MANAGING HIGH ELEVATION RESERVOIR-STREAM SYSTEMS

Michael D. Marcus

May 1987 WWRC-87-08

Department of Zoology and Physiology
University of Wyoming
Laramie, Wyoming

Co-Principal Investigators

Robert W. Brocksen
Wyoming Water Research Center

Harold L. Bergman
Department of Zoology and Physiology

Final Report

Submitted to

Wyoming Water Research Center
University of Wyoming
Laramie, Wyoming

ABSTRACT

Increasing demands for water in the western United States are leading to increasing construction of reservoirs on high elevation headwater streams. Through appropriate management, these reservoirs present new chances to expand and enhance potentially unique fishing opportunities. This report presents diverse information on the characteristics of lakes, streams, and reservoirs, and on the effects of reservoirs on streams. A number of management options are suggested as having potential applicability to enhancing fishery potentials both within and downstream of high elevation reservoirs. These include (1) construction of reservoir outlets that permit discharge of water from greater than one level from the reservoir, enabling regulation of water quality and temperatures both within and downstream of these reservoirs; (2) maintenance of minimum pools within reservoirs; (3) scheduling of downstream flushing flows; (4) severely regulating or prohibiting additional sources of potential environmental stress within watersheds above headwater reservoirs, which include logging, cattle grazing, and mining; (5) installing artificial structures, which offer potentials for enhancing fishery habitats both within and downstream of reservoirs; and (6) direct manipulation of fisheries through selective stocking, species and strain selection, and harvest regulations, which can lead to "wild" and/or trophy fisheries in some of these systems. All implemented management actions should be monitoring to evaluate the actions achieve the stated objectives.

ACKNOWLEDGEMENTS

For their continued support and expert assistance, I thank my Co-Principal Investigators, Bob Brocksen and Harold Bergman. I also thank my wife, Debbie Finch, who had to endure my late nights finishing this project; Steve Wolff for sharing his knowledge about flow dynamics and habitat models in streams; Paula Guenther and Wayne Hubert for sharing their review on recommended minimum pools for reservoirs; and Beth Mullan, for helping to keep some of my verb tenses correct, at least until my next revision.

This work was supported by Wyoming Water Research Center Grants 5-38669 and 5-38677.

TABLE OF CONTENTS

Section		Page

LIST OF TABLES

SECTION 1

INTRODUCTION

Streams are dammed and reservoirs are filled for many reasons. Most often in the western United States the primary purposes are to store and regulate downstream discharges to maximize municipal and agricultural water supplies. Reservoirs also are constructed for flood control, hydroelectric power, and recreation, particularly to increase manageable fisheries and boating opportunities. Reservoirs can benefit agricultural, forestry, and livestock management (Canter 1985). And reservoirs can be managed to increase wetlands, enhancing habitat for migratory waterfowl and other resident wildlife.

In the arid West water quality and quantity problems intensify as water needs grow accompanying municipal expansion, energy development, and agricultural requirements (Howe 1980). Often the only remaining sources of surface water available for these increasing demands are found in headwater streams. Thus, these headwater areas are often developed to meet the existing or perceived demands for water. For example, in 1986 seventeen new water developments were proposed in Wyoming that included potentials for construction or improvement of headwater reservoirs (Wyoming Water Development Commission 1986).

OVERVIEW OF CONCERNS ABOUT RESERVOIRS

Development of headwater reservoirs often encounter resistance in some public sectors due to concerns about potential environmental impacts resulting from their development. Many concerns are, in fact, the same as are associated with construction of any reservoir. Therefore, it can be useful to briefly introduce environmental characteristics of reservoirs and their effects; later sections expand on many of these topics.

Water quality in reservoirs, as in any surface-water body, is related to (1) climate, especially the quantity and quality of incoming precipitation; (2) the chemical nature of the geologic formations within the watershed basin through which waters drain before entering surface waters; (3) vegetation within the basin and its influences, along with influences of climate, on the structure and chemistry of basin soils; and (4) influences of man within the watershed, i.e., contributions to surface runoff produced by surface disturbances or effluents discharged directly into the surface waters.

Construction and operation of reservoirs variously alter the natural quality of water. Factors within reservoirs that can influence the extent of these alterations include its shape (morphometry), retention time for waters, its age, thermal or chemical stratification of the waters, biological activities, and discharge depth(s) and timing.

A number of downstream hydrological changes are produced by all impoundments, but the extent and timing of these changes vary among reservoirs: (1) evaporation from reservoirs reduces the total average annual runoff from watersheds; (2) seasonal flows become less variable;

(3) annual extremes in flow are altered; (4) magnitudes for floods attenuate; and (5) pulses occur that are unnatural in terms of timing and duration (Petts 1984). Additional downstream effects often resulting from reservoir construction and operation include alteration of (1) daily and annual thermal patterns; (2) nutrient, dissolved gas, suspended sediment, and salinity levels; (3) composition of bottom sediments; (4) shoreline stability; and (5) composition of the biological community (Canter 1985, Petts 1984).

The multiple effects on the downstream biological communities include alterations to the riparian vegetation, instream aquatic vegetation, composition and diversity of resident organisms, migration routes for the stream's inhabitants, biotic productivity, life cycles for the resident organisms, and trophic relationships among these organisms.

Dams also can cause permanent physical changes by increasing seismic tendencies; altering groundwater flows and water tables; inundating settled areas; destroying wildlife habitat; interfering with migrations of fish and other aquatic organisms; increasing extensive aquatic weed growths; and potentially increasing the spread of communicable diseases, particularly in tropical areas (Canter 1985).

GOALS AND OBJECTIVES OF THIS REPORT

This report has three general goals:

1. To examine our present understanding of the general characteristics of lakes and streams, which can help to increase the understanding of relationships within reservoirs and their downstream effects.

2. To examine our present understanding of reservoirs and their downstream effects, which can help determine appropriate design and management strategies for these systems.

3. To present options for managing high elevation reservoir-stream systems, particularly emphasizing fisheries and effects of water quality on fisheries.

This report is not intended to be a comprehensive, original review of all research completed on the dynamics of reservoirs and their effects on streams. Many excellent reviews and compilations exist for these subjects (e.g., Neel 1963; American Fisheries Society 1967; Hall 1971; Baxter 1977, 1985; Ward and Stanford 1979a, Petts 1984). This report, in fact, draws heavily from such reviews, while attempting, where possible and appropriate, to extrapolate from these reviews to management of headwater reservoir-stream systems.

It is hoped that this report will aid professional fisheries and aquatic scientists, who work with high elevation reservoir stream systems, by introducing and summarizing the literature on concepts for these systems. I also hope that it will aid those who are not specialists in aquatic systems by introducing many of the considerations necessarily involved in managing reservoir fisheries. Finally, it is hoped that this document will encourage viewing reservoirs and downstream receiving waters as a management unit, leading to integrated management plans for these systems.

SECTION 2

GENERAL CHARACTERISTICS OF LAKES AND STREAMS

LAKES

Thermal cycles and stratification

Thermal characteristics are, perhaps, the single most important physical influence in lake environments. Typically, temperate lakes are warmed by solar radiation during the spring. As the cold (about 0 C) surface waters from winter are heated to 4 C (the temperature at which water has its maximum density), these dense surface waters sink to the bottom, pushing the colder, less dense bottom waters toward the surface. Following this spring turnover of lake water, the lake first develops uniform temperatures extending through the vertical profile of the water column. Then, as surface waters are gradually warmed, lakes often become thermally stratified with a layer of warm water nearest the surface (the "epilimnion"), a middle layer where temperatures decrease at about 1 C per meter of water depth (the "metalimnion" or "thermocline"), and a layer of cool water nearest the bottom (the "hypolimnion"). Warming of surface waters tends to cause the metalimnion to sink through the summer.

With the onset of autumnal cooling, the reverse of the above process occurs. Surface waters cool until 4 C is again reached and these dense waters sink to the bottom, displacing the warmer water upwards toward the surface where it cools. The fall mixing of the lake leads again to isothermal conditions.

Through the winter, as ice cover forms over the lake, lake waters often tend to stagnate with relatively warm, dense (3-4 C) water on the bottom and cold, lighter (ca. 0 C) water near the ice-covered surface. During this winter stagnation period a chemical stratification, of sorts, can develop, as will be discussed briefly in the next section.

Lakes that turnover twice yearly, such as just discussed, are termed "dimictic lakes." Lakes also exist that do not mix ("amictic"), mix once ("monomictic"), or multiple times ("polymictic"). But dimictic lakes are the most common and are the ones most studied in the north temperate zone. A comprehensive discussion of the thermal properties of lakes is presented in Hutchinson (1957); good discussions are also found in most limnology texts.

Nutrient cycling

The manner nutrients are used by organisms, how they are transferred between organisms at different trophic levels, and how inputs of chemical pollutants influence lake ecosystems have been researched and discussed in great detail. Good compilations of this work are again found in Hutchinson (1957) and in other limnological texts. But a brief review on the general cycle of nutrients in lakes can contribute to later discussions and to understanding the chemical dynamics of reservoirs and the management potentials for reservoirs.

Most nutrients dissolved in the upper illuminated layers of a lake (the euphotic zone) are incorporated into plant tissue through the process of photosynthesis by small, free-floating algae, the phytoplankton. Subsequently, many of these algae are consumed by small floating animals (zooplankton) or fish, and many of the zooplankton are consumed by larger animals, including fish.

Other phytoplankton and zooplankton die without being consumed. Carcasses of these dead organisms and fecal wastes from living organisms slowly sink toward the lake's bottom. As these materials pass through the water column, chemical and bacterial decomposition release some of the bound nutrients into the water column, where they become readily available for reuse. The bulk of the settling material, however, passes through the euphotic zone into the deep, often dark (aphotic), waters of the hypolimnion. Eventually, much of the material settles in organic layers on the bottom, where it can become permanently stored.

Decomposition of organic materials continues in the deep waters and on the bottom. Because of low light intensities in these deep waters, photosynthesis usually cannot occur. Without the continual input of oxygen through photosynthesis and with the continual use of oxygen, through both decomposition of settled organic materials and respiration by the resident organisms, concentrations of oxygen continually decrease in the deep-water environment.

During periods of summer and winter stratification, oxygen can be totally depleted in the deep waters. This depletion produces reducing conditions for chemicals in the waters and in the sediments. Under such conditions, chemically bound oxygen present in the settled organic materials can be used during bacterial metabolism. For example, bacteria metabolize the oxygen contained in nitrate (NO_3) and nitrite

(NO_2) compounds, which chemically reduces these substances into ammonia (NH_3) and ammonium (NH_4) containing compounds (Wetzel 1975). In this form the dissolved nitrogen containing compounds readily move from sediments into the water column.

Phosphorus is present in aerobic sediments as both organic and inorganic compounds, including apatite and orthophosphate ions covalently bonded to hydrated iron oxides. In the reducing conditions produced by oxygen depletion, this organic bound phosphorus is decomposed and phosphorus in ferric hydroxides and complexes are reduced. Consequently, ferrous iron and absorbed phosphate are mobilized and released into the overlying water. In contrast to the importance of bacteria in recyclying nitrogen compounds, they are relatively unimportant in the movement of phosphorus compounds from sediments into the water column (Wetzel 1975).

Once nutrients become dissolved in the deeper layers of the water column, the spring and fall turnovers mix the redissolved nutrients through the entire column, where these nutrients again become available for use by the phytoplankton. Once exposed to oxygenated waters, however, phosphates rapidly react to form insoluble compounds and complexes, in which form they again begin to precipitate to the sediments. Thus, the process of recycling nutrients between the sediments and the biota in the upper water column is repeated twice annually in dimictic lakes.

STREAMS

Stream waters are intrinsically interconnected to their surrounding watersheds (Motten and Hall 1972, Hynes 1975, Likens 1984). Stream waters mostly originate as runoff from overland flows or as groundwater seepage from the watershed. Hence, the quality and quantity of water reaching streams depend upon the infiltration capacity of the soil and on other physical and chemical attributes of the watershed's geology and lithology.

As water percolates over and through the soils, chemical constituents are dissolved and other chemical reactions occur that define the water quality characteristics in the stream. Within the stream, other mechanisms further affect the composition of dissolved substances. For example, dissolved ions can be rapidly adsorbed onto inorganic particles or absorbed by living organisms (e.g., Likens 1984).

The relationships of streams with their watersheds were perhaps best summarized by Hynes (1975, p. 12): "We may conclude then that in every respect the valley rules the stream. Its rock determines the availability of ions, its soils, its clay, even its slope. The soil and climate determine the vegetation, and the vegetation rules the supply of organic matter. The organic matter reacts with the soil to control the release of ions, and the ions, particularly nitrogen and phosphorus, control the decay of litter, and hence lie right at the root of the food cycle."

Flows and Sediment Transport

Runoff volumes to streams usually follow patterns of precipitation, with greater amounts at higher elevations, and with great overall variation on a seasonal basis (e.g., Parrett and Hull 1985). In mountainous headwater areas in the West, snowmelt provides most of the annual stream flows, with most occurring from May to July. Minimum stream flows occur during the fall and winter, when stream flows are basically groundwater influxes.

Few continuously recording stream-flow gauging stations have been established in small mountain drainage areas having less than 100 sq mi. In one approach to estimate stream discharges for such systems, Hunt (1963) and Riggs (1969) determined that annual mean discharge at an ungauged site could be estimated with an error of ±10% using ratios between point measurements from ungauged streams and measurements for the same day from a nearby gauged stream site.

In an alternative approach, Parrett and Hull (1985) used regression to relate both mean and long-term annual discharges to drainage area and annual precipitation. Equations were also developed to predict probabilities of flows that would exceed predicted annual flows. While equations develop by such approaches tend to have localized application, similar approaches can be used to estimate stream flows in other ungauged drainages.

Stream flow velocities may be thought of in terms of "subcritical" and "supercritical" flows (Heede 1980). Subcritical flows exert relatively low energies on banks and beds, while supercritical flows can produce highly erosive forces and cause channel damage. Standing waves are commonly associated with supercritical flows.

Erosion and deposition of sediments by moving water are two of the primary processes in developing land forms. Sediment transport in streams is a very complex relationship involving at least 30 variables (Heede 1980). But, in general, the mean particle size transported and sediment mass discharged is proportional to the water volume discharged and the gradient of the streambed. Also, water temperature can influence sediment transport. For example, 40 F water is able to carry two to three times more sediment than 80 F waters (Heede 1980).

Steep drainages in mountain regions pose potentials for high erosion and production of deeply incised channels and greatly steepened valley slopes (Heede 1980). To counter these potentials, natural mechanisms exist that allow streams to adjust channel slopes, which helps to protect streambeds. These mechanisms include (1) bed armoring by gravel and boulders, (2) gravel bars that form transverse to stream flows, and (3) log steps that incorporate fallen timber and associated debris into the streambed. Through such mechanisms streams can reach a "dynamic equilibrium" with their channels (Heede 1981).

During his study of mountain streams in the western United States, Heede (1981) found that transverse gravel bars and log steps created stepped longitudinal profiles with the steps spaced at regular intervals; and when log steps were lacking, the incidence of transverse gravel bars increased. Each of these mechanisms were so efficient at reducing the erosive capacity of the flowing water that flows were nearly stilled in places. Based on this study, Heede suggested that while such natural flow adjustment structures were effectively temporary, their continual, natural replacement created a "dynamic equilibrium" between the stream's erosive forces and the stream's bed and banks. By manipulating numbers of dead and dying trees in the streamside forest, the manager can influence the hydraulic nature of small streams.

Thermal cycles

Water temperatures in streams vary with air temperatures. This means that stream temperatures have daily and seasonal cyclic patterns paralleling air temperatures, but because of the greater density of water and the fact that water freezes at 0 C, temperature extremes are less. Thus, in the northern temperate zone, stream temperatures tend to be cooler than air temperatures during the summer and warmer during the winter. Also, water temperatures tend to warm with distance downstream in response to solar radiation and warm air; the rate of the warming is approximately proportional to the distance traveled (Hynes 1970). In stream reaches predominated by groundwater contributions, stream temperatures tend to reflect the often cooler or warmer groundwater temperatures.

Both cooling and warming trends can be important in triggering either developmental or reproductive changes in aquatic organisms. Thus, daily and seasonal cycles in stream temperature are frequently important in the development and growth of aquatic invertebrates and fish.

Nutrient spiralling

In lakes and terrestrial ecosystems, nutrients move in cycles from organisms to soils or sediments and then back to organisms again. Often these cycles can be repeatedly completed within close proximity of each other. But in rivers and streams, the flow of water causes the cycles to be completed at progressive intervals downstream. This downstream movement of these cycles, which include nutrient processing and transport, has been termed "nutrient spiralling" (Webster and Patten 1979).

While moving downstream, materials can be transferred among environmental compartments where nutrients can be "stored" for varying periods, effectively altering rates for downstream movements. Storage times for different compartments can be relatively long or short. For example, some nutrients incorporate mostly into the tissue of organisms and are released back into the water at relatively slow rates; while other nutrients can be rapidly excreted by organisms, soon becoming available again for use by other organisms. Such differences produce spirals of different spatial lengths for different nutrients.

