SUSITNA HYDROELECTRIC PROJECT

WATER QUALITY EFFECTS RESULTING FROM IMPOUNDMENT OF THE SUSITNA RIVER

DECEMBER 1982

PREPARED BY:

R&M CONSULTANTS, INC.

PREPARED FOR:

ACRES

ALASKA POWER AUTHORITY

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TASK 3 - HYDROLOGY

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TABLE OF CONTENTS

Page

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1 - INTRODUCTION

The characterization of stream water quality is of vital importance in making sound watershed management decisions. An assessment of the physical and chemical quality of Susitna River water and the potential impacts resulting from its impoundment are discussed in this report.

If the Susitna Hydroelectric Project is developed, the Watana Dam will be 860 feet high and will retain 9.6 million acre-feet of water at full storage. The area of the reservoir at the maxium operating level will be 38,000 acres. The Devil Canyon Reservoir will be smaller, covering 7,800 acres at the maximum operating level, with 1.1 million acre-feet at full storage. The Devil Canyon Dam will be 650 feet high. Either multi-level outlets or single outlets at a depth no greater than 120 feet will be used. Consequently, neither dam will have an outlet near the reservoir bottom.

2 - IMPOUNDMENT EFFECTS

The environmental conditions in an impoundment differ from those in a flowing stream in many ways, including change in water depth, increased detention time, and the possibility of stratification, eutrophication, increased evaporation, sedimentation, and leaching. Each of these has an influence on the normal physical, chemical, and biologic processes that cause changes in water quality. Additional detention time allows natural processes to proceed far beyond the extent feasible in a flowing stream. If stratification exists, the bottom portion of the water (hypolimnion) is trapped and does not contact aerated water or the atmosphere, thereby having a marked effect on the natural processes that occur in water, and leading to creation of poor-quality water. Leaching also results in poor quality of the hypolimnion. Eutrophication leads to algal blooms and nuisance conditions throughout an impoundment, and evaporation concentrates the dissolved fractions of minerals in water.

2.1 - Impoundment Processes and Interactions

The impoundment processes (stratification, eutrophication, evaporation, sedimentation, leaching, and ice cover) are defined in this section, and a summary of their interactions is presented. The degree to which each of these processes is important in a particular reservoir is variable, depending on geographical location, climate, hydrologic regime, allocthonous nutrient inputs, lake morphometry, biotic community activity, and inorganic sediment inputs, as well as other parameters.

2.1.1 - Processes

Stratification. Stratification is a layering of water because of density differences, which can be caused by temperature or sediment load. Stratification occurs in the summer due to the warming of the surface water by short and long-wave radiation, by conduction, and by advection (Roesner, et al. 1971). Winter stratification can occur in cold regions because 0°C water at the surface is lighter than the warmer water below (Kittrel, 1965). Winter stratification is not as stable as summer stratification, which is stable with cold, dense water at the bottom and warmer, lighter water at the top, with very little vertical mixing. The top layer of the reservoir (the epilimnion) is of nearly uniform temperature, with the region of changing temperature below the epilimnion called the thermocline, and the bottom layer the hypolimnion (Symons, 1969). Only weak thermal stratification has been observed in glacial lakes (R&M, 1982b). Seasonal water temperature fluctuations coupled with environmental factors such as wind velocity, direction and duration, and the geometry of the reservoir basin create internal flow patterns that influence water quality. Several distinct and independent currents may exist simultaneously within a stratified reservoir. When river water enters an impoundment it mixes and descends to a depth at which the inflowing water has the same temperature or density as the resident water. Depending on the reservoir's temperature profile, inflowing water may travel and remain as a surface-overflow current, an intermediate-interflow current or a bottom-underflow current. Reservoirs discharging water from the epilimnion encourage the development of surface-type currents (Turkheim, 1975).

Love (1961) noted seasonal temperature gradient variations and related them to seasonal salinity distribution and circulation patterns in a reservoir he studied. At the time of spring runoff, the temperature of inflowing water was about the same as the surface of the reservoir and sediments settled out rather quickly. Thus, in spring, inflow proceeds into the reservoir as an overflow current. With the onset of stratification, a vertical gradient of temperature and salinity became established. Flow along the surface created cellular circulation below the 150-foot level causing an upstream flow along the bottom. In summer, inflow decreases in volume, salinity increases, and river water proceeds as an interflow at a depth of about 80 feet. In the fall, cool river temperatures cause greater sinking of inflow water along the bottom until it comes in contact with heavier layers, then spreads horizontally. As reservoir surface waters become cooler than deeper waters in the fall, vertical convection currents are created, mixing reservoir waters until an isothermal state is achieved (Smith and Justice, 1976).

Eutrophication. Eutrophication is a term meaning enrichment of waters by nutrients, either man-induced or through natural means (Mackenthun, 1969). Phosphorus and nitrogen are the fertilizing elements most responsible for lake eutrophication. However, carbon and silica are important nutrients in some systems. Iron and other trace elements may also be important to a limited extent.

Evaporation. The major effect of evaporation on water quality is the resulting higher concentration of dissolved substances (Symons, 1969; Love, 1961). In Alaska, cool temperatures and abundant water indicate that evaporation may not be critical. Sublimation from ice and snow, however, may cause significant water loss (Smith and Justice, 1976).

Sedimentation. The quiescent conditions in impoundments indicate that sedimentation will occur. The type of material that will settle is dependent on upstream conditions. The rate of settling is a function of particle size, shape, and density (Weiss et al., 1973).

Leaching. Leaching is the exchange of chemicals from an impoundment bottom to the water mass. The process of exchange is more rapid under reducing conditions than under oxidizing conditions (Mortimer, 1941, 1942).

Ice Cover. An ice cover has one direct effect and some indirect effects on impoundment water quality. The direct effect was noted by Mortimer (1941, 1942), who discussed the increased concentration of solutes just below the ice. As water freezes, the dissolved solids are exuded from the ice and concentrated. The indirect effects include (1) the prevention of atmospheric reaeration, (2) stratification, and (3) a reduction in light penetration after snow covers the ice. In addition, long periods of ice and snow cover will prevent the addition of allocthonus nutrients to the reservoir (Ryder, 1978).

2.1.2 - Interactions

Each of the six processes defined above interact with one another in impoundments. The cumulative effect of these interactions on water quality is usually to further degrade it. The process interactions described below are from Smith and Justice (1976) unless otherwise indicated.

Stratification-Leaching. In a stratified reservoir, no vertical mixing occurs between the epilimnion and the hypolimnion; thus, no oxygen is transferred to the lower water. If anaerobic conditions result, the redox potential will decrease, and leaching rates will increase.

Stratification-Sedimentation. Stratification causes inflowing water to enter at a depth with equal density, thereby controlling the distance the sediment load has to fall before being effectively removed from the incoming water. In some cases, stratification determines whether or not the suspended material will be removed at all. Also, loss of sediment reduces the water density, which can affect stratification.

Stratification-Evaporation. Stratification increases evaporation because the warm, less dense water remains near the surface. On the other hand, surface cooling by evaporation and heat loss can cause convective currents if the heat loss is greater than the energy added by the sun's radiation. The convective currents keep the epilimnion isothermal and mixed.

Stratification-Eutrophication. Nutrient-rich hypolimnion water is prevented from mixing with surface water, thus controlling algae growth if the concentration of the limiting nutrient is controlled.

Eutrophication-Leaching. The dying and settling of algae adds organic matter to the bottom. Upon degradation, the organic matter depletes oxygen and releases chemicals. Nutrients are released by the leaching of detritus material. If the nutrients are transported to the surface, algae growth may be stimulated.

Eutrophication-Sedimentation. Dead algae settle to the bottom. Settling of dead algae and of some precipitates removes nutrients from the zone of algae growth. Increased light penetration due to turbidity removal can stimulate algae growth.

Eutrophication-Evaporation. Active algae growths at the water surface cause an increase in evaporation rates.

Sedimentation-Leaching. Settling of inorganic material will reduce leaching by covering or diluting organic deposits. If the settled material is high in organic content, anoxic conditions will increase, and thus leaching will be favored.

Ice Cover-Evaporation. Water loss by evaporation will be reduced if an ice cover exists, but sublimation will still occur and will remove some of the ice and snow cover.

