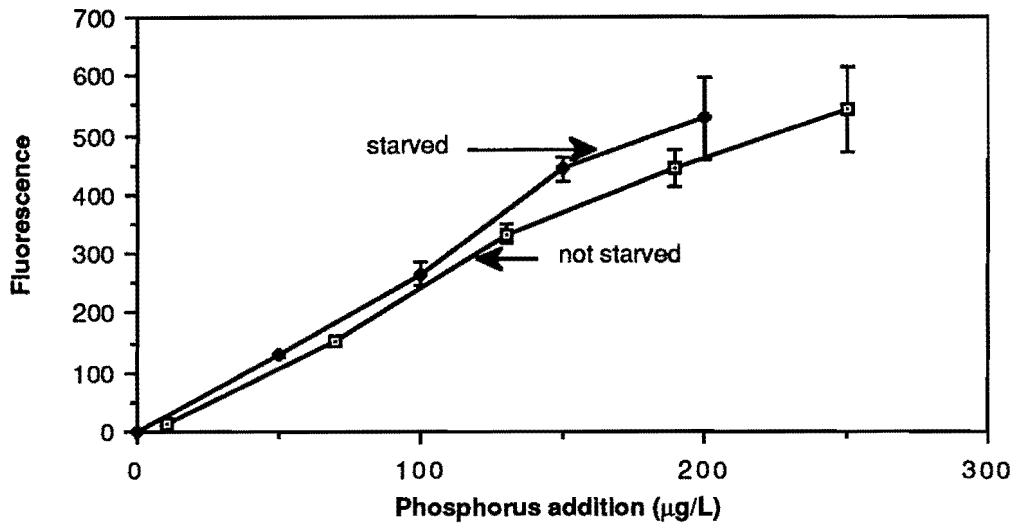
Figure 7. Effect of algal starvation on maximum fluorescence with phosphorus addition (95% confidence interval error bars drawn).
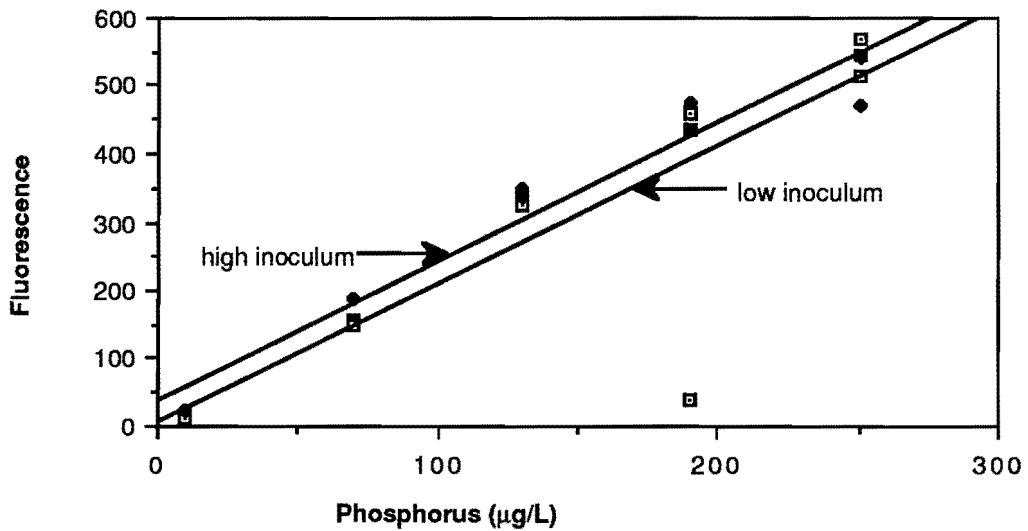
Figure 8. Maximum fluorescence response of a high and low inoculum size of *S. capricornutum* to phosphorus additions to a February 22, 1988 sample.
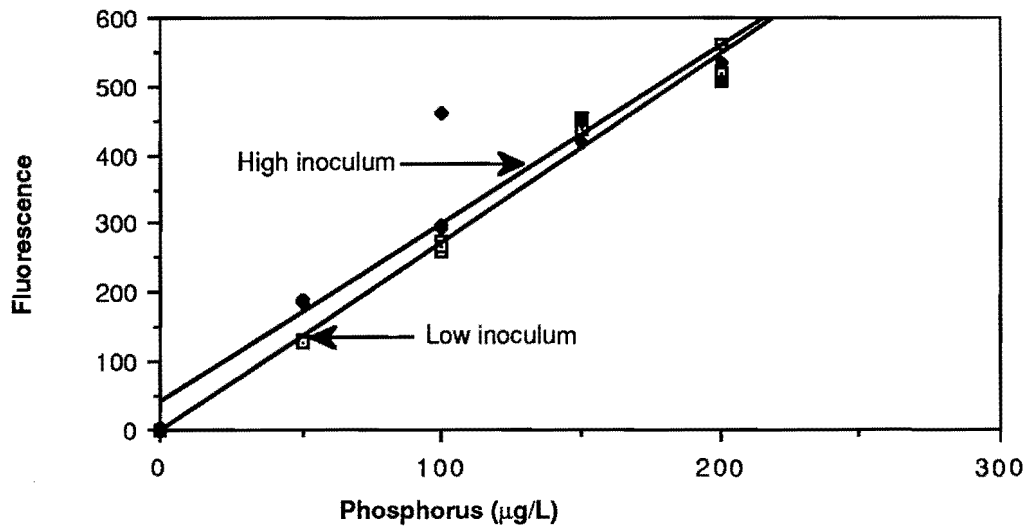
Figure 9. Maximum fluorescence response of a high and low inoculum size of *S. capricornutum* to phosphorus additions to a March 22, 1988 sample.
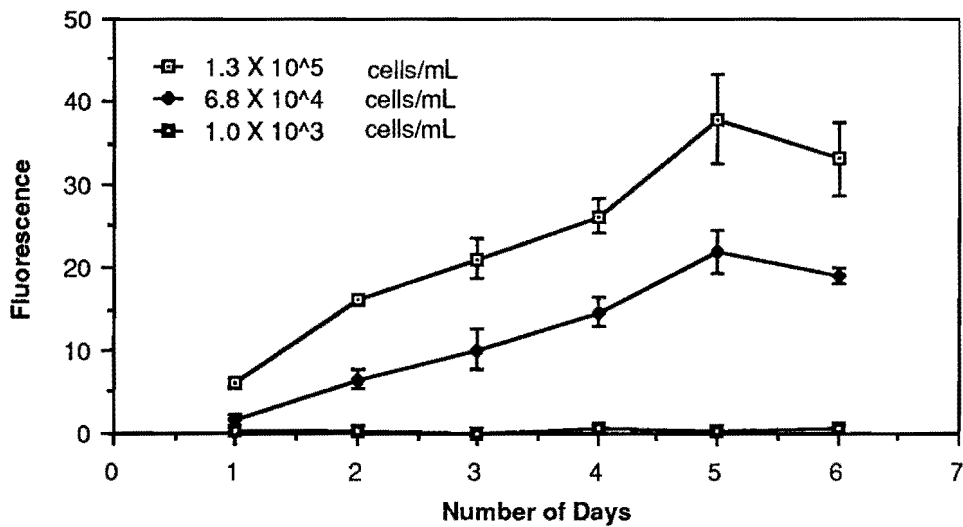
Figure 10. Fluorescence over time for three inoculum sizes in a February 22, 1988 Bear River sample without added phosphorus. (95% confidence interval error bars drawn).
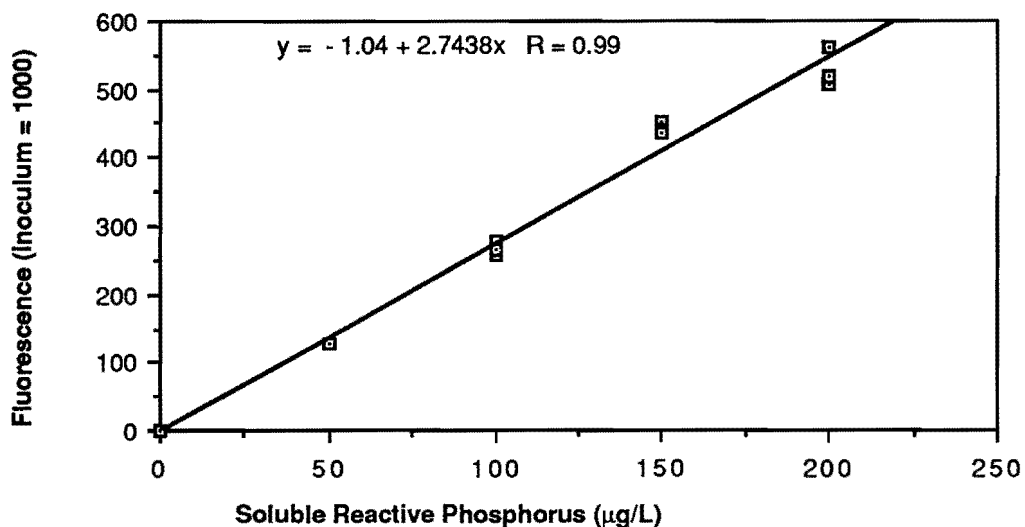
Figure 11. Linearity of external standard curve conducted on March 22, 1988.

Lower fractions of the added phosphorus could actually precipitate from any phosphorus addition since a portion of the calcium would be used by the algae and would be unavailable for precipitation. The linearity of the standard curve seen in Figure 11 indicates that PO4-P precipitation was not a factor in algal response to PO_4-P addition because the line does not curve to suggest an increasing fraction of the phosphorus precipitated at lower P concentrations.

The minimal detectable change in fluorescence when using the external standard curve was determined to be 8 μg BAP/L. This value was equal to the 95 percent prediction confidence interval of the 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 regression equation for calculating BAP from the fluorescence of the sample culture without phosphorus addition was:

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

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 the ability of samples to support algal growth suggest 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. Soluble copper inhibits S.

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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 percent of total copper for water ranging between pHs 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

The standard Algal Assay Bottle Test (Miller, et al., 1978) was primarily developed to determine the limiting nutrient and the presence of any toxicants for algae. This procedure has been modified to allow BAP estimation. Miller et al.(1978) suggest 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 effects 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/or pH control by enrichment 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. Using pH control and luxury uptake would probably produce the maximum BAP estimate for the Bear River system. This method may not be the most accurate, but it would give the largest BAP estimate and a margin of safety in making management decisions. A more accurate but more difficult procedure might include maintaining the sample pH at the natural river pH.

Although pH control with CO_2 and/or use of luxury phosphorus uptake may eliminate many or all of the problems with standard additions of phosphorus in the determination of BAP, these procedures were developed too late in the project to be extensively used and the standard additions procedure was abandoned in favor of the external standard procedure. All BAP estimates reported here were made using the external standard (Figure 11) procedure with an inoculum of 10^3 cells/mL and no pH control.

Evaluation of Bioavailable Phosphorus Indicators

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 prediction of BAP.

Initially, total and soluble reactive phosphorus were investigated. Table 8 and Figures 12, 13, 14, 15, 16, and 17 present the phosphorus data obtained for the Bear River system. Figure 12 is a plot of TP plotted versus BAP for river water; the data have 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 percent 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 13 and 14 do not show a statistically significant ($p > 0.05$) correlation of SRP

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Table 8. Soluble reactive, total and bioavailable phosphorus data for the Bear River system

River System	Sample Location	Date	SRP µg/L	TP µg/L	BAP µg/L
Bear River	Weston Creek	5/27/87	52.	224.	<8.
	W. of Fairview, USGS gauge	5/11/87	0.	388.	<8.
	W. of Fairview, USGS gauge	7/12/87	20.	100.	<8.
	W. of Fairview, USGS gauge	8/18/87	19.	67.	<8.
	Abv Cutler Res., W. Benson	5/11/87	0.	138.	<8.
	Abv Cutler Res., W. Benson	5/27/87	37.	224.	<8.
	Abv Cutler Res., W. Benson	7/12/87	18.	130.	<8.
	Blw Cutler Res., UPL gauge	5/11/87	14.	125.	13.
	Blw Cutler Res., UPL gauge	5/27/87	34.	208.	<8.
	Blw Cutler Res., UPL gauge	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 River	South of Richmond	8/18/87	17.	128.	18.
	Worm Creek*	6/21/87	129.	274.	156.
Little Bear River	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, at 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.
Blacksmith Fork	Blw Anderson Ranch, gauge	5/11/87	7.	17.	<8.
Logan River	Blw Logan lagoon outfall	7/12/87	14.	490.	302.
Point Sources	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.

*Receiving stream for Preston City WWTP effluent.
†Receiving stream for Hyrum City WWTP effluent and E. A. Miller Inc. treatment plant effluent.
§Below the confluence with the Logan River and Spring Creek; above the Bear River confluence.
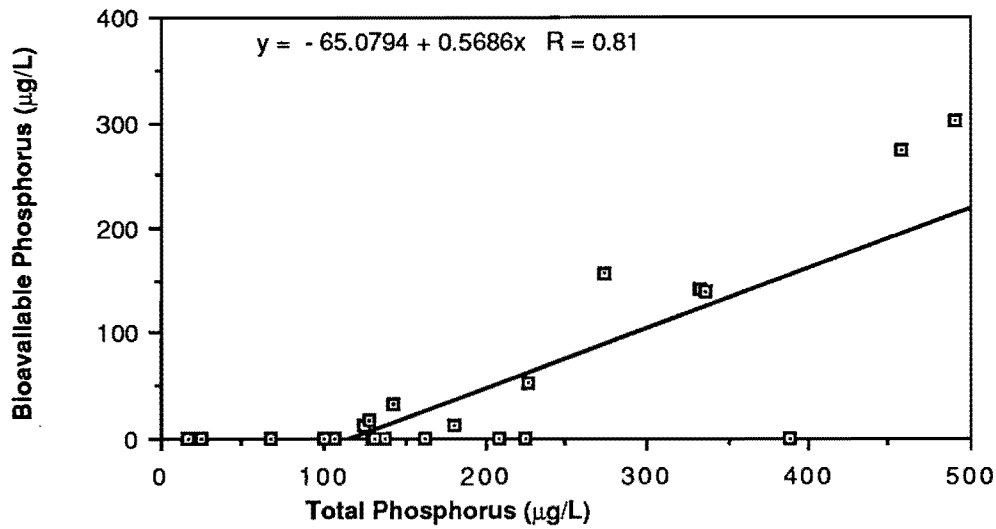
Figure 12. Total phosphorus data plotted against bioavailable phosphorus data in river water (Table 8).
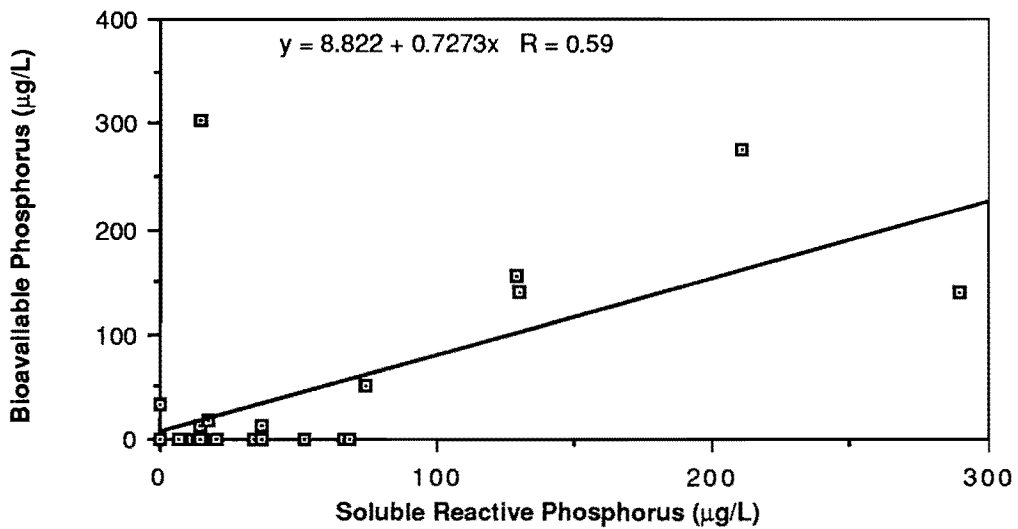
Figure 13. Soluble reactive phosphorus data plotted against bioavailable phosphorus data in river water (Table 8).
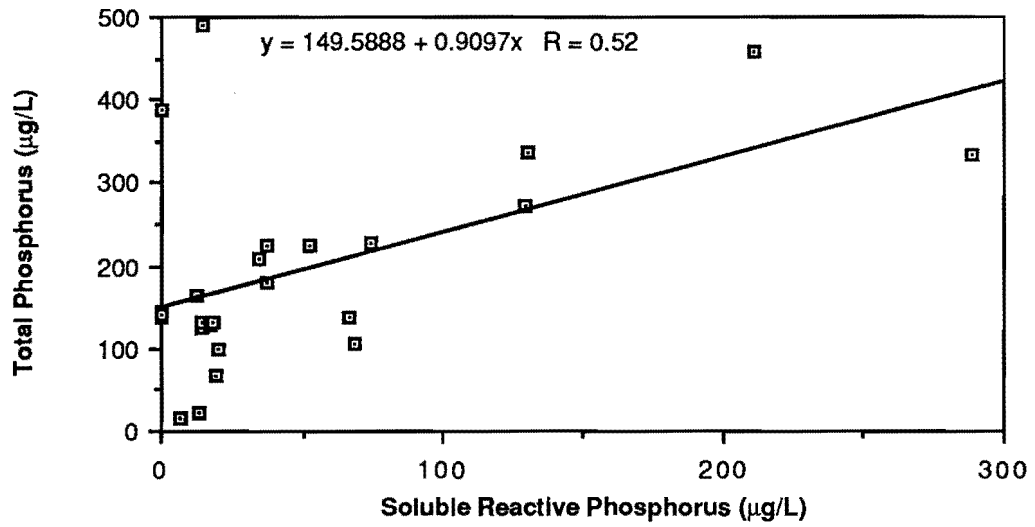
Figure 14. Total phosphorus data plotted against soluble reactive phosphorus data in river water (Table 8).
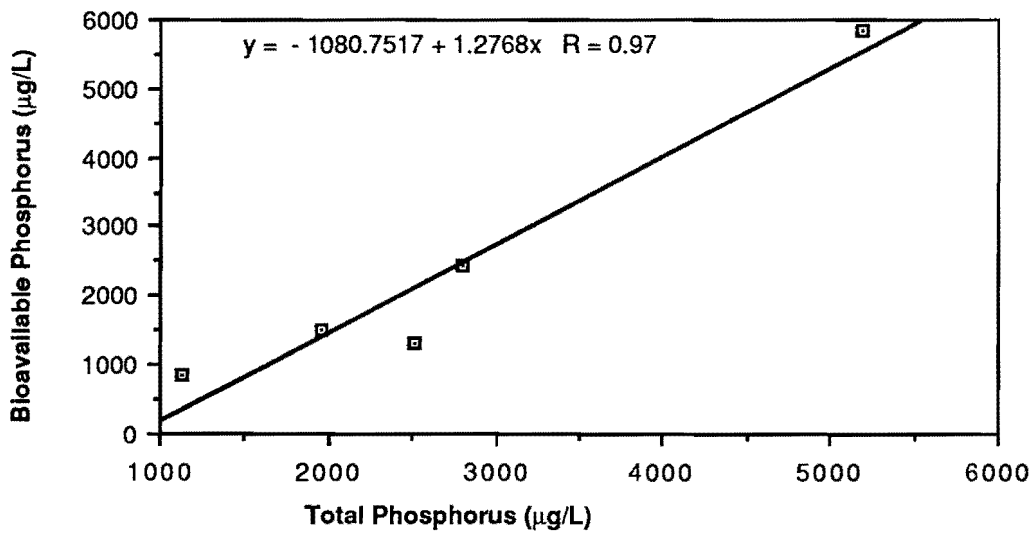
Figure 15. Total phosphorus data plotted against bioavailable phosphorus data for wastewater sources (Table 8).