It has been suggested that shorter nutrient spirals help to establish constancy in stream ecosystems, leading to potentially increased biomasses, spatial heterogeneity through the stream continuum, resistance to external stresses, and ability to rapidly recover from perturbations (O'Neill et al. 1979). Disturbances will tend to disrupt storage mechanisms, increase losses of dissolved nutrients and/or nutrients in sediments, and to increase spiral lengths within streams (Webster and Patten 1979, Newbold et al. 1983, Mulholland et al. 1985).

Currently, a major research emphasis in stream ecology involves determining how different factors can alter spiral lengths in streams.

River Continuum Theory

The "River Continuum Concept" was introduced by Vannote et al. (1980) to describe changes in structure and function through the lengths of river systems. Most simply, the concept proposes that biological systems occupying river systems develop in response to the physical forces present in the drainage. It suggests that biological structure and function at a particular location conform to the river's physical characteristics, which are defined by how the river uses its kinetic energy in achieving a dynamic equilibrium.

The concept further proposes that lotic communities can be divided into three groups based on stream size or stream order: headwater streams (order 1-3), medium-sized streams (order 4-6), and large rivers (order >6). Progressive downstream changes in the physical characteristics associated with the three stream sizes were suggested to produce a continuum of change in sources of energy, sizes of organic (food) particles, and types of organism. According to the concept, headwater reaches tend to be dominated by riparian vegetation, which reduces production by plants in the streams by shading and contributes large amounts of allochthonous (from outside of the system) materials to the energetic base of these systems. These contributions cause the carbon stores in headwater systems to be dominated by coarse particulate organic material. Consequently, invertebrate communities in headwater streams tend to be dominated by organisms that shred coarse matter into finer particles, and by those that collect these drifting particles. Due to the preponderance of materials (and energy) originating outside the system in headwater streams, the ratio of community photosynthesis to respiration (P/R ratio) is generally less than one. This means that more carbon compounds are used by the community than are produced by it; when P/R ratios exceed one, more carbon compounds are produced than are used by the community.

In contrast to headwater streams, algae and other aquatic plants are thought to have greater importance in medium-sized streams. With the increase in autotrophic production in these systems, P/R ratios often exceed one. This shift in the energy base of the system shifts the dominance in invertebrate communities to those organisms capable of collecting particles released by upstream processes, and to those able to graze on the plants produced in the stream.

Further downstream, the energetics of large rivers are increasingly dominated by fine particles released from upstream sources. Here the systems again have P/R ratios of less than one and the invertebrate communities tend to be dominated by collectors.

Through the River Continuum Concept it was further suggested that stability within lotic ecosystems "is achieved by a dynamic balance between forces contributing to stabilization (e.g. debris dams, filter feeders, and other retention devices; nutrient cycling) and those contributing to its instability (e.g. floods, temperature fluctuations, microbial epidemics)" (Vannote et al. 1980, p. 134). Streams that have low physical variability tend to have low biological diversities and high ecological stabilities. Whereas, streams with high physical variability tend to have high biological diversities and complexities that also tends to maintain ecological stability. For example, biological diversity may be relatively low in headwater streams because of low temperatures and/or low diversity in the nutritional base, while the stability may be maintained due to the low daily or seasonal fluctuations in temperatures.

Stability developed by biological communities in stream systems may be thought of as a strategy through which energy or nutrient loss to downstream reaches is minimized; i.e., spiral lengths for nutrients are shortened. Disturbances within streams can disrupt the overall stream continuum by changing conditions over the length of the disturbance to conditions more similar to those occurring either upstream or downstream of the disturbed reach (Vannote et al. 1980).

Subsequent to the introduction of the River Continuum Concept, its original authors plus others have tempered some of its original tenets (e.g., Minshall et al. 1985, Statzner and Higler 1985). Specifically, local influences of climate, geology, tributary inflows, and soil conditions, plus cumulative long-term effects due to man can produce alterations not included within the river continuum as initially proposed. For example, local conditions may preempt the presence of riparian vegetation at some headwater streams. Without sources of coarse particulate matter in headwater reaches, invertebrates adapted to shredding such particles are not likely to be present.

SECTION 3

CHARACTERISTICS OF RESERVOIRS AND THEIR EFFECTS ON STREAMS

MODIFICATION OF RIVER FLOWS AND SEDIMENTATION PATTERNS

Reservoirs convert portions of flowing water environments into new standing water, lake-like environments. Their construction disrupts and disperses the historical kinetic energy patterns in the downstream flow of water. With this dispersal of energy, virtually all sediments carried by river inflows can settle, causing aggradation (lifting of the river valley) upstream from dams (Simons 1979, Heede 1980, Newbury 1985). Often deltas are created by the inflows and reservoirs gradually fill with sediments (Neel 1963). Rates of filling depend on the sediment loads in the influent waters and retention time for these waters in the reservoir.

Waters discharged from reservoirs tend to be nearly devoid of suspended inorganic particles (Neel 1963, Petts 1984). During construction of dams, however, masses of sediments released downstream may be increased by greater than 50% over historical mass flows (Petts 1984). Without proper sediment control practices (cf., Beasley et al. 1984), these construction-related sediments can clog downstream river beds and adversely impact the resident stream biota, as discussed below.

Below reservoirs, historical daily and seasonal flow rates are dramatically altered, as maximum flows are stored to augment minimum flows. With reservoirs constructed for hydroelectric power generation, for example, there is an increased frequency and amplitude in daily downstream pulses, while there is a decrease in the seasonal amplitudes. Also, evaporation from reservoir surfaces reduces the total volumes of water draining from watersheds (Neel 1963).

These alterations in downstream water flows redistribute the historical downstream supply of kinetic energy. In turn, this redistribution can substantially alter the morphology of the downstream channel as it accommodates the new energy regime in the regulated stream. In effect, a new dynamic equilibrium is forced between the stream channel and the regulated flows (Heede 1980, Newbury 1985). In most cases, however, a new equilibrium cannot be reached over the relatively short economic life (ca. 60 years) of most North American water projects, and instabilities in the downstream environments persist (Newbury 1985). Where instabilities cannot be controlled, major shifts to new streambed geometries occur.

Various instream forces contribute to changing the geometry in the downstream channel. "Sediment-hungry" waters discharged from reservoirs rapidly entrain sediments to achieve equilibrium loads; this degrades stream beds and erodes river banks (Simon 1979, Heede 1980). If uncontrolled, the degradation and erosion can endanger downstream structures, including bridges and roads. The extent of streambed degradation, however, is limited by the extent of bed armoring, which is a layer of cobble or rubble too large to be moved by existing hydraulic conditions and which, in turn, protects finer particles that accumulate below the protective armor layer (Simons 1979). With this protection, potentially greater sediment masses erode from stream banks, decreasing bank stability and often destroying resident riparian vegetation. With destruction of river banks, downstream sediment loads can further alter the downstream geometry of stream systems. In extreme cases, such alterations can extend over 300 miles downstream and the severity of their consequences may not appear for at least 30 years following closure of the dam (Newbury 1985).

Beyond these problems, stream regulation can lead to clogging of the bed gravels with fine sediments. Fine sediments are naturally transported by both stream flows and intra-gravel flows through the upper layers of poorly graded gravel stream beds. Gradually, these fine particles tend to sink deeper into the bottom, due both to gravity and to sieving by intra-gravel flows. In unregulated streams, natural peak flows during seasonal or storm related runoff events mix the upper bed layers and flush accumulated fine sediments from the deeper layers. But where natural peak stream flows are regulated, fine sediments can accumulate in the deep layers, clogging the free flow of water. This can adversely impact the intra-gravel habitat, habitat important to benthic invertebrates, incubating fish eggs, and rearing larval fish, as discussed in a later section.

THERMAL AND HYDROLOGICAL PATTERNS WITHIN RESERVOIRS

While thermal stratification is often the dominant physical force in lakes, water flow is often the dominant physical force within reservoirs. This is not to say that thermal characteristics of reservoirs are not important. In fact, thermal patterns in reservoirs can be very similar to those seen in natural lakes; and the flow of water through reservoirs is often guided by thermal patterns within reservoirs. Also similar to lakes, reservoir waters are primarily heated both by solar radiation and through heat advection from inflowing waters; about 20% of the heat within a reservoir can result from the latter mechanism (Whalen et al. 1982).

One approach to estimating whether a reservoir will stratify is to use the densimetric Froude number (F). When F is less than 1/Pi, no stratification is expected (Canter 1985):

$$F = (320 \times L \times Q) / (D \times V)$$

where L = reservoir length (m),

D = mean depth of reservoir (m),

Q = volumetric discharge through the reservoir (m^3/sec), and

V = volume of reservoir (m^3).

Thermal stratification is common in reservoir waters (e.g., Neel 1963), particularly during the summer. In fact, thermal stratification has been observed even in high-elevation, headwater reservoirs (e.g., LaBounty et al. 1984, Marcus 1987a). Thermal patterns in reservoirs and lakes with surface outlets, however, can contrast markedly with those found in reservoirs with deep-water (or hypolimnetic) outlets (Wright 1967, Martin and Arneson 1978). In systems with surface-level outlets, the heat contained in surface water layers, which are heated by solar radiation, is relatively rapidly discharged downstream. In contrast, those systems with deep-water outlets act as "heat traps" (Wright 1967). Here, solar radiation again heats the surface waters, but these heated waters are stored as the deeper, cooler waters are discharged. The process can cause greater masses of warmer waters to accumulate in the upper layers of deep-release reservoirs than occur in similar reservoirs or lakes having only surface outlets. One consequence of these different patterns is the relatively more rapid downward movement of the thermocline during the summer in those reservoirs with deep-water outlets.

Perhaps the most significant aspect of summer and winter thermal stratification in reservoirs is its influence on water flow through reservoirs (Neel 1963, Wright 1967, Marcus 1987a). Upon entering a reservoir, tributary waters may flow through the reservoir in a "river" near the surface as "overflows," near the bottom as "underflows," or midway through the water column as "interflows"; or they may move down the reservoir enmasse as discrete bodies of slowly moving water following prior inflows, while mingling somewhat with waters both ahead and behind (Neel 1963, Wunderlich 1971).

Which path inflows follow through reservoirs depends on the relative densities of both the inflow waters and the existing reservoir waters. These densities are, in turn, mostly defined by their relative temperatures. But factors other than temperature, including salinity (total dissolved solids), can also influence water density and flow patterns (Wunderlich 1971). Also other factors, in particular wind, can have considerable periodic influences on the movement of water in lakes and reservoirs (Hutchinson 1957).

In a study of a montane, headwater reservoir, Marcus (1987a) found that underflows and overflows both occurred at times during summer stratification. At other times the inflows appeared to mix with reservoir waters shortly after entering the reservoir. In general, though, the movement of influent waters appeared most related to thermal densities. The relatively low dissolved solids concentrations in influent waters and the relatively short holding times, which eliminate salinity increases due to evapoconcentration, both precluded establishment of salinity related density currents. Strong-wind induced mixing disrupted the thermal stratification in mid-summer by longitudinally mixing the reservoir.

In summary, high-elevation headwater reservoirs can stratify, developing an identifiable metalimnion for at least a portion of the summer season (LaBounty et al. 1984, Marcus 1987a). But these reservoirs are also subject to substantial wind-induced mixing. Thus, extended durations of stratification are unlikely to persist during the summer. During the winter, stratification can again occur in these reservoirs, but little is known of its potential effect. Due to the lack of data, questions remain about whether these systems will develop strong stratification. If so, stratification potentially will be most severe during the winter under ice cover or during the summer under long periods of calm winds.

MODIFICATIONS TO DOWNSTREAM THERMAL PATTERNS

The heat content of waters flowing into reservoirs is modified continually up to the time of discharge from the reservoir. This substantially modifies historical temperature regimes downstream. The extent of this modification and of its influence in the stream below the reservoir depend primarily on the depth of the reservoir outlet.

As is also the case for the volume of water flowing downstream of reservoirs, amplitudes of both daily and seasonal temperature fluctuations are moderated due to strem impoundment. In fact, no daily thermal cycles exist for waters in the initial stream reaches below reservoirs having deep-water outlets. Daily fluctuations, however, do tend to increase with distance downstream below the dam, especially as tributary inflows contribute greater influence on the streams (Ward 1974).

Because reservoirs and natural lakes that have surface water outlets release warmed water relatively soon after heating, temperatures in downstream waters can often be considerably warmer than upstream waters during much of the spring, summer, and early fall (Wright 1967, Martin and Arneson 1978). But in those reservoirs having deep-water outlets, while heated surface waters accumulate in the upper layers during the spring and summer, the cooler, deeper waters are discharged (Wright 1967, Martin and Arneson 1978). Then during the fall and winter, cool influent waters tend to be stored, while the relatively warmer waters are discharged. Consequently, compared to temperatures in tributary inflows, river waters downstream of reservoirs tend to be cooler in the spring and summer, while being warmer in the fall and winter. This warming of downstream waters during the winter can alter or even prevent downstream winter ice formation (Neel 1963).

In one example of alterations in the seasonal thermal cycle, the Bighorn Reservoir in Wyoming and Montana delayed the seasonal heating and cooling. The downstream cycle was about two months behind the upstream cycle, and the summer peak temperatures were 8 to 14 C cooler and the winter temperatures 2 to 3 C warmer than temperatures of the inflows (Soltero et al. 1973). Finally, it should be noted that despite the alterations in downstream temperatures, mean annual temperatures may not be substantially different as a result of impoundment (Ward and Stanford 1979b).

In summary, modifications that reservoirs produce on thermal regimes in downstream waters may be grouped into six general catagories (Ward and Stanford 1979b): (1) increased diurnal consistency, (2) increased seasonal consistency, (3) summer depression, (4) summer elevation, (5) winter elevation, and (6) pattern alteration. Existing evidence indicates that these patterns are all likely to occur in high elevation streams below headwater reservoirs (Ward 1974, Marcus et al. 1978, Schillinger and Stuart 1978).

CHEMICAL DYNAMICS WITHIN RESERVOIRS

Salinity

Effects of reservoir passage on dissolved salt concentrations appear to vary among reservoirs. A number of early research efforts on reservoirs reviewed by Wright (1967) indicated that the influence of both evapoconcentration and irrigation return flows tended to increase salinity in reservoirs. Based on this review, Wright (1967) suggested that appropriate placement and operation of outlets for reservoir discharges can prove valuable in manageing salinity in discharge waters. Other reviews, however, contrast with the findings of Wright and conclude that natural salinity removal processes (e.g., precipitation and settling) may predominate in reservoirs (Neel 1963, Messer et al. 1981).

Research that indicates reservoirs can act as salinity traps include studies on Bighorn Lake, Wyoming; Hebgen Lake, Montana; and Tongue River Reservoir, Montana (Soltero et al. 1973, Martin 1967, Whalen et al. 1982). In fact, Messer et al. (1981) suggested that natural salinity reduction processes in reservoirs might be used to decrease salinity levels in the lower Colorado River drainage. Included among potential processes that they suggested as causing salinity reductions are (1) precipitation of calcium carbonates, associated with reductions in dissolved carbon dioxide concentrations due to photosynthesis; (2) co-precipitation of calcium with clay following ion exchange of calcium with sodium adsorbed to clay particles; and (3) coagulation of colloidal calcium carbonate and precipitation of larger particles. They concluded that the degree and mechanism of natural salinity removal in reservoirs depend on site-specific physical, chemical, and biological characteristics for the reservoir in question.

Nutrients

Cycles for nutrients in reservoirs can be quite similar to those in lakes, discussed above. Oxygen contents in the hypolimnion of some productive reservoirs can be depleted by the decomposition of organic materials settling from the epilimnion (Neel 1963). In fact, reduced oxygen concentrations have been observed in the hypolimnion of one montane, head-water reservoir that had no major man-made source of nutrients (Marcus 1987a).

Differences do exist, however, between reservoir and lake chemistries. First, construction and filling of reservoirs flood the former terrestrial environments. Subsequent leaching of chemicals from the flooded soils and from rotting organic forest debris can profoundly affect water quality in the reservoir. Decaying forest materials consume dissolved oxygen and elevate carbon dioxide concentrations, while leaching can extract dissolved nutrients and organic compounds from the flooded plants and soils. As a result, heavy algae growths can be supported, undesirable levels of color and odorous substances may be produced, and conditions that enhance aquatic productivity or that can even be toxic to aquatic life may result (Sylvester 1965, Canter 1985).

Comparisons between areas in a reservoir where forest vegetation was left standing to neighboring areas where it was removed showed the former to be more eutrophic than the latter (Hendricks and Silvey 1977). These researchers also noted that increases in the overlying water depths tended to decrease the influences of flooded terrestrial materials on reservoir water quality. Over time, influences from the flooded terrestrial environment on the overlying water quality decrease (Sylvester 1965).