Ice Cover-Eutrophication. The decreased light penetration due to snow and ice cover will slow the growth of algae. However, rapid algae growth has been observed under ice cover. The intensity and distribution of solar radiation penetrating an ice covered lake depends on the reflectance, light-scattering and absorptive-optical properties of the ice sheet and the water column. In addition, the geographical location of the lake controls the duration and elevation of solar radiation received during different seasons. Climatic factors which are peculiar to individual water bodies also influence the stratigraphy and duration of an ice cover (Adams, 1978).

Ice Cover-Stratification. Winter stratification is protected from wind mixing by an ice cover. However, the frictional effect of surface ice on inflowing water may influence thermal-density water stratification and increase the rate of sediment deposition as the colder inflowing water descends and mixes with reservoir waters (Turkheim, 1975).

Ice Cover-Leaching. Winter reaeration can only come from advection of oxygen-rich water or through cracks in the ice cover. If anaerobic conditions develop, leaching will increase. The reduced eddy diffusion coefficient of water under ice inhibits mixing, causing a slower spread of nutrients released from bottom sediments and a shallower anaerobic region. Chemical and biological reactions are also inhibited in bottom sediments due to lower temperatures. However, the lack of atmospheric aeration in winter may lead to severe water quality problems if chemical or biological processes utilize all available dissolved oxygen.

Evaporation-Sedimentation. Loss of water by evaporation leaves the dissolved solids more concentrated, thus forcing precipitation reactions to the solid phase with possible settling of reaction products.

2.2 - Sedimentation/Turbidity

When a turbulent, sediment-laden stream such as the Susitna River enters a reservoir, the quiescent conditions will allow much of the material to settle to the bottom. Weiss et al. (1973) Wright and Soltero, (1973), Love (1961) and Churchill (1957), substantiate the reduction of turbidity by the impoundment of a sediment-laden river.

Turbidity caused by suspended sediments may have both beneficial and detrimental effects in northern reservoirs. High turbidity levels which prevent solar radiation penetration also restrict photosynthetic activity to a relatively shallow zone just below the air-water interface. As a consequence, nutrient rich lakes are rendered unproductive if turbidity levels remain high throughout the growing season (Ryder, 1978). Suspended sediments also have the capacity to bind nutrients and toxic pollutants, and remove them from the water column as they settle to the bottom. Furthermore, high turbidity levels limit flowing water aeration capacities and aquatic fauna reproduction by inhibiting the productivity of oxygen-producing organisms such as phytoplankton and rooted aquatic plants (Turkheim, 1975). Low turbidity levels, on the other hand, may either encourage phytoplankton productivity by increasing light penetration or kill photosynthetic algae if they are intolerant of elevated light conditions (Smith and Justice, 1976).

Due to the fact the turbidity is caused mainly by inorganic suspended solid loads, it can be used to trace the fate of river water flowing into a reservoir (St. John et al., 1976). The depth at which the inflowing water enters the reservoir will dictate the distance that suspended sediments must fall before being removed from the inflow. Temperature-density stratification will determine whether suspended solids will be removed at all (Smith and Justice, 1976). When the inflowing stream is laden with suspended sediments, the inflowing water may be heavier than resident water, thus causing it to move as an under-current near the bottom of the reservoir (Smith and Justice, 1976). The following seasonal turbidity pattern was noted in a glacial lake in Canada. Turbidity increased in early spring, limiting light penetration near the surface. A turbid intermediate layer developed in June indicating river water interflow. Turbidity remained low near the bottom throughout the summer but increased in September and October due to re-suspension of bottom sediments (St. John et. al., 1976). Vertical mixing may be responsible for the re-suspension of bottom sediments into the water column and a corresponding increase in turbidity (Carmack and Gray, 1982).

Color (Drachev, 1962), particularate phosphorus (Wright and Soltero, 1973), dead microorganisms such as plankton and algae (Erickson and Reynolds 1969), and precipitated chemicals (Mortimer; 1941, 1942) are removed in the sedimentation process.

According to reservoir sedimentation studies, 70-97 percent of sediment entering Watana Reservoir would settle, even shortly after filling of the reservoir starts (R&M Consultants, 1982b). The Devil Canyon Reservoir would have a slightly lower trap efficiency than Watana due to its smaller volume. However, most sediment will be deposited in Watana, the upstream reservoir. Turbidity levels and suspended solids concentrations downstream from the reservoir will decrease sharply from natural levels during the summer months due to the sediment trapping characteristics of the reservoirs. The turbidity of water released during winter will be substantially reduced from summer conditions, as suspended sediment in near-surface waters should rapidly settle once the reservoir ice cover forms and essentially quiescent conditions occur. However, it is possible that glacial flour that entered the reservoir during summer will pass through and out the reservoir during winter. If this occurs, the turbidity of water released during winter, although low, will be higher than pre-project levels.

2.3 - Metals

A reduction of metal concentrations such as iron, manganese, and trace elements occurs in reservoirs as these elements are precipitated and settle to the bottom (Neal, 1967). Oligotrophic lakes having a low pH and containing low dissolved salt concentrations generally contain higher concentrations of metals in surface waters than oligotrophic lakes having a high pH and high dissolved salt concentrations. The higher concentration of metals is due to the absence of dissolved salts which react with and precipitate metal ions. Since oligotrophic lakes are generally well aerated and low in productivity, there is little variation in metal content with depth relative to mesotrophic and eutrophic lakes. In eutrophic lakes, very high local concentrations of certain metals are often present as a result of an acidic or reducing environment near the bottom (Williams et al., 1976).

2.4 - Leaching

Anaerobic bottom conditions can harm aquatic life and cause the reduction and release of undesirable chemicals into the water (Fish, 1959). The leaching process, which is more efficient under anaerobic conditions, degrades bottom water quality by releasing such chemicals as alkalinity, iron, manganese (Symons et al., 1965), hydrogen sulfide, and nutrients (Turkheim, 1975). Also, leaching problems become more severe as the organic content in the soils increase. The potential for leaching at the Watana Reservoir should decline over time as the inorganic glacial sediment carried in by the river settles and blankets the reservoir bottom.

The products of leaching are not anticipated to be abundant enough to affect more than a small layer of water near the reservoir bottom. Also, leaching products will not degrade downstream water quality over the long-term because water will be released from the reservoir surface. A short-term increase in dissolved solids, conductivity, and most of the major ions may be evident immediately after closure of the dam. The magnitude of increase cannot be quantified with available data, but it is anticipated that the increase will not result in detrimental effects to freshwater aquatic organics. Bolke and Waddell (1975) report that the highest concentration of all major ions, except magnesium, occurred immediately after closure of the dam they were studying. They attributed the increase in concentration to the initial inundation and leaching of rocks and soils in the reservoir area. However, effects such as these are temporary and diminish as the reservoir matures (Baxter and Glaude, 1980).

2.5 - Heat Transfer/Evaporation

The four principle mechanisms of heat transfer in a reservoir include evaporation, convection, radiation loss, and solar radiation gain. It is possible to predict what temperature changes will occur in an impoundment before its construction, based on estimates of these variables. The addition of heat to an impoundment from solar radiation on large surface areas may reduce dissolved oxygen and increase evaporation and microbial activity unless the volume of warmed water is small in relation to the total reservoir volume (Love, 1961).

The range and seasonal variation in temperature of the Susitna River will change after impoundment. Bolke and Waddell (1975) noted in an impoundment study that the reservoir not only reduced the magnitude of variation in temperature but also changed the time period of the high and low temperature. This will also be the case for the Susitna River, where pre-project temperatures generally range from 0°C to 13°C with the lows occurring in October/November through March/April and the highs in July or August. After closure of the dam gates, the temperature range will be reduced and low temperatures will occur in November through March. The period of highest temperature will be July and August, as is the pre-project case. Reservoirs releasing water from the surface are "heat exporter" reservoirs (Turkheim, 1975), and both Susitna River reservoirs fall into this category. Post-project reservoir temperatures are discussed elsewhere by Acres (1982).

Thermal stratification is likely to occur in both reservoirs during summer and winter as a result of temperature density differences within the water column, although stratification is often relatively weak. Winter stratification would be less stable than summer stratification because the maximum temperature difference would be 4°C, the temperature of water at its maximum density. It is expected that vertical mixing will occur in the spring as a result of the large input of water, wind effects, and surface-water warming. During stratified conditions, vertical mixing is inhibited or eliminated. Thus, the transport of oxygen from the surface, where reaeration occurs, to the bottom, where biologic and chemical processes use oxygen, is severely inhibited.