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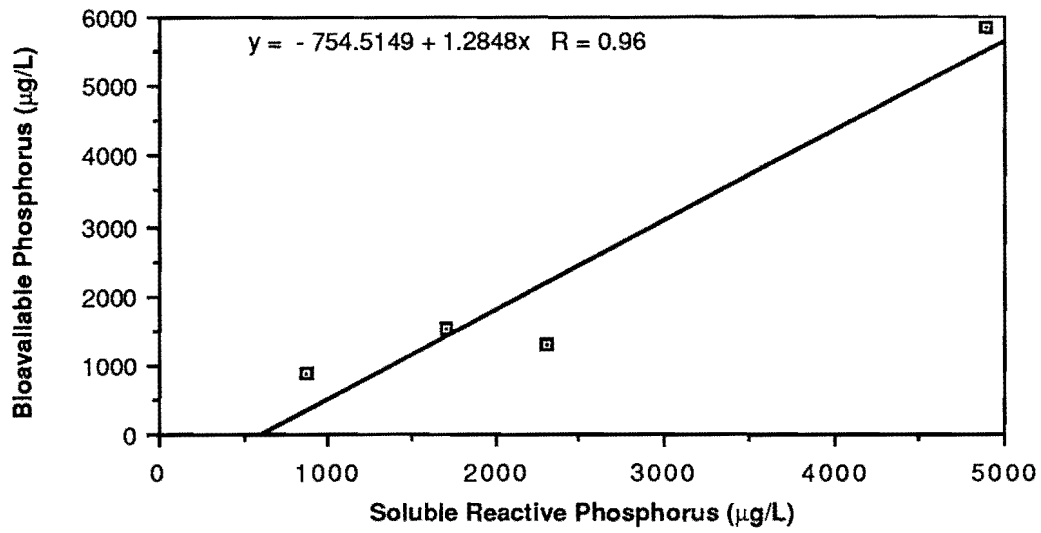
Figure 16. Soluble reactive phosphorus data plotted against bioavailable phosphorus data for wastewater sources (Table 8).
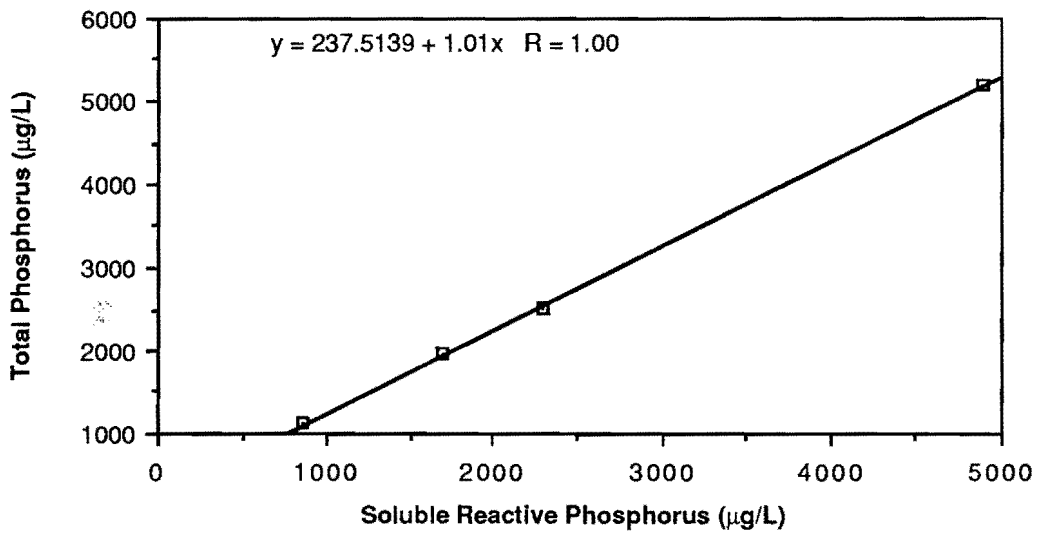
Figure 17. Soluble reactive phosphorus data plotted against total phosphorus data for wastewater sources (Table 8).

with BAP (r = 0.59) and SRP with TP (r = 0.52) respectively for river water. There does not appear to be a reliable correlation between BAP and SRP or between BAP and TP for the Bear River system. This probably results from the control of phosphorus solubility by relatively insoluble calcium-phosphate minerals which form in the hard, alkaline waters. The formation of these minerals during the algal assay for determination of BAP may also interfere with accurate measurements of BAP. In some areas of the U.S., total phosphorus is a good estimator of BAP but for the Bear River this appears not to be the case.

Sample data from wastewater effluents show that TP plotted against BAP (Figure 15) and SRP plotted against BAP (Figure 16) 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 17 shows a statistically significant (p ≤ 0.05) correlation (r ≈ 1.00) between SRP and TP for wastewater samples.

Comparison of the fractions of BAP, SRP, and TP between river water and wastewater effluent samples is shown in Table 8. BAP was approximately 88 percent of TP and 98 percent of SRP for wastewater samples. For river samples with BAP greater than 8, BAP was approximately 41 percent of TP and 120 percent of SRP. The WWTP's effluent phosphorus is highly available because the phosphorus found in complex materials in the sewage is converted by the treatment process into simple, available forms. Also, a low particle load in the WWTP's effluent, relative to river water in the Bear River basin, could explain the higher ratio of TP to SRP in Bear River water.

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 appeared reasonable because land and livestock runoff can contain large concentrations of organic matter and organic phosphorus may be readily converted to BAP by microbial processes.

Table 9 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 18. The regression analysis of these data suggests that there is a statistically significant (p ≤ 0.05) relationship (r = 0.82) between these measurements. However, all but one of the data points with BAP > 8 µg/L are from stream or reservoir sample sites that have wastewater discharged 1 to 7 miles upstream of them. This suggests that the relationship may not be applicable to river waters in general. It is noteworthy that the correlation between TOC and BAP in wastewater effluents is not statistically significant (p ≤ 0.05; r = 0.05; Figure 19). 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. Since there does not appear to be a reliable correlation and the precision of the TOC analysis (± 1 mg C/L) may not be adequate to allow resolution of changes in TOC relative to BAP changes in river water, the TOC and BAP relationships were not further investigated.

Particle Size Range

The correlation between BAP and particle size was also investigated. Several researchers have proposed that BAP may be approximated by adding 20 percent of particulate phosphorus to orthophosphorus (Dorich et al., 1980; Rast and Lee, 1982; U.S. Environmental Protection Agency, 1980a). The high surface area of clays, fine silt, and

Table 9. Total organic carbon and bioavailable phosphorus data for the Bear River system.

River	Sample Location	Date	TOC (mg/L)	BAP (µg/L)
Weston Creek	Near Bear R. confluence	5/27/87	5.	<8.
Bear River	West of Fairview, UT-ID gauge	7/12/87	6.	<8.
Worm Creek	West of Franklin, ID	6/21/87	8.	156.
	Richmond Lagoon effluent	6/21/87	27.	2450.
Bear River	Abv Cutler Res., W. of Benson	5/27/87	5.	<8.
Bear River	Abv Cutler Res., W. of Benson	7/12/87	7.	<8.
Bear River	Blw Cutler Res., UPL gauge	5/27/87	7.	<8.
Bear River	Blw Cutler Res., UPL gauge	7/12/87	7.	<8.
Little Bear R.	S. Fork Blw Davenport Creek	5/27/87	9.	12.
	Whites Trout Farm effluent	6/21/87	4.	<8.
	Hyrum WWTP effluent	6/21/87	4.	5830.
Little Bear R.	Blw Hyrum Res., Hwy 101	7/12/87	5.	<8.
	Wellsville Lagoon effluent	6/21/87	10.	860.
Little Bear R.	Abv. Logan R. confluence, 6th S.	7/12/87	5.	<8.
Little Bear R.	Benson Marina	5/27/87	6.	52.
Little Bear R.	Benson Marina	7/12/87	10.	274.
	Logan Lagoon effluent	6/21/87	10.	1518.
Logan River	Blw Logan Lagoon outfall	7/12/87	11.	302.
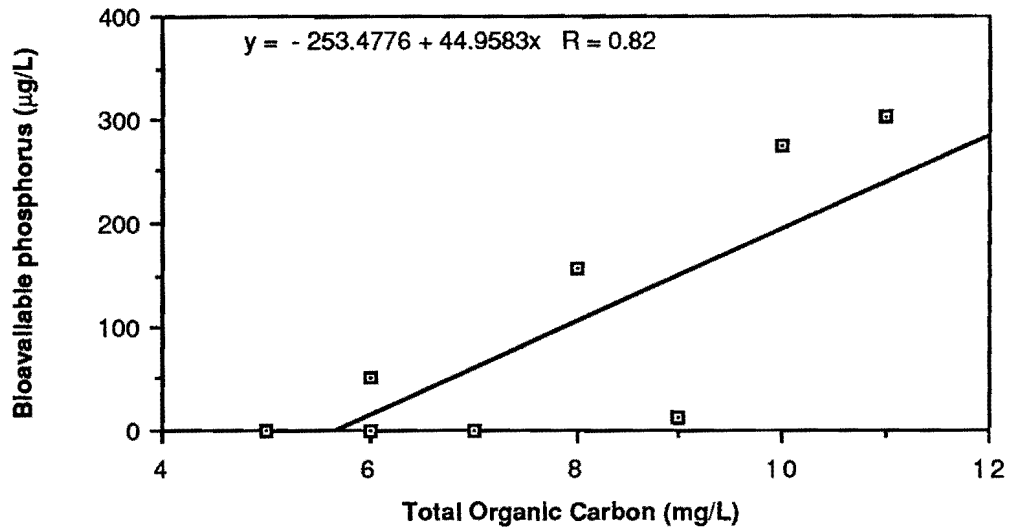
Figure 18. Total organic carbon data plotted against bioavailable phosphorus data in river water (Table 9).
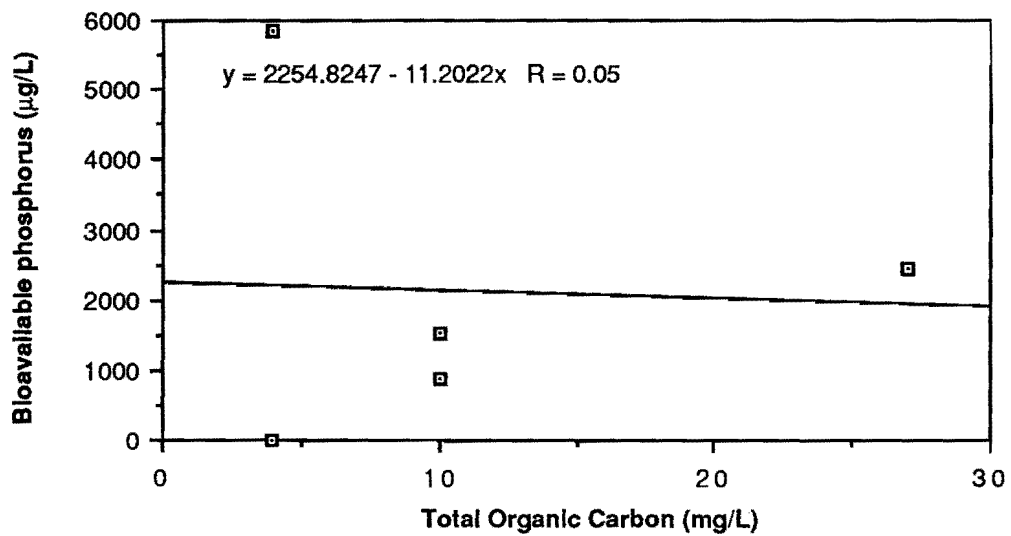
Figure 19. Total organic carbon data plotted against bioavailable phosphorus data for wastewater point sources (Table 9).

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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 regressed with BAP estimates in corresponding samples 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 10). The data in Figure 20 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 20), the suspended solids concentrations in all of the size fractions were quite variable. There also appears to be no correlation between total phosphorus or SRP and suspended solids as shown by the data in Figures 21 and 22.

For 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.

The Interaction of Water Hardness, pH and BAP

Algal bioassays for the determination of BAP can take up to 14 days to complete. During this time a portion of the initially available phosphorus can become unavailable due to precipitation, adsorption, or settling. During bioassays, the sample pH was observed to increase by as much as two units (e. g., from 8 to 10). This increase in pH can cause phosphorus to precipitate from solution with hardness metal ions (e. g., Ca^{2+}, Mg^{2+}) resulting in a low BAP estimate.

In the bioassay, the pH can increase from natural conditions because of the algal photosynthetic consumption of carbon dioxide. REDEQL-EPAK (Ingle et al., 1980) was used to determine the effects that a pH increase would have on phosphorus. Bear River data was used in the model as the pH was increased from 8 to 10. The model calculated that 99.9 percent 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 actually 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, thus 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 naturally low.