Similar to differences between reservoir and lake patterns for heat transport, nutrient regimes in reservoirs can differ markedly from those found in lakes depending on the relative vertical location of the outlet through which reservoir waters are discharged (Wright 1967, Martin and Arneson 1978). In effect, reservoirs with deep-water outlets act as nutrient sources, while those with surface outlets act as nutrient sinks with respect to the downstream waters. Recalling the discussion for lakes, nutrients contained in organisms and their wastes mineralize as they sink through the water column, with the bulk of the mineralized nutrients becoming stored in the deeper waters and sediments. Then with spring and fall mixing, these nutrient are recycled up to water layers nearer the surface, and again become available for organic production. In contrast, reservoirs having deep-water outlets continually discharge deeper waters and the nutrients they contain (Wright 1967). This continually removes nutrients from the reservoir, preventing them from being recycled to producers in the upper water layers.

ALTERATIONS IN DOWNSTREAM WATER QUALITY

Passage of water through reservoirs tends benefit water quality by reducing turbidity, hardness, and coliform bacteria levels; and by oxidizing organic materials within the reservoir, potentially reduces downstream biochemical oxygen demand (Canter 1985). But reservoir passage also can be detrimental to water quality by lowering reaeration rate for the water during storage, allowing buildup of inorganic chemicals in the hypolimnion that can be released to enrich downstream waters, and enhancing potentials for algae blooms (Baxter 1977, Canter 1985).

Perhaps the worst potential consequences of reservoir storage occur in the hypolimnion during thermal stratification when dissolved oxygen decreases, anaerobic waters develop, and iron, manganese and hydrogen sulfides can dissolve from the bottom deposits (Neel 1963, Canter 1985). Also, by allowing surface discharges from reservoirs to drop considerable distances into plunge pools, reservoir effluents can develop supersaturated concentrations of dissolved gases, which can have substantial adverse effects on resident stream biota (Petts 1984).

BIOLOGICAL RESPONSES TO THE PHYSICAL-CHEMICAL ALTERATIONS

Biological processes can have relatively little importance in reservoirs with high flushing rates (Toetz 1976). But, as discussed above, reservoir environments are different from either streams or lakes. Consequently, they provide new opportunities for development of biological communities that differ from communities found in either of the other systems. Thus, while slowing the flow of river waters causes river-bourn sediments to settle, it also permits development of reservoir phytoplankton and zooplankton communities, communities otherwise limited to lakes and large, slowly flowing rivers. Many of these planktonic organisms are discharged from reservoirs, where they can contribute to the food chain and organic enrichment in the downstream reaches (Petts 1985).

Often, in the first few years following filling, reservoir waters become highly productive, yielding relatively thick growths of algae (Neel 1967, Funk and Gaufin 1971). This productivity largely results from high nutrient inputs to reservoir waters through leaching from soils and from decaying plant materials inundated by the reservoir. Within a few years after filling, when most of these nutrients have leached from the flooded terrestrial environment, and in the absence of other external nutrient inputs, productivities often drop.

Geological chemistries within watersheds also can produce conditions leading to degradation of water quality in reservoirs. For example, watersheds dominated by volcanic rocks tend to be relatively low in most dissolved minerals except phosphorus (Dillon and Kirchner 1975, Marcus 1980). Drainages from such geologic formations can cause relatively high concentrations of dissolved phosphorus in reservoirs. Then, in the relatively warm lentic conditions produced by the reservoir, nuisance growths of nitrogen-fixing algae or bacteria can develop, resembling algae blooms normally associated with organic eutrophication. Such events apparently led to large blooms of Aphanizomenon and Anabaena in a montane headwater reservoir in Montana (Marcus 1987a). Since these species are not adapted to growth in streams, problems produced by them do not develop until reservoirs are filled.

In addition to these source of nutrients to reservoirs, enrichment can result also from shoreline septic systems, cattle grazing, timber harvesting, mining, plus other industrial activities (cf., Baxter 1977, Schillinger and Stuart 1978). Discharge of nutrients leached from the decaying terrestrial vegetation and from mineralized organic materials settling from the epilimnetic to hypolimnetic zones can enhance productivities in downstream environments (e.g., Pfitzer 1954, McConnell and Sigler 1959, Neel 1963, Marcus 1980).

Growths of emergent aquatic macrophytes (plants) along shorelines, which are common in many lakes, are uncommon in most reservoirs because fluctuations in reservoir water levels expose and disturb the near-shore areas inhibiting plant establishment and propagation. Also, the suspended materials settling from river inflows can restrict light penetration and coat submerged aquatic plants, reducing or prohibiting photosynthesis and growth by submerged plants. In other reservoirs where sediment loads are less, nuisance growths of submerged plants can develop; in fact, harvesting of these plants has been proposed as a method for reducing nutrients in shallow, highly productive reservoirs (Boyd 1971).

River impoundment can eliminate from the flooded areas most of those benthic macroinvertebrates particularly adapted to flowing water. Development of benthic invertebrate communities in reservoirs can be limited further by (1) deposition of silt onto the bottom substrates, (2) stilling of flowing waters, (3) fluctuations in shoreline water levels that prevent development of near-shore (littoral) communities, (4) depletion of oxygen in hypolimnetic waters, (5) increased hydrostatic pressure, and (6) decreased light intensities that decrease primary production and visibility (Isom 1971). Even under the most adverse conditions, however, some seasonal colonization can occur in shallow water littoral areas.

Despite exclusion of most invertebrates adapted to flowing waters, reservoirs often develop abundant communities of benthic invertebrates adapted to standing water environments (Baxter 1977). Frequently these communities are limited primarily to forms that burrow into the sediments. In particular, members of the midge family (Chironomidae) are often the first and most dominant colonizers of the reservoir invertebrate community (Baxter 1977). Also, high densities of aquatic insects can develop among periphyton growths colonizing submerged trees and those aquatic plants that do colonize reservoirs (Cowell and Hudson 1967). In her study of drawdown effects on the fauna in small mountain reservoirs, McAfee (1980) found no statistically significant differences among benthic invertebrate communities in reservoirs having substantial drawdown versus those having little drawdown. Interpretation of her data was complicated, however, by high variability and the relatively low densities for invertebrates inhabiting the studied reservoirs.

Benthic macroinvertebrates downstream of reservoirs appear predominately affected by alterations in seasonal temperatures, especially those produced by reservoirs having deep water outlets (Ward 1974, 1976; Ward and Stanford 1979b). Specifically, three alterations appear to produce the greatest impact on downstream benthic macroinvertebrates:

1) Greater seasonal consistency of temperatures reduces the number of resident stream species by providing competitive advantages to one or a few species. In terms of ecological theory, environments with greater thermal consistency may approach "equilibrium" conditions where fewer species are able to exist; whereas, environments with greater environmental variability permit relatively more species to exist because conditions under which competition occurs are always changing.
2) Seasonally colder summer waters can slow nymphal development and prevent maturation and life cycle completion for some invertebrates. For example, this can delay insect emergence by months, causing insects that would normally emerge in late summer or fall to emerge when the air is cold and the ground is snow covered, significantly interfering with normal feeding and mating activities in adult insects.
3) Similarly, the seasonal warm winter temperatures can speed morphogenic development in some invertebrates and again cause

species to emerge into a cold, snow-covered environment. But warm winter waters also can inhibit development in other invertebrates by preventing occurrence of sufficiently low temperatures necessary for some physiological cues during their development.

High yield fisheries also are often noted for some reservoirs, particularly during periods of high overall biotic productivities in the first years following initial filling of reservoirs (e.g., Varley et al. 1971). The high fisheries productivity is often associated with high planktonic productivities, high densities of benthic invertebrates, and extensive cover provided by any terrestrial vegetation left within the reservoirs (Baxter 1977). But reservoirs also can harm fisheries by blocking migration routes, by lowering shoreline water levels below nesting and spawning areas, and by developing temperatures too warm and/or oxygen levels too low for fish (Baxter 1977). Yet, despite these potential harms, reservoirs present great opportunities to develop fisheries that are potentially more productive and diverse than found in the pre-impounded river systems.

In addition to fish, other vertebrates can be affected by reservoir construction. In particular, when reservoirs are managed appropriately waterfowl can benefit from wetlands produced when terrestrial areas are flooded; and reservoirs can provide valuable watering opportunities and grazing areas for other wildlife during drawdown (Baxter 1977). But reservoirs also destroy habitat for the resident terrestrial organisms. For example, reservoirs planned for construction in Wyoming are projected to destroy valuable waterfowl nesting areas, plus flood valuable summer and winter ranges, and block migration route for big game animals (Wyoming Game and Fish Department 1984, 1985).

DISRUPTION OF THE STREAM CONTINUUM

In an attempt to integrate our current understanding and to help direct future research on the effects by reservoirs on rivers and streams, Ward and Stanford (1983) have introduced the "serial discontinuity concept of lotic ecosystems." Through this concept it is proposed that when a reservoir is constructed on a flowing water system, the characteristics of the stream below the reservoir are shifted longitudinally to more resemble characteristics occurring some "displacement distance" (DD) either upstream or downstream (lower or higher stream order) prior to the construction of the reservoir. The direction and intensity of DD vary dependent on the position of the dam along the longitudinal river continuum and on the specific stream characteristics in question.

For reservoirs on headwater streams, Ward and Stanford (1983) proposed that the downstream environments would become more like upstream, lower order streams in terms of environmental predictability and environmental heterogeneity. The streams would become more like downstream, higher order streams in terms of the ratio of community photosynthesis to respiration, plankton populations, biotic diversity, nutrient levels, and the diversity of organic compounds. And they would remain relatively unchanged in terms of light reaching the stream bottom, substrate size, daily changes in temperature, seasonal changes in temperature, annual water discharge, growth of submerged aquatic plants, and availability of nutrients. Research on a montane, headwater reservoir in Montana suggests, however, that substrate size and both daily and seasonal temperature changes may more reflect upstream conditions, while nutrient concentrations may be more similar to downstream conditions (Marcus et al. 1978; Schillinger and Stuart 1978; Marcus 1980, 1987a).

For all reservoirs, Ward and Stanford (1983) suggest that maximum spiral lengths will be shortened in the downstream waters. Newbold (1985), however, has suggested that stream impoundments can either increase or decrease spiral lengths for nutrients in streams, depending on the downstream water velocities and plankton densities in discharged waters. Slower water velocities will tend to increase nutrient exchange rates between the water column and the sediments, causing spiral lengths to shorten; while higher plankton densities will tend to retain nutrients in water columns longer, causing spiral lengths to lengthen. Certainly, additional research on reservoir-stream systems is needed to better evaluate these proposed relationships and to advance the theoretical understanding of these systems, which, in turn, can improve the bases for managing these systems.

SECTION 4

DESIGN AND MANAGEMENT CONSIDERATIONS FOR RESERVOIR-STREAM SYSTEMS

Construction and operation of reservoirs necessarily involve consideration of various design and management criteria. Those important for developing fisheries potentials in reservoir-stream systems can include determinations for the height of the outlet from the reservoir, potential downstream changes in channel morphology, needs for flow to flush potentially harmful downstream accumulations of sediments, possible deterioration of shoreline riparian communities, concerns about endangered species, factors potentially limiting the biotic productivities, potential management alternatives for the reservoir-stream systems, and potentials for both within reservoir and downstream fisheries. This section discusses these considerations, beginning first with a brief introduction to the approach through which many reservoirs are currently developed in Wyoming.

DEVELOPMENT OF RESERVOIRS IN WYOMING

The Wyoming Water Development Program

Most proposed developments of reservoirs within Wyoming proceed under The Wyoming Water Development Program, which is administered by two state agencies--the Wyoming Water Development Commission (WWDC) and the Water Development Division of the Economic Development and Stabilization Board (EDSB). WWDC assembles water development plans, administers the Ground Water Grant Program, and manages the New Development and Rehabilitation Programs through the planning stages. It also produces and compiles data necessary to evaluate project feasibilities, recommends project developments, and presents the results of these efforts to the Governor and Legislature (WWDC 1986).

For projects approved by the Governor and Legislature, the EDSB becomes the lead state agency for completing water development projects, developing and administering project contracts and loans, inspecting projects during their various stages of development, and fulfilling the State's ownership and operational responsibilities for the developments.

In the New Developments Program, WWDC water projects typically follow four development levels:

- Level I--Reconnaissance Studies. Area- or basin-wide development plans or preliminary analyses are prepared and compared to previously identified development alternatives.
- Level II--Feasibility Studies. Sufficiently detailed information is compiled and evaluated to determine the feasibility of and the ranking of developments relative to other proposed water projects.
- Level III, Phase I--Conceptual Design and Development. The operation plan, configuration, budgets, and schedules are developed adequately to request construction funding, prepare permit applications,

finalize water rights applications, identify benefits, and acquire land and easements.

- Level III, Phase II--EIS and Final Design. After commitment by the State and project sponsors to complete a project, the environmental impact statements, construction documents, contract documents, bidding plans, and applications for construction permits are completed. In addition, titles or options on the necessary lands and easements are obtained. During this time, construction and water supply contracts for the project can be formalized.
- Level IV--Construction. Under this phase, agreements between the State and sponsor(s), land acquisition, permitting, bidding, engineering, and actual construction of the project are completed.

Similar phases of development are included in the Rehabilitation Program, which apply to improvement of water projects completed and in use prior to 1970. Suitable projects include those to improve dam safety, to decrease operation and maintenance costs, and to provide more efficient uses for existing water supplies.

In 1986, new water development projects proposed in Wyoming included five at Level I, 11 at Level II, 15 at Level III, and five at Level IV. Under the rehabilitation program there were four projects at Level II, one at Level III, and 15 at Level IV. Seventeen of the new water development projects include development or potential development of headwater reservoirs. Characteristics of several projects that include potentials for reservoir construction or expansion are shown in Table 1.

Potential Conflicts with Wildlife in Development of Wyoming Reservoirs

Wyoming is a state renowned for its wildlife resources, but many potential sites for reservoirs, particularly for head-water reservoirs, are in areas used historically by wildlife. Consequently, various potential conflicts exist between reservoir development and wildlife use.

In recognizing that public wildlife resources could be lost as a result of reservoir development, the Wyoming Game and Fish Commission (WGFC 1985) has established a mitigation policy directing the Wyoming Game and Fish Department (WGFD) to "actively pursue the resolution of conflicts arising between human activities and wildlife habitat." While acknowledging that various legitimate demands on land and water resources exist, the Commission emphasizes that under statutory responsibility the WGFC is "the principal proponent for the maintenance and perpetuation of wildlife in Wyoming" (WGFC 1985).

Under this mitigation policy the WGFD is directed to (1) identify and quantify potential impacts associated with developments; (2) recommend measures to avoid impacts to wildlife; (3) develop and recommend specific measures to offset unavoidable impacts; and (4) develop and recommend appropriate mitigation for individual developments. Mitigation is defined by the Commission "as the avoidance, minimization or compensation of adverse impacts that arise as a result of a given development action." Two categories of mitigation

Table 1. Data for representative reservoirs proposed in Wyoming.

RESERVOIR	DRAINAGE	VOLUME (Acre Feet)	MAX. DEPTH (Feet)	CREST HEIGHT (Feet)	REFERENCE
Lake Adelaide	Big Horn River	4,550		9,287	ESA Geotechnical Consultants 1986a
Clarks Fork	Yellowstone River	522,850	235	4,500	ESA Geotechnical Consultants 1986b
Middle Fork	Powder River	59,600	190	5,049	Harza Enginerring Co. 1986.
Crazy Woman	Powder River	80,000	100	4,173	Harza Enginerring Co. et al. 1983.
Mid/Lower Clear Creek	Powder River	19,200	126	3,640	Harza Enginerring Co. et al. 1983.
Upper North Laramie River	Laramie River	3,944	52	7,124	USDA Soil Conservation Service 1980
Deer Creek	North Platte River	65,785	280	6,720	Banner Associates Inc. 1984
Jack Creek	Little Snake River	38,410	200	8,475	Banner Associates Inc. 1980
Upper Savery	Little Snake River	59,575	125	7,172	Banner Associates Inc. 1980
Sandstone	Little Snake River	52,040	200	6,965	Banner Associates Inc. 1980
Pot Hook	Little Snake River	61,550	160	6,709	Banner Associates Inc. 1980
Three Forks	Little Snake River	131,980	250	7,168	Banner Associates Inc. 1980
Upper Smiths Fork	Bear River	125,000	281	6,896	G.B.R. Consultants Group Inc. 1985
West Fork	Bear River	12,000	125	8,362	Rollins Brown and Gunnell Inc. and Forsgren-Perkins Engineering 1985

actions are listed: (1) resource maintenance--measures avoiding adverse impacts through project planning, and (2) resource compensation--measures restoring, rehabilitating, replacing, or directly compensating for unavoidable impacts to wildlife.

The Commission's mitigation policy further establishes a matrix of habitat criteria, descriptions, and mitigation categories to be followed by the Department in evaluating proposed projects potentially conflicting with wildlife use (Table 2). For each of the four mitigation categories, specific habitat management objectives are stated: (1) irreplaceable habitat--no loss of existing habitat value; (2) high value habitat--no net loss of in-kind habitat value; (3) moderate value habitat--no net loss of habitat value while minimizing loss of in-kind habitat value; and (4) low value habitat--minimize loss of habitat value. It is the position of the Commission that "it is more desirable to maintain fish and wildlife resources rather than attempting to compensate for adverse impacts on them" (WGFC 1985).