Due to an aggradation process whereby reservoir water levels are increased in an upstream direction, a reservoir can increase the amount of evaporation from a river (Turkheim, 1975). However, the amount of evaporation from the river will be a small percentage of the total evaporation from the Watana and Devil Canyon reservoirs, and evaporation at these reservoirs will be minimal. The average annual evaporation predicted for May through September at Watana is 10.0 inches, and at Devil Canyon it is 11.1 inches. There is no evaporation during the period of ice cover, November through May. The percentage of the reservoirs' volume lost to evaporation during summer will be 0.3 percent at Watana and 0.6 percent at Devil Canyon. Although evaporation has been noted to cause an increase in dissolved solids concentrations in reservoirs (Love, 1961; Symons, 1969), the potential effect of a less-than 1 percent concentration increase is not significant. Sublimation may also cause some water loss, creating local effects, but this is not anticipated to be significant at the Watana or Devil Canyon impoundments.

In cold climates, reduced current velocities upstream from reservoirs favor earlier freezeup and later breakup than in unregulated rivers (Baxter and Glaude, 1980). Ice first forms in quiescent waters such as a reservoir, whereas the faster flowing reaches remain open initially. When reservoir ice accumulations at the reservoir inlet have sufficiently raised the water level and decreased the local water velocity, upstream rapids freeze and the ice cover builds further upstream. At breakup, ice jams may occur at the reservoir inlet, causing higher water stages upstream (Turkheim, 1975).

2.6 - Dissolved Oxygen

Many changes in the chemical constituents of an impoundment are related to oxygen concentrations of the water (Mortimer, 1941, 1942). Reservoirs decrease dissolved oxygen concentrations by increasing depth, decreasing turbulence and increasing surface temperatures, thus lowering oxygen saturation values (Smith and Justice, 1976). Slowing down of water transport by dams allows more time for biochemical oxygen demand to consume the available oxygen and also reduces the rate at which water is reaerated. If gas exchange at the air-water interface cannot supply enough oxygen to meet the respiratory demand, oxygen concentrations in reservoir waters become very low (Ruggles and Watt, 1975). When phytoplankton die or move out of the photosynthetic zone, oxygen is consumed as a result of algal degradation and respiration processes. These processes usually take place in the hypolimnion (Smith and Justice, 1976). If anaerobic conditions develop, the redox potential may decline, causing insoluble materials to be reduced to soluble states (Ingols and Wilroy, 1962). Smith and Justice (1976) note that carbonaceous biochemical oxygen demand and nitrification of ammonia caused anaerobic conditions in reservoirs. They further noted two impoundments which contained a zone of oxygen depletion just below the thermocline in addition to one near the bottom. The oxygen demand in the upper zone was attributed to 60-80 percent phytoplankton, 15-20 percent biochemical oxygen demand, and 1-10 percent fish respiration. St. John et al., (1976) report that decreases in summer dissolved oxygen levels in epilimnetic waters of Kamloops Lake, B.C. were a response to higher temperatures and corresponding decreases in oxygen solubility. Conversely, oxygen concentrations above saturation have been found at the surface of impoundments during summer as a result of the photosynthetic activity of phytoplankton (Smith and Justice, 1976). If surface waters become supersaturated with oxygen, some will be lost to the atmosphere (Symons et al., 1965). Churchill (1957) reports that warm water interflows remained near the surface of a reservoir where pollutants were degraded aerobically, decreasing dissolved oxygen. Such layers remain near the surface until fall when they enter the hypolimnion just prior to overturn. In at least one instance, cold, well-aerated water entered an impoundment as an underflow, forcing warmer low-oxygen water to the surface.

Water level fluctuations may cause low-oxygen problems at hydroelectric dams. During dry years with low water levels, lakeside vegetation may be prolific on exposed shores. When this organic mass is inundated during periods of high water, it exerts a high oxygen demand causing severe oxygen-related water quality problems in the reservoir (Ingols, 1959). High water levels may also cause low groundwater-oxygen levels (Turkheim, 1975).

In some cases a lake's trophic state can be related to areal oxygen depletion rates St. John et al. (1976). Eutrophic lakes are generally characterized by areal depletion rates greater than 1.5 mg. O_2/cm^2/month, which is sufficient to totally deplete oxygen in the hypolimnion. The areal depletion rate is calculated by obtaining the difference between the dissolved oxygen content of two samples from a known depth in the hypolimnion taken on two different occasions, and dividing the result by the time interval. However, there are two reasons why it may be misleading to relate the areal oxygen deficits to trophic states in all lakes. First, lakes with depths greater than 50 meters, as well as those which are narrow and steep sided, display high sediment surface areas relative to hypolimnion-ceiling surface areas. Dividing oxygen deficit values by the hypolimnion ceiling area results in an areal depletion rate that is higher than would be obtained if the sediment surface area were used. Second, the rate of allocthonous organic matter input from the drainage basin into the hypolimnion is unrelated to the production of organic matter in the epilimnion. Hence, decomposition of allocthonous materials in the upper zones increases the areal depletion rate near the bottom.

Another technique used by St. John et al. (1976) involved measuring the volumetric oxygen depletion rate of a lake. This was accomplished by dividing the difference between the average dissolved oxygen concentrations in the hypolimnion determined on two different occasions, by the time interval between them. The average concentrations were calculated in turn by adding the dissolved oxygen concentrations at 20 meter intervals in the hypolimnion and dividing the sum by the total volume of the measured intervals. The volumetric depletion rate was expressed in terms of milligrams per liter per day. It was noted, however, that if the movement of the river plume mixes oxygenated water with hypolimnetic water, the measured net depletion rate will considerably underestimate true absolute rates.

Oxygen demand in the Susitna River is typically low. Chemical oxygen demand levels measured in 1980 and 1981 at Vee Canyon have averaged 16 mg/l. Consequently, dissolved oxygen levels within the Susitna impoundments are anticipated to remain sufficiently high to support a diverse population of aquatic organisms.

2.7 - Trophic Effects

2.7.1 - Introduction

The process of eutrophication is defined as the increase in nutrient enrichment that causes increased productivity in lakes. This enrichment is expressed in terms of nutrient supply or load. Nutrient supply is the concentration of a nutrient per unit volume of water received by a lake, expressed in terms of mg/m^3. Nutrient load, on the other hand, is the concentration of a nutrient per unit of lake surface area, expressed in terms of mg/m^2.

Lake trophic status is an expression of the eutrophication process in a particular lake of a known mean depth, flushing rate, and nutrient retention capacity. The major characteristics used to quantify the trophic status of a water body are nutrient concentration, chlorophyll "a" concentration, and Secchi disc transparency. Phosphorus concentrations measured at spring overturn best represent the nutrient supply for plankton algae in the approaching growing season. The critical level of dissolved inorganic phosphorus is the level that, if exceeded, will produce nuisance blooms of algae. Chlorophyll "a" concentration is the best measure of algal biomass. Thus the spring phosphorus concentration in epilimnetic (surface) waters is an index of the trophic status of a lake. Trophic status can be generally classified as: (1) oligotrophic -- 0-10 mg phosphorus/m^3, (2) mesotrophic -- 10-20 mg phosphorus/m^3, and (3) eutrophic -- greater than 20 mg phosphorus/m^3. These conditions correspond to summer chlorophyll "a" concentrations of 0-2 mg/m^3, 2-6 mg/m^3, and greater than 6 mg/m^3, respectively, in clear water lakes. Secchi disc transparency is a convenient way of expressing the depth of light penetration in relation to algal biomass in epilimnetic waters. High Secchi disc transparencies reflect oligotrophic conditions whereas low transparencies may reflect eutrophic conditions. However, because Secchi disc transparencies also reflect high levels of turbidity and suspended solids concentrations in silt-laden waters, their use as trophic status indicators is limited to clear water lakes.

The mean depth of a lake is a convenient index of a lake's volume used to estimate the nutrient concentration from a given nutrient supply.

The flushing rate expresses the rate at which water is transported through a lake. This factor is important because it determines the period of time that nutrients will be available for use by algae.