Table 10. Ranges of suspended solids and bioavailable phosphorus data for the Bear River system.

Stream	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 River	West of Fairview, UT-ID gauge	7/12/87	31.8	14.0	19.3	<8.	100.	20.
Bear River	West of Fairview, UT-ID gauge	8/18/87	24.7	8.7	13.0	<8.	67.	19.
Bear River	Abv Cutler Res., W. of Benson	7/12/87	63.2	29.0	38.6	<8.	130.	18.
Bear River	Blw Cutler Res., UPL gauge	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 River	South of Richmond	8/18/87	26.6	7.7	12.9	18.	128.	17.
Little Bear R.	Blw Hyrum Res., Hwy 101	7/12/87	2.4	0.9	1.9	<8.	106.	68.
Little Bear R.	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.
Little Bear R.	Benson Marina	7/12/87	51.2	4.0	39.0	274.	458.	211.
Little Bear R.	Benson Marina	8/18/87	38.4	5.0	35.0	139.	336.	130.
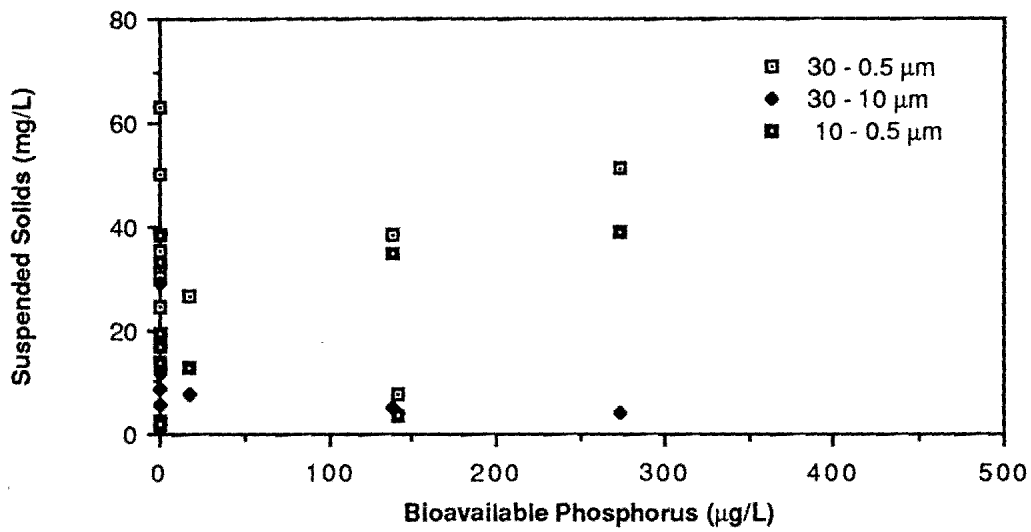
Figure 20. Suspended solid concentration for three particle size ranges plotted against bioavailable phosphorus data in river water (Table 10).
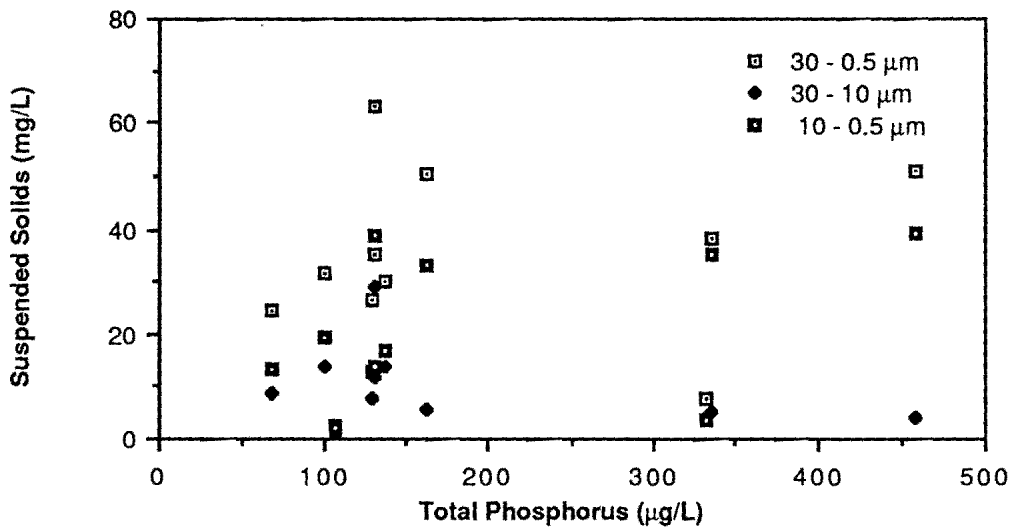
Figure 21. Suspended solid concentration for three particle size ranges plotted against total phosphorus data in river water (Table 10).
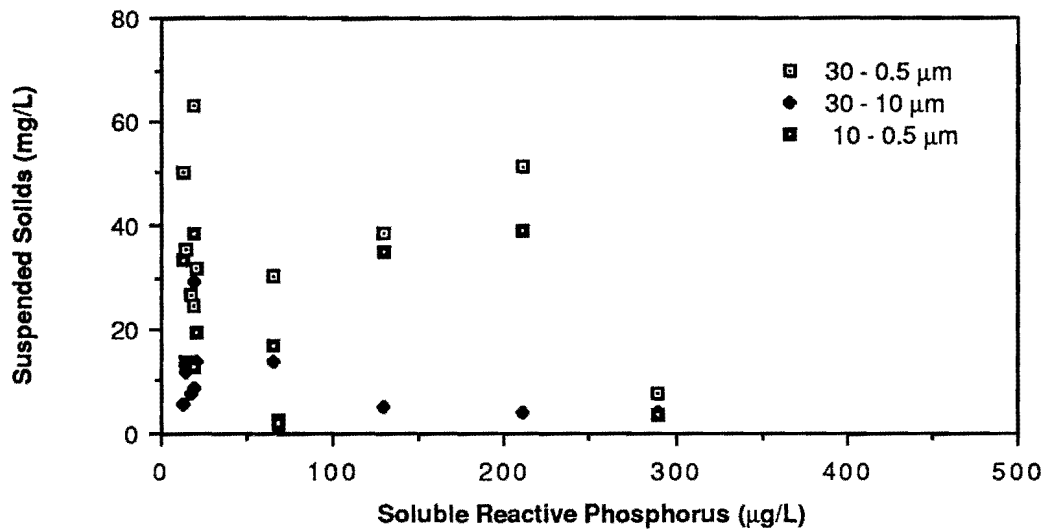
Figure 22. Suspended solid concentration for three particle size ranges plotted against soluble reactive phosphorus data in river water (Table 10).

In using the REDEQL model, both pH and calcium were important factors in the precipitation of phosphorus and its resulting unavailability. To determine if there was a relationship between BAP, calcium, and pH in river waters, a multiple regression analysis (using stepwise procedures {Feldman and Gagnon, 1986}) 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 as the Bear River, and the data had a relatively broad range of pH, BAP, and total hardness. More data were available from the Snake River study than have 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 from their data by using the algal yields given by Greene et al. (1975) in the following ratio:

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

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 14 sample results where either nitrogen or phosphorus limited algal growth. Total phosphorus (TP), SRP, total hardness (TH), hydrogen ion concentration (H^+) and iron concentration (Fe), all of which might affect BAP concentrations, were also given for these samples.

The variables and their interactions used in the regression analysis 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: the product (interaction) of SRP, total hardness and hydrogen ion concentration (SRP*TH*H^+); and total hardness.

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

The r^2 value of the equation with both of these variables was 0.941.

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

A multiple regression analysis was then performed using the same parameters for the Bear River data as were used for 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 ($r^2 = 0.879$).

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

This analysis indicates 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 and 180, with a few areas getting as high as 240 mg $CaCO_3$ hardness/L (Geraghty, et al., 1973). The difference in surface water hardness across the U.S. could be one reason why, in some areas of the country, TP correlates closely with BAP while in other parts there is no correlation. In hard water areas, the TP would be expected to be 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 have been conducted to estimate BAP.

Sources of Bioavailable Phosphorus

Phosphorus contributed by point sources

Cache Valley has five WWTP's 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 Inc. are the only significant industrial waste dischargers to surface water in the Cache Valley.

Table 11 shows that, based on the average BAP concentration and average flow, point sources in Cache Valley contribute approximately 30,000 kg of BAP/year to the Bear River system. The Logan lagoons contribute 73 percent of the total while Preston and Hyrum WWTP's contribute 6 and 21 percent, respectively. This means that if the Logan lagoons effluent phosphorus were completely eliminated from entering the Bear River system, the BAP load from WWTP's would be reduced to approximately 8,200 kg/yr. If Logan lagoons and Preston and Hyrum WWTP effluents were all removed from the Bear River system, the point source BAP load would be reduced by 99.5 percent 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 percent of the PO_4-P--the form which is highly available. If a 90 percent reduction of PO_4-P from the three major plants were instituted then PO_4-P, and hence BAP from the point sources, would be reduced by 89.5 percent to 3,200 kg/yr. A 90 percent reduction in phosphorus for the Logan lagoons alone could reduce the BAP contribution from WWTP's by 65.5 percent to 10,400 kg/yr to the Bear River system. The lack of detectable BAP in the White's Trout Farm effluent is noteworthy (Table 11). This wastewater is treated through a lagoon system. It is possible that phosphorus in this effluent is mostly combined in unavailable mineral forms by the time it is discharged.

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 percent) of total phosphorus, this range would be typical for effluents (Tchobanoglous, 1979). The TP concentration from the Cache Valley WWTP's ranges from 1.1 to 5.2 mg/L (Table 8). 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 12). The calculated loading comes within a factor of two of the measured values (Table 11), a reasonable variance.

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 2,500 kg/yr of total phosphorus was contributed to the Bear River system from feedlots. This was equivalent to approximately 0.6 percent of the estimated total phosphorus (440,000 kg/y) that passed by Cutler Reservoir in 1985. A similar calculation for BAP can now be made. The average BAP concentration below Cutler Reservoir in 1987 was ≤ 12.4 µg/L (Appendix A). Assuming an annual flow of approximately 3,000,000 acre-ft/yr., the estimated BAP which passes Cutler Reservoir would be ≤ 46,000 kg/yr. If all the total phosphorus (2,500 kg/yr) contributed from feedlots were bioavailable then it would compose ≤ 5 percent of all the BAP that passes below Cutler Reservoir. Feedlot phosphorus could be highly available since 5 to 25 percent of the total phosphorus is orthophosphate and much of the remaining phosphorus fraction is decomposable organic material (Wieneke et al., 1980).

Table 11. Cache Valley point sources that discharge into the Bear River system.

				Concentrations			Average mass discharge		Percentage
River	Effluent Samples	Date	Flow cu. ft/s	TP µg/l	SRP µg/l	BAP µg/l	TP kg/yr	BAP kg/yr	of total BAP kg/yr
Bear	Preston WWTP	6/21/87	1.0	2210	1470	1560	2697	1800	6.0
		2/22/88		3830	3290	2500			
Little Bear	White's Trout Farm	6/21/87	31.0	*	*	< 8			
		2/22/88	25.0	57	23	< 8	1272	< 179	<0.6
	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.

Table 12. Phosphorus loading calculation for the three major wastewater plants in Cache Valley using the factor of 1.1 kg P/capita-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

Phosphorus Contributed By
Livestock Runoff

Wieneke et al. (1980) found that there was not a linear correlation between TP and SRP in feedlot runoff. This might be because of phosphorus 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 residence time of Cutler Reservoir. More studies need to be conducted to determine BAP transport along the river and reservoirs and the potential impact upon downstream reservoirs from livestock runoff.

Phosphorus Contributed by Land
Runoff and Stream Bank 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 snow melt 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. In February 1986 during a high runoff event, a Bear River sample taken at the Utah-Idaho state border 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 are 11 and 14 percent, 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 acreage (Cundy and Conant, 1982) of Cache County (Table 13). It was estimated that 55,000 to 65,000 kg of TP/year was contributed to the 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 water body. Sorensen et al. (1987) estimated that 440,000 kg 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 estimate the percentage of TP that was BAP in runoff. Since rain is a 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 14). The sites were: a wheat field, a recently planted barren field, and two range land sites in the Cache Valley. The sample data 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 snow pack would not affect the loss of top soil and plant litter. The rain simulation apparatus provided better experimental control but some error may have occurred due to the small number of sample plots done and their 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); thus, the simulated rainfall intensity was 10 times greater than actual rainfall. This excess rainfall intensity may have produced a lower BAP estimate because of its higher erosive power which would wash away larger particles, lower the fraction of suspended clays and organic matter, and result in a lower BAP fraction per unit mass of suspended solids than would be expected from natural runoff.

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, 0.5, 0.02, and 0.4 µg BAP/µg TP for the runoff sites at the Blacksmith Fork, Weston Creek, Cub River, and above Cutler Reservoir, respectively. The large variation may be because the Blacksmith Fork site soil has less than a detectable concentration of $CaCO_3$ and is a range land with an accumulated organic layer on the surface. The Cub River, however, has 15.3 percent $CaCO_3$ and is a crop land--the organic matter is tilled into the soil. The average of the ratios is 0.5 which was multiplied by the TP high and low export estimates (Table 13) to obtain an estimate for BAP export. The average value would provide the best BAP estimate due to the large range and a limited data set. The estimated BAP loads ranged from 28,600 to 33,800 kg BAP/yr using this ratio. These estimated BAP loads may 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. The estimate of the total phosphorus from the runoff needs to be cautiously accepted since the export coefficients are national averages and may not be representative of a calcareous area such as Cache County.

Table 13. 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.

Table 14. 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 µg/L	µg/ g SS	TP 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 Creek	3.91	130	18	11.6	7.84	1.95	932	735.	788.6	1366	1466
Cub River	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 Creek	7.97	710	32.5	10.2	8.13	0.74	2560	15.	5.9	2529	988
Weston Creek	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.

Phosphorus contributed from stream banks was also investigated because bank erosion and landsliding was obvious in many locations in Cache Valley (Table 14). The relative phosphorus contribution from stream banks with respect to runoff can be seen by dividing the stream bank BAP concentrations by the suspended solids concentrations. This allows a common basis comparison with the runoff samples. This comparison shows that the runoff samples contained several orders of magnitude more BAP than the stream bank samples. Runoff samples come from top soils that have continual phosphorus replenishment from atmospheric deposition, fertilizer addition, and/or decaying vegetation. The stream banks are naturally low in phosphorus and continually lose phosphorus by root uptake. In agreement, Taylor and Kunishi (1971) state that stream banks can actually act as phosphorus sinks because they are usually low in phosphorus content.

Analysis of the data in Table 14 showed that no correlations existed for either the runoff or stream bank samples between BAP and percent clay, percent $CaCO_3$, and pH. The Blacksmith Fork runoff sample had the lowest pH, lowest percent $CaCO_3$, and average clay percentage, and this sample gave the highest BAP estimate. The lowest BAP estimate for runoff samples was obtained for the Cub River sample which had a higher pH, the highest percent $CaCO_3$, and the highest clay content.

The Relative Contribution of BAP
From Each of the Sources

Cache County was used as the common base to compare the relative contributions of BAP from the various sources (Table 15). The point sources (not including Preston WWTP) in Cache County contribute approximately 28,200 kg BAP/yr. It was assumed that all of the 2,500 kg BAP/yr contributed from livestock runoff to the Bear River enters the 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. From these data, point sources contribute 46 percent, livestock runoff contributes 4 percent and land runoff (assuming 31,100 kg/year of BAP) contributes 50 percent of the bioavailable phosphorus load in Cache County. The majority of phosphorus from point sources comes from only three WWTP's. The livestock runoff comes from approximately 200 feedlot areas and the land runoff is very diffuse, encompassing 744,000 acres.

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 proposed reservoirs should be ranked highest (Sorensen et al., 1987), and a management plan should apply the best management practices to those sources which will reduce the BAP load to the reservoirs in the most cost effective manner.

The proposed Honeyville reservoir will be immediately below Cutler reservoir and is the main reservoir impacted by the Cache Valley sources. The Logan, Hyrum, and Wellsville wastewater effluents are the closest point sources to the Honeyville reservoir and, therefore, should be highly ranked since they have the potential of contributing high amounts of BAP. The discharge from the E. A. Miller Inc. treatment plant needs to be studied to determine if it fits into this category.