Based on the this policy, any stream mileage lost in Wyoming due to reservoir development must be compensated for by enhancement of the undisturbed stream. No compensation credit for stream habitat losses results from new habitat created within a reservoir. To evaluate potential fish and wildlife impacts, Department personnel survey the habitat associated with each proposed project and develop individually applicable recommendations to mitigate potential project impacts. Included among the procedures used to evaluate present and projected values for aquatic habitats are surveys of present fishery stocks, analyses using U.S Fish and Wildlife Services Instream Flow Incremental

Table 2. Criteria established by the Wyoming Game and Fish Commission (1985) and used by its Game and Fish Department in evaluating projects for appropriate mitigation.

CRITERIA	DESCRIPTION	MITIGATION CATEGORY
Species Present	Federally listed threatened or endangered species	Irreplaceable
	State rare fish	High
	Native game fish	High
	State protected fish	High
	Non-native game fish	Moderate
	Trophy game mammals	High
	Big game animals	High
	Furbearers	Moderate
	Migratory birds of high Federal interest	Moderate
	Game birds	Moderate
	Non-game fish and wildlife	Low
	Small game animals	Low
Wyoming Game and Fish Department Stream Classes	1	Irreplaceable
	2	High
	3	Moderate
	4	Low
	5	Low
Fisheries Management Concept	Species, Trophy	High
	Wild (native trout)	High
	Wild (non-native game fish)	Moderate
	Basic yield	Moderate
	Put-and-take	Low
Special use	Irreplaceable State or Federal function	Irreplaceable
	Intensive public use	High
	Replaceable State or Federal function	Moderate
	Moderate to high public use	Moderate
	Little public use	Low
Habitat Type	Habitat for Critical or endangered species	Irreplaceable
	Critical habitat	Irreplaceable
	Riparian habitat	High
	Raptor nesting area	High
	Non-critical seasonal habitat	Moderate
	Parturition area	Moderate

Method to evaluate instream flows (Milhous et al. 1981), Binns' (1979, 1982) Habitat Quality Index to evaluate stream fisheries habitat, and Whitworth's (1984, 1985) Reservoir Quality Index.

HEIGHT AND DESIGN OF RESERVOIR OUTLETS

The principal strategy available to manage water quality in streams below reservoirs and, to some extent, within reservoirs themselves is by discharging water from those levels within the reservoir that contain the water quality appropriate for meeting management objectives. To have maximum flexibility in applying this strategy requires installation of continuously variable, multi-level outlets. But a variety of outlet options exist.

Outlets designed to spill water over the tops of dams have the advantages of being relatively inexpensive and easy to construct, but they also limit the ability to draw waters down below the spillway crest. Perhaps the most common outlet design is to place a single level outlet near the base of the dam. This strategy has several potential or perceived advantages as a result of permitting complete or nearly complete discharge of all reservoir waters (Nelson et al. 1978). With total drawdown of the reservoir possible, water supply, safety, and management options are maximized; migratory fish such as kokanee salmon can be released downstream; procedures to control or eliminate undesirable fish populations are simplified; and cold waters can be discharged, enabling establishment of cold water fisheries in stream reaches where only warm water species might otherwise exist (Nelson et al. 1978). But deep-water outlets also present potential disadvantages through the discharge of waters possibly devoid of oxygen, enriched in nutrients, or too cold for normal development of downstream aquatic residents (Wright 1967, Ward 1974, Brocksen et al. 1982). Deep-water outlets also can cause unwanted discharge and damage to reservoir fisheries and plankton populations (Nelson et al. 1978).

In addition to management options available through selecting discharge depths, two other structural options in outlet designs are available for use in effecting water quality changes downstream of reservoirs (Nelson et al. 1978). The first option is to construct stilling basins to intercept discharged reservoir waters, helping dissipate high energies contained in these waters, reducing excessive scour and erosion in downstream channels, and helping to control levels of dissolved gases in downstream waters.

A second technique, used mostly to control dissolved gas concentrations, is construction of spillway deflectors to mix discharged waters with the atmosphere. Without control of dissolved gases, reservoir discharges can have either low concentrations of dissolved oxygen or supersaturation of gases. Supersaturation results when waters are discharged at high velocities from reservoirs into deep plunge pools, entraining high concentrations of atmospheric gases, particularly nitrogen. Supersaturation can produce a condition called "gas-bubble" or "pop-eye disease" that often leads to death in fish and some aquatic invertebrates; this problem is particularly prevalent below reservoirs on the Columbia River, where it has been intensively researched (Petts 1984).

For most high-elevation, headwater reservoirs, due to their relatively shallow depth and homogenous water qualities, the rather costly discharge design using continuously variable, multi-level outlets is probably not necessary. In these systems all that may be needed are options to discharge waters at fixed points from near the surface and from one deeper level; or from one, two, or three deeper levels. Selection of the appropriate design will, in most cases, be based on site specific characteristics, depending largely on the reservoir's maximum depth and on expected water qualities within the reservoir. Management options for reservoir discharges that can potentially benefit both within reservoir and downstream habitats are discussed in later sections.

CHANGES IN CHANNEL MORPHOLOGY

Flood frequencies and size distributions of suspended particles are the two dominant forces defining stream channel morphologies (Petts 1984). A primary impact, then, produced by dams as well as by diversion structures is disruption of these two dynamic forces. But, while dams and diversions do disrupt the energetics of stream systems, we are very limited in our abilities to predict accurately how stream morphologies will respond to changing flows (Simons and Milhous 1981).

We know, as discussed in a previous section, that increases in flow velocities will proportionally increase erosion and the mass of sediments carried, while decreases in flow velocities will similarly decrease the ability of streams to carry sediments. We also know that when flowing waters are devoid of sediments, they scour sediments from the stream banks and beds until an equilibrium load is suspended in the water. And we can quantify how the kinetic energy of flowing water will change with changes in flow velocities, and how these changes can interact with stream width and depth to change the abilities of streams to transport sediments (Leopold and Maddock 1953, Osterkamp et al. 1983, Knighton 1984). Despite this knowledge, we currently cannot quantitatively predict how these changes will alter the morphology of the channel downstream following a disturbance, even though such alterations can be substantial (Heede 1980, Simon and Milhous 1981, Petts 1984).

Certainly, many techniques exist that can be used to estimate potential changes in stream channels below impoundments and diversions. For example, some models currently available to estimate stream bed degradation rates include the Morphological River Model, the U.S. Army's Corps of Engineers HEC-6, and variable exponent sheer-stress models (Bettess and White 1981, HEC 1977, Petts 1984, Osterkamp et al. 1983). But the lack of consensus among experts on the best approach to predict changes in downstream channel morphologies is illustrated in the diversity of approaches applied and conclusions derived by twenty professional hydrologists using three examples of reservoir and diversion projects (Simon and Milhous 1981). Despite this lack of agreement on appropriate methods to quantify such changes, it is important that resource managers have a qualitative understanding about the types of morphological changes likely to occur in downstream channels following impoundment.

Streambeds respond to reservoir construction with downstream degradation (characterized by increased stream depth and increased straightening) and/or upstream aggradation (characterized by decreased stream depth and increased sinuosity). Following impoundment, degradation of the downstream channel normally occurs, even when stilling pools are used to reduce energy in discharged waters (Petts 1984). In comparison, erosion is always greater downstream of reservoirs than in natural streams having otherwise similar characteristics (Petts 1984). Typically this erosion begins in the stream reaches nearest the dam and continues until limited by factors within the channel. In mountain streams erosion processes can progress gradually downstream at the rate of ten kilometers per year (Petts 1984).

Downstream erosion of the stream bank and bed can be slowed by the accumulation of materials resistant to erosion, including large particulates and/or cohesive silts and clays that clog the streambed. Strongly established riparian plant communities can be important deterrents of bank degradation (Graf 1979). Erosion rates are also slowed when stream flow energies are reduced. Flows can be slowed and energies dispersed through (1) erosional cutting that produce shallower stream gradients or large cross-sectional areas; (2) development of roughened stream beds that increase frictional resistance; and (3), most often, restrictive regulation of discharges from reservoirs. Tributary inflows to regulated streams also can contribute necessary suspended materials, helping the stream to achieve an equilibrium sediment load, hensce reducing the erosive nature of reservoir discharges. But in general, stream bed armoring, which usually begins near the dam and then extends downstream, is frequently the principal factor limiting continued stream bed erosion (Petts 1984).

FLUSHING FLOWS AND DOWNSTREAM SEDIMENT ACCUMULATIONS

Downstream dewatering and desiccation are undoubtedly the worst of the possible adverse impacts on the stream and riparian habitats resulting from stream impoundment. A second potentially destructive impact on downstream habitats is sediment clogging of the stream bed. Relationships between accumulation of fine sediments in stream beds and the ability of stream flows to flush out these sediments have been thoroughly reviewed by Reiser et al. (1985). This section draws liberally from that review.

Gravels stream beds provide important protective habitats for benthic invertebrates, for incubation of fish eggs, and for rearing of larval fish. Unrestricted intra-gravel flows carry not only suspended materials but also carry dissolved oxygen to and metabolic wastes away from the organisms within the sediments. As bed densities of fine sediments increase, abilities to maintain life within the stream bed decrease. For example, accumulations of fine sediment in spawning and rearing areas have been repeatedly found to be inversely related to fish survival and abundances in streams (Reiser et al. 1985).

Overall, periodical high stream flows that flush fine sediments from deeper bed layers are necessary to maintain the channel and riparian habitats. They prevent vegetation encroachment into the channel, and maintain and enhance fisheries habitat. Movement of sediments in a stream is dependent on (1) the availability of sediment in the drainage, and (2) the ability of streams to transport sediment (competency). Water development projects can alter either, but competency is most affected by regulated flows (Reiser et al. 1985).

Considering the biology of the stream, flushing flows are needed whenever sediment accumulations in streams exceed historic levels and begin to adversely affect aquatic habitats, interfering with life history functions.

A wide variety of methods are available to measure accumulations and changes in accumulations of fine sediments within the streambed. While many are good, no method or group of methods presently stand out as the best approach (Reiser et al. 1985).

Similarly, at least 15 potentially useful methods exist to determine magnitude, timing, and duration of flows necessary to flush deep sediments from streambeds, but again no method is clearly superior (Reiser et al. 1985). One of the simplest approaches is the Tennant (1975) method, which is the most common method used in the western U.S. (Reiser et al. 1985). Under this method an adequate flushing flow is defined as 200% of the average annual flow. However, no recommended duration for an adequate flushing flow is provided by this technique. Many state resource agencies recommend a period of 14 days (Reiser et al. 1985).

An alternative approach has been applied in Wyoming by Wesche et al. (1977). Using information from the literature plus their own field data and observations, they recommended flushing flows for six Wyoming streams. Basically, they recommended that bankfull flows needed to be maintained for three days to obtain adequate flushing of sediments from these stream. Subsequent to these recommendations, Nature fortunately "provided" to one of the streams a series of three flushing flows that met or exceeded the magnitude and duration prescribed by Wesche et al.

(1977). Analysis of sediment content in the stream prior to and following these events indicated that the recommended flushing flows were "somewhat successful" in removing the deposited sediments (Wesche et al. 1985).

When considering needs for flushing flows, it must be remembered that while high flow rates may be necessary to remove sediments from deep deposits in some streambeds, these high flow rates can be very stressful and damaging to biota resident in the stream channel. Also, not all streams develop clogged beds as a result of stream regulation. For example, if large sediment sources, particularly those with large deposits of silts and clays, are lacking within the watershed, flushing flows may be unnecessary.

In determining the potential need for flushing flows for any water development project, a number of points need consideration (Reiser et al. 1985):

1. Physical location of the water development project in relation to major sediment sources.
2. Topography and geology of the project area.
3. Susceptibility of the drainage to catastrophic events.
4. Sensitivity of target fish species and their life history stages to sediment depositional effect.
5. Extent of man-induced activities within the drainage.
6. Operational characteristics of the project.

After needs for flushing flows are established, the problem becomes selection of the best time for implementation. Here, considerations must be given to the species of fish present in the system; life histories of these species; the natural, historical runoff periods that the species have adapted to; and the potential flow available for flushing after project completion (Reiser et al. 1985). Ideally, the best time for implementing flushing flows is when greatest potential benefits to the biological communities can be derived. In fact, the most difficult aspect of determining flushing flows is to determine the timing and magnitude of required flow. As noted above, no single, standard approach for determining this necessary flow exist. Until appropriate approaches are established, the most reliable method for establishing required flushing-flow rates is to observe the effects of various test flow releases in the stream of interest.

Reiser et al. (1985) recommended the following guidelines in conducting site-specific, flushing-flow studies.

1. The studies should involve interdisciplinary team members, including at least a hydraulic engineer and a fisheries biologist.
2. Determination of the actual need of flushing flows should precede the detailed study.
3. The assessment should fit the specific needs and character of the stream and the project; office and field personal both may be needed.
4. More than one method should be used to compare derived flow recommendations.
5. Recommendations should be stated in terms of magnitude, timing and duration of flushing flows.

6. Follow-up studies should be completed to evaluate the effectiveness of the recommendations.

Results from research currently underway by Wesche et. al. (1983) undoubtedly will help improve our understanding of appropriate methods to determine flushing flows necessary for maintaining stream channels.

INTERACTIONS BETWEEN RIPARIAN AND AQUATIC COMMUNITIES

Riparian communities dispense multiple benefits to stream systems. They contribute decaying vegetation, adding importantly to a stream's energetic base; provide structural materials, creating pools and protective cover; shade streams, lowering water temperatures; stabilize banks, reducing erosion and suspended sediment loads; and establish stream-side cover, important both to fish and to the terrestrial insects that also can contribute to the energetic bases for streams (Meehan et al. 1977, Vannote et al. 1980). In general, the kinds of riparian vegetation that can be most important to aquatic communities are first shrubs and bushes, then trees, and finally grasses and forbes (Platt et al. 1983).

Stuber (1983) found that stream reaches with well developed riparian communities had narrower, deeper, channels with less stream bank erosion than did adjacent channels where the riparian vegetation was heavily grazed. In fact, considerable erosional down-cutting of streams and associated dewatering of associated terrestrial habitats in much of the western U.S. appears directly associated with the destruction of riparian vegetation (Platts 1986, Parker 1986).

Stuber (1983) further found that stream channels with ungrazed riparian vegetation had twice the standing stocks of trout as were found in adjacent grazed stream reaches, and that the grazed reaches had higher proportions of rough fish than did the ungrazed area. Further, Wesche (1980) found that variation in cover supplied by riparian vegetation accounted for nearly 40% of the variation in the biomasses for trout, primarily brown trout, at 27 study sites in 11 reaches of eight montane and foothill streams in Wyoming.

Periodic flooding of the shoreline primarily limits establishment of terrestrial plants in riparian communities around reservoirs. Teskey and Hinkley (in Canter 1985) suggest that survival of terrestrial plants in riparian zones of reservoirs is governed by whether anoxic conditions develop when soils surrounding roots are flooded and by whether proper root function can be maintained during these times. Plants that possess adaptations to maintain concentrations of oxygen in their roots (e.g., transportation of oxygen from the stem) have improved abilities to survive flood conditions. Flooded soils also tend to prevent dispersal of metabolic products released by the roots. This can lead to mortality for some plants. But other tolerant plants incorporate mechanisms that permit reusing some of the less toxic metabolic end products. Consequently, concentrations of these products are reduced, which increases survivalship in such plants during flooded conditions (Canter 1985).

Considering the recognized importance that riparian communities have in maintaining the integrity of stream systems throughout the western U.S., the Western Division of the American Fisheries Society (1982) compiled a selection of best management practices for managing and protecting these riparian-stream ecosystems. Among the general methods recommended for dams and reservoirs, which also can apply to high-elevation reservoir-stream systems, are the use of staged or incremental filling, an option recommended for use when the immediate need for impounded water is less than available storage or initial demand. The document suggests that incremental filling can delay the ultimate loss of downstream stream and riparian habitat, permitting its extended use by the public and by wildlife. Also, it notes that because this practice tends to gradually inundate vegetation and slowly release nutrients, it can help to sustain a reservoir's sport fishery at a higher level for an extended time. Additionally, they recommended use of multi-level reservoir outlet structures to enable management of downstream water quality; stilling basins to reduce flow energies and reduce downstream scouring and erosion; and application of appropriate instream flow regulations. Potential benefits derived from each of these techniques are discussed in other sections of this report.

RESERVOIR CONSTRUCTION AND ENDANGERED SPECIES

Habitat changes due to water development, in general, and reservoir construction, in particular, are largely responsible for the decline in populations of native fishes in the American Southwest (Minckley and Deacon 1968). Because of these declines, many native fish species in the West are now protected by the Endangered Species Act of 1973. Under this act, the Secretary of Interior (or the Secretary of Commerce for some commercially important species) can categorize a species as "endangered" or as "threatened." The former category is any species "in danger of extinction throughout all or a significant portion of its range" and the latter includes "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range" (Environmental Law Institute 1977). The category of threatened is intended to offer protection to species before they become endangered.