Phosphorus retention is an expression of the fraction of phosphorus not lost through the outlet or by settling. Thus, the greater the retention factor, the greater the amount of phosphorus retained for use by algae.

The eutrophication process in reservoirs is similar to that in lakes. If water is released from a reservoir surface, the reservoir is a "nutrient trap" (Turkheim, 1975), much like a lake. Both Susitna River reservoirs will release water from at or near the surface. Hence, they can be expected to accumulate nutrients. However, the probability of eutrophic conditions developing in these reservoirs is not necessarily high because they are nutrient traps. The trophic status of the Watana and Devil Canyon reservoirs have been predicted to be oligotrophic on the basis of spring phosphorus concentrations derived from estimates of phosphorus supply, mean depth, flushing rate, and phosphorus retention capacity at each reservoir.

Chlorophyll "a" data are unavailable from the Susitna River at Vee Canyon. However, the high suspended sediment and turbidity levels in the Susitna River indicate that chlorophyll "a" values will be low. In addition, Secchi disc transparencies in the Susitna River will not accurately reflect the level of algal biomass that will result from impoundment. Consequently, the determination of chlorophyll "a" concentrations and Secchi disc transparencies resulting from impoundment have been disregarded in this study.

2.7.2 - Results

Results from the application of a trophic status model in the Susitna Hydroelectric Project are presented in this section. Spring phosphorus concentrations, in combination with the factors of mean depth, flushing rate, and phosphorus retention capacity, indicate that both Watana and Devil Canyon Reservoirs will be oligotrophic. A technical discussion of the various aspects of two nutrient models is presented in Attachment A.

Spring C:Si:N:P mass and atomic ratios were calculated for the Susitna Project by assuming a worst case dissolved orthophosphate value (0.01 mg/l) and by using values of total carbon, dissolved silicon and inorganic carbon measured in June at Vee Canyon. The average 1980-81 June C:S:N:P ratio at Vee Canyon was 1080:340:28:1 which corresponds to an average atomic ratio of 3000:403:68:1. Hence, from among the nutrients considered to be important to algal growth, it is apparent that phosphorus will be the limiting nutrient.

Successful use of the Dillon and Rigler (1975) equation depends on the accurate determination of a phosphorus retention coefficient for a particular water body. The reliability of determining a retention coefficient in glacially-influenced lakes in Alaska has not been established. Therefore, the use of this model is questionable in relation to the Susitna Project. Along these lines, St. John et al. (1976) found that Kamloops Lake in British Columbia had a measured phosphorus retention capacity 760 times greater than the predicted value using the Dillon and Rigler (1975) retention coefficient equation. As a result, they suggest that the use of this coefficient may lead to similar discrepancies in other glacially-influenced lakes. The only known application of a phosphorus model to glacially-influenced lakes in Alaska involves the use of Vollenweider's (1976) equation. More recently, Koenings and Kyle (1982) successfully utilized the Vollenweider equation in calculating the annual total phosphorus load to Crescent Lake in south-central Alaska. In this instance, the measured in-lake total phosphorus concentration at the time of spring overturn was used in determining an annual loading value. Theoretically, the same equation may be used to predict the spring phosphorus concentration in a lake with a known annual loading value. However, the reliability with which Vollenweider's model may be applied to Alaska lakes may be complicated by the fact that a high percentage of the total phosphorus load may be non-reactive in lakes which are fed by silt-laden glacial rivers.

As a result of the retention coefficient limitation in the Dillon and Rigler (1975) model, and the successful use of Vollenweider's model at Crescent Lake, the latter of the two models was selected for use in the Susitna Project. At this time there are no known limitations to the application of Vollenweider's model in Alaska which are not common to both models.

Application of the equation,

$$[P] = \frac{L}{\bar{z}\,p} \times \frac{1}{1 + \sqrt{1/p}}$$

to the Susitna Hydroelectric Project was made using the following rationale.

Natural Land Loading: According to Vollenweider (1976), the spring concentration of total phosphorus in a lake is the critical quantity used in evaluating a lake's trophic status. On this basis, Koenings and Kyle (1982) utilized the measured June phosphorus concentration in Crescent Lake to calculate an annual areal phosphorus loading value. Using this same logic, June phosphorus concentrations at Vee Canyon were used to project spring areal loading values at Watana and Devil Canyon. Loading values were in turn used in predicting the June phosphorus concentration in both reservoirs. In calculating spring areal phosphorus loads for the Susitna Project, dissolved orthophosphate is considered the form of the total phosphorus pool which is available for the use by microflora. The average 1980-1981 dissolved orthophosphate concentration measured at Vee Canyon during June was below the detectable limit (0.01 mg/l). However, the "worst case" value of 0.01 mg/l was assumed because it was felt that a value of zero was inappropriate for the Susitna Project area. Upon conversion of this value, the average June dissolved orthophosphate concentration becomes 10 mg/m^3. This concentration was multiplied by the average annual flow at each damsite (7.0×10^9 m^3/yr at Watana and 8.0×10^9 m^3/yr at Devil Canyon) to derive the phosphorus supply at each reservoir. Upon dividing the supply by the surface area of each reservoir (153,786,000 m^2 at Watana and 31,566,600 m^2 at Devil Canyon), June areal phosphorus loading from the land is obtained. The June natural land load to the surface at Watana is 456 mg/m^2, and at Devil Canyon is 2533 mg/m^2, if only one dam or the other is built. The loading to Devil Canyon would be lower if Watana is in place, because Watana will act as a nutrient trap.

Natural Precipitation Loading: The phosphorus concentration in precipitation was taken as 0.03 mg/l -- the maximum phosphorus concentration reported in snow and rain samples collected at Fairbanks, Alaska, by Peterson (1973).

Conversion of the area used to collect samples and the volume of sample collected, and using the normal annual precipitation at Talkeetna, indicates that natural precipitation loading will be 22 $mg/m^2/yr$.

Total Natural Loading: The total natural phosphorus load at each reservoir equals the sum of the natural land load and the natural precipitation load. At Watana the total natural load is 478 mg/m^2 and at Devil Canyon it is 2555 mg/m^2 (without Watana). Recalling the Vollenweider (1976) critical loading equation, the natural load below which results in oligotrophic status is 1057 mg/m^2 at Watana and 3763 mg/m^2 at Devil Canyon. Upon inspection, the surface specific load (Lp) is below the critical surface specific load (Lc) at both reservoirs. The calculated Lp/Lc ratio is 0.45 at Watana and 0.62 at Devil Canyon (when only one reservoir is in place). If both reservoirs are constructed, the surface specific load at Devil Canyon may be reduced, resulting in a smaller Lp/Lc ratio at this site.

Artificial Loading: This is assumed to be zero since there are no man-induced sources of phosphorus in the study area. Additional phosphorus loading to a reservoir will cause a subsequent increase in the steady-state phosphorus concentration, which may result in a change in water quality. Therefore, artifical loading must be incorporated into the phosphorus model if the capacity for residential dwelling or summer cottage development is to be determined.

z, Mean Depth: The mean depth was calculated for both reservoirs as the "full pool" volume divided by the reservoir surface area. This is the same method used to determine mean depths at Crescent Lake and Kamloops Lakes by Koenings and Kyle (1982) and St. John et al. (1976), respectively. The mean depth at Watana and Devil Canyon will be 76 meters and 43 meters, respectively (R&M Consultants, 1982c).

p, Flushing Rate - The flushing rates at Watana and Devil Canyon are 0.61 year and at 6.25 years, respectively (R&M Consultants, 1982b).

The above values indicates that both Susitna River reservoirs will be oligotrophic under existing conditions, as the spring phosphorus concentration [P] will be 4.5 mg/m^3 at Watana and 6.8 mg/m^3 at Devil Canyon. The concentration at Devil Canyon would be reduced if the Watana Reservoir is in place because Watana will act as a nutrient trap, thereby reducing the natural land loading.

The above values of [P] plot in the same area as oligotrophic water bodies (Figure 1) with similar phosphorus loading, mean depth, and flushing rate values. Levels below 10 mg/m^3 are indicative of oligotrophic conditions, 10 to 20 mg/m^3 are in the mesotrophic range, and levels above 20 mg/m^3 are considered eutrophic (Vollenweider 1976).

Although both reservoirs initially will be oligotrophic, artificial loading could shift the trophic status of one or both reservoirs at some future time. Because of this concern, an analysis of response time and artificial loading was made.