Table 15. Total and bioavailable phosphorus contributions from various sources in Cache County, UT

	Total Phosphorus kg/yr	Bioavailable Phosphorus kg/yr
Three (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 $\approx$ 1.0

Bioavailable Phosphorus Transport in the Bear River

It is noteworthy that on May 11, 1987, BAP was only 10 μg/L at the Bear River sampling site above Cutler Reservoir (Appendix A). On July 15, 1987, October 21, 1987, and February 22, 1988 BAP concentration was below detection (< 8 μg/L) at this site. This site would receive most of the land runoff in Cache Valley. These data suggest that even though land runoff can contribute 50 percent of the BAP, only a small fraction may remain bioavailable by the time it reaches Cutler Reservoir. More data is needed to verify this possibility.

Concentrations of bioavailable phosphorus through Cutler Reservoir

Understanding the transport of phosphorus through Cutler Reservoir and the impacts it will have upon the downstream Honeyville Reservoir is vital to accurate predictions of algae production in that reservoir. Vollenweider (1968) estimated that, based on the nutrient budget of 15 eutrophic lakes, an average of 49 percent of the phosphorus entering the lakes was retained in them, presumably in the sediments. Porcella and Bishop (1975) found that a similar ratio, 54 percent, 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 two and five days based on an average 5 foot depth (approximated from random measurements) and an approximate surface area of 5.4 X 10^7 ft^2. Probably much less than 54 percent of the influent phosphorus remains in Cutler Reservoir because its retention time is shorter than Hyrum's. An approximate mass balance for TP calculated from data collected October 20, 1987 (Table 16) indicated that 69 percent

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

Date	Ab. Cutler Res. Bear River Flow cfs	TP µg/L	BAP µg/L	Benson Marina--inflow of L. Bear and Logan R. flow cfs	TP µg/L	BAP µg/L	Below Cutler Res. Calculated values flow cfs	TP* µg/L	(1) <8=8 BAP* µg/L	(2) <8=0 BAP* µg/L	Below Cutler Res. measured values 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.

of the influent TP can be stored in Cutler Reservoir. However, similar data collected October 3, 1987 indicated that 193 percent of the influent phosphorus was released from the reservoir. Cutler 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 approximately 50 percent of the influent phosphorus since its expected retention time will be similar to that of Hyrum Reservoir.

Logan, Hyrum, and Wellsville WWTP's discharge to the Little Bear and Logan River systems within four, nine, and nine miles, respectively, of Cutler Reservoir. Table 17 presents the October 3, 1987 phosphorus and flow data that is graphed in Figure 23. The outflow from Cutler Reservoir was measured at the site below Cutler Reservoir. Table 16 shows that the Bear River usually had 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 had lower or equal TP concentrations to those in the Logan and Little Bear rivers, and the Bear River had BAP concentrations so low they were unmeasurable (Table 17). The higher flows and lower TP's and BAP's of the Bear River can dilute the phosphorus concentration in the Little Bear and Logan rivers and explain most of the decrease in concentration. At mile 12, BAP was reduced to below measurable concentrations while TP concentration was reduced by 20 percent. One possible explanation for the lack of a decrease in phosphorus concentration at mile nine, 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 rivers, 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 (ft³/s)	Total Phosphorus (µg/L)	Bioavailable Phosphorus (µg/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
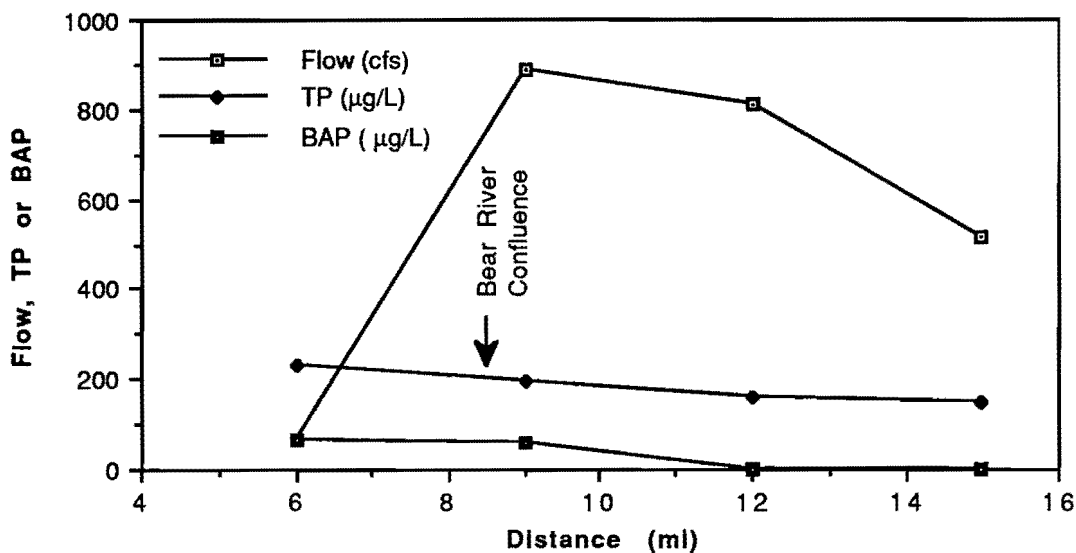
Figure 23. Changes in flow, total and bioavailable phosphorus with distance through Cutler Reservoir on October 3, 1987.

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 two, while the TP concentrations are within 33 percent 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 the reservoir due to power peaking. Two scenarios were used to calculate the BAP concentrations below Cutler Reservoir: the BAP above Cutler Reservoir is assumed to be eight, or the BAP above Cutler Reservoir is assumed to be zero. 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 may be another removal mechanism for BAP. This removal mechanism might be plant uptake, chemical precipitation, settling, or a combination of these processes.

Figure 24 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 ($r^2 = 0.95$); BAP, however, 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 are stored within the reservoir. If the percent change is positive then Cutler reservoir is releasing flow and TP. It appears that water volume and TP are generally conserved through Cutler Reservoir, but the data suggest that BAP may not be conserved and is actually removed through the reservoir.

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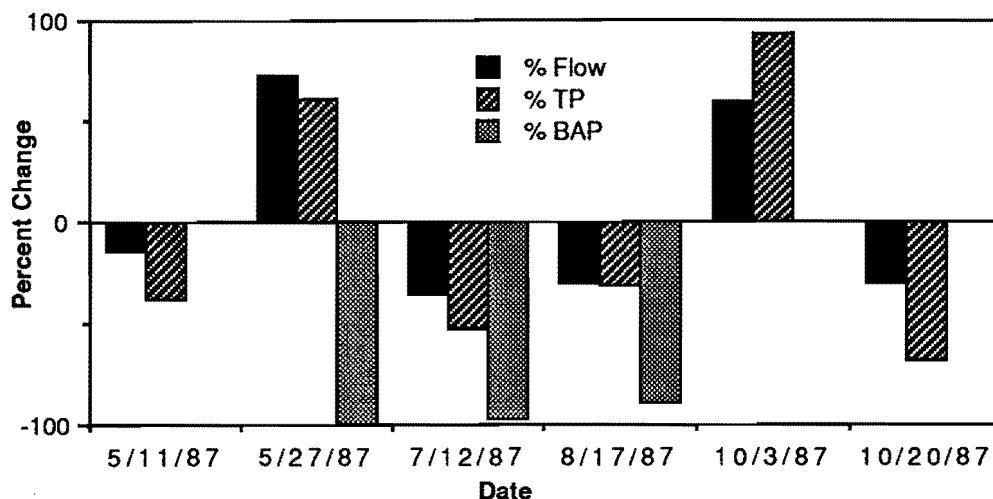
Figure 24. Percent change [((output - input)/output)*100] of total phosphorus, bioavailable phosphorus (BAP) and flow through Cutler Reservoir.

Assuming an annual flow of 3,000,000 acre-ft passing Cutler Dam and a BAP concentration of < 8 µg/L, less than 30,000 kg (< 33 tons) of BAP would pass Cutler Dam per year. If ≤ 12.4 µg BAP/L (observed 1987 average, Appendix A) is assumed, then ≤ 46,000 kg (51 tons) of BAP would pass Cutler Dam per year.

Evaluation of the Eutrophication Potential
for the Proposed Honeyville Reservoir

If the BAP concentrations entering the proposed Honeyville Reservoir are as low as the data analysis suggests, the eutrophication potential of the reservoir needs to be reassessed. Sorensen et al. (1986) used the computerized riverine reservoir water quality model RESEN (Messer et al., 1982) to predict chlorophyll a concentrations in Honeyville Reservoir. Based on the assumption that approximately 50 percent of the total phosphorus flowing into the reservoir would be available for algal growth (i. e., 54 µg BAP/L), the model predicted summertime chlorophyll a concentrations ranging from 9.5 to 23 µg/L, depending on the development of grazing protozoan populations near the dam. These chlorophyll concentrations are typical of eutrophic conditions (Jones and Lee, 1982). Empirical models, based on the correlation of normalized total phosphorus loading with observed summertime chlorophyll concentrations in lakes and reservoirs (Jones and Lee, 1982), also predicted eutrophic conditions in the proposed reservoir. Using the same hydrologic conditions and reservoir temperature profile established for the predictions of Sorensen et al. (1986), the RESEN model was run assuming that the BAP concentration was at the 8 µg/L detection limit of the BAP assays. Assuming no zooplankton grazing, the maximum predicted chlorophyll a concentration was 3.6 µg/L, indicating oligotrophic to mesotrophic conditions.

The potential for relatively low algal production and hence high water quality (e. g., high transparency, low benthic oxygen demand, and low taste and odor production) in the

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Honeyville reservoir is encouraging but must be viewed with caution. The water temperature model used by Sorensen et al. (1986) predicted summer and early fall thermal stratification of the reservoir at depths ranging from 20 to 36 ft (six to 11 meters). Since the reservoir is expected to have an average depth of about 32 ft, the hypolimnion may extend over more than half of the reservoir. If the benthic oxygen demand is sufficient to make the hypolimnion anaerobic, iron bound phosphate in the sediments may become dissolved in the hypolimnion. Then, during the late fall and spring when isothermal conditions develop in the water column, the phosphate will be distributed throughout the water column when it is ixed by the wind. If this happens the eutrophic conditions predicted by the empirical model may develop in the fall and/or spring. The potential input of BAP from the sediments was not simulated by the computer model. Much of the total phosphorus that would enter the Honeyville Reservoir is in insoluble forms dominated by apatites and other calcium and magnesium minerals and only small amounts are likely to be associated with iron, suggesting that relatively small amounts of phosphate may be solubilized under reducing (anaerobic) conditions in the sediments. This suggests that the eutrophication potential may be low to moderate.

Perhaps the best indication of eutrophication potential can be gained by studying similar existing reservoirs. Hyrum Reservoir (Figure 1) has similar hydraulic residence time to the proposed reservoir (66 vs. 58 days for Honeyville), is only slightly deeper (average = 39 ft), has averaged ~78 percent of the area based total phosphorus loading of the proposed reservoir ($7,600$ mg $P \cdot m^{-2} \cdot y^{-1}$ vs. $9,800$ mg $P \cdot m^{-2} \cdot y^{-1}$), becomes strongly thermally stratified most summers, has water with similar hardness (260 vs. 290 mg total hardness as $CaCO_3$/L; Sorensen et al. 1986), but has had a somewhat lower pH (average pH = 7.6 vs. 8.2). Hyrum Reservoir experiences algal blooms of eutrophic proportions at least once in most years. Willard Reservoir, fed by Weber River water, is relatively shallow (≤ 22 ft), frequently mixed by wind action, and morphologically very different from the proposed Honeyville Reservoir but is also eutrophic (Sorensen et al., 1987). Cutler and Oneida reservoirs are not eutrophic but algae production is apparently limited by the poor translucence of their turbid waters (Sorensen et al., 1986). The information available from comparison with neighboring reservoirs suggests that the Honeyville Reservoir will become eutrophic. This conclusion is supported by empirical models based on other lakes and reservoirs.

The implication that the low BAP load of the Bear River and its tributaries below Bear Lake will not support eutrophication of the proposed reservoirs is in conflict with the observation of eutrophic conditions in Hyrum Reservoir and with empirical model predictions of eutrophic conditions (Sorensen et al., 1986). Since the empirical information integrates nutrient availability and cycling factors with all other environmental factors affecting algal growth, it is difficult to conclude that the proposed reservoirs, especially the Honeyville Reservoir, will not be eutrophic. Closer analysis of potential phosphorus availability from sediments and in-reservoir nutrient cycling processes is needed before the eutrophication potential of the proposed reservoirs can be predicted with confidence.

Phosphorus Control and Treatment

Technology is available that can reduce the phosphorus from WWTP's by 90 percent. Reckhow et al. (1980) discuss the economics of chemical precipitation, biological, and land treatment for phosphorus removal. The construction, operation, and maintenance costs from Reckhow et al. (1980) were updated to June, 1988 by using the ENR construction cost index of 4531.58 (ENR, 1988) (Table 18).

Table 18. Costs for construction and maintenance for various treatment systems for phosphorus removal from wastewater. (Data from Reckhow et al. (1980) updated using ENR construction cost index = 4531 for June, 1988.)

Treatment Process	Total Phosphorus Removal (%)	Wastewater Flow (MGD)	Construction Cost ($)	Annual Maintenance Cost ($)
Wetlands (Aquaculture)	0-94	1	26,000	8,300
		10	140,000	37,000
Rapid Infiltration, Underdrained	0-90	1	440,000	47,000
		10	3,400,000	2,400,000
Rapid Infiltration, Not Underdrained	0-90	1	370,000	37,000
		10	3,100,000	2,300,000
Land Treatment, Slow Rate, Sprinkler (Center Pivot), Underdrained	80-99	1	1,200,000	64,000
		10	9,800,000	440,000
Land Treatment, Slow Rate, Sprinkler (Center Pivot), Not Underdrained	80-99	1	1,200,000	64,000
		10	9,800,000	440,000
Land Treatment, Slow Rate, Gravity (Boarder Strip Application), Not Underdrained	80-99	1	88,000	39,000
		10	370,000	240,000
Land Treatment, Slow Rate, Gravity (Boarder Strip Application), Underdrained	80-99	1	1,200,000	51,000
		10	5,900,000	270,000
Overland Flow, Gravity, Not Underdrained	30-60	1	980,000	22,000
		10	8,800,000	150,000
Activated Sludge (Phostrip)	>90	1	820,000	64,000
		10	1,600,000	100,000
Alum Addition	94	1	48,000	55,000
		10	180,000	400,000
Ferric Chloride Addition	56-97	1	46,000	40,000
		10	160,000	280,000
Lime Clarification of Raw Wastewater	75	1	47,000	20,000
		10	210,000	200,000

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Alum, ferric chloride, and lime addition are the three most common phosphate precipitation methods. Removal efficiencies ranged from 75 percent for lime to 97 percent for ferric chloride addition. These processes produce sludges which, because of their toxicity, may require additional treatment depending on disposal methods. For a 1 million gallon per day (MGD) WWTP, ferric chloride treatment had the lowest construction costs of $46,000, while alum addition was the most costly at $48,000. The operation and maintenance costs for a 1 MGD WWTP showed that lime clarification was the least expensive at $20,000 per year, while alum addition was again the highest at $55,000. For a 10 MGD WWTP, ferric chloride addition had the lowest construction costs of $160,000 and lime clarification was highest at $210,000. The operation and maintenance for a 10 MGD WWTP showed that lime clarification was the least expensive at $200,000 per year with alum addition being the most expensive at $400,000 per year (Table 18). Even though lime clarification has relatively low construction costs compared to the other strategies, it should only be recommended with caution because of its low (75 percent) removal efficiency and high operation and maintenance costs.

Land treatment technology ranges from wetlands to overland flow with removal efficiencies ranging between 0 to 99 percent depending on the site and design characteristics. Construction costs and annual operation and maintenance costs show that wetlands are the least expensive for treating both 1 and 10 MGD while land treatment options are the most expensive (Table 18).

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 percent of the phosphorus. Estimated construction costs are $820,000 for 1 MGD and $1,600,000 for 10 MGD. Annual operation and maintenance costs are $64,000 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 percent of the total BAP, but its control is difficult. Land runoff also contributes significant quantities of sediment which fills channels and reservoirs. 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. Only 0.2 percent and 9.9 percent, respectively, of Cache County's agriculture used no-till and low-till practices in 1987 (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 in 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; as of December 1988, 17,250 acres (38 percent) had been put into this program.