The stated goal of the Endangered Species Act is to protect the ecosystem upon which endangered or threatened species depend. Under Section 7 of the Act, the Secretary has the duty to "insure" that federal activities neither "jeopardize" the continued existence of threatened or endangered species, nor destroy or modify habitat determined as "critical" for these species (Environmental Law Institute 1977). Section 7(a)(1) of the Act states the any programs authorized by a Federal agency should not jeopardize the continued existence of any threatened or endangered species. Reservoir construction is included under the federal permitting process, particularly the Corps of Engineers 404 permit program that regulates dredging and filling activities in the nation's waters (Carnevale 1986). Therefore, any reservoir project is subject to cancellation if, during consultation with representatives of the Secretary of Interior as required under Section 7 of the Act, the project is determined to pose a significant threat to the continued existence of a threatened or endangered species, or to its habitat.

When the historical range for a threatened or endangered species overlaps that of a proposed project, it does not mean that the project will be cancelled. Often under these circumstances a project will not be found to pose a significant risk to the species or its habitat. Or, if no recent occurrence of the species has been reported within the area of the project, the species can be judged as already extinct in that area.

Even when a project is proposed within an area defined as "critical habitat" for a threatened or endangered species, the 1982 Amendments to the Act offer the possibility of excluding the area from being defined as "critical." Under Section 424.19 of the 1983 proposed rules for the 1982 Amendments (U.S. FWS 1983),

> "The Secretary may exclude such an area from Critical Habitat if the benefits of such exclusion outweigh the benefits of specifying the area as part of the Critical Habitat. The Secretary shall not exclude any such area if, based on the best scientific and commercial data available, the failure to designate that area as Critical Habitat will result in the extinction of the species concerned."

Thus, if scientific study shows that use of an area as intended by a project is not likely to cause extinction of the species, it is possible to petition for the exclusion of that area from any present or future classification as "Critical Habitat" or as management areas for the species.

In Wyoming the species of particular concern as threatened, endangered, rare, or sensitive that could be impacted by water developments include Columbian sharp-tailed grouse, sandhill cranes, bald eagles, black-footed ferrets, Colorado River and Bonneville cutthroat trout, bluehead suckers, flannelmouth suckers, and leatherside chubs. Concerns must also be addressed on potential impacts of water development on downstream endangered populations of Colorado squawfish and humpback chubs in the Colorado River (Wyoming Game and Fish Department 1984, 1985).

IDENTIFYING FACTORS LIMITING PRODUCTION

The productivity potential for any environment is always limited by some variable within the environment. When developing a management plan for a reservoir-stream system, or for that matter any other system, it is important to identify what is or may be limiting the potential biological production in the system. In general, productivity limits may be defined by climatic, edaphic, morphometric, and/or biological influences. A fifth important influence on potential productivities in reservoir-stream systems is the reservoir drawdown regime.

For high elevation reservoir-stream systems, cold temperatures shorten growing seasons, and slow metabolic and, consequently, production rates for most resident aquatic plants and animals. Low temperatures can be a principal limitation on aquatic productivity at high elevations (Rawson 1942, Fabris and Hammer 1975, McAfee 1980). However, high elevation reservoirs can effect temperature changes that can influence productivities both within the reservoir and downstream. Stratification of high elevation lakes and reservoirs can lead to an accumulation of warm water masses in the epilimnion (Rawson 1942, Marcus 1987a). These waters can provide habitats having greater potentials for productivity than possible in the previous stream systems. In turn, discharge of cold, hypolimnetic waters resulting from reservoir stratification can extend cold seasonal temperatures downstream of reservoirs and adversely limit productivity by the downstream aquatic residents.

Wind is a second climatic influence that can have substantial influences in reservoirs. Strong winds, common in much of Wyoming, can thoroughly mix reservoir waters, producing nearly homogeneous solutions (Baxter 1977, Marcus 1987a). Such mixing can disrupt and limit those reservoir management strategies that are based on expected reservoir stratification. In addition, high winds can increase suspended sediments in reservoirs by eroding shorelines and limiting potentials for the development of shoreline (littoral or riparian) communities. It should be added, however, that reservoir valleys with steep slopes and reservoirs having deep maximum water depths are less likely to be affected by wind induced mixing.

Another frequent limitation to productivity in high elevation aquatic systems is low nutrient concentrations. Nutrient concentrations, as indicated by alkalinities and total dissolved solid concentrations, tend to decrease in lakes and reservoirs with increasing elevation; at the highest elevations, surface water concentrations of nutrients often approximate that of the incoming rain (Fabris and Hammer 1975, Baron 1983, Turk and Adams 1983, Nelson 1985).

Physical characteristics of mountain reservoir and stream environments can additionally limit biological productivities. For example, both canyon reservoirs and those with widely fluctuating water levels offer little suitable habitat for shoreline plant or benthic invertebrate communities, or for necessary spawning areas for some fish species (McAfee 1980).

Reproductive potentials for fish in mountain streams are typically limited by seasonally violent spring flows, frequent scarcities of suitable spawning substrates, and low flow during egg incubation periods (Hudelson et al. 1980, Nelson 1985). Fish productivity also can be limited in these waters by the low productivities of food organisms, again caused by the substrate and flow limitations (Hudelson et al. 1980).

In a survey of eleven western states to determine the limiting factors for natural reproduction for trout in reservoirs, all eleven states listed the lack of adequate spawning area as the most common limitation (Gebhards 1975). A reservoir often may occupy the prime spawning reaches in the stream system, which are usually characterized as a broad floodplain having a relatively flat gradient with a meandering channel punctuated by alternating pools and riffles. Dam construction tends to displace the spawning population into stream reaches with steeper gradients, larger rubble sizes, colder waters, and increased competition for space.

After spawning limitations, six states listed overfishing due to hatchery stocking programs and access development as limiting development of natural reproduction. Five states listed fluctuating water levels. Competition with nongame and hatchery stocked fish species was also listed by five states, and unsuitable water temperatures were listed by two states as limiting natural reproduction (Gebhards 1975). To establish optimally productive fisheries in reservoir-stream systems, habitat limitations should be recognized in management plans and efforts made to lessen their potential influences.

MANAGEMENT OF RESERVOIR FISHERIES

Stocking impoundments with fish may be the most common management strategy for fisheries. Research has shown that stocking is most effective for (1) establishing fisheries in new or reclaimed reservoirs; (2) introducing desirable new species; (3) restoring "balance" by introducing populations of large predator species or by replacing year class failures for existing important species; and (4) supplying catchable fish in otherwise suitable reservoirs where natural reproduction does not occur (Jenkins 1970).

Because new reservoirs are largely empty environments until species are introduced, a major theme for fisheries management in reservoirs has been to establish "balanced" communities of predator and prey species (e.g., Swingle and Swingle 1967). Introduced predators have provided trophy fishing, angler satisfaction, and have altered fishing pressures, while impacts on prey species have varied from unnoticeable to severe depletion (Kieth 1986). In general, vulnerabilities of prey appear related more to characteristics of the prey than to characteristics of the predators (Lewis 1967, Summerfelt 1986).

Often, in trying to establish predator-prey systems, too little consideration has been given to the forage base for the prey species. This link in the food chain is particularly weak in reservoirs because most fish reproduction is poorly synchronized with the availability of plankton for food, and because littoral communities are unstable due to fluctuating water levels (Noble 1986). Efforts at managing lower trophic levels in reservoirs have had only limited success (Noble 1986). Also, population dynamics of important predator and prey fish are poorly understood for reservoirs.

All fish species introduced into reservoirs are, in fact, "exotics," as the systems themselves are man-created exotic environments. And for many of these fish, no potential for spawning exists. Therefore, these systems often are maintained through maintenance stocking, with the stocking strategies based on the informal experience and observations of fisheries managers (Kieth 1986).

When developing a stocking strategy, considerations must be given to the management objectives and habitat limitations. For example, if winterkill conditions develop or are expected to develop in a reservoir, catchable size fish should be planted. Whereas, if over-winter survival is not a problem, planting of subcatchable sizes may be a more economical stocking strategy.

Another consideration regarding the appropriate size of fish for planting depends on catch rates. In coldwater reservoirs in Colorado, catch rates for rainbow trout planted as fingerlings ranged from 4 to 38%, while the catch rates for plants of creel-sized rainbow trout ranged from 31 to 59% (McAfee 1984). Based on hatchery cost data, it was determined that catch rates for fingerling fish must be at least 10% to be more cost effective than a catch rate of 40% obtained using creel-size plants of rainbow trout. These data show that the economic basis favoring plants of either size fish varied among reservoirs. This indicates the need for site specific studies in establishing management programs for many such waters.

Stocking strategies have chiefly focused on supplying predator species for anglers, which have traditionally been salmonids in the western U.S. Limited consideration has also been given to supplying forage species for the predators. Kokanee salmon are often the principal forage species supplied for salmonids in coldwater lakes and reservoirs, a species that can often go on to become a fishery itself (Wydoski and Bennett 1981). Other forage species used in western reservoirs include opossum shrimp (*Mysis relicta*) and redside shiners; but the success of these introductions depends on the outcome of the competition for food with resident species (Wydoski and Bennett 1981). Native forage species, particularly those that have co-evolved with salmonid species, have been under used as introduced forage species in reservoirs (Wydoski and Bennett 1981).

When introducing any group of species into a new reservoir or a single species into a species complex existing in an older reservoir, consideration also must be given to how the introduced species will interact among themselves and with any previously existing resident species (Li and Moyle 1981). New species combinations often create more problems than they solve. Li and Moyle (1981) suggest an approach using loop analysis to predict the behavior of new species combinations when only limited information exists. After applying the approach to two case studies, they concluded that an "ideal" species for introduction is coadapted with some members of the new species complex, has a relatively narrow spectrum of resource requirements, can easily be controlled if it escapes from the management area, and is free of exotic diseases and parasites.

After selection of stocking strategies and species, a third approach to the management of fisheries is through manipulation of harvest regulations. Two basic regulative approaches prevail: minimum size limits and slot length limits (Redmond 1986). While the minimum size limits have been historically used to protect fish up through the time of first spawning, the philosophy has shifted to protecting fish up to a balanced size--one large enough to maintain predation pressure on the forage base, but not so large as to allow excessive natural mortality before the fish become available for harvest (Redmond 1986). Use of slot limits allows harvesting of fish in the intermediate size classes. This thereby tends to maintain better size distributions for the regulated population (Redmond 1986). When properly applied, this approach permits more fish in the population to reach larger sizes, and is often used to establish trophy fishing waters. Under some slot limit regulations, a limited number of larger fish may be harvested in addition to a larger number of small to intermediate sized fish, as defined by the "slot."

In her analysis of fishing use and stocking strategies for coldwater reservoirs larger than 20 acres in Colorado, McAfee (1984) concluded

1. Total use for a given water appears related to accessibility and visibility of the reservoir, and to the recreational facilities available there.

2. Total harvest of rainbow trout responded directly to increases and decreases in stocking rates.
3. Total harvest and mean catch of creel sized rainbow trout increased directly with increased use.
4. First year returns on creel sized fish were at or above levels needed to justify stocking creel sized rainbow trout.
5. Planting fingerling sized rainbow trout did not apparently alter catch rates for planted catchable trout.
6. Both plants of fingerling and catchable fish contributed substantially to total harvest. Neither size alone appeared able to support the fishing pressure possible through plants using both sizes.
7. Stocking of creel sized fish did not appear to affect size distributions of stocked fingerling fish.

While tailwater fisheries below reservoirs can also be managed through manipulations of stocking strategies, species selection, and harvest regulations, the additional approach of regulating reservoir releases is the principal technique used to influence these fisheries (Ruane et al. 1986). Previous sections have already discussed relationships among reservoir water quality, placement of the reservoir outlet, and downstream water quality. Management actions implemented to alter the water quality of reservoir discharges, can also be used to enhance the habitat for downstream fisheries. The effectiveness of most of these techniques, however, has not been adequately documented or evaluated (Ruane et al. 1986).

EVALUATING POTENTIALS FOR WITHIN RESERVOIR FISHERIES

When relationships are defined between productivities of fisheries and characteristics of their habitat, these relationships can then be used to project fisheries productivities based on measured or anticipated habitat characteristics. Nearly fifty years ago Rawson (1942) announced that fish productivities were primarily affected by edaphic, morphologic, and climatic variables. His subsequent research showed a close correlation between mean depth and long-term fish production in large alpine lakes in western Canada (Rawson 1952).

Later research showed that fish productivity was highly correlated to the total dissolved solids concentration (mg/l) in a lake divided by its mean depth (m). This ratio is termed the morphoedaphic index (MEI, Ryder 1965). The relationship of this ratio to fish production has subsequently been investigated in a diversity of lakes and reservoirs (e.g., see overviews by Ryder et al. 1974, Ryder 1982, Jenkins 1982, Oglesby 1982). The MEI often accounts for greater than 60% of the variation in productivities by fisheries; and in an analysis of 290 reservoirs, the MEI defined 47 to 72% of the variation in productivities for fisheries (Jenkins 1982). This study also showed that when measured fisheries productivities in reservoirs most deviated from predicted values using MEI based relationships, the measurements were from the first 10-15 years after impoundment.

Investigating relationships of fish productivities with physical, chemical, and biological variables in thirteen Wyoming reservoirs, Whitworth (1984, 1985) found that the MEI accounted for 72-79% of the variance in trout population densities and biomasses on both aerial and volumetric bases for the reservoirs. But the two variables that best predicted trout densities per hectare of reservoir (D_t), accounting for 97% of the total variance in trout densities, were maximum depth (Z_M, in meters), which accounted for 82% of the variance, and total dissolved solids (TDS, in mg/l), which accounted for 15% of the variance:

$$D_t = e^{[4.0016 + 0.0004(TDS) - 0.0241(Z_M)]}$$

This model has been termed Whitworth's Reservoir Quality Index (RQI).

Because the RQI was developed using Wyoming reservoirs and for the lack of any better model, I suggest that the RQI can be used to provide at least rough estimates of potential trout densities in Wyoming's high-elevation, headwater reservoirs. But estimates obtained through use of this model should be extrapolated with caution. First, the highest elevation for reservoirs included in Whitworth's sample was 2298 m, so estimates for trout densities above this elevation may have a "low" elevation bias. Second, the mean age for reservoirs included in his sample was 40 y, so density estimates made using the RQI will be for "aged" reservoirs; these estimates may not be appropriate for the estimated trout densities occurring in the first 10 to 15 y following filling of the reservoirs (cf., Jenkins 1982). Third, the RQI may be biased by the inclusion of a preponderance of reservoirs having high TDS. So, while the RQI may be most appropriate for foothills and high plains reservoirs having TDS concentrations in the range for reservoirs he studied, it may not be appropriate for the water quality encountered in many headwater reservoirs.

Finally, the RQI equation shown here was selected as "best" by Whitworth using a stepwise regression routine. This approach to selecting the "best" equation is potentially flawed due to the effects of high multicollinearity (multiple high correlations) among the independent variables used by Whitworth (1984, see his Table 12). Under these conditions, correlated variables can mask the influence of other potentially more important variables (e.g., see discussions in Fausch et al. 1987 and Marcus 1987b). In fact, Whitworth (1984) noted that "maximum depth" may be acting as a "surrogate variable," masking the influence of the potentially more important "mean depth" or "surface area" variables. (A general discussion of the many potential statistical problems in most existing habitat based models is presented in the next section.)

To address the question of whether the cited RQI equation is in fact <u>best</u>, Whitworth's data set should be reanalyzed using an all possible subset regression routine, available in the BMDP package used by Whitworth. To address the question of whether the RQI is appropriate for application to high-elevation, headwater reservoirs will require collection and analysis of additional data from these systems, with perhaps a new RQI derived for such reservoirs in Wyoming.

ESTIMATING POTENTIALS FOR DOWNSTREAM FISHERIES

Maximum habitat for stream fisheries depends ultimately on the volume of water flowing down the stream. Consequently, much research has focused on relationships among stream fisheries, habitats, and flows. This research is thoroughly reviewed by Wesche and Rechard (1980), Loar and Sale (1981), EA Engineering, Science, and Technology, Inc. (EA ES&T 1986), and Faush et al. (1987). Included in the review by EA ES&T (1986) were 54 instream flow and habitat quality methods, while 99 models were reviewed by Faush et al. (1987). The following discussion draws heavily from these two latest reviews.

Most instream flow and habitat investigations endeavor to develop techniques and models through which standing crops and/or other measures of biological productivity, generally as pertaining to fish, can be described or predicted using a set of habitat variables. Underlying all of the resulting models is the premise that for flow, as well as for other environmental measures, there are definable limits, beyond which conditions become unsuitable for fisheries. Somewhere between these upper and lower extremes, optimal conditions exist that grade predictably to the unacceptable conditions. Ultimately, after appropriate relationships are derived, it is hoped that a few well chosen, easily obtained measurements made for a stream can be entered into a model to predict the stream's potential carrying capacity for fish.