<u>Response Time</u> - The time required for a lake having an initial loading rate (L_1) to response to a change in loading to a new level (L_2) may be described by the half-life change in concentration (Dillon and Rigler 1975). The half-life change is the time required for a lake's phosphorus concentration to move from the original steady-state concentration to the advanced steady-state concentration. The half-life can be estimated as:

$$t_{\frac{1}{2}} = \frac{0.69}{p + 10 / \bar{z}}$$

Where: $t_{\frac{1}{2}}$ = half-life time

p = flushing rate

$\bar{z}$ = mean depth

Watana will have a half-life time of 0.93 year and Devil Canyon a half-life time of 0.11 year. However, Dillon and Rigler (1975) note that 3 to 5 times the half-life time can be used as an indicator of a lake's response time to additional phosphorus loading. Thus the time required to approach a new steady-state phosphorus concentration following an increase in loading will be 2.8 to 4.6 year in the Watana impoundment and 0.3 to 0.6 year in the Devil Canyon impoundment.

2.7.3 - Population Equivalent

Results from 13 studies in North America and Europe concluded that the average per capita phosphorus supply (excrement plus household waste) is 800,000 mg/yr from domestic sources (Dillon and Rigler 1975). By dividing the average per capita supply by the surface area of each reservoir, the average per capita phosphorus load can be obtained, which will be 0.005 $mg/m^2/yr$ and 0.025 $mg/m^2/yr$ at Watana and Devil Canyon, respectively. The maximum permissible artificial loading resulting in oligotrophic status is calculated as the difference between the critical surface load and the natural surface load at each reservoir. Thus, the maximum permissible artificial loads are 579 $mg/m^2/yr$ and 1208 $mg/m^2/yr$ at Watana and Devil Canyon, respectively. The loading at Devil Canyon could be higher if Watana is in place because Watana will trap nutrients. The permissible number of permanent (year round) residents at each reservoir is obtained by dividing the maximum permissible artificial load at each reservoir by its corresponding average per capita artificial phosphorus load. Assuming that all residents will be permanent, Watana will accommodate a maximum of 115,800 individuals. Similarly Devil Canyon will accommodate a maximum of 48,300 individuals if it is the only reservoir. The maximum number of permanent dwelling unit equivalents around each reservoir may be calculated by dividing the number of permissible residents by the number of residents at each dwelling unit. For example, if three individuals occupy each dwelling unit for the entire year, the maximum permissible number of permanent dwelling units will be 38,600 at Watana and 16,100 at Devil Canyon. In a situation where dwelling units are (on the average) occupied for less that 365 days per year (i.e. summer cottages), the permissible number of "seasonal" units may be calculated by multiplying the number of permissible permanent units at each reservoir by 365 days divided by the average number of days spent at each unit per year. If both permanent and seasonal dwellings are constructed, the total combined artificial phosphorus load should not exceed the equivalent amount generated by 115,800 permanent residents at Watana or 48,300 permanent residents at Devil Canyon, if oligotrophic conditions are to be maintained.

Artificial loading from a 3000 person construction camp would amount to 15 $mg/m^2/yr$ at Watana and 75 $mg/m^2/yr$ at Devil Canyon. These loading levels represent about 3 percent (Watana) and 6 percent (Devil Canyon) of the maximum permissible artificial loading required to maintain oligotrophic conditions.

2.7.4 - Conclusions

It has been determined that under natural conditions, both the Watana Reservoir and the Devil Canyon Reservoir will be oligotrophic. It was further determined that Watana and Devil Canyon will maintain oligotrophic status if provided with a maximum additional phosphorus supply equivalent to 115,800 permanent residents and 48,300 permanent residents, respectively. Additional loading from a 3000-person construction camp would amount to less than 3 percent of the maximum permissible artificial phosphorus load at Watana and 6 percent of the maximum permissible artificial phosphorus load at Devil Canyon.

2.7.5 - Summary

To summarize, reservoir trophic status is determined in part by the relative amounts of carbon, silicon, nitrogen and phosphorus present in a system as well as the quality and quantity of light penetration. The C:Si:N:P ratio indicates which nutrient limits algae productivity. The nutrient which is least abundant will be limiting. On this basis, it was concluded that phosphorus will be the limiting nutrient in the Susitna impoundments. Vollenweider's (1976) model was considered to be the most reliable in determining phosphorus concentrations at the Watana and Devil Canyon impoundments. However, because the validity of this model is based on phosphorus data from temperate clear water lakes, predicting trophic status of silt-laden water bodies with reduced light conditions and high inorganic phosphorus levels may overestimate actual trophic status. The spring phosphorus concentration in phosphorus limited lakes is considered the best estimate of a lake's trophic status. Bio-available phosphorus is the fraction of the total phosphorus pool which controls algae growth in a particular lake. The measured dissolved orthophosphate concentration at Vee Canyon was considered to be the bio-available fraction in the Susitna River. Accordingly, the average June dissolved orthophosphate concentration was multiplied by the average annual flow at each reservoir to calculate spring phosphorus supplies from the land. These values were in turn combined with phosphorus values from precipitation and divided by the surface area of each impoundment. The resultant spring phosphorus loading values at Watana and Devil Canyon were below the maximum loading levels required for the maintenance of oligotrophic conditions. Likewise, upon incorporating spring loading values into Vollenweider's (1976) phosphorus model, the volumetric spring phosphorus concentration at both reservoirs fell into the same range as oligotrophic lakes with similar mean depths, flushing rates, and phosphorus loading values.

The aforementioned trophic status predictions depend upon several assumptions that cannot be quantified on the basis of existing information. These assumptions include:

(1) the C:Si:N:P ratio does not fluctuate to the extent that a nutrient other than phosphorus becomes limiting,

(2) no appreciable amount of bio-available phosphorus is released from the soil upon filling of the reservoirs,

(3) phosphorus loading levels are constant throughout the peak algal growth period,

(4) June phosphorus concentrations measured at Vee Canyon correspond to the time of peak algal productivity.

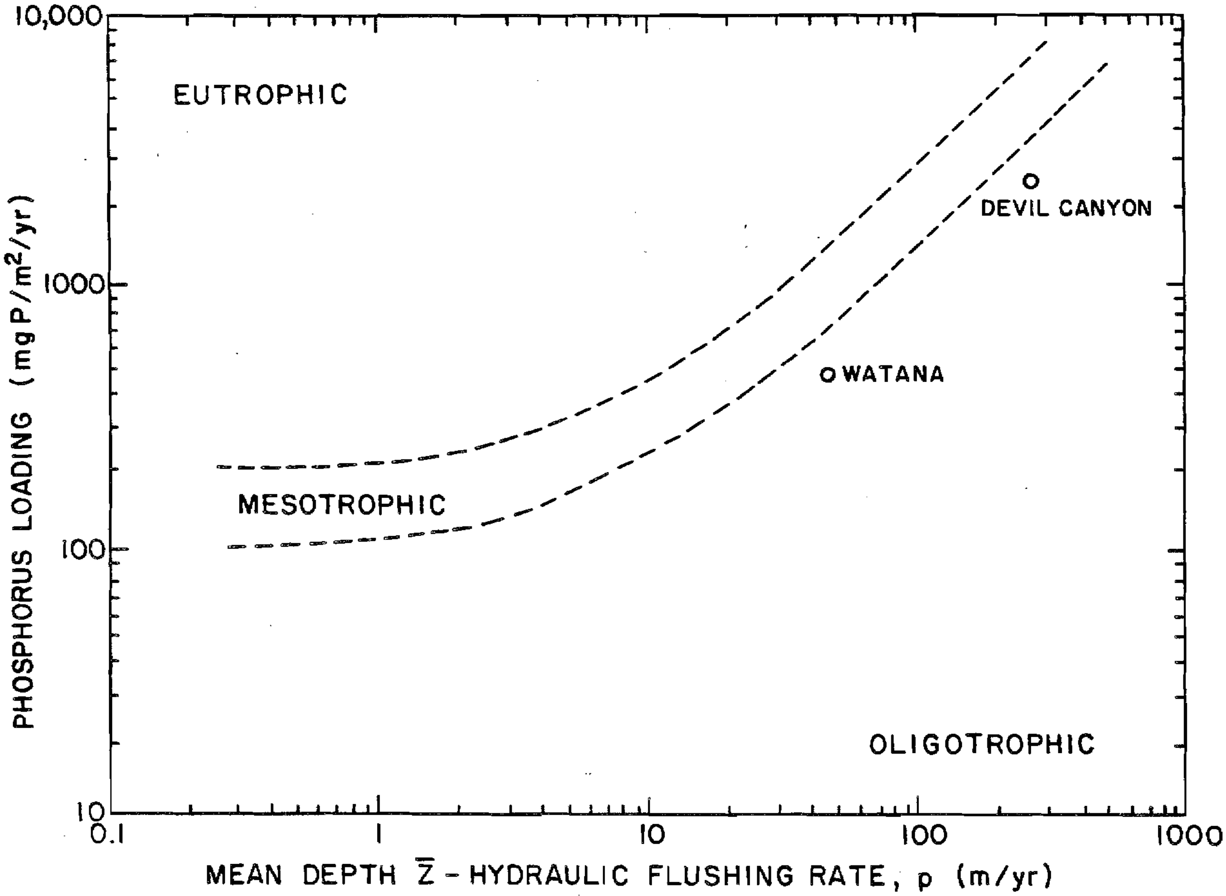
(5) phosphorus species other than dissolved orthophosphate are not converted to a bio-available form,