Stream bank erosion should be controlled to reduce the sediment load but not necessarily to control BAP. Stream banks 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 four percent of the BAP. Wieneke et al. (1980) stated that separation of cattle from the receiving stream by approximately 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 water table, consistent with federal, state, and local laws and regulations." (Chapter 13-A, Agriculture Zone, amended November 15, 1983, Section 13-6-3).

A benefit-cost analysis is needed to determine whether it is more cost effective to control the sources of phosphorus to the proposed reservoirs or to treat the reservoir water before use. Sorensen et al. (1986) determined that it would cost from $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 need to be determined and subtracted from the costs of source treatment before a meaningful cost analysis can be determined.

The comprehensive benefits of high quality water resources are difficult to measure in monetary value alone. Because of this, decisions to manage water resources to maintain high water quality cannot be made from a benefit/cost analysis alone. Traditionally, management to maintain the quality of natural resources has been a matter of government policy.

An Economic Evaluation of Phosphorus Control

Costs of Treatment Measures for Reduction
of Bioavailable Phosphorus

The most promising alternatives for reducing the bioavailable phosphorus loads in the lower Bear River Basin appear to be associated with treatment of municipal wastewater. Logan City, by far the largest single contributor of municipal wastewater and BAP in the basin, is used as an example in this cost analysis.

For developing the treatment option considered in this example, two basic methods were selected: evaluation of irrigation and wetland use, and consideration of chemical treatment.

For irrigation, sprinkler irrigation technology was selected because of more uniform control of application, especially considering the fragile clay soils within a reasonable distance from the Logan City wastewater lagoons. Center pivot irrigation and wheel line systems were both evaluated.

Two separate land application areas were evaluated for the study. The first, called area A, is near the lagoons (Figure 25) and is mostly pasture land. It would be expected to remain in pasture. The soils in this area are primarily silty clays and silty clay loams (USDA, 1974). Currently, the area is partially flood irrigated and is generally a fairly wet, poorly drained area. Costs for investment and operation would be somewhat lower for this

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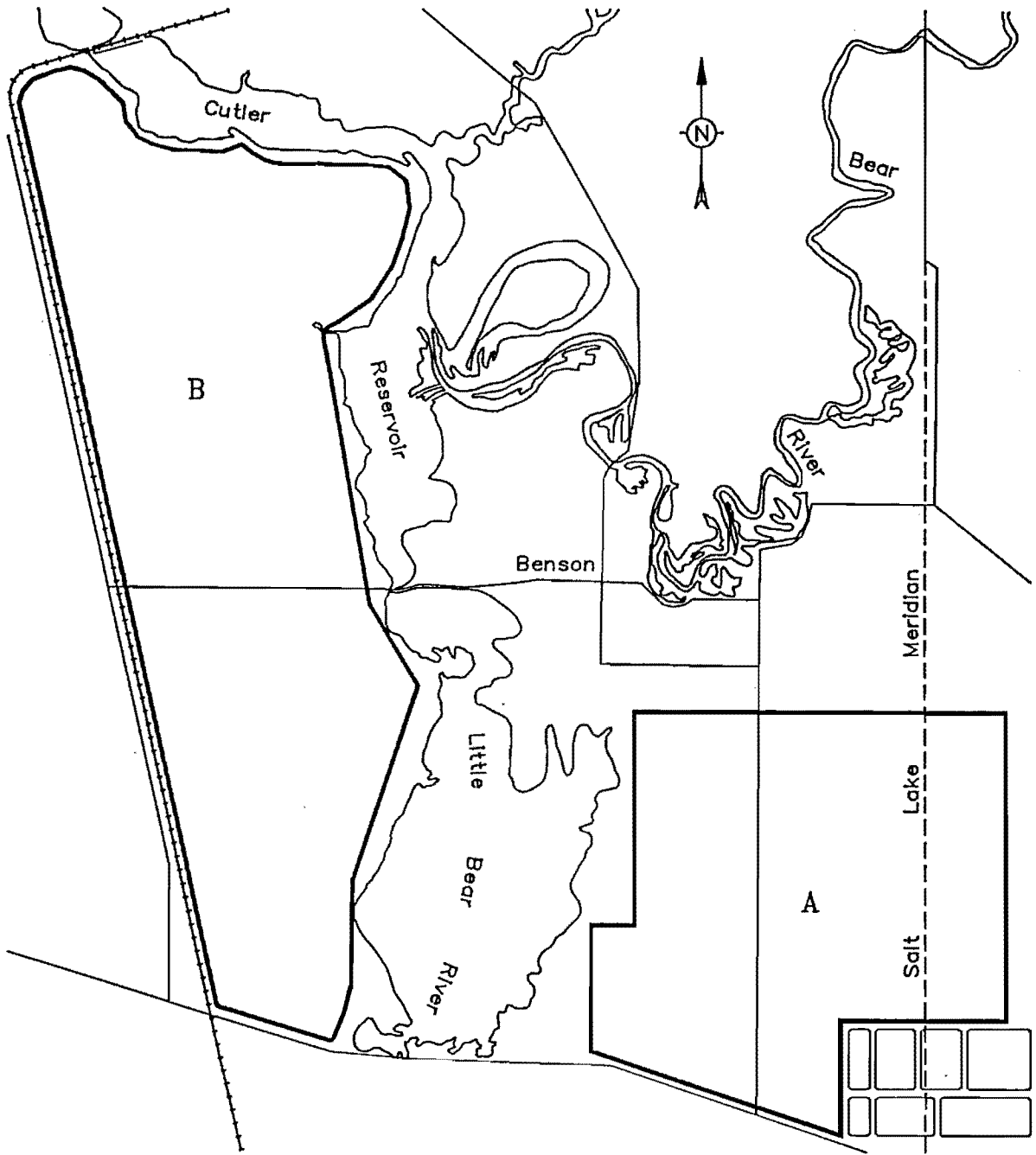
Figure 25. Map of proposed wastewater land application areas A and B near the Logan City wastewater lagoons.

area. Area B (Figure 25) is about five miles away from the lagoon outfall. The soils in this area are also primarily silty clay loams and silty clays (USDA, 1974) but could sustain a variety of crops grown in the area. Because of distance, investment and operating costs for irrigating this area are higher (Table 19).

As shown in Table 19, only 77 days of irrigation could be used at the rate of maximum water use. Thus, only 21 percent of the available water could be used per year. Nine center pivot systems, each irrigating 135 acres, are required to use 77 days of water. Eighteen such pivots are required to use 154 days effluent. It was calculated that by using the storage capacity in the lagoons to store water prior to and during the irrigation season, the amount of land that could be irrigated was twice the 77 days of direct flow that could be used. Thus, the use of 154 days, or 42 percent of the year, was also examined. Thirty or 60 wheel lines would be used with and without the storage option, respectively.

The land and crop selections available in the area are very limited. Irrigation and soils specialists express urgent concern that the soils must not be over irrigated. The irrigation system and design was specified, to a major extent, by irrigation specialists (Neil Allen and Robert W. Hill, Agricultural and Irrigation Engineering Department, Utah State University, personal communication, 1988). Care was taken that crops could use the amount of water applied.

A wetland treatment was also developed using an application rate of five inches per day. This required a fairly complex set of level areas, dikes, and weirs. Eighty acres would be required and this system was assumed to be operational for six months of the year without encountering climate problems such as snow cover, freezing, and icing which would limit the system effectiveness.

Table 19 itemizes the cost for each of the systems. Note that the annual investment cost plus the annual operating cost range from a little more than $69,000 for wetland treatment to over $512,000 per year for center pivot irrigation of Area B with storage. Of course, this is only a partial analysis. Account must be taken of the proportion of all water treated and thoroughness or efficiency of the treatment. The proportions of water subjected to each treatment are shown in Table 20. Additionally, the costs of ferric chloride treatment for water not applied to irrigation or wetlands are shown. Treatment by ferric chloride appeared to have lower investment and operating cost than other documented chemical methods (Reckhow et al., 1980).

Ranking of the cost-effectiveness of combined irrigation or wetland treatments with chemical treatment, as well as only chemical treatment is shown in Table 21. Ranges of effectiveness are shown for all forms of treatment. These ranges were combined in the analysis to provide an aggregated range. Ranking of the options was according to the mid-point of each range. These were used with cost elements to determine the cost per percent of removal of bioavailable phosphate. In any case, the costs are quite high. Decision makers may choose to do nothing, to use only chemical treatment, to use the wetland/chemical combination, or to use one of the irrigation/chemical options. Some individuals believe there are certain political, social or other values associated with consumptive use of water in irrigating crops.

Table 19. Investment and operating costs using Logan City municipal wastewater for irrigating two areas.

Item	Units	Area A					Area B			
		Center Pivot	Center Pivot with Storage	Wheel Line	Wheel Line with Storage	Wetlands	Center Pivot	Center Pivot with Storage	Wheel Line	Wheel Line with Storage
Amount of water	MGD	10	10	10	10	10	10	10	10	10
Number of days in year taking all outflow	days	77	154	77	154	180	77	154	77	154
Total capacity	gpm	6,944	13,888	6,944	13,888	6,944	6,944	13,888	6,944	13,888
Total horsepower required	hp	294	628	294	628	101	498	1,092	489	1,092
Number of 100 hp motors and pumps	No.	3	6	3	6	1	5	11	5	11
Acres irrigated	acres	1,170	2,340	1,200	2,400	80	1,170	2,340	1,200	2,400
Number of center pivots	No.	9	18				9	18		
Number of wheel lines	No.			30	60				30	60
Investment Costs:										
Motors & pumps (25 yr.)	$	45,000	90,000	45,000	90,000	15,000	75,000	165,000	75,000	165,000
Pipe network (25 yr.)	$	324,730	742,160	384,788	887,907	66,317	922,070	2,182,100	990,428	2,358,747
Center pivot with wiring and valving (25 yr.)	$	418,500	837,000				418,500	837,000		
Wheel lines (12 yr.)	$			210,000	420,000				210,000	420,000
Land forming, dikes, ditches (25 yr.)	$					263,000				
Total investment	$	788,230	1,669,160	639,788	1,397,907	344,317	1,415,570	3,184,100	1,275,428	2,943,747
Annualized cost @ 10%	$	86,839	183,891	70,485	154,007	37,933	155,953	350,792	140,514	324,313
Operating Costs:	$									
Energy @ $0.08/kwh	$	36,812	73,624	36,812	73,624	28,684	61,353	134,978	61,353	134,978
Operating & Maintenance:	$									
Pumps & motors	$	266	532	266	532	207	443	976	443	976
Irrigation system	$	6,026	12,052	3,024	6,048	1,300	6,026	12,052	3,024	6,048
Labor	$	6,739	13,478	12,960	25,920	1,080	6,739	13,478	12,960	25,920
Labor/Operating Cost	$	49,843	99,686	53,062	106,124	31,271	74,561	161,484	77,780	167,922
Total Annual Cost	$	136,682	283,577	123,547	260,131	69,204	230,514	512,276	218,294	492,235

Table 20. Costs for combined systems for treatment of Logan, Utah, municipal wastewater by land application and ferric chloride treatment.

Option	Percentage Wetland or Sprinkler Operating Time During Year	Percentage Ferric Chloride Operating Time During Year	Wetland or Sprinkler Total Annual Cost*	Ferric Chloride Annualized Investment Cost	Ferric Chloride Annual Operating Cost	Total Cost†	Net Benefits to Agriculture from Irrigation	Total Costs Less Net Benefits to Agriculture
Area A:								
Center Pivot	21.1	78.9	136,681	16,525	244,590	397,796	26,586	371,210
Center Pivot with Storage	42.2	57.8	283,574	16,525	179,180	479,279	53,136	426,143
Wheel Line	21.1	78.9	132,383	16,525	244,590	393,498	26,586	366,912
Wheel Line with Storage	42.2	57.8	277,802	16,525	179,180	473,507	53,136	420,371
Wet Lands	50	50	69,203	16,525	155,000	240,728	-2,000	242,728
Area B:								
Center Pivot	21.1	78.9	230,511	16,525	244,590	491,626	35,006	456,620
Center Pivot with Storage	42.2	57.8	512,270	16,525	179,180	707,975	70,012	637,963
Wheel Line	21.1	78.9	227,128	16,525	244,590	488,243	35,006	453,237
Wheel Line with Storage	42.2	57.8	509,902	16,525	179,180	705,607	70,012	635,595
Ferric Chloride		100		16,525	310,000	326,525	0	326,525

*From Table 19.

†Sum of total wetland or sprinkler, ferric chloride investment, and ferric chloride operating annual costs.

Table 21. A comparison and ranking of costs of bioavailable phosphorus control by alternative methods of land application and ferric chloride treatment for Logan, Utah, municipal wastewater.

| Option | Percentage Effectiveness of Irrigation or Wetlands When Used | Percentage of Bioavailable Phosphorus Controlled by Irrigation* | Percentage of Bioavailable Phosphorus Controlled by Ferric chloride* | Range of Total Control | | | Cost Net of Benefits to Agriculture in Dollars per Percent of Control | | | Rank in Dollars per Percent of Control |
				Low	Mid-point	High	Low	Mid-point	High	Midpoint
Area A:										
Center Pivot	80-99	17-19	44-77	61	78	96	6,085	4,759	3,867	4
Center Pivot with Storage	80-99	34-38	32-56	66	80	94	6,457	5,327	4,533	6
Wheel Line	80-99	17-19	44-77	61	78	96	6,015	4,704	3,822	3
Wheel Line with Storage	80-99	34-38	32-56	66	80	94	6,369	5,255	4,472	5
Wet Lands	0-94	0-47	28-48	28	62	95	8,669	3,915	2,555	1
Area B:										
Center Pivot	80-99	17-19	44-77	61	78	96	7,486	5,854	4,756	8
Center Pivot with Storage	80-99	34-38	32-56	66	80	94	9,666	7,975	6,787	11
Wheel Line	80-99	17-19	44-77	61	78	96	7,430	5,810	4,721	7
Wheel Line with Storage	80-99	34-38	32-56	66	80	94	9,630	7,945	6,762	9
Ferric Chloride			56-97	56	76	97	5,830	4,296	3,366	2

*Combining percentage effluent treated by irrigation, wetlands, or ferric chloride; Table 20.
The effectiveness of ferric chloride is assumed to be 56-97%.

Benefits from Irrigation

Two types of benefits are associated with irrigation that has municipal effluent: increased production as measured by net income, and reduction in costs of production associated with nutrients applied as part of the effluent.

The first can be measured best by changes in rental value of land. In area A, the rental value of pasture land can be expected to change by approximately one animal unit month (AUM) per acre at a price of $9 per AUM. Thus, the annual value for 1,200 acres is $10,800. Additionally, the effluent is calculated to contain sufficient nutrients to negate the need for any fertilizer application. As shown by Davis and Bond (1986) this amounts to $6.50 for nitrogen, $2.64 for phosphate, and application costs of $4 per acre. Therefore, the fertilizer replacement value is $13.14 per acre. So the value in Area A is 1,200 times $13.14, or $15,768. Thus, benefits from sprinkling area A are ($10,800 + $15,768) $26,568, or $53,136 for 2,400 acres if effluent is stored so that the larger acreage can be irrigated. For the wetlands, there would be loss of all agricultural production from 80 acres. The rental value per acre is approximately $25, so the total loss is $2,000. For area B, the increased production value is the difference between the rental value of dry farmland and irrigated land. This can be shown to amount to about $16 (Utah State Tax Commission, 1988) per acre per year for the class III or IV land involved. Thus, the value is $19,200 for 1,200 acres. Additionally, the value of fertilizer and cost for application for the mix of crops (alfalfa, 5-years; barley 2-years) is $13.17 per acre, or $15,806 for 1,200 acres for a total benefit of $35,006 on 1,200 acres, or $70,012 on 2,400 acres.

Certainly, these irrigation benefits are not sufficient to justify an irrigation project based on the irrigation values but they do offset up to 20 percent of costs in area A and 16 percent of costs in area B. Of course, a more careful examination of yield responses and values is needed before proceeding with further planning.

Downstream Benefits of Bioavailable Phosphorus Reduction or Removal

The estimates of the economic benefits of environmental improvements associated with phosphate removal are beyond the scope of the study. This section briefly describes the types of studies and considerations needed in order to estimate these benefits.