Present models, which have been developed using both qualitative and quantitative approaches, include as few as one to as many as 21 input variables. Some variables included in some models are transformed from or derived (recombined) using the originally measured variables (EA ES&T 1986). Overall, the variables include details on basin morphology, channel morphology, flow rates, habitat structure, species present, and other physical and chemical measurements.

Assorted real and potential problems are associated with all existing habitat models (EA ES&T 1986, Fausch et al. 1987). The problem potentially most damaging in the long term is that no standard methods exist for the measurement of habitat variables (Wesche 1983, Fausch et al. 1987). Without standardized measurements, it is impossible to compare or synthesize data from different investigators. Moreover, because methods used in collecting data are reflected in the resulting models developed, single data sets can not be explored using otherwise similar models that are based on alternative sampling methods.

Various problems associated with many of these models have statistical bases (Fausch et al. 1987). First, presentations for most models lack information necessary to critically evaluate how the model was statistically selected or how the model may perform in general application. To evaluate the statistical worth of models, sufficient information should be included to enable evaluation the correlation coefficients (r); coefficients of determination (r^2, R^2, or adjusted R^2; R^2 is the r^2 for multiple regression relationships and adjusted R^2 is corrected for the number of variables included in the equation); standard errors for the regression coefficients (i.e., is the coefficient for each variable included in the model significantly different from 0); and/or the confidence interval for the presented models (Fausch et al. 1987).

Another statistical problem in many of the models is small sample sizes. This limits the potential applicability of the model to limited ranges in habitat variability. If models are used to extrapolate outside these limits, the resulting predictions can be biased and unreliable.

Also, errors associated with measuring the various habitat variables have rarely been evaluated during model development. If measurements upon which the model is based are biased, the model will yield similarly biased predictions. Most models have not been tested with data that was not used in developing the models. Thus, we know little of the overall realism, precision, or generality of the models (Levins 1966).

Various potentially unreasonable assumptions about habitat relationships are also implicit in many of these models, including, for example, the U.S. Fish and Wildlife Service's Instream Flow Incremental Methodology (IFIM). These potentially erroneous assumptions include (1) fish primarily respond to average water velocities at some defined depth (e.g., 0.6 of the depth below the surface); (2) stream depths, velocities, and substrates are not related to each other (i.e., an underlying assumption for regression analysis is that independent variables are uncorrelated); and (3) large amounts of suboptimal habitat are equivalent to a small amount of optimal habitat (EA ES&T 1986, Fausch et al. 1987). Yet, studies show that fish respond more to flow difference in microhabitats; that stream depths, velocities, and substrates are often highly correlated; and that suboptimal habitats can often be uninhabitable (Fausch et al. 1987).

Finally, most models include at least one additional and perhaps also unreasonable assumption about the relationship between measured habitat variables and measured fisheries densities or biomasses used to derive the models. That is, it is often assumed that the measured density or biomass for fish is at the carrying capacity for the habitat, and that this carrying capacity is defined by those physical and chemical conditions measured in the habitat. This assumption precludes such effects as predation (including fishing), or competition as having any potential influence on the population.

Only one existing model apparently provides good predictions of standing crops for fish, and the efficacy of this model has been documented only in unregulated streams (EA ES&T 1986). This model, the Habitat Quality Index (HQI), was developed by Binns (1979, Binns and Eiserman 1979) to predict trout standing crops in Wyoming. An R^2 = 0.966 was reported for the correlation between the HQI and measured standing crops for trout in 36 Wyoming streams (Binns and Eiserman 1979). Subsequently, Annear and Conder (1983, 1984; Conder and Annear 1986) applied the HQI to ten additional Wyoming streams and obtained an R^2 = 0.872 for the relationship between the HQI and measured standing stocks of trout.

While a plethora of models relating stream flows to fish are indeed available, no model currently available has been verified as reliably predicting effects of stream flow alterations on standing crops of fish (EA ES&T 1986). The only experimental investigations of the effects from varying flow regimes on fish indicate a lack of any statistically significant relationship of flow with either production, distribution or growth using rainbow and brown trout (Rimmer 1985, Irvine 1985). But interpretation of the results from these experiments are complicated both by experimental problems and by inadequate sample sizes for strong statistical comparisons.

Despite these results, I have little doubt, and ecological theory tends to support, that low and high stream flows, as well as the low and high ends of ranges for other variables, can be stressful to organisms and adversely affect population sizes (e.g., Huston 1979, Marcus 1980). Therefore, methods are needed to estimate how changes in stream flows caused by stream impoundment will affect downstream fish populations.

Considering the current availability and understanding of instream flow and related habitat assessment models for streams, the best available method, particularly in terms of applicability to streams in Wyoming, is Binns' HQI. And there is at least some circumstantial evidence that the HQI model may have at least limited abilities to simulate trout standing crops for regulated streams (Binns 1982, Annear and Conder 1983).

While a detailed presentation of the HQI model is beyond the scope of this report, a brief introduction to its approach follows. In developing the HQI, data on physical, chemical, and biological variables were collected or compiled for 44 sites on 36 streams (Binns and Eiserman 1979). For each variable an index ranking (0-4) was selected based on criteria presented by Binns and Eiserman (1979). Then, using correlation and multiple regression techniques, variables were selected that constituted the "best" multiple regression equation. The HQI model suggested by Binns and Eiserman (1979) as best for predicting trout standing crop in Wyoming streams, and that which is currently used to for habitat evaluation work in Wyoming is

$$\log(Y+1) = 1.12085\ [-0.903 + 0.807 \log(X_1+1) + 0.877 \log(X_2+1) + 1.233 \log(X_3+1) + 0.631 \log(F+1) + 0.182 \log(S+1)]$$

where Y = predicted trout standing crop (kg/ha)

X_1 = late summer stream flow (m^3/sec)

X_2 = annual stream flow variation (m^3/sec)

X_3 = maximum summer stream temperature (^oC)

F = food index = $X_3(X_4)(X_9)(X_{10})$

S = shelter = $X_7(X_8)(X_{11})$

X_4 = nitrate nitrogen (mg/l)

X_7 = cover (%)

X_8 = eroding stream banks (%)

X_9 = substrate (% coverage by submerged aquatic vegetation)

X_{10} = water velocity (m^3/sec)

X_{11} = stream width (m)

Potential difficulties in computing correct HQI values can occur when individuals who lack experience in evaluating fisheries habitat attempt to apply this method. Elements of qualitative judgement are involved in variables X_7 (cover), X_8 (eroding stream banks) and X_9 (substrate). Such inexperience has potentially contributed to the poor performance of the HQI when it has been applied by other investigators (EA ES&T 1986). Additionally, because ranges for characteristics at the study sites used in developing the regression equation were limited, the HQI model is potentially limited in its application for estimating standing stocks of fish in streams having habitat characteristics outside the ranges for those in the original samples. And because the model was principally developed using streams where the principal resident trout were *Salmo* species, the model may not be appropriate for most other trout, char, or non-trout species.

For estimating standing stocks for brook trout (char), another model may be more useful than the HQI. Chisholm and Hubert (1986), using environmental and standing stock data for brook trout from 24 reaches in 13 montane streams, developed a multiple regression equation to predict standing stocks for brook trout. Their equation accounted for about 69% of the variation in brook trout standing crop and may be useful in estimating changes in standing stocks of brook trout in response to flow alterations below impounded streams:

$$S = -9.88 - 3.43G + 1246.8W/D + 12.28D - 321.5W$$

where S = standing stock (kg/km of stream)

G = gradient over stream reach (%)

W = width (m)

D = depth (m)

W/D = width to depth ratio.

Finally, using any existing habitat based model to predict changes in fish populations due to stream regulation below impoundments is currently only a "best guess" approach. No method has proven reliability. Additional research is needed to evaluate their realism in predicting effects of alternate flow regimes on downstream standing stocks of trout and non-trout fish species. Undoubtedly, new equations must be developed for most species, particularly for non-trout species omitted in the existing models. Also, such equations are particularly needed for streams having environmental characteristics lying outside those found in the streams on which present models are based.

SECTION 5

MANAGEMENT OPTIONS FOR HIGH ELEVATION RESERVOIR-STREAM SYSTEMS

Management of fisheries to enhance or optimize fisheries in high elevation reservoir-stream systems can be approached through a variety of options. These include indirect management of the fishery through manipulating water quality or the physical habitat, and direct management by manipulating the fisheries themselves. This section considers these management options after first introducing necessary considerations for establishing management needs and objectives in formulating overall management plans for fisheries management in high elevation reservoir-stream systems. The section concludes with a brief discussion on the importance of including pre- and post-action monitoring in all management plans.

MANAGEMENT PLANNING FOR HIGH ELEVATION RESERVOIR-STREAM SYSTEMS

Establishing Management Needs for Reservoir Fisheries

Prior to proposing or establishing any management program, the need for that program must be established. Similarly, needs for fisheries management in high elevation reservoir-stream systems must be also established. Reservoirs are, in fact, artificial environments that lack community histories (Noble 1986). In natural lakes and streams, species associations have developed in which species can survive with their environment and with the other species living around them. Over time, these animals have learned, in a sense, which species are good to eat and which species are good to avoid being eaten by. Other learning processes have taught characteristically similar species how to divide up similar environmental resources, permitting these species to coexist in competitive systems. But reservoirs lack these histories. While they are waterbodies placed where streams one flowed, they are similar to but different than either lakes or streams. Reservoirs are created by man, the waters they contain are usually regulated by man, and, for fisheries to exist in them, they will necessarily be managed by man.

Nationally, reservoirs provide more days of fishing and promote more fishing related spending than any other water type (Fisher et al. 1986). The Corps of Engineers estimate that about 25% of total use for Corps reservoirs is by fishermen (Spencer 1986).

Similar figures have not been computed for reservoir fishing within Wyoming. But the last fishing pressure survey for the state indicated that average resident anglers spend 17.6 days fishing, with about eight days fishing on flowing waters and 9.6 days on standing waters; while nonresident seasonal licence holder averaged about 15 days angling, with about seven days fishing on streams and rivers and eight days on standing waters (Marcus et al. 1985). Since the state's most heavily fished standing waters are reservoirs (Marcus et al. 1985), reservoir fishing constitutes perhaps about 40-45% of the total fishing pressure in Wyoming. This indicates high public interest in Wyoming for maintaining quality reservoir fisheries. With increased access to new headwater reservoirs, the proportion of fishing on standing waters could increase.

Regarding specific needs for fisheries management in headwater reservoir-stream systems, this report has emphasized how impounding rivers and streams replaces naturally established ecosystems with artificially regulated ones. With these disruptions, two choice are left. The first, which dominated much of the early history of reservoir construction, is to ignore what happens to the biology associated with the impounded system. But with increasing public distress over actions that degrade the environment, options under this choice have diminished. Thus, left predominantly with the second choice, these man-made systems must be managed with goals to minimize adverse impacts of reservoir construction and, when possible, to enhance productivities for the remaining resources. Since these latter goals are becoming a given in reservoir construction, it is increasingly important that water user groups and fisheries biologists work closely together in planning, designing, and operating reservoirs. Perhaps, it will soon become the that the only way new reservoirs will get built is through this cooperation.

Establishing Management Objectives

Since fisheries management constitutes a discernable need during the construction and operation of reservoirs, steps must be taken to incorporate fisheries management objectives into the overall management plans for reservoirs. In some cases and at some times, fisheries management may necessarily become the primary management objective for operation of a reservoir (Summerfelt 1986). But in general, management plans for fisheries must balance (1) ecological limits for the fishery;

(2) social, economic, and political pressures; (3) monetary, personnel, and equipment resources available; and (4) the quality and quantity of data required to evaluate the success of the management program (Taylor and King 1983).

Wesche (1985) discusses a series of steps useful in managing restoration of streams and rivers for fisheries. These steps are equally useful in establishing management plans for reservoir-stream systems. Following his approach, first an interdisciplinary team should be formed to establish the overall management objectives and operation plans for the reservoir. This team should consist at minimum of a representative from each water user group, a hydrologist, a hydraulic engineer, and a fisheries biologist. Next, each team member should compile a list of questions or concerns regarding their specific expertise and interests regarding the proposed project. Wesche (1985) provides nearly thirty such questions to help stimulate thought processes under this second step. Questions potentially important during development of headwater reservoirs could include consideration of factors potentially limiting survival and productivity in the fishery, appropriate drawdown regimes to minimize impact to the fishery, and access by the public to the fishery. After such questions have been posed and satisfactory answers obtained, the final management plan can be drafted and then implemented. Finally, careful monitoring of the implemented management plan must be conducted to determine whether the results obtained meet the stated goals and objectives of the plan. Wesche (1985) emphasizes that whenever and wherever possible, postimplementation evaluations be conducted so that we can learn from our past mistakes and improve our future actions.

Most construction of headwater reservoirs in Wyoming will occur on Class 2 and 3 streams, as defined by the State's Stream Fishery Classification (Wyoming Game and Fish Department 1977). Under the Wyoming Game and Fish Commission's (1985) mitigation policy, habitat values on Class 2 and 3 streams are rated, respectively, as high--with no net loss of in-kind habitat, and moderate--with no net loss of habitat and minimum loss of in-kind habitat. This means considerable instream habitat enhancement, including improvements to the riparian and stream channel habitats, will be necessary during development of headwater reservoirs. At this time, no credit is given for enhancement of the reservoir habitat as a trade off for loss of stream habitat (M. Stone, Wyoming Game and Fish Department, personnal communication).

Currently, due largely to limitations in spawning habitats, most headwater reservoirs in Wyoming are managed under the basic yield concept of fisheries management (J. Baughman, Wyoming Game and Fish Department, personal communication). With increased management directed at reducing limiting factors in reservoirs, perhaps higher level management concepts can be implemented.

In developing fisheries management plans for headwater reservoir-streams systems, a combination of activities must be included that influences both the basic productivity and the structure of the fish populations. Habitat modifications may include hydrological manipulations, erosion control, vegetation control or enhancement, manipulation of stratification and water quality characteristics, and addition or improvement of spawning areas (Summerfelt 1986). Finally, any management plan must include and integrate management objectives for both the reservoir and the downstream system. Operation of all water projects should fully consider effects on downstream fisheries (Peters 1986).

MANIPULATING WATER QUALITY IN RESERVOIR-STREAM SYSTEMS

A number of water quality alterations likely to occur during passage of waters through reservoirs can be anticipated. Based on these expected alterations, management plans can be implemented to mitigate potential reservoir impacts and to enhance habitats for aquatic and terrestrial residents. One important limitation in designing management plans for high elevation headwater reservoir systems results from the present scarcity of data from these systems. This limits our certainty about how these systems can alter water quality. But existing data do suggest that all alterations that have been observed in lower elevation reservoir-stream systems can potentially occur, at least to some extent, in high elevation systems (Marcus 1987a). Therefore, until additional data prove otherwise, we can assume that most management options available for altered water quality in lower elevation systems will also be available, at least to a limited degree, in high elevation systems.

When developing management plans for water quality in these systems, one of the first considerations should address management of present and planned activities in the watershed. Since many land-use activities produce direct and measurable effects on water quality (e.g., Likens et al. 1977), these activities should be appropriately restricted to limit potentially adverse impacts on water quality. For example, both logging and mining can directly increase sediment runoff (Likens et al. 1977, Schillinger and Stuart 1978). Therefore, both logging and mining should be restricted, or more probably prohibited, in any portion of a drainage above a headwater reservoir. Similarly, cattle grazing has been shown to directly degrade stream riparian communities (Platts 1986). Since reservoirs are generally acknowledged to have stressed and unstable riparian communities within their basins, and to degrade downstream riparian communities, cattle grazing should be severely restricted, or perhaps prohibited, in any watershed containing a headwater reservoir, limiting any additional stress on these communities.

Also, recreational developments associated with headwater reservoirs should include appropriate controls to limit input of any resulting septic wastes and sediment runoff. But not all watershed sources of potential water quality degradation are necessarily associated with man or his livestock. I have previously discussed how natural geologic influences above headwater reservoirs can lead to blooms of nitrogen fixing algae and bacteria resembling those normally associated with organic eutrophication (Marcus 1987a).

If nuisance growths of algae or other plants develop within reservoirs and if decomposition of this biomass leads to depletion of oxygen in the deep waters of the reservoir, methods of hypolimnetic aeration are available to alleviate this condition (Lorenzen and Fast 1977, Nelson et al. 1978, Henderson-Sellers 1984). However, due to their relatively low water temperatures and short growing seasons, I expect that such problems are unlikely in most high elevation headwater reservoirs.

Difference in the thermal and chemical regimes between surface-release lakes and reservoirs and deep-release reservoirs have been previously discussed, as have potential constraints that water temperatures and chemistries place on reservoir production. In developing management plans for high elevation reservoir-stream systems, tradeoffs likely will have to be made between maintaining warm reservoir waters to enhance reservoir productivities and releasing reservoir-warmed waters for enhancing downstream productivities. Perhaps the optimal management strategy with respect to temperature would be to release surface warmed reservoir waters in the spring and early summer to enhance downstream productivities; then shift to mid-water column reservoir water releases in mid-summer to encourage warming of surface waters and potential productivity enhancement in the reservoir. Ideally, the release regime would permit downstream warming that may be slightly advanced (ca. one month) of the normal warming cycle in the early portion of the normal growing season, and then return to approximately normal temperatures for the rest of the growing season.