(6) flushing rates and phosphorus sedimentation rates are constant,

(7) phosphorus losses occur only through sedimentation and the outlet, and,

(8) the net loss of phosphorus to sediments is proportional to the amount of phosphorus in each reservoir.

Artificial phosphorus loading represents any additional input from domestic or industrial sources. The difference between critical loading and natural loading is defined as the maximum allowable artificial load. The maximum artificial phosphorus load at the Watana impoundment is equivalent to 115,800 permanent residents and at Devil Canyon it is equivalent to 48,300 permanent residents. These estimates are conservative in that the effect of low light penetration has been neglected in their calculation. In addition, artificial phosphorus loading may be reduced if septic filter bed or other treatment systems are employed at each dwelling, thereby increasing the number of permissible residents.



SUSITNA PROJECT DATA APPLIED TO VOLLENWEIDER (1976)

PREPARED BY:

PREPARED FOR:

R&M CONSULTANTS, INC.

PHOSPHORUS LOADING & MEAN DEPTH HYDRAULIC FLUSHING RATE RELATIONSHIP

FIGURE 1.

ACRES

3 - DOWNSTREAM EFFECTS

3.1 - General

Construction of hydroelectric dams and their reservoirs has a profound effect on the river regime of downstream reaches. The effects are summarized here; more detailed discussion is presented in "River Morphology Studies" (R&M Consultants, 1982b). Since the rate of reservoir water outflow is controlled, the downstream reach is no longer subject to the fluctuations of a normal river regime, with the consequence that the flow becomes more seasonally uniform throughout the year (Kellerhals and Gill, 1973; Turkheim, 1975). The minimum flow rate is significantly increased, and peak flows are decreased. The decrease in spring flood magnitude, especially during the initial impoundment, may result in negative effects on the downstream environment. It is reasonable to expect that the interference with natural Susitna River flows will cause a change in stream levels and bank storage levels for some distance downstream from the dams.

3.2 - Suspended Solids/Turbidity

Low turbidity waters immediately below a reservoir are accompanied by an increased sediment transport potential and erosion potential as well as a high nutrient level. These phenomena may result in considerable streambed material removal, to the extent which the affected channel is scoured to bedrock or becomes armored. Consequently, downstream waters immediately below a dam often become unsuitable for the breeding of upstream species such as arctic char, grayling, inconnu, lake chub, and longnose suckers (Turkheim, 1975). Bed material below the Susitna dams is large enough that scour is not expected at the regulated flow levels (R&M Consultants, 1982b).

Farther downstream from a dam, sediments eroded from stream banks and channels are re-deposited. Heavy silting of gravel interstices result in decreased intergravel water flow and oxygen content. Riverflow regulation associated with hydroelectric dams eliminates spring freshets which normally cleanse the downstream channel substrate of such silt deposits. Thus, changes in natural turbidity and sedimentation processes due to impoundment may have a detrimental impact on fish spawning habitat many miles downstream from a hydroelectric dam, even when reservoir discharges are low in turbidity and suspended solid loads (Turkheim, 1975).

Operation of the Watana Reservoir will sharply reduce summer suspended solids concentrations and turbidity levels downstream from the reservoir. The velocity of water entering the impoundment will be reduced, which will allow all but the finest particles to settle. The turbidity and suspended solids levels of water released during winter will be lower than summer conditions because of quiescent conditions occurring under an ice-cover. However, glacial flour that entered the reservoir during summer may pass through and out the reservoir during winter, resulting in turbidity levels which, although low, may be higher than pre-project levels. Addition of the Devil Canyon Dam will have little additional effect on downstream turbidity levels because any particles carried through the Watana Reservoir will also pass through the Devil Canyon Reservoir.

The range and seasonal variation in temperature of the Susitna River will change after impoundment. The magnitude of temperature variation will change from the pre-project range of 0°C to 13°C to a range having slightly warmer temperatures in winter and slightly colder temperatures in summer. Under existing conditions, the lows occur in October/November through March/April and the highs appear in July or August. After impoundment, low temperatures will occur in November through March, but high temperatures will still occur in July or August.

3.3 - Heat Transfer

When water is released from the epilimnion of a deep reservoir, there is likely to be a warming effect on the stream below the dam (Turkheim, 1975; Baxter and Glaude, 1980). There may also be an increase in the amount of organic detritus originating from plankton in the reservoir. Furthermore, this type of release will encourage the accumulation of poor quality bottom water in the reservoir until the time of fall overturn when water becomes mixed (Baxter and Glaude, 1980).

Comparatively little work has been done in Alaska pertaining to the effects that impoundment-related temperature changes have on stream biota. However, there is evidence indicating that impoundment-related water temperature changes alter resident fish distribution and abundance, fish food productivity, and organism development rates downstream from dams (Baxter and Glaude, 1980). The timing and extent of temperature changes is of importance since the breeding seasons and life cycles of most stream organisms are integrated with annual temperature changes. Water temperature changes may also alter the pattern of anadromous fish migration below a dam (Turkheim, 1975).

Baxter and Glaude (1980) report that certain insect species which are important fish food sources are particularly sensitive to changes in the downstream thermal regime because their metamorphosis is induced by temperature changes. If these changes do not occur or occur at the wrong time, their life cycles will be disrupted. Furthermore, change in seasonal temperature patterns may change the timing of anadromous fish spawning to their detriment (Baxter and Glaude, 1980).

In reservoirs at high latitudes, large-scale downstream river icings have been known to occur when winter flows are greatly reduced for the purpose of hydroelectric peaking. These icings are thick accumulations of bottom-fast ice resulting from low flow, extreme cold and the constriction of bedrock or permafrost below the channel bed (Turkheim, 1975). Downstream channel icings may significantly increase erosion with consequent increase in sediment loads downstream (Baxter and Glaude, 1980). Furthermore, flow regulation may delay spring breakup downstream if breakup is otherwise accelerated by a rapid increase in discharge (Turkheim, 1975).

The range and seasonal variation in temperature of the Susitna River will change after impoundment. The magnitude of temperature variation will change from the pre-project range of 0°C to 13°C to a range having slightly warmer temperatures in winter and slightly colder temperatures in summer. Under existing conditions, the lows occur in October/November through March/April and the highs appear in July or August. After impoundment, low temperatures will occur in November through March, but high temperatures will still occur in July or August.

3.4 - Dissolved Oxygen

Water released from near the surface of an impoundment generally provides a higher dissolved oxygen content in downstream waters than waters released from the deeper levels (Love, 1961). Turkheim (1975) notes that open water downstream from an impoundment created by water discharge through a dam, produces a significant increase in winter dissolved oxygen concentrations.

Dissolved oxygen concentrations below the reservoirs will be relatively high. Turbulence created by spillage over the dams and transit through the power tunnels will aerate water to or slightly above saturation levels.