The benefits can be appropriately divided into two categories. First is the "increased availability of certain goods and services." They represent an increase in some activities and are evaluated as the value of future uses. Many of these activities are not in usual market transactions; therefore, value may represent a consumer surplus or willingness to pay rather than a price. The other category is "damage reduction benefits." Examples of the types of benefits are listed below:

Increased production of goods and services:

Swimming
Fishing
Boating
Hunting
Wildlife habitat
Visual and aromatic amenities

Damage reduction benefits:

Municipal and industrial treatment costs (removal of suspended solids, taste and odor, chlorinated hydrocarbon precursor compounds, etc. {Sorensen et al., 1986}).
Mosquito and disease control.
Use of lower Bear River water rather than more expensive alternatives.

To evaluate these potential benefits, there are some realities of the physical environment that must be addressed. First, the lower Bear River is not a premium environment for some types of specialized recreation. Second, the changes proposed by a project may or may not alter the conditions for activities such as waterfowl hunting. Third, damage reduction benefits for municipal water treatment may not apply if the water must be treated for reasons not related to phosphorus control (i. e., eutrophication control) or if an alternative supply of water is used.

It is useful to review the standard procedures for estimating recreation benefits (U. S. Water Resources Council, 1983). In these procedures, it is stressed that "benefits for projects (or project features) that alter willingness to pay (e. g., through quality changes) are measured as the difference between the without- and with-project willingness to pay. Willingness to pay includes entry and use fees actually paid for site use plus any unpaid value (surplus) enjoyed by consumers."

Figure 26 from the U. S. Water Resources Council (1983) is a flowchart for selecting the evaluation method. From reviewing this flowchart, it seems likely that the unit day value (UDV) would be the method determined. Following this, it would be recognized that changes in recreation use and value must be based on the with- and without-project differences. The changes can be estimated according to the flowchart in Figure 27 from the U. S. Water Resources Council (1983). Other details are available in this document.

In following the likely determination to use the unit day values method, it is appropriate to review the suggested application of unit day values to estimated use. Two general categories of recreation are listed. "General" activities are those enjoyed by a majority of outdoor users; they require development and maintenance of convenient access and adequate facilities. "Specialized" refers to activities which are limited use, intensity is low, and high skills and appreciation of the activity is often involved. In Table 22, the conversion of points to dollar value is tabulated. The points, in turn, are tabulated for general recreation in Table 23 and for special recreation Table 24.

Generally, a review of these point and value systems, as applied to the lower Bear River, would likely produce estimates in the lower range of point values and lower range of dollar estimates. These estimates would have to be done in some detail, but it appears that changes in use and in recreation and the values associated with these changes would be fairly limited and would not extend far in offsetting costs associated with phosphorus removal. Other benefits besides recreation would require specific analysis, but these benefits may also be fairly limited.

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Table 22. Conversion of points to dollar values for recreational use of water resources (from U.S. Water Resources Council, 1983, p. 84).

Activity Categories	Point values										
	0	10	20	30	40	50	60	70	80	90	100
General recreation*	1.60	1.90	2.10	2.40	3.00	3.40	3.70	3.90	4.30	4.60	4.80
General fishing and hunting*	2.30	2.60	2.80	3.10	3.40	3.70	4.10	4.30	4.60	4.70	4.80
Specialized fishing and hunting**	11.20	11.50	11.70	12.00	12.30	13.50	14.70	15.60	16.80	18.00	19.00
Specialized recreation other than fishing and hunting**	6.50	6.90	7.40	8.00	8.50	9.60	10.60	12.80	14.90	17.00	19.00

*Points from Table 23.
**Points from Table 24.
Note.--Adjust dollar value for subsequent years to reflect changes in the Consumer Price Index after July 1, 1982.

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Table 23. Guidelines for assigning points for general recreation (from U.S. Water Resources Council, 1983, p. 85)

Criteria	Judgment factors				
(a) Recreation experience [1]	Two general activities [2]	Several general activities	Several general activities; one high quality value activity [3]	Several general activities; more than one high quality high activity	Numerous high quality value activities; some general activities
Total points: 30 Point value:	0–4	5–10	11–16	17–23	24–30
(b) Availability of opportunity [4]	Several within 1 hr. travel time; a few within 30 min. travel time	Several within 1 hr. travel time; none within 30 min. travel time	One or two within 1 hr. travel time; none within 45 min. travel time	None within 1 hr. travel time	None within 2 hr. travel time
Total points: 18 Point value:	0–3	4–6	7–10	11–14	15–18
(c) Carrying capacity [5]	Minimum facility development for public health and safety	Basic facilities to conduct activity(ies)	Adequate facilities to conduct without deterioration of the resource or activity experience	Optimum facilities to conduct activity at site potential	Ultimate facilities to achieve intent of selected alternative
Total points: 14 Point value:	0–2	3–5	6–8	9–11	12–14
(d) Accessibility	Limited access by any means to site or within site	Fair access, poor quality roads to site; limited access within site	Fair access, fair road to site; fair access, good roads within site	Good access, good roads to site; fair access, good roads within site	Good access, high standard road to site; good access within site
Total points: 18 Point value:	0–3	4–6	7–10	11–14	15–18
(e) Environmental quality	Low esthetic factors [6] exist that significantly lower quality [7]	Average esthetic quality; factors exist that lower quality to minor degree	Above average esthetic quality; any limiting factors can be reasonably rectified	High esthetic quality; no factors exist that lower quality	Outstanding esthetic quality; no factors exist that lower quality
Total points: 20 Point value:	0–2	3–6	7–10	11–15	16–20

[1] Value for water-oriented activities should be adjusted if significant seasonal water level changes occur.
[2] General activities include those that are common to the region and that are usually of normal quality. This includes picnicking, camping, hiking, riding, cycling, and fishing and hunting of normal quality.
[3] High quality value activities include those that are not common to the region and/or Nation and that are usually of high quality.
[4] Likelihood of success at fishing and hunting.
[5] Value should be adjusted for overuse.
[6] Major esthetic qualities to be considered include geology and topography, water, and vegetation.
[7] Factors to be considered to lowering quality include air and water pollution, pests, poor climate, and unsightly adjacent areas.

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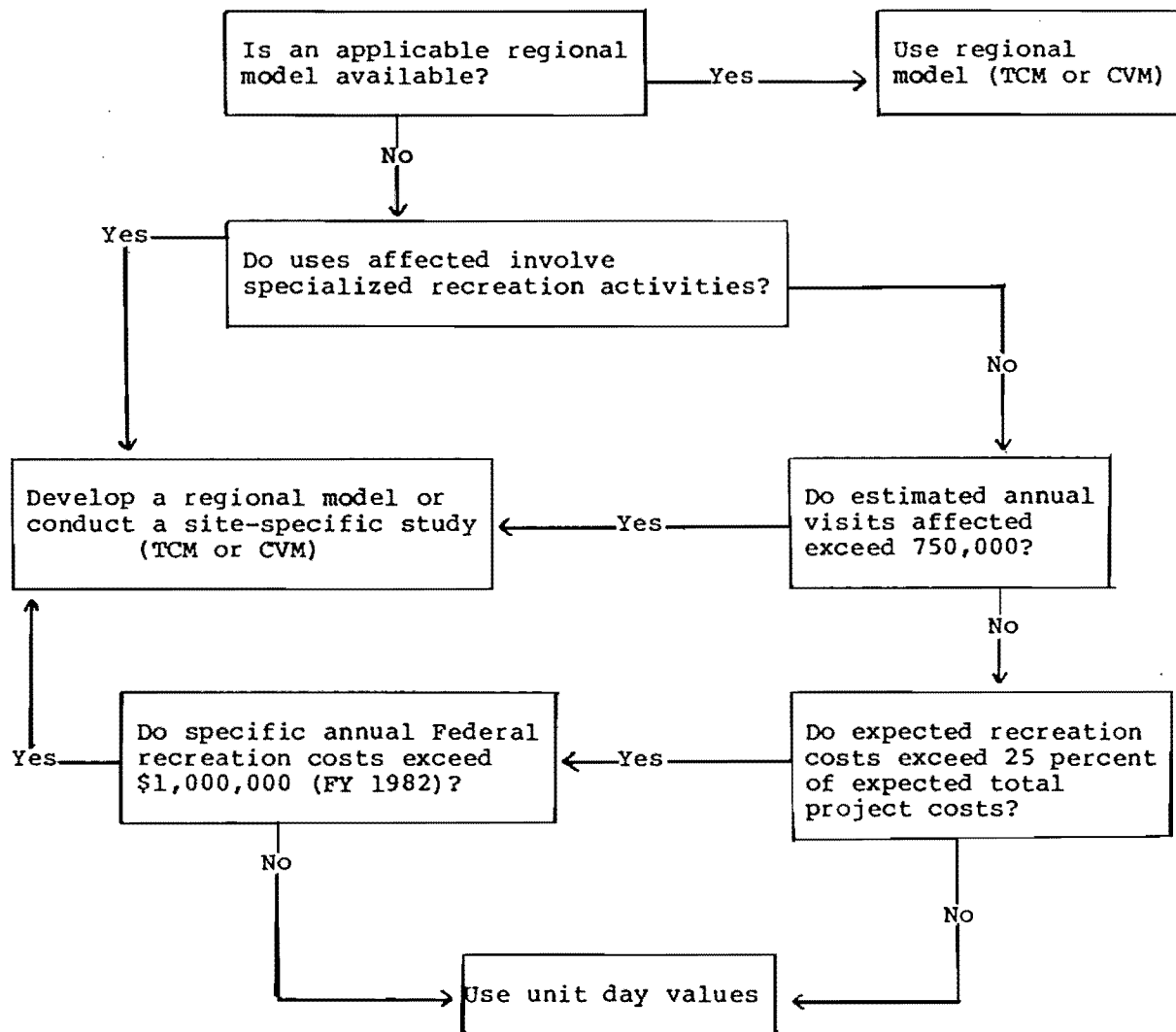
Figure 26. Criteria for selecting procedures for evaluating recreation benefits (from U. S. Water Resources Council, 1983, p. 69).

Table 24. Guidelines for assigning points for special recreation (from U.S. Water Resources Council, 1983, p. 85-86).

Criteria	Judgment factors				
(a) Recreation experience [1] Total points: 30 Point value:	Heavy use or frequent crowding or other interference with use 0-4	Moderate use, other users evident and likely to interfere with use 5-10	Moderate use, some evidence of other users and occasional interference with use due to crowding 11-16	Usually little evidence of other users, rarely if ever crowded 17-23	Very low evidence of other users, never crowded 24-30
(b) Availability of opportunity [2] Total points: 18	Several within 1 hr. travel time; a few within 30 min. travel time	Several within 1 hr. travel time; none within 30 min. travel time	One or two within 1 hr. travel time; none within 45 min. travel time	None within 1 hr. travel time	None within 2 hr. travel time
(c) Carrying capacity [3] Total points: 14 Point value:	Minimum facility development for public health and safety 0-2	Basic facilities to conduct activity(ies) 3-5	Adequate facilities to conduct without deterioration of the resource or activity experience 6-8	Optimum facilities to conduct activity at site potential 9-11	Ultimate facilities to achieve intent of selected alternative 12-14
(d) Accessibility Total points: 18 Point value:	Limited access by any means to site or within site 0-3	Fair access, poor quality roads to site; limited access within site 4-6	Fair access, fair road to site; fair access, good roads within site 7-10	Good access, good roads to site; fair access, good roads within site 11-14	Good access, high standard road to site; good access within site 15-18
(e) Environmental quality Total points: 20 Point value H	Low esthetic factors [4] exist that significantly lower quality [5] 0-2	Average esthetic quality; factors exist that lower quality to minor degree 3-6	Above average esthetic quality; any limiting factors can be reasonably rectified 7-10	High esthetic quality; no factors exist that lower quality 11-15	Outstanding esthetic quality; no factors exist that lower quality 16-20

[1] Value for water-oriented activities should be adjusted if significant seasonal water level changes occur.
[2] Likelihood of success at fishing and hunting.
[3] Value should be adjusted for overuse.
[4] Major esthetic qualities to be considered include geology and topography, water, and vegetation.
[5] Factors to be considered to lowering quality include air and water pollution, pests, poor climate, and unsightly adjacent areas.

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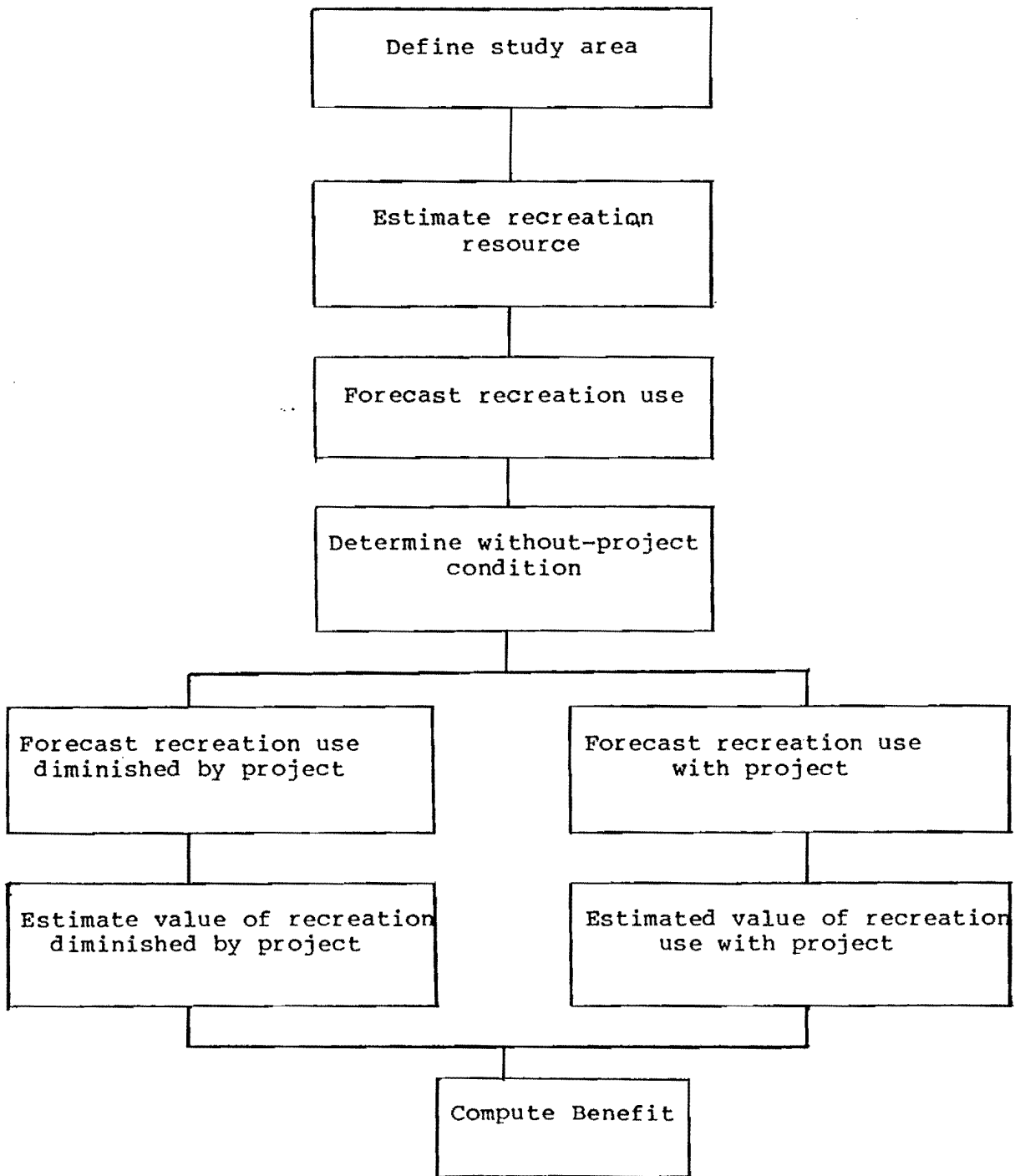
Figure 27. Flowchart of recreation benefit evaluation procedures (from U.S. Water Resources Council, 1983, p. 71).

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APPENDIX A

Table A-1. Data from April 19-21, 1987, (spring) sampling of the Bear River and it tributaries.