Recalling that reservoirs with deep-water outlets, including high elevation headwater reservoirs, can act as downstream nutrient sources (Wright 1967, Martin and Arneson 1979, Marcus 1987a), altering discharge levels can similarly be used to manage downstream productivities. When productivities in streams below reservoirs are primarily limited by low nutrient concentrations, one management option is to discharge from the reservoirs deep-water layers. When these layers contain elevated nutrient concentrations, these discharges can enrich and enhance downstream productivities (Marcus 1980).

But a cautionary note is needed: the deep-water layers of reservoirs also can become anoxic as dissolved oxygen concentrations are depleted through decomposition of organic materials. Potentials for this conditions are greatest when large sources of nutrients or organic runoff are present in the watershed, or when high densities of terrestrial vegetation are left in a reservoir's basin. Discharge of anoxic reservoir waters can kill organisms downstream. Although total depletion of oxygen in the deep waters of high elevation reservoirs is unlikely, consideration of this possibility should be given during planning of the reservoir discharge level. If conditions capable of deoxygenating hypolimnetic waters are present, the depth of the discharge should be set at an elevation above where the anoxic waters may occur. Alternatively, a spillway deflector to aerate discharge waters could be incorporated into the outlet design. But, again, wind induced mixing can disrupt reservoir stratification and eliminate potential accumulation of anoxic waters.

MANIPULATING THE PHYSICAL HABITAT OF RESERVOIRS

Water Level Fluctuation

Duration and timing of drawdowns can enhance or damage reservoir populations (McAfee 1980). Limiting maximum drawdown to relatively short periods during the cool time of the year can minimize dehydration and death of many burrowing insects. If reservoirs are partially refilled prior to freezing weather, plankton and benthic invertebrates can partially recolonize the new waters and reinundated sediments. Limiting drawdowns during freezing weather can minimize the potentially high rate of freezing deaths for benthic fauna burrowed within the reservoir sediments. Limiting discharges during the spring and fall spawning periods can minimize exposure of reservoir spawning beds and/or the discharge of floating fish eggs (McAfee 1980).

Water-level changes are perhaps the most common manipulation of physical habitat used to manage reservoir fisheries. A recent review indicted that to maintain fisheries habitat most reservoir fishery managers, when possible, (1) draw down water levels in late summer or fall; (2) establish terrestrial vegetation on the exposed basin by seeding or allowing recolonization; (3) flood this vegetation in the spring, providing spawning habitat and productive cover for rearing fish; and (4) maintain high water as long through the growing season as possible (Ploskey 1986).

An alternative strategy used in Kansas typically consists of (1) a spring rise to flood terrestrial vegetation; (2) about a 4-ft summer drawdown to allow regrowth of the vegetation and to concentrate the predator and prey species, enhancing growth of the game species; (3) a partial 2-ft water-level increase in the fall to flood some terrestrial vegetation for use by waterfowl; and (4) a winter drawdown to again concentrate the predator and prey species, and to protect the vegetation during the winter (Willis 1986). Where drawdown amounted to 20% or more of the reservoir's basin area, population densities for most predatory species have increased.

It is doubtful that either of these strategies are appropriate for high elevation headwater reservoirs. For these systems, spring comes late, growing seasons are short, and late summer and fall water demands are often high, while flows are low. Consequently, these reservoirs will often be filled in late June to mid-July, and most of the drawdown will likely occur in August through October, with minimum pools maintained from late fall to early spring. Under such a regime, perhaps the only potential for enhancing habitat with water fluctuations would be to encourage growth of terrestrial vegetation as the basin becomes exposed. Then, during good water years this vegetation could be flooded in mid- to late-fall, potentially benefiting reservoir productivities and migratory water fowl. This established vegetation could possibly reduce sediment erosion and reduce turbidity within the reservoir during the late fall to early spring. And, when the sediments were flooded in later spring, leached nutrients could enhance productivities within the reservoir. Ideally, further investigation could identify species of vegetation that develop rapidly with good root development, yet have potential to survive one to two months while their roots are inundated with reservoir waters. Perhaps, this community would be dominated or supplemented by rapidly colonizing annual species.

McAfee (1980) presents several site characteristics that can increase the suitabilities of reservoirs for fish and to potentially lessen effects of drawdown. These include (1) fertile watersheds that can enrich reservoirs with moderate nutrient loadings; (2) gentle basin slopes that permit establishment of rooted littoral vegetation; (3) shallow mean depths that permit photosynthesis in a high proportion of the reservoir waters; (4) a morphometry that permits the water mass to completely circulate; (5) relatively warm water temperatures maintained over extended intervals; (6) the absence of high winds and waves which erode the shore; and (7) adequate refuges to protect fish and invertebrates during drawdown periods. Most of these characteristics tend to be rare at most potential sites for high elevation, headwater reservoirs.

Minimum Pool for Fisheries

Fish need water to survive. After accepting this statement the next consideration is how much water is needed for fish to survive. This is, in fact, the essence of the problem: How much water must be left within a reservoir to assure the survival of a viable fishery? What proportion of the total capacity of the reservoir should be retained as the "minimum pool" to assure fish survival? But here, because of the scarcity of good data, we have no consensus.

Drawing heavily on previous reviews by Ploskey (1983, 1986), minimum pool requirements for maintenance and enhancement of reservoir fisheries has recently been discussed by Guenther and Hubert (1987). They report that increasing concern has arisen about the importance of minimum pools for reservoirs, and that arbitrary, unproven minimum pools are being recommended and enforced. They found, however, no documentation quantifying benefits of current minimum pool practices.

Of perhaps greatest concern about minimum pools is their relationship to overwinter survival of fish. In turn, the greatest concern here is the total depletion of oxygen in waters under the surface ice (see Guenther and Hubert [1987] for an excellent review of methods used to predict low oxygen conditions in lakes). A second concern about minimum pools is what reservoir level is necessary to maintain adequate habitat to support a productive fisheries.

A problem likely to persist in defining minimum pools is, to a large extent, that the necessary size for a minimum pool is site dependent. That is, it is dependent on the prevailing climate, geology, soils, morphology, age, species present, and management goals for the reservoir.

Among the approaches reported by Guenther and Hubert (1987) that have been used in defining appropriate minimum pools are

1) For warm water, low elevation reservoirs, the water below 50% of the reservoir's surface area must remain at least 12 feet deep.
2) For cold water, high elevation reservoirs, the water below 50% of the reservoir's surface area must remain at least 20 feet deep.
3) The surface area during drawdown must remain at least 30% of that normally found for the high water level.
4) The minimum water volume must remain 20-25% of normal maximum storage.
5) Minimum pool is defined as the break point in the plot of storage volume versus the morphoedaphic index (Ryder 1965).
6) Minimum pool is set at the "best guess" of the area fishery manager.

Again, it must be emphasized that all of these approaches are, in fact, best guesses, as the effectiveness of any of these approaches have not been quantified (Guenther and Hubert 1987). Since depth is a key determinate for risk of winter kill in ice covered lakes and reservoirs, winter drawdown is not advised in severe climates (Mathias and Barica 1980, Guenther and Hubert 1987).

Guenther and Hubert (1987) are currently investigating the effectiveness of minimum pools in maintaining and enhancing salmonid fisheries in nine small, coldwater Wyoming reservoirs. Notably, this effort plans to quantify relationships for variables affecting survival of salmonids in reservoirs, and identify critical limiting factors. They then plan to build a decision model that can help guide management decisions, first, for minimizing risks from winterkill and, second, for enhancing salmonid production in reservoirs. Until their work is completed, minimum pools for Wyoming reservoirs can only be defined using existing qualitative methods. These difinitions should be completed by a planning team, including at minimum a fisheries biologist and a representative from the water user groups.

Basin Structure

Increased structural diversities within the physical habitats of reservoirs are often directly correlated with increasing densities and standing stocks for fish (Brown 1986). Habitat structure can provide cover for fish, as well as substrate for periphyton and cover for invertebrates (Nelson et al. 1978). The physical habitat of reservoirs can be altered and their fisheries enhanced using a diversity of options.

One early consideration necessary when planning the physical habitat in reservoirs is whether and how much timber within the reservoir basin should be removed prior to filling. Retaining submerged trees and brush can provide valuable feeding areas for fish, and exposed tree limbs can provide roosting branches for birds (Nelson et al. 1978). Standing timber can increase angling success, while decreasing actual catch rates (Brown 1986). Selective clearing of timber tends to preserve and produce good fish habitat, but it can also produce obstacles for boaters and adversely impact visual aesthetics with dead tree branches.

When suitable standing timber does not exist within the reservoir, artificial structures can be added to the basin. Perhaps the most common artificial structures are constructed using branches and trunks from small trees (Nelson et al. 1978). These structures can be very durable, with some hardwood shelters lasting for over 30 years in Michigan (Brown 1986). When these structures are suspended above the bottom, they tend to be more durable than when placed on the bottom, and less likely affected by accumulations of settling silt. Brush shelters may be most effective of all structures in attracting fish (Brown 1986).

Rock reefs and automobile tires also can be used to enhance reservoir structure, and both will last indefinitely. But, while tires are considered biologically non-polluting, they can be aesthetically unapealling (Brown 1986). Some fisheries biologists consider tire shelters to be most productive biologically (Nelson et al. 1978).

Spawning areas can be enhanced with artificial gravel beds. But the potential value of this option can be limited by sediment deposition and by drawdown regimes.

Seasonal plantings on exposed shores using, for example, winter wheat, may be possible for some reservoirs. The resulting growth can stabilize the shore, reduce erosion, and when reflooded can provide spawning habitat, nursery areas, and nutrient releases that enhance nutrient recycling and productivities in reservoirs (Brown 1986).

Finally, it is important to note that most of the physical habitat enhancement efforts have been directed at non-salmonid species in low-elevation, warm-water lakes and reservoirs. Brown trout, however, particularly appear to benefit from cover in streams (Wesche 1980) and brook trout have also been reported to respond positively to cover improvements in mountain streams (Burgess 1985). So enhancing habitat structure in high-elevation reservoirs may benefit these species. Still, the overall efficacy of these techniques with salmonids in high-elevation headwater reservoirs requires additional investigation. In addition, many questions remain regarding the overall use of structures for any species (Brown 1986): what types of structures are preferred, where they are best located, what is the best technique for placement, how much structure is needed, and why they are used by fish? But even with all of these unknowns, we do know that they often work in lower elevation reservoirs, and there is good reason to think they will work in high elevation reservoirs.

MANIPULATING THE PHYSICAL HABITAT OF STREAMS BELOW RESERVOIRS

Two important approaches to manipulating physical habitats in streams below reservoirs are available to resource managers. First, since the volume of water in the stream ultimately defines the habitat available for fish, appropriate regulation of discharge waters can help reduce physical limitations in downstream habitats. Holding back portions of the annual high spring flows and limiting discharge velocities during intervals where sensitive life stages of fish are present can reduce potentially adverse impacts accompanying high flows. And then, if at least some of these stored waters are released to supplement low flows, adverse effects normally encountered during this period can be lessened. Also, as discussed, the frequency and duration of high water (flushing) flows define the character of the stream bottom sediments and the availability of spawning habitat. And the height of the outlet(s) above a reservoir's bottom affects downstream water quality in terms of both chemical constituents and temperatures.

The second approach to manipulating the downstream physical habitat is by altering the actual appearance of the stream channel by placing artificial structures within the stream. Many of these structures can be similar in some respects to those discussed in the previous section. In use, these structures improve fisheries habitat by reducing bank erosion and increasing habitat diversity through creating new series of riffles and pools. For example, installing small rock dams and deflectors in a small mountain stream in Quebec not only increased the biomass of brook trout, but also increased biomasses of aquatic invertebrates and increased usage of the stream by mink and raccoons (Burgess 1985).

Wesche (1985) has recently presented a comprehensive discussion on the relationship of fish with components of their habitat, including stream velocity, substrate, depth, invertebrate drift, spawning areas, and protective cover. In reviewing the use of artificial structures, he concluded that the most commonly used within-channel structure are current deflectors, low profile overpour dams and weirs, bank cover and boulder placement. Table 3 lists the commonly used options available for artificially modifying the physical structures of stream habitats. Nearly all of these techniques are potential options that can be used to enhance stream habitats below reservoirs. Both Wesche (1985) and Nelson et al. (1978) provide guidance on appropriate application, construction, and installation techniques for the structures.

DIRECT MANIPULATION OF FISHERIES

Direct management of fisheries in high elevation reservoir-stream systems can incorporate manipulations of stocking strategies, species selection, and harvest regulations. Stocking will be necessary in all reservoirs where access to spawning areas is limited. Where over-winter survival is a problem, stocking of creel-size fish will be necessary; the fishery will exist on a put-and-take basis. If over-winter survival is not a problem but spawning habitat is limited, planting subcatchable fish may be a more cost effective approach (McAfee 1984). For some reservoirs, planting both subcatchable and creel-size fish may be the optimal strategy (McAfee 1984).

Species selection for headwater reservoirs requires thought and planning. The general stocking strategy used for many cool water

Table 3. Common artificial structures and management techniques used to enhance habitats for stream fisheries (Nelson et al. 1978, Wesche 1985).

Current Deflectors
Rock-boulder deflectors
Gabion deflectors
Double-wing deflectors
Underpass deflectors
Half log deflectors
Boulder placement
Trash Catchers
Low-Profile Check Dams
Rock-boulder dams
Single and multiple log dams
Plank or board dams
Gabion check dams
Beaver introduction
Bank Cover Treatments
Log overhangs
Artificial overhangs
Tree retards
Bank revegetation
Riprap
Erosion-control matting
Stream bank fencing
Grazing control

reservoirs tends to emphasize introduction of rainbow trout in attempts to develop a "fast, family fishery." With some headwater reservoirs this approach may be totally appropriate. But these headwater systems constitute unique environments with new opportunities for management.

For some reservoirs, available management options may be limited by the presence of rare, threatened, or endangered species, such as the Colorado River cutthroat trout or the bluehead sucker. However, for most reservoirs, species selection options could permit introductions of other species and specialized strains to provide potentially unique fishing experiences. For example, one high elevation headwater reservoir in Montana is inhabited by a wild, naturally reproducing fishery, consisting primarily of grayling and the McBride Lake strain of cutthroat trout (Zubik 1983). Both species spawn successfully in tributaries upstream of the reservoir. The grayling, which may be descendants of the original ancestral stocks in Hyalite Creek, competes successfully with both cutthroat and brook trout in the reservoir. And the spawning chronology of the grayling is reproductively advantageous, with the upstream spawning runs occurring after peak runoff, a timing that minimizes dislodging their broadcast eggs by high spring flows (Wells 1976).

MONITORING TO EVALUATE RESULTS OF MANAGEMENT ACTIONS

Whenever a management action is implemented that is intended to produce an expected environmental response, it is important to determine whether the action indeed produced the intended effect(s). Unfortunately, follow-up monitoring for management actions are too infrequently undertaken. For example, of the massive environmental data collection effort in response to the National Environmental Act of 1969,

> ". . . precious few (data) were collected after the projects were completed. Without these end data there was no way to compare the initial conjectural impact statement with the true impact of the project on the environment. We lost the opportunity to evaluate a great many different environmental management schemes and wasted monitoring resources. Fishery managers should recall this lesson and design all monitoring programs to allow full evaluation of the efficacy of each management program relative to its objectives, but to collect only those data needed" (Taylor and King 1983, page 347).

Monitoring should be considered a necessary cost for any environmental management action. The first action of any management scheme should be to establish the need for the action and the baseline conditions on which the management action is expected to have an effect. The appropriate environmental variables to monitor depend upon the management action implemented. In some cases it may be more appropriate to evaluate the action by monitoring responses in the habitat. In other situations it may be more appropriate, and perhaps easier, to monitor responses in the target population.

The actual methods used during monitoring should follow standardized techniques; this will enable needed comparisons between data generated in different studies (cf., Wesche 1983). Various appropriate methods used for sampling plankton, periphyton, macrophytes, macroinvertebrates, and fish have been presented by Weber (1973). Specific approaches to analyze and monitor stream and riparian habitats were presented in 1983 by the USDA Forest Service (Platts et al. 1983), the US Fish and Wildlife Service (Armour et al. 1983), and the US EPA (1983).

Monitoring programs should be designed to permit rigorous statistical analysis of the resulting data. Green (1979) presents ten principles that should be followed by environmental biologists in the design and analysis of data collection programs. Basically, he recommends the approach followed by all good experiments: establish need for study; pose understandable questions; define statistically testable hypotheses; design the study plan and select appropriate monitoring variables with help of a statistician experienced in the nature of the specific problem; gather data needed under the design; analyze data using statistical procedures specified under the study design; evaluate tests of the statistical hypotheses; and interpret and present the results. Often, results from one study will establish needs for another study; then, the experimental design and analysis process can begin again.