3.5 - Gas Supersaturation

Turkheim (1975) reports that nitrogen supersaturation of water below a dam is possible in certain seasons, extending an unknown distance downstream. This is certainly a possibility below both Susitna dams. Data from the Columbia River indicate that at least 75 kilometers may be required before nitrogen equilibrium conditions are re-established below an impoundment (Geen, 1975). The work of Beiningen and Ebel (1970) revealed that downstream supersaturation levels only dropped from 135% to 114% over a distance of 120 kilometers below a dam they were studying. It is expected that equilibrium conditions are achieved more rapidly the more turbulent the water is downstream (Geen, 1975). The ultimate impact of nitrogen supersaturation is its effect on fish. Supersaturation has a serious impact on adult and young salmon below a reservoir (Geen, 1975). If dissolved gases reach levels of supersaturation, lethal gas embolisms in fish may result for miles downstream of an impoundment (Turkheim, 1975). The death of Atlantic salmon by "gas-bubble" disease was directly attributable to nitrogen supersaturation below a dam in Canada (Ruggles and Watt, 1975). Potential nitrogen supersaturation problems will be solved structurally at the Susitna dams through the use of Howell-Bunger valves, eliminating plunging spills up to the 1:50 year flood. Portage Creek, just below Devil Canyon, is essentially the upstream limit for spawning salmon under natural conditions, although some salmon may negotiate Devil Canyon Rapids at low flow. With Watana only, power releases will travel through several sets of rapids as well as Devil Canyon before reaching Portage Creek. It is reasonable to expect that, will the natural plunging and turbulence of the canyon, post-project nitrogen supersaturation levels with Watana only will be the same as the pre-project levels at the downstream end of Devil Canyon. However, nitrogen supersaturation caused by spills from Devil Canyon Dam during an extreme event will not have as much opportunity to attain natural levels.

4 - CONCLUSIONS

Impoundment of the Susitna River will affect water quality in the reservoir area(s) and downstream from the reservoir(s). The major changes will result from impoundment of the flowing river, and minor changes will occur below the impoundment(s). The above assessment of the effects of impounding the Susitna River is summarized herein.

(1) The reservoirs will be oligotrophic. The oligotrophic status can be maintained at both reservoirs if they are provided with less than a maximum additional phosphorus supply equivalent to that produced by 115,800 year-round residents at Watana and 48,300 year-round residents at Devil Canyon (if only one dam is built). The Devil Canyon Reservoir could support a large population if Watana is in place because Watana will trap nutrients, reducing the natural supply to Devil Canyon. Additional loading from a 3000-person construction camp would amount to less than 3 percent of the maximum permissible artificial phosphorus load at Watana and 6 percent of the maximum permissible artificial load at Devil Canyon.

(2) A short-term increase in dissolved solids, conductivity, and most of the major ions may be evident immediately after closure of the dam(s). Inundation and leaching of rocks and soil in the impoundment area promote this situation. The magnitude of increase cannot be quantified with available data, but it is anticipated that the increase will not result in detrimental effects to freshwater aquatic organisms. The leaching effects will diminish for two reasons as the reservoir(s) matures. First, the most soluble elements will dissolve into the water rather quickly and the rate of leachate production will decrease with time. Second, much of the inorganic sediment carried by the Susitna River will settle in the Watana Reservoir. The formation of an inorganic sediment blanket on the reservoir bed will retard leaching.

(3) About 70-97 percent of the suspended solids load in the river is expected to settle in Watana Reservoir. Consequently, the reservoir and downstream area will be significantly less turbid than the pre-project condition during summer. Glacial flour entering the reservoir during summer may still be passing through during winter. Consequently, winter turbidity values, although low, may be higher in the reservoir and downstream area than under pre-project conditions, where turbidity is essentially zero.

(4) The percentage of the reservoirs' volume lost to evaporation during summer will be less than 1 percent at both reservoirs.

The potential effect of a less-than 1 percent concentration increase in dissolved solids is not significant.

(5) The range and seasonal variation in temperature of the river will change after impoundment. There will be a reduction in the magnitude of temperature variation and some shift in the time period of low temperatures.

(6) Many metals' concentrations will be reduced in Watana Reservoir as these elements will be precipitated and will settle to the bottom.

(7) The reservoir(s) will maintain relatively high oxygen levels and low algal productivity. Productivity will be limited by low light conditions. Turbidity in the summer and an ice cover in the winter (especially if snow covers the ice) will reduce the depth of the photic zone. Dissolved oxygen should remain high because the existing oxygen demand is low.

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ATTACHMENT A

METHODOLOGY FOR THE PREDICTION OF THE POTENTIAL FOR EUTROPHICATION

Two widely recognized models for the prediction of trophic status in phosphorus limited lakes are presented in this attachment. In addition, the applicability of each model in Alaskan waters is discussed.

The relative quantities of carbon, silicon, nitrogen, and phosphorus (i.e. C:Si:N:P ratio) in a lake or reservoir indicates which of these nutrients controls the eutrophication process. A particular C:Si:N:P ratio that exists at any given time in a lake may be subject to change, depending upon the rate at which available forms of these nutrients from both internal and external sources are supplied to a water body, as well as the rate of their utilization or transformation to available forms. Among these nutrients, phosphorus and nitrogen most often limit the growth of algae in aquatic systems. Since algae production occurs rapidly over a short period of time, it is the bio-available fraction of these nutrients present, rather than the quantities of total nitrogen or phosphorus, that determine the limiting nutrient. The form of phosphorus that is bio-available in natural fresh water consists of the dissolved orthophosphate fraction, and the bio-available form of nitrogen consists of the inorganic fraction. An inorganic nitrogen to dissolved orthophosphate mass ratio between 5:1-10:1, corresponding to an atomic ratio of between 11:1-22:1, is the critical range above which phosphorus is limiting and below which nitrogen is limiting (Rast and Lee, 1978).

Phosphorus is usually the least abundant nutrient controlling algal growth in lakes. Phosphorus loading -- the amount of phosphorus added to a lake per unit area per unit time -- is recognized as the best measure of the degree of eutrophication that may be predicted in a phosphorus limited lake. Two models for predicting the spring total phosphorus concentration in a lake appear below.

The phosphorus imported to a lake in runoff, when combined with input directly to the lake's surface through precipitation and dry fallout, gives a measure of the natural total phosphorus load. The total natural phosphorus load can be combined with the total artifical phosphorus load, the mean depth of the lake, the lake's water budget expressed as the flushing rate, and the phosphorus retention coefficient of the lake, to predict spring total phosphorus concentration in the lake. The predicted spring total phosphorus concentration can then be used to predict trophic status, which is directly related to summer chlorophll "a" concentration, and secchi disc transparency. Dillon and Rigler (1975) present an equation to predict the total, steady-state phosphorus concentration which is expressed by:

$$[P] = \frac{L\ (1-R)}{\bar{z}\ p}$$

Where: [P] = steady-state phosphorus concentration

L	=	total loading (natural and artifical)
$\bar{z}$	=	mean depth of the lake (lake morphometry)
p	=	flushing rate (water budget of the lake)
1-R	=	retention coefficient (the fraction of the loading that is not lost via the outflow)

According to the equation, the total concentration of phosphorus may be predicted for a lake. The correlation coefficient between measured and predicted phosphorus concentrations for eleven lakes in southern Ontario ($r = 0.90$) indicates that lake phosphorus concentrations can be accurately predicted in at least some lakes (Dillon and Rigler, 1974). However, because the equation for determining the retention coefficient was developed for a homogeneous set of lakes, its application may be limited to lakes in similar geographical areas.

Subsequently, a model was developed by Vollenweider (1976) which is considered to be an improvement over the Dillon and Rigler model. In this model, the retention coefficient is replaced by a factor which incorporates the flushing rate ($1 / \; 1 + \sqrt{1/p}$).

Theoretically, either of these models may be used to predict the trophic status of a reservoir following the impoundment of a stream or river, based on the assumptions that: (1) phosphorus will be the factor controlling phytoplankton productivity, (2) the influx of phosphorus is constant, (3) phosphorus losses occur through sedimentation and the outlet, and (4) the net loss of phosphorus to sediments is proportional to the amount of phosphorus in the reservoir (Utturmark and Hutchins, 1978). Also, phosphorus concentrations tend to increase with the age of a reservoir (Smith and Justice, 1976), and peak algae biomass and productivity levels may occur under spring ice rather than in summer (LaPerriere et al., 1978), making both models inapplicable.