Stream	Sample Location	River mile upstream from Honeyville site	Field Measurements			Concentrations			Mass Transport	
			Flow cubic ft/s	Temp. °C	Cond. (25°) umhos	NO3-N+ NO2-N mg/l	Total Phosphorus ug/l	Ortho-Phosphorus ug/l	Total Phosphorus mg/s	Ortho-Phosphorus mg/s
Bear River	Bear Lake Outlet Canal at US-89	192.9	41.	7.0	692.0	0.02	73	<5	85	6
Bear River	Pescadero at USGS gauge	182.6	187.	6.0	627.0	0.22	99	<5	521	26
Bear River	Above Soda Point Res. USGS gauge	153.8	313.	5.5	518.0	0.10	39	<5	347	44
Bear River	At Grace Dam UPL gauge	138.3	353.	8.0	624.0	0.19	52	<5	517	0
Bear River	Above Alder Creek at Bridge	125	-	7.0	611.0	0.35	118	8	-	-
Cottonwood Creek	Near Cleveland, ID at I-34		50.	4.0	282.0	0.03	31	<5	43	6
Bear River	Above Oneida Reservoir	109	519.	7.0	656.0	0.39	64	<5	943	134
Bear River	Below Oneida Reservoir UPL gage	104.2	95.	10.5	649.0	0.37	57	17	154	45
Mink Creek	Near Bear R. confluence		2.	8.0	451.0	0.58	92	55	5	3
Bear River	Above Battle Creek at US-91	86.8	2288.	11.0	701.0	0.21	171	<5	11080	324
Battle Creek	Near Bear R. confluence		18.	-	1349.0	0.74	1300	121	679	355
Deep Creek	Near Bear R. confluence		50.	-	947.0	0.82	126	24	179	255
Five-mile Creek	Near Bear R. confluence		8.	-	816.0	1.37	144	83	34	8
Weston Creek	Near Bear R. confluence		33.	-	1559.0	3.56	51	9	48	44
Bear River	West of Fairview, ID at USGS gauge	72.4	407.	-	750.0	0.35	64	<5	739	58
Bear River	West of Richmond, UT	60	-	-	768.0	0.40	78	<5	-	-
Bear River	Above Cutler Reservoir, West of Benson	37.6	1145.	-	1050.0	0.44	125	<5	4053	162
Bear River	Below Cutler Reservoir at UPL gauge	26.8	2.	-	1290.0	0.34	124	<5	8	0
West Wheelon Canal	Below Cutler Reservoir	26.8	651.	-						
East Wheelon Canal	Below Cutler Reservoir	26.8	121.	-						
Bear River	West of Deweyville at Bridge	11.1	841.	-	619.0	0.21	166	<5	3955	119
Bear River	West of Honeyville at Bridge	0	1745.	-	697.0	0.25	144	<5	7117	247
Cub River	North of Franklin, ID at Bridge		-	-	280.0	0.33	44	16	-	-
Cub River	West of Franklin, ID at Bridge		44.	-	313.0	0.36	59	19	74	23
Worm Creek	West of Franklin, ID		7.	-	643.0	2.28	504	179	105	37
High Creek	At US-89/91		-	-	278.0	0.15	15	<5	-	-
Cub River	North of Richmond at Bridge		-	-	346.0	0.58	164	48	-	-
Cub River	South of Ricmond at Bridge		-	-	345.0	0.56	151	48	-	-
Logan River	Logan Lagoon Effluent		2.	-						
Logan River	Below Logan Lagoon Effluent		-	-	430.0	0.22	483	439	-	-
Davenport Creek	At road abv. confluence with L. Bear R.		97.	9.0	262.0	0.13	58	6	160	17
Little Bear River	So. Fork Below Davenport Cr.		90.	9.5	239.0	0.13	46	6	116	16
Little Bear River	Below Hyrum Reservoir		-	-	407.0	0.33	27	<5	-	-
Little Bear River	Above Logan R. confluence, E. Mendon		266.	-	448.0	0.56	97	<5	727	
Mill Creek	Near Blacksmith Fork confluence		24.	8.0	313.0	0.16	56	19	38	13
Sheep Creek	Near Blacksmith Fork confluence		12.	7.5	380.0	0.39	10	<5	3	2
Blacksmith Fork R.	Above Sheep Cr.		70.	8.0	390.0	0.28	12	6	24	13
Blacksmith Fork R.	At Anderson Ranch		153.	9.0	385.0	0.31	16	6	69	25

Table A-2. Data from May 11, 1987, (late spring) sampling of selected locations on the Bear, Little Bear, and Blacksmith Fork Rivers.

Stream	Sample Location	River mile upstream from Honeyville site	Flow cubic ft/s	Concentrations Total Phosphorus µg/l	Ortho-Phosphorus µg/l	Mass Transport Total Phosphorus mg/s	Ortho-Phosphorus mg/s	Concentration Bioavailable Phosphorus µg/l	Transport Bioavailable Phosphorus mg/s
Bear River	Below Oneida Reservoir UPL gage	104.2	250.0	50.	18.	351.	129.	<20	
Bear River	West of Fairview, ID at USGS gage	72.4	277.0	388.	<5	3044.		<8	<63
Bear River	Above Cutler Reservoir, West of Benson	37.6	813.3	138.	<5	3179.		10.	230.
Bear River	Below Cutler Reservoir UPL gage	26.8	2.4	125.	14.	8.	1.	30.	2.
Bear River	West Wheelon Canal	26.8	712.0	-	-	-	-	-	-
Bear River	East Wheelon Canal	26.8	151.0	-	-	-	-	-	-
Bear River	West of Honeyville at Bridge	0	69.7	142.	<5	280.		32.	63.
Little Bear River	Below Davenport Cr. at USGS gauge		80.0	24.	13.	54.	29.	<8	<18
Little Bear River	Benson Marina		206.9	299.	49.	1752.	287.	-	-
Blacksmith Fork R.	Below at USGS gage		77.0	17.	7.	37.	15.	<8	<17

Table A-3. Data from June 21, 1987, sampling of point sources and streams in Cache Valley.

Stream	Sample Location	Field Measurements			Concentrations		Mass Transport		Concentration	Transport
		Flow cubic ft/s	Temp. °C	Cond. (25°) μmhos	Total Phosphorus μg/l	Ortho-Phosphorus μg/l	Total Phosphorus mg/s	Ortho-Phosphorus mg/s	Bioavailable Phosphorus μg/l	Bioavailable Phosphorus mg/s
	Preston WWTP Sec. clarifier (8/17/87)	1.3	17.0	608.0	2520.	2300.			1320.	48
Worm Creek	West of Franklin, ID	0.0	25.0	457.8	274.	129.	0.	0.	134.	0
	Richmond Lagoon effleunt	0.0(+)	26.0	681.6	2800.				2080.	
	White's Trout Farm effluent	31.0	17.0	446.4	-	-			< 8	
	Hyrum WWTP effluent	1.6	18.5	806.6	5190.	4890.	228.	215.	5160.	227
	Hwy 101 crossing L. Bear R.	6.0	18.0	536.2	53.	27.	9.	5.		
	Wellsville Lagoon effluent	0.1	20.0	568.0	1130.	862.	4.	3.	1130.	4
Little Bear River	At 4400 S. Crossing	111.8	18.0	556.6	127.	54.	402.	171.		0
Spring Creek	At 4400 S. Crossing	-	22.0	502.0	376.	184.				
	Irrigation ditch to Spring Cr. at 4400 S.	-	24.0	548.7	344.	224.				
Little Bear River	Benson Marina	383.8	26.0	429.4	266.	55.	2891.	598.		
Little Bear River	At Hwy 30 crossing	-	24.0	468.0	221.	56.				
	Logan Lagoon effluent +irrigation water	0.8	24.0		1960.	1700.	43.	37.	2860.	62
Logan River	Below Logan Lagoon effluent	-	27.0		821.	14.				

(+) 0.034 cfs but doesn't reach river

Table A-4. Data from July 12, 15, 16, 1987, (summer) sampling of the Bear River and its tributaries.

| Stream | Sample Location | River mile upstream from Honeyville site | Field Measurements | | | Concentrations | | Mass Transport | | Concentration | Mass Transport |
			Flow cubic ft/s	Temp. °C	Cond. (25°) μmhos	Total Phosphorus μg/l	Ortho-Phosphorus μg/l	Total Phosphorus mg/s	Ortho-Phosphorus mg/s	Bioavailable Phosphorus μg/L	Bioavailable Phosphorus mg/s
Bear River	Bear Lake Outlet Canal at US-89	192.9	696.0	21	695	109.	15.	2148.	290.		
Bear River	Pescadero at USGS gauge	182.6	976.7	21	696	103.	15.	2849.	407.		
Bear River	Above Soda Point Res. USGS gauge	153.8	940.0	21	672	129.	13.	3434.	349.		
Bear River	At Grace Dam UPL gauge	138.3	0.0	20	698	64.	13.	0.	0.		
Bear River	Above Alder Creek at Bridge	125	-	21	777	100.	42.				
Cottonwood Creek	Near Cleveland, ID at I-34		0.0	(no sample taken)							
Bear River	Above Oneida Reservoir	109	904.9	21	768	113.	20.	2896.	500.		
Bear River	Below Oneida Reservoir UPL gage	104.2	2580.0	19	775	64.	7.	4705.	490.		
Mink Creek	Near Bear R. confluence		5.3	17.5	451	64.	47.	10.	7.		
Bear River	Above Battle Creek at US-91	86.8	1213.7	19	698	58.	16.	1990.	560.		
Battle Creek	Near Bear R. confluence		5.4	19	1163	624.	109.	96.	17.		
Deep Creek	Near Bear R. confluence		34.8	19	858	122.	40.	120.	40.		
Five-mile Creek	Near Bear R. confluence		9.5	17	795	164.	110.	44.	30.		
Weston Creek	Near Bear R. confluence		8.2	20.5	1079	81.	21.	19.	5.		
Bear River	West of Fairview, ID at USGS gauge	72.4	421.4	19	909	100.	20.	1193.	235.	<8	<95
Bear River	West of Richmond, UT	60	-	22	859	135.	15.				
Bear River	Above Cutler Reservoir, West of Benson	37.6	1049.3	19	808	130.	18.	3863.	541.	<8	<238
Bear River	Below Cutler Reservoir at UPL gauge	26.8	2.4	19	1443	131.	14.	9.	1.	<8	<1
West Wheelon Canal	Below Cutler Reservoir	26.8	651.0					0.	0.		
East Wheelon Canal	Below Cutler Reservoir	26.8	125.0					0.	0.		
Bear River	West of Deweyville at Bridge	11.1	8.3	26	881	103.	15.	24.	3.		
Bear River	West of Honeyville at Bridge	0	140.8	26	1219	196.	16.	782.	65.		
Cub River	North of Franklin, ID at Bridge		-	22	418	61.	20.				
Cub River	West of Franklin, ID at Bridge		5.0	21	585	354.	228.	50.	32.		
Worm Creek	West of Franklin, ID		0.0	22	482	296.	80.	0.	0.		
High Creek	At US-89/91		0.0	(Creek was dry)							
Cub River	North of Richmond at Bridge		-	21	553	229.	51.				
Cub River	South of Ricmond at Bridge		25.0	22	517	148.	15.	105.	10.		
Logan River	At 6th. South		113.3								
Logan River	Logan Lagoon Effluent		0.0								
Logan River	Below Logan Lagoon Effluent		-	23	442	490.	14.			11.	
Little Bear River	So. Fork Below Three mile Cr. at Bridge		-	16	303	29.	13.				
Davenport Creek	At road abv. confl. with L. Bear R.		17.3	18	350	29.	7.	14.	3.		
Little Bear River	So. Fork Below Davenport Cr., USGS ga.		31.3	19	386	29.	7.				
Little Bear River	Below Hyrum Reservoir, Hwy 101		26.8	20	533	106.	68.	80.	52.	5.	4.
Little Bear River	Above Logan R. confluence, E. Mendon		158.0	17	577	137.	66.	613.	294.	5.	22.
Little Bear River	Benson Marina		163.7	21.5	541	458.	211.	2123.	978.	10.	46.
Mill Creek	Near Blacksmith Fork confluence		11.2	16	454	50.	30.	16.	9.		
Sheep Creek	Near Blacksmith Fork confluence		11.0	15	364	26.	8.	8.	3.		
Blacksmith Fork R.	Above Sheep Creek		77.0	11	406	29.	11.	62.	24.		
Blacksmith Fork R.	At Anderson Ranch		88.0								
Blacksmith Fork R.	At USGS gauge		96.2	14	380	29.	8.	78.	22.		

Table A-5. Data from August 17, 1987, sampling of selected location on streams in Cache Valley.

Stream	Sample Location	River mile upstream from Honeyville site	Field Measurements			Concentrations		Mass Transport		Concentration Bioavaible Phosphorus μg/l	Mass Transport Bioavailable Phosphorus mg/s
			Flow cubic ft/s	Temp. °C	Cond. (25°) μmhos	Total Phosphorus μg/l	Ortho-Phosphorus μg/l	Total Phosphorus mg/s	Ortho-Phosphorus mg/s		
Bear River	Below Oneida Reservoir UPL gage	104.2	825	17	768	57.	15.	1332.	350.		
Bear River	West of Fairview, UT-ID border gage		300	17	842	67.	19.	569.	161.	<8	<68
Bear River	Above Cutler Reservoir, West of Benson	37.6	808	20	841	124.	16.	2838.	366.		
Bear River	Below Cutler Reservoir UPL gage	26.8	3	26	1676	114.	15.	8.	1.		
West Side Canal	Below Cutler Reservoir	26.8	131	25	790	163.	12.	605.	45.	17.	63.
East Side Canal	Below Cutler Reservoir	26.8	603	19	816	169.	14.	2886.	239.		
Cub River	South of Richmond at Bridge		5	19	551	128.	17.	17.	2.	12.	2.
Logan River	At 6th South gauge		129	18	469	37.	14.	135.	51.		
Logan River	Irrigat. ditch into Logan R. at 6th South		-	19	384	27.	14.				
Logan River	Outlet from small lake at 200 N.		-	22	442	98.	27.				
Logan River	Logan Lagoon Effluent		0								
Logan River	Below Logan Lagoon Effluent		-	25	458	425.	15.				
Little Bear River	Above Logan R. confluence, E. Mendon		295	17	567	166.	82.	1387.	685.		
Spring Creek	Spring Cr. at 6th South		-	17	553	332.	289.			83.	
Little Bear River	Irrigat. ditch entering Spring Cr., 6th S.		-	24	702	280.	211.				
Little Bear River	Benson Marina		243	22	552	336.	130.	2312.	895.	122.	840.

Table A-6. Data from October 3, 4, 1987, sampling in and around Cutler Reservoir.

Stream	Sample Location	Field Measurements			Concentrations		Mass Transport		Concentration	Mass Transport
		Flow cubic ft/s	Temp. °C	Cond. (25°) μmhos	Total Phosphorus μg/l	Ortho-Phosphorus μg/l	Total Phosphorus mg/s	Ortho-Phosphorus mg/s	Bioavailable Phosphorus μg/L	Bioavailable Phosphorus mg/s
Bear River	Above Cutler Reservoir, West of Benson	1291	14	882	115	15.6	4204.1	570		
Bear River	Below Cutler Reservoir UPL gage	1590	16	902	147	21.6	6619.2	973	<8	<360
Bear River	East Wheelon Canal (flow only)	75								
Bear River	West Wheelon Canal (flow only)	491								
Little Bear River	At 6th South	141	14	562	98	34.3	392.5	137		
Spring Creek	At 8th South	76	13	550	318	257	684.4	553		
Little Bear River	Valley View Highway	299	16	514	237	147	2006.8	1245	48	406
Little Bear River	Benson Marina	67	15	589	231	59.7	438.0	113	67	127
Logan River	Below Logan Lagoon	--	16	472	271	20.9			63	
Logan River	Logan R., 6th South (flow only)	124								
Logan River	Logan Lagoon effluent	17	16	634	2192	102	1055.3	49		
Logan River	Irrigation ditch near Lagoon effluent	1	12	518	95	44.8	1.3	1		
Within Cutler Res.	41° 49.59' North; 111° 57.5' West	887	16	837	196	21.6	4923.5	543	60	1507
Within Cutler Res.	Railroad bridge	808	17	837	159	15.6	3638.3	357	<8	<183
Within Cutler Res.	41° 51.38' North; 112° 2.18' West	515	17	887	150	15.6	2186.0	227	<8	<117

Table A-7. Data from October 20, 21, 24, 1987, (fall) sampling of the Bear River and its tributaries.