When values for loading and flushing rate are estimates (as opposed to direct measurements) then it is likely that the uncertainty of these estimates will be quite large (Reckhow, 1979). The uncertainty of natural phosphorus loading figures can in themselves result in a 100 percent error in the calculation of the natural phosphorus budget in a reservoir, while other factors are only approximations. Nevertheless, these estimates are valuable because of their simplicity and quantitative nature, provided that caution is taken in interpreting the results (Dillon and Rigler, 1975).

Loading

Total phosphorus loading is calculated by totalling the phosphorus load from the land (runoff), the phosphorus load from precipitation, and the artifical phosphorus load (from human development). Total phosphorus load from the land is equal to the total area of each watershed or drainage basin contributing runoff to the lake multiplied by the phosphorus export coefficient. This coefficient is the phosphorus exported from each m^2 of land in the watershed per year, and is calculated by combining the measured amount of phosphorus carried by each stream in the watershed with the total discharge for a given period of time. Dillon and Kirchner (1975) measured the total phosphorus export for 34 southern Ontario watersheds. The total phosphorus export for all watersheds was tabulated along with additional information on the geology, land use, and population density of each watershed. Upon inspection of these data, it was apparent that the watersheds could be grouped according to whether they were forested or consisted of pasture as well as forest, and according to whether they were on igneous or sedimentary formations. The range and mean phosphorus export values ($mg/m^2/yr$) obtained for each two-way (land use-geology) classification were:

Land Use	Igneous	Sedimentary
Forest		
Range	2.5 - 7.7	6.7 - 14.5
Mean	4.8	10.7
Forest & Pasture		
Range	8.1 - 16.0	20.5 - 37.0
Mean	11.7	28.8

Changing land use from "forest" to "forest and pasture" in watersheds on igneous rock apparently more than doubles the phosphorus export. Similarly, the export from a sedimentary forested watershed is about double that from an igneous forest watershed. The advantage of using export coefficients is that the total phosphorus load to a lake can be estimated without extensive field investigations. However, an export coefficient must be selected which was determined for a land use similar to the area under consideration. In areas where no export coefficient has been established, the total phosphorus load to a water body may be determined by combining the measured amount of phosphorus carried by each contributing stream ($mg/m^3/yr$) with the total stream discharge (m^3/yr). In effect, this is the method by which Dillon and Kirchner (1975) initially determined export coefficients for 34 southern Ontario watersheds. Until such regional coefficients are determined for Alaska land uses, the latter of the two methods probably offers a higher degree of total phosphorus export reliability.

The total phosphorus load from the land is combined with the total phosphorus load from precipitation to calculate the total phosphorus load. Vollenweider (1976) established a method for determining the maximum phosphorus load to a lake which will result in oligotrophic status. This method is expressed by:

$$Lc = 10 \left[\bar{z}p \left(1 + \sqrt{1/p} \right) \right]$$

Where: Lc = critical areal phosphorus load ($mg/m^2/yr$)

$\bar{z}$ = mean depth
p = flushing rate

The fraction of the phosphorus load which is biologically available may be significantly different than the total measured phosphorus load. Accordingly, the steady-state concentration of total phosphorus in a lake may be different than the concentration of phosphorus which is available to the growth of lake algae. For example, Kamloops Lake is an oligotrophic lake in British Columbia with a very high phosphorus load (22,800 $mg/m^2/yr$) and mean depth of 75 m (St. John et al., 1976). Without differentiating between the various forms of phosphorus, the predicted phosphorus concentration in the lake was above what is considered to be oligotrophic. When the appropriate corrections were made for available phosphorus forms, and the lake's flushing rate was taken into account, the steady-state phosphorus concentration fell well within the oligotrophic category. It was determined that 80 percent of the total phosphorus in Kamloops Lake occurred as particulate phosphorus, of which 80 percent was biologically unavailable over a wide range of particle sizes in the water column. The dissolved fraction was not analyzed for its orthophosphate content, which is considered the most important algae nutrient in lakes (St. John et al., 1976).

The recent investigation of a glacially influenced lake in Alaska (Koenings and Kyle, 1982) indicates that 50 percent of the total phosphorus concentration in the lake was biologically inactive, owing to the fact that the greatest percentage of the lake's total phosphorus occurred in the inorganic particulate form. In addition, non-particulate (soluble) inorganic phosphorus was to a great extent converted to particulate phosphorus in the epilimnion in mid-summer. It was concluded that the nutrient dynamics in glacially influenced lakes which have high silt inputs are different than those of clear water lakes.

Mean Depth

The mean depth of a lake or a measure of the lake's morphometry is required in using the predictive model for phosphorus concentrations.

Flushing Rate: The flushing rate is the reciprocal of the detention time of a unit volume of water within a lake and is the equivalent of the lake's annual water budget expressed as the total outflow volume per year divided by the lake volume.

Estimation of a reservoir's water budget from precipitation and evaporation data may be quite inaccurate, especially for small lakes and small drainage areas. Actual measurement rather than estimation of the water budget undoubtedly provides much more accurate results, and should be undertaken where possible (Dillon and Rigler, 1975).

Flushing rate is significant in that it holds down algal production in lakes which may otherwise be highly productive. In lakes with high flushing rates, biologically available phosphorus accumulates much slower than in lakes with relatively low flushing rates. In lakes containing a high percentage of suspended phosphorus, the flushing rate may not reflect the rate at which total phosphorus is removed. Instead it may be an expression of the percentage of dissolved phosphorus removed from the system, if particulate phosphorus settles to the bottom before it can be removed (St. John et al., 1976).

Flushing rate is an important variable since it regulates both the degree and regime of phosphorus loading. In reservoirs, the flushing rate is controlled by the amount of water allowed to escape relative to the reservoir volume (Ryder, 1978).

Calculations of flushing rate may be complicated by the varying hydrologic regime of a reservoir. For example, St. John et al. (1976) report that flushing rates may vary 20-fold over one year, and at times inflowing water may pass directly through a lake, thus increasing the flushing rate even further.

Retention Coefficient: The retention of phosphorus in a reservoir is dependent upon factors such as the dissolved oxygen concentration and the pH at the sediment-water interface, and upon major cations that combine with phosphorus and transport or retain it in sediments. Thus, the phosphorus retention coefficient is actually a simplification of the phosphorus retention process (Reckhow, 1979). For instance, phosphorus contained in reservoir sediments may be released and redistributed under reducing conditions during fall overturn (Hutchison, 1957). Successful use of this coefficient is dependent on an accurate estimate of the phosphorus retention in the lake in question. A large discrepancy between predicted phosphorus retention (0.1 percent) and measured phosphorus retention (76 percent) was found by St. John et al. (1976) in Kamloops Lake. This discrepancy was attributed to a situation where most of the phosphorus load was in the form of inorganic particulate phosphorus, which is converted to dissolved phosphorus at a much slower rate (if at all) than particulate organic phosphorus. Phosphorus retention in a lake is a function of sedimentation--specifically, the amount of phosphorus that is retained by sedimentation. This amount is difficult to calculate. However, a model was by Kirschner and Dillon (1975) developed for lakes in southern Ontario relating the areal water load (qs) to phosphorus retention as expressed by:

$$Rp = 1 - [0.426 \exp (0.271\ qs) + 0.574 \exp (-0.00949\ qs)]$$

Areal loading (qs) in m/yr is the surface overflow rate and is calculated as the lake outflow volume divided by the lake surface area. Values for phosphorus retention in 15 Ontario lakes, using the measurement model and the values derived from the theoretical model, were in close agreement ($r = 0.94$). "The fact that the retention coefficient of phosphorus is more closely related to the areal water load (qs) than the volumetric water load (i.e. water renewal time) is not readily explainable, but in light of the above advantages we feel that this model warrants acceptance on purely empirical grounds" (Kirschner and Dillon, 1975).

An alternative method of estimating phosphorus retention

$$\left(\frac{1}{1 + 1/p} \right)$$

was derived by Vollenweider (1976) which is an expression of the relationship between phosphorous residence time and the residence time of water in a lake. This equation was used in calculating phosphorus concentrations in 60 heterogenous northern temperate lakes and plotted against chlorophyll "a" concentrations in the same lakes. The resultant correlation coefficient (0.868) verified the "unquestionable" relationship between Vollenweider's model and algae production among lakes in the north temperate zone (Vollenweider, 1976).

ATTACHMENT A

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