Stream	Sample Location	River mile upstream from Honeyville site	Field Measurements Flow cubic ft/s	Temp. °C	Cond. (25°) μmhos	Concentrations NO3-N+ NO2-N mg/l	Total Phosphorus μg/l	Ortho- Phosphorus μg/l	Mass Transport Total Phosphorus mg/s	Ortho- Phosphorus mg/s	Concentration Bioavailable Phosphorus μg/L	Mass Transport Bioavailable Phosphorus mg/s
Bear River	Bear Lake Outlet Canal at US-89	192.9	0.0	9	721	<0.04	30.1	7.3	0.00	0.00		
Bear River	Pescadero at USGS gauge	182.6	100.5	9	712	0.29	34.6	7.3	98.48	20.78		
Bear River	Above Soda Point Res. USGS gauge	153.8	160.0	9	569	0.43	21.2	8.7	96.06	39.42		
Bear River	At Grace Dam UPL gauge	138.3	-	9	845	0.73	96.2	19.1	-	-		
Bear River	Above Alder Creek at Bridge	125	-	12	893	1.06	42.3	22.1	-	-		
Cottonwood Creek	Near Cleveland, ID at I-34		dry	-	-	-	-	-	-	-		
Bear River	Above Oneida Reservoir	109	150.7	11	807	1.02	43.5	14.7	185.65	62.74		
Bear River	Below Oneida Reservoir UPL gage	104.2	36.1	9	815	0.42	35.7	19.1	36.50	19.53		
Mink Creek	Near Bear R. confluence		4.9	4	482	0.41	63.5	50.3	8.81	6.98		
Bear River	Above Battle Creek at US-91	86.8	53.5	7	845	<0.04	25.1	14.7	38.02	22.26		
Battle Creek	Near Bear R. confluence		5.6	3.5	1662	1.31	341	51.7	54.30	8.23		
Deep Creek	Near Bear R. confluence		12.7	5.5	1124	2.34	116	71	41.72	25.53		
Five-mile Creek	Near Bear R. confluence		10.8	6	886	2.18	19.7	75.5	6.02	23.06		
Weston Creek	Near Bear R. confluence		9.1	6	1468	3.58	37.9	16.2	9.78	4.18		
Bear River	West of Fairview, ID at USGS gauge	72.4	119.0	9.5	1606	0.74	42.3	19.1	142.55	64.37	<8	<27
Bear River	West of Richmond, UT	60	-	9	1685	0.7	54	19.1	-	-		
Bear River	Above Cutler Reservoir, West of Benson	37.6	284.1	10	801	0.74	79.6	14.7	640.42	118.27	<8	<64
Bear River	Below Cutler Reservoir at UPL gauge	26.8	14.0	-	-	-	-	-	-	-		
West Wheelon Canal	Below Cutler Reservoir	26.8	202.0	-	819	0.37	126	20.6	720.80	117.85	<8	<46
East Wheelon Canal	Below Cutler Reservoir	26.8	45.4	-	-	-	-	-	-	-		
Bear River	West of Deweyville at Bridge	11.1	19.8	-	794	0.31	90.7	13.2	50.86	7.40		
Bear River	West of Honeyville at Bridge	0	188.9	-	877	0.19	113	14.7	604.41	78.63		
Cub River	North of Franklin, ID at Bridge		-	5.5	456	0.36	29.6	16.2	-	-		
Cub River	West of Franklin, ID at Bridge		344.8	5.5	439	0.36	20.1	13.2	196.25	128.88		
Worm Creek	West of Franklin, ID		3.4	6	529	0.09	26.8	14.7	2.59	1.42		
High Creek	At US-89/91		dry	-	-	-	-	-	-	-		
Cub River	North of Richmond at Bridge		-	-	497	0.8	140	19.1	-	-		
Cub River	South of Ricmond at Bridge		5.5	10	491	0.67	86	20.6	13.41	3.21	<8	<1
Logan River	Logan Lagoon (flow only)		-			-						
Logan River	Below Logan Lagoon Effluent		-	11.5	438	<0.04	191	23.6	-	-		
Logan River	At 6th South (flow only)		212.0			-						
Little Bear River	So. Fork Below Three Mile Cr. at Bridge		-	10	321	0.11	14.6	13.2	-	-		
Davenport Creek	At road abv. confl. with L. Bear R.		8.6	10.5	397	0.06	7.9	8.7	1.92	2.12		
Little Bear River	So. Fork Below Davenport Cr.		23.9	10.5	418	0.1	11.5	8	7.78	5.41	<8	<5
Little Bear River	Below Hyrum Reservoir, Hwy 101		11.2	-	596	0.16	40.1	32.5	12.74	10.33		
Little Bear River	Above Logan R. confluence, 8th S.		42.4	-	593	0.9	63.5	35.4	76.25	42.51		
Spring Creek	At 8th South		74.1	-	556		358	329				
Little Bear River	Benson Marina		351.2	12	521	0.37	242	157	2407.00	1562.00	33	328
Mill Creek	Near Blacksmith Fork confluence		-	8.5	471	0.21	45.1	29.5	-	-		
Sheep Creek	Near Blacksmith Fork confluence		0.6	6	406	0.41	6.22	14.7	0.10	0.24		
Blacksmith Fork R.	Above Sheep Cr.		-	8.5	404	0.38	11.8	14.7	-	-		
Blacksmith Fork R.	At Anderson Ranch (flow only)		124.5			-						
Blacksmith Fork R.	At USGS gauge		52.0	8.5	393	0.31	9.6	10.2	14.14	15.02	<8	<12

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Table A-8. Data from February 20, 22, 23, 1988, (winter) sampling of the Bear River and its tributaries.

Stream	Sample Location	River mile upstream from Honeyville site	Field Measurements Flow cubic ft/s	Temp. °C	Cond. (25°) µmhos	Concentrations NO3-N+ NO2-N mg/l	Total Phosphorus µg/l	Ortho-Phosphorus µg/l	Mass Transport Total Phosphorus mg/s	Ortho-Phosphorus mg/s	Concentration Bioavailable Phosphorus µg/L	Mass Transport Bioavailable Phosphorus mg/s
Bear River	Bear Lake Outlet Canal at US-89	192.9	-	1.0	832	0.36	25.7	7.8	-	-		
Bear River	Pescadero at USGS gauge	182.6	274	0.5	687	1.34	31.5	10.8	244.4	83.8		
Bear River	Above Soda Point Res. USGS gauge	153.8	247	1.0	614	0.8	51.6	13.8	360.9	96.5		
Bear River	At Grace Dam UPL gauge	138.3	-	2.0	874	1.29	37.2	25.8	-	-		
Bear River	Above Alder Creek at Bridge	125	-	6.0	765	1.18	138	101				
Cottonwood Creek	Near Cleveland, ID at I-34		11.1	6.0	322	0.09	17	9.3	5.3	2.9		
Bear River	Above Oneida Reservoir	109	100.9	6.0	804	1.29	94.9	37.8	271.3	108.1		
Bear River	Below Oneida Reservoir UPL gage	104.2	1860.0	3.0	919	1.31	86.3	39.3	4545.9	2070.1		
Mink Creek	Near Bear R. confluence		15.0	1.0	443	0.86	60.3	18.3	25.6	7.8		
Bear River	Above Battle Creek at US-91	86.8	872.8	3.0	808	0.84	51.6	21.3	1275.5	526.5		
Battle Creek	Near Bear R. confluence		16.9	1.0	1497	2.22	1370	161	654.0	76.9		
Deep Creek	Near Bear R. confluence		54.0	2.0	994	1.52	865	101	1322.5	154.4		
Five-mile Creek	Near Bear R. confluence		11.7	3.0	859	2.42	187	107	62.0	35.5		
Weston Creek	Near Bear R. confluence		21.5	4.0	1456	3.51	441	46.8	268.3	28.5		
Bear River	West of Fairview, ID at USGS gauge	72.4	623.0	3.2	1452	1.51	199	59.6	3511.0	1051.5	< 8	< 141
Bear River	West of Richmond, UT	60	-	5.0	1077	1.34	132	43.8	-	-		
Bear River	Above Cutler Reservoir, West of Benson	37.6	ice	1.0	-	1.38	124	48.3	-	-	< 8	-
Bear River	Below Cutler Reservoir at UPL gauge	26.8	2410	6.0	798	0.87	187	105	12763.0	7166.4	< 8	< 546
West Wheelon Canal	Below Cutler Reservoir	26.8	-	-	-	-	-	-	-	-		
East Wheelon Canal	Below Cutler Reservoir	26.8	-	-	-	-	-	-	-	-		
Bear River	West of Deweyville at Bridge	11.1	1527.6	2.5	897	0.97	179	91.9	7743.6	3975.6		
Bear River	West of Honeyville at Bridge	0	-	0.0	888	1.02	150	91.9	-	-		
Cub River	North of Franklin, ID at Bridge		-	4.0	459	0.63	83.4	31.8	-	-		
Cub River	West of Franklin, ID at Bridge		-	2.0	468	0.78	69	33.3	-	-		
Worm Creek	West of Franklin, ID		-	3.0	713	3.12	658	575	-	-		
High Creek	At US-89/91		-	-	-	-	-	-	-	-		
Cub River	North of Richmond at Bridge		-	3.0	544	1.63	322	161	-	-		
Cub River	South of Ricmond at Bridge		-	4.0	555	1.66	213	153	-	-		
Logan River	Logan Lagoon (flow only)		0	2.0	-	-	-	-				
Logan River	Below Logan Lagoon Effluent		-	5.5	598	0.22	1180	979	-	-		
Logan River	At 6th South (flow only)		269.6	6.0	414	0.29	31.5	9.3	240.5	71.0		
Little Bear River	So. Fork Below Three Mile Cr. at Bridge		-	7.0	309	0.16	34.3	15.3	-	-		
Davenport Creek	At road abv. confl. with L. Bear R.		20.3	0.0	393	0.3	54.5	7.8	31.3	4.5		
Little Bear River	So. Fork Below Davenport Cr.		23.6	5.0	394	0.19	34.3	9.3	23.0	6.2		
Little Bear River	Below Hyrum Reservoir, Hwy 101		9.1	6.0	546	0.48	63.2	22.8	16.3	5.9		
Little Bear River	Above Logan R. confluence, 8th S.		153.3	5.0	568	0.88	118	52.8	512.0	228.9		
Spring Creek	At 8th South		-	-	-	-	-	-	-	-		
Little Bear River	Benson Marina		902.0	3.0	484	0.41	196	125	5006.7	3193.1	62	1584
Mill Creek	Near Blacksmith Fork confluence		62.0	-	445	0.29	51.6	25.8	90.6	45.3		
Sheep Creek	Near Blacksmith Fork confluence		7.8	1.5	402	0.45	25.7	7.8	5.7	1.7		
Blacksmith Fork R.	Above Sheep Cr.		99.2	6.2	396	0.32	28.6	9.3	80.3	26.1		
Blacksmith Fork R.	At Anderson Ranch (flow only)		107.0	5.2	-	-	-	-	-	-		
Blacksmith Fork R.	At USGS gauge		45.0	6.0	393	0.33	28.6	9.3	36.4	11.9		

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Table A-9. Data from February 20, 22, 23, 1988, sampling of point sources and other waters in Cache Valley.

| Stream | Sample Location | River mile upstream from Honeyville site | Field Measurements | | | Concentrations | | | Mass Transport | | Concentration | Mass Transport |
			Flow cubic ft/s	Temp. °C	Cond. (25°) μmhos	NO3-N+ NO2-N mg/l	Total Phosphorus μg/l	Ortho-Phosphorus μg/l	Total Phosphorus mg/s	Ortho-Phosphorus mg/s	Bioavailable Phosphorus μg/L	Bioavailable Phosphorus mg/s
(Miscellaneous	White's Trout Farm influent		40.0	3.5	464	0.51	29	12	32.4	13.9		
Samples)	White's Trout Farm effluent		25.0	3.5	451	0.49	57	23	40.6	16.1	< 8	
	Irr. ditch above Hyrum outfall		-	3.0	1222	<0.04	2334	1925				
	Hyrum WWTP effluent		1.16	6.0	1365	1	6921	7002	227.4	230.0	4180	137
	Groundwater well 6th west-airport		-	18.0	508	2.18	20	20				
	Preston WWTP secondary clarifier		0.11	-	648	3.6	3832	3291	11.9	10.3	2500	8
	Logan Lagoons effluent		-	2	697	0.2	3168	2528			1710	

Table A-10. Data from May 19, 20, 21, 1988, (late spring) sampling of the Bear River and its tributaries.

| Stream | Sample Location | River mile upstream from Honeyville site | Field Measurements | | | Concentration Ortho-Phosphorus μg/l | Mass Transport Ortho-Phosphorus mg/s |
			Flow cubic ft/s	Temp. °C	Cond. (25°) μmhos		
Bear River	Bear Lake Outlet Canal at US-89	192.9	-	14.0	627	6	
Bear River	Pescadero at USGS gauge	182.6	431.7	13.5	613	11	134
Bear River	Above Soda Point Res. USGS gauge	153.8	-	-	592	29	
Bear River	At Grace Dam UPL gauge	138.3	<10	14	596	6	
Bear River	Above Alder Creek at Bridge	125	-	16	627	0	
Cottonwood Creek	Near Cleveland, ID at I-34		19.62	-	265	7	4
Bear River	Above Oneida Reservoir	109	625.80	14.5	613	11	195
Bear River	Below Oneida Reservoir UPL gage	104.2	212.00	13	641	10	60
Mink Creek	Near Bear R. confluence		53.58	9	258	28	42
Bear River	Above Battle Creek at US-91	86.8	647.70	15	620	8	147
Battle Creek	Near Bear R. confluence		19.40	13	1254	48	26
Deep Creek	Near Bear R. confluence		47.46	14	976	61	82
Five-mile Creek	Near Bear R. confluence		8.17	15	711	132	31
Weston Creek	Near Bear R. confluence		3.74	17	948	8	1
Bear River	West of Fairview, ID at USGS gauge	72.4	862.00	15	655	7	171
Bear River	West of Richmond, UT	60	-	18	659	10	
Bear River	Above Cutler Reservoir, West of Benson	37.6	1056.37	17	766	15	449
Bear River	Below Cutler Reservoir at UPL gauge	26.8	3680.00	17	564	26	2710
West Wheelon Canal	Below Cutler Reservoir	26.8	616.00	-	-	-	
East Wheelon Canal	Below Cutler Reservoir	26.8	115.00	-	-	-	
Bear River	West of Deweyville at Bridge	11.1	44.64	18.5	976	9	11
Bear River	West of Honeyville at Bridge	0	88.20	18	1087	8	20
Cub River	North of Franklin, ID at Bridge		-	15	251	19	
Cub River	West of Franklin, ID at Bridge		119.00	13	265	27	91
Worm Creek	West of Franklin, ID		10.44	14	362	182	54
High Creek	At US-89/91		-	11	258	18	
Cub River	North of Richmond at Bridge		-	16	300	51	
Cub River	South of Ricmond at Bridge		-	13	307	41	
Logan River	Logan Lagoon (flow only)		-		648	97	
Logan River	Below Logan Lagoon Effluent		-	22	309	14	
Logan River	At 6th South (flow only)		414.54	-	-	-	
Little Bear River	So. Fork Below Three Mile Cr. at Bridge		-	11	188	11	
Davenport Creek	At road abv. confl. with L. Bear R.		74.24	10	184	9	19
Little Bear River	So. Fork Below Davenport Cr.		89.80	10	209	8	20
Little Bear River	Below Hyrum Reservoir, Hwy 101		37.74	18	348	11	12
Little Bear River	Above Logan R. confluence, 8th S.		70.76	17.5	474	25	50
Spring Creek	At 8th South		84.70	16	488	103	247
Little Bear River	Benson Marina		-	21	390	106	
Mill Creek	Near Blacksmith Fork confluence		-	10	365	24	
Sheep Creek	Near Blacksmith Fork confluence		2.16	7	334	6	0
Blacksmith Fork R.	Above Sheep Cr.		-	9	348	8	
Blacksmith Fork R.	At Anderson Ranch (flow only)		131.12	-	-	-	
Blacksmith Fork R.	At USGS gauge		56.00	8	334	7	11