Specific to adequately quantifying alterations resulting from river impoundments, Canter (1985) has proposed twelve steps important in designing and conducting pre- and post-impoundment monitoring programs. The following twelve step program is based on his initial presentation.

1. Determine management and monitoring objectives and the relative importance of each.
2. Define objectives in statistical terms.
3. Determine the budget available for monitoring and the budget allocated to each objective.

4. Determine characteristics of the area in which the monitoring is to take place.
5. Determine appropriate water quality and aquatic biology variables to be monitored, include variables that can be manipulated using management options.
6. Identify sampling locations and frequencies.
7. Adjust original monitoring design objectives (1-2) using subjective management considerations (3-7).
8. Develop feedback mechanisms to adjust the reservoir operations design.
9. Initiate monitoring procedures and implement reservoir operations.
10. Collect data and information needed to evaluate the operations design.
11. Evaluate data with respect to the defined management objectives (1).
12. Prepare new reservoir management goals and/or operation designs, if necessary, and reevaluate (1-11).

In completing the statistical analyses, it is important to evaluate not only whether the statistical hypotheses can be or cannot be rejected, but also whether the decision to reject or not reject is robust. The problem here is based on the fact that environmental data, by its nature, tends to violate some of the assumptions used in formulating many of the statistical tests for hypotheses. During statistical analysis of collected data, it is important to establish which assumptions are violated by which data for which variables. Appropriate techniques to test the compliance of data to various assumptions, including normal, equal, and independent distributions for error, are available from many sources (e.g., Green 1979, Sokal and Rohlf 1981, Armour et al. 1983). But many statistical tests can provide good results even when their underlying assumptions are violated, that is, the models are robust to violation of their assumptions.

One approach to evaluate the robustness of conclusions derived from the statistical analysis is to apply multiple tests of data using models that vary in their underlying assumptions; when these different models lead to similar conclusions, the conclusions can be viewed as robust, i.e., true (Levins 1966, Green 1979, Marcus 1987b).

SECTION 6

CONCLUSIONS

Increasing demands for water in the West will lead to increasing construction of reservoirs on high elevation headwater streams. Through appropriate management, these reservoirs present the chance to expand and enhance potentially unique fishing opportunities. This report brings together a diversity of information on the characteristics of lakes, streams, reservoirs, and the downstream effects of reservoirs. Its aim has been to present management options for headwater reservoir-stream systems.

Impoundment of streams and rivers produces a diversity of effects potentially influencing fisheries. Included among these are modifications of flow, sedimentation, thermal, chemical, and biological patterns in streams and rivers. A variety of new relationships are established in reservoirs, which differ in perceivable ways from the those found in lakes. Among those potentially important to fisheries are the thermal and hydrological patterns, flows of water through the reservoirs, and potential alterations to water qualities during reservoir passage.

Among the design and management considerations discussed for reservoir-stream systems, those potentially important for headwater reservoirs include changes in channel morphology, necessary flows to flush downstream accumulations of fine sediments, effects on riparian communities, the need to identify limiting factors, options in the management of reservoir and stream fisheries, methods to evaluate potentials for reservoir and stream fisheries, and potential conflicts with wildlife and endangered species.

A number of the discussed management options are suggested as having potential applicability to high elevation reservoir-stream systems that can enhance fishery potentials in both the reservoir and the downstream waters:

- The first step in the development of a management plan requires establishing the need for that plan; then management objectives can be defined. These objectives should include identifying factors limiting biological productivity and actions directed at reducing these limitations. The general goal of the management plan with regard to fisheries should be directed at influencing the basic productivity and structure of the fishery.
- Reservoir outlets should be constructed to include options to discharge waters from greater than one level within the reservoir to enable manipulation of water quality both within the reservoir and downstream.
- The ability to manipulate water levels in headwater reservoirs is limited due to the nature of the water supply to these systems. But, management plans should include a stated minimum pool depth, as defined by a management team.
- A schedule of periodic releases of waters with high flow velocities should be established to flush accumulations of

fine sediments from the downstream beds. Desirable flushing flow regimes should be establish by an interdisciplinary team, including at least a hydraulic engineer and a fisheries biologist.

- With options available to discharge waters from various levels from reservoirs, discharges regimes could be managed to release seasonally warm waters in the winter, spring, and early summer, enhancing downstream productivity rates. To enhance within reservoir warming and production rates, intermediate or lower level waters could be discharged during the summer and fall. Attention also needs to paid to alterations in the chemical quality of reservoir waters to assure that potentially deleterious conditions do not develop.
- Because of the potential stresses that headwater reservoirs place on aquatic systems, additional potential sources of stress, including logging, cattle grazing, and mining, should be severely restricted or prohibited in any watershed above a headwater reservoir.
- The best available model to predict potential standing stocks for trout in high elevation reservoirs is Whitworth's (1984) RQI. But this model should be applied cautiously, with an understanding of it potential flaws and limitations.
- The best general model available to predict potential standing stocks of trout in high elevation streams is Binns' (1982) HQI. For estimating potential standing stocks of brook trout in these streams, however, the model developed by Chisholm and

Hubert (1986) may be more appropriate. But, again, neither these or any other models should be used without understanding their potential flaws and limitations.

- o While the efficacy of artificial structures as enhancements for fishery habitat has not been evaluated for high elevation systems, these structures do present a wide selection of techniques for potentially enhancing fisheries habitats, both within these reservoirs and in the downstream sections.
- o Direct manipulation of the fishery in these systems is possible through stocking, species selection, and harvest regulations. In these high elevation systems, stocking strategies will likely include both fingerling and creel-size fish, with the mix dependent on a number of environmental and economic factors. Species and strain selection may lead to establishing "wild" fisheries in some of these reservoirs. Harvest regulations could be used to upgrade management plans in some of these systems from basic yield to trophy fisheries.

Finally, relatively few data have been collected from high elevation reservoir-stream systems. Consequently, nearly all management actions implemented for these systems could be considered "best guess." In fact, most of the implemented actions will be, in effect, applied research. Therefore, it is of utmost importance that management plans for these systems include monitoring to determine whether the management actions implemented achieve the stated objective(s). This monitoring must be planned and conducted with a mix of good science, statistics, and common sense.

SECTION 7

LITERATURE CITED

American Fisheries Society. 1967. Reservoir Fishery Resources Symposium. American Fisheries Society, Bethesda, MD.

Annear, T.C. and A.L. Conder. 1984. Relative bias of several fisheries instream flow methods. North American Journal of Fisheries Management 4:531-539.

Annear, T.C. and A.L. Conder. 1983. Evaluation of instream flow methods for use in Wyoming. Completion report. Wyoming Game and Fish Department, Fish Division, Cheyenne, WY.

Armour, C.L., K.P. Burnham and W.S. Platts. 1983. Field methods and statistical analyses for monitoring small salmonid streams. FWS/OBS-83/33. U.S. Department of the Interior, Fish and Wildlife Service, Washington, DC.

Banner Associates, Inc. 1980. Little Snake River Water Management Project Feasibility Study. Wyoming Water Development Commission, Cheyenne, WY.

Banner Associates, Inc. 1984. Deer Creek--Level III, Executive summary. Wyoming Water Development Commission, Cheyenne, WY.

Baron, J. 1983. Comparative water chemistry of four lakes in Rocky Mountain National Park. Water Resources Bulletin 19(6):897-902.

Baxter, R.M. 1977. Environmental effects of dams and impoundments. Annual Review of Ecology and Systematics 8:255-283.

Baxter, R.M. 1985. Envirnomental effects of reservoirs. Pages 1-26 in D. Gunnison (editor). Microbial processes in reservoirs. Dr. W. Junk Publishers, Boston, MA.

Beasley, R.P., J.M. Gregory and T.R. McCarty. 1984. Erosion and sediment pollution control. Iowa State University Press, Ames, IA.

Bettess, R. and W.R. White. 1981. Mathematical simulation of sediment movement in streams. Proceedings of the Institution of Civil Engineers 71:635-641.

Binns, N.A. 1979. A habitat quality index for Wyoming trout streams. Fishery research report, Monograph series number 2. Wyoming Game and Fish Department, Cheyenne, WY.

Binns, N.A. 1982. Habitat Quality Index procedures manual. Wyoming Game and Fish Department, Cheyenne, WY. 209 pp.

Binns, N.A. and F.M. Eiserman. 1979. Quantification of fluvial trout habitat in Wyoming. Transactions of the American Fisheries Society 108:215-228.

Boyd, C.E. 1971. The limnological role of aquatic macrophytes and their relationships to reservoir management. Pages 153-166 in G.E. Hall (editor). Reservoir fisheries and limnology. Special publication number 8, American Fisheries Society, Bethesda, MD.

Brocksen, R.W., M. Fraser, I. Murarka and S.G. Hildebrand. 1982. The effects of selected hydraulic structures on fisheries and limnolgy. CRC critical reviews in environmental control, March 1982:69-89.

Brown, A.M. 1986. Modifying reservoir habitat with artificial structures. Pages 98-102 in G.E. Hall and M.J. Van Den Avyle (editors.). Reservoir fisheries management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, MD.

Burgess, S.A. 1985. Some effects of stream habitat improvement on the aquatic and riparian community of a small mountain stream. Pages 223-246 in J.A. Gore (editor). The restoration of rivers and streams, theories and experience. Butterworth Publishers, Boston, MA.

Canter, L. 1985. Environmental impacts of water resources projects. Lewis Publishing, Inc., Chelsa, MI.

Carnevale, M.A. 1986. The role of the 404 program in riparian wetland protection. Pages 70-78 in Wyoming Water 1986 and Streamside Zone Conference, Casper, WY. Wyoming Water Research Center, University of Wyoming, Laramie, WY.

Chisholm, I.M. and W.A. Hubert. 1986. Influence of stream gradient of standing stock of brook trout in the Snowy Range, Wyoming. Northwest Science 60(2):137-139.

Conder, A.L. and T.C. Annear. 1986. A test of weighted usable area estimates derived from a PHABSIM model for instream flow studies on trout streams. Draft manuscript. Wyoming Game and Fish Department, Cheyenne, WY.

Cowell, B.C. and P.L. Hudson. 1967. Some environmental factors influencing benthic invertebrates in two Missouri River reservoirs. Pages 541-555 in Reservoir Fishery Resources Symposium. American Fisheries Society, Bethesda, MD.

Dillon, P.J. and W.B. Kirchner. 1975. The effects of geoolgy and land use on the export of phosphorus from watersheds. Water Research 9:135-148.

EA Engineering, Science, and Technology, Inc. 1986. Instream flow methodologies. Research Project 2194-2. The Electric Power Research Institute, Palo Alto, CA.

Environmental Law Institute. 1977. The evolution of national wildlife law. Prepared for the Council on Environmental Quality. 485 pp.

ESA Geotechnical Consultants. 1986a. Shell Valley watershed, Level III, Phase I: Interium report on conceptual design, Lake Adelaide enlargement. Wyoming Water Development Commission, Cheyenne, WY.

ESA Geotechnical Consultants. 1986b. Big Horn Basin--Clarks Fork, Level II, Final report on project analysis. Wyoming Water Development Commission, Cheyenne, WY.

Fabris, G.L. and U.T. Hammer. 1975. Primary production in four small lakes in the Canadian Rocky Mountains. Verh. Internat. Verein Limnol. 19:530-541.

Fausch, K.D., C.L. Hawkes and M.G. Parsons. 1987. Models that predict standing crop of stream fish from habitat variables (1950-1985). Wildlife and Fishery Ecology Unit, USDA Forest Service, Ft. Collins, CO.

Fisher, W.L., J.J. Charbonneau and M.J. Hays. 1986. Economic characteristics of reservior fisheries. Pages 5-10 in G.E. Hall and M.J. Van Den Avyle (editors). Reservoir fisheries management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, MD.

Funk, W.H. and A.R. Gaufin. 1971. Phytoplankton productivity in a Wyoming cooling-water reservoir. Pages 167-178 in G.E. Hall (editor). Reservoir fisheries and limnology. Special publication number 8, American Fisheries Society, Bethesda, MD.

G.B.R. Consultants Group, Inc. 1985. Smiths Fork Project, Level II Study, Final report. Wyoming Water Development Commission, Cheyenne, WY.

Gebhards, S. 1975. Wild trout, not by a damsite. Pages 31-33 in W. King (editor). Proceedings of the wild trout management symposium. Trout Unlimited, Inc., Washington, DC.

Graf, W.L. 1979. The development of montane arroyos and gullies. Earth Surface Processes 4:1-14.

Green, R.H. 1979. Sampling design and statistical methods for environmental biologists. John Wiley and Sons, New York, NY.

Guenther, P.M. and W.A. Hubert. 1987. Minimum pool requirements for maintainence and enhancement of salmonid fisheries in small, coldwater Wyoming impoundments. Draft proposal, Department of Zoology and Physiology, University of Wyoming, Laramie, WY.

Hall, G.E. (editor). 1971. Reservoir fisheries and limnology. Special publication number 8, American Fisheries Society, Bethesda, MD.

Hannon, H.H. 1979. Chemical modification in reservoir-regulated streams. Pages 75-94 in J.V. Ward and J.A. Stanford (editors). The ecology of regulated streams. Plenum Press, New York, NY.

Harza Engineering Company. 1986. Middle Fork Powder River Dam and Reservoir project, Conceptual design report, Executive summary. Wyoming Water Development Commission, Cheyenne, WY.

Harza Engineering Company, Wright Water Engineers and Simons, Li & Associates. 1983. Storage developments for water supply, Powder River Basin in Wyoming, Level I Reconnaissance study, Volume I - Main report. Wyoming Water Development Commission, Cheyenne, WY.

HEC. 1977. HEC-6. Scour and deposition in rivers and reservoirs. User's manual. U.S. Army Corps of Engineers, Hydrologic Engineering Center, Davis, CA.

Heede, B.H. 1980. Stream dynamics: An overview for land managers. General Technical Report RM-72, Rocky Mountain Forest and Range Experiment Station, Forest Service, United States Department of Agriculture, Fort Collins, CO.

Heede, B.H. 1981. Dynamics of selected mountain streams in the western United States. Z. Geomorph. N. F. 25:17-32.

Henderson-Sellers, B. 1984. Engineering limnology. Pitman Advanced Publishing Program, Boston, MA.

Hendricks, A.C. and J.K.G. Silvey. 1977. A biological and chemical comparison of various areas of a reservior. Water Research 11:429-438.

Howe, C.W. 1980. The coming conflict over water. Pages 15-38 in Western water resources: coming problems and the policy alternatives. Westview Press, Boulder, CO.

Hudelson, R., G. Boyer and J. McMillan. 1980. High mountain lake and stream survey of the Bridger Wilderness Area: 1969-1975. Completion report: D.J. Report F-1-R-8; F-1-R-9; F-1-R-10; F-1-R-11; F-1-R-12. Wyoming Game and Fish Department, Fish Division, Cheyenne, WY.

Hunt, O.P. 1963. Use of low-flow measurements to estimate fow-duration curves. United States Geological Survey Professional Papers 475:C196-C197.

Huston, M. 1979. A general hypothesis of species diversity. American Naturalist 113:81-101.

Hutchinson, G.E. 1957. A treatise on limnology. Volume 1, geography, physics, and chemistry. John Wiley and Sons, Inc., New York, NY.

Hynes, H.B.N. 1970. The ecology of running water. University of Toronto Press, Toronto, Ontario, Canada.

Hynes, H.B.N. 1975. The stream and its valley. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie 19:1-15.

Irvine, J.R. 1985. Effects of varying flows in manmade streams on rainbow trout (Salmo gairdneri) fry. Pages 5-6 in Abstracts, Third International Symposium on Regulated Streams. University of Alberta, Edmonton, Alberta, Canada.

Isom, B.G. 1971. Effects of storage and mainstream reservoirs on benthic macroinvertebrates in the Tennessee Valley. Pages 179-191 in G.E. Hall (editor). Reservoir fisheries and limnology. Special publication number 8, American Fisheries Society, Bethesda, MD.

Jenkins, R.M. 1970. Reservoir fisheries management. Pages 173-182 in A century of fishing in North America. Special publication number 7, American Fisheries Society, Bethesda, MD.

Jenkins, R.M. 1982. The morphoedaphic index and reservoir fish production. Transactions of the American Fisheries Society 111:133-140.

Keith, W.E. 1986. A review of introduction and maintenance stocking in reservoir fisheries management. Pages 144-148 in G.E. Hall and M.J. Van Den Avyle (editors). Reservoir fisheries management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, MD.

Knighton, D. 1984. Fluvial Forms and Processes. Edward Arnold, London. 218 pp.

LaBounty, J.F., J.J. Satoris and D.M. Lieberman. 1984. Effects of operating Mt. Elbert Pumped-storage Powerplant on Twin Lakes, Colorado: 1982 report of findings. Report REC-ERC-84-23, Bureau of Reclamation, Engineering Research Center, Denver, CO.

Leopold, L.B. and T. Maddock, Jr. 1953. The hydraulic geometry of stream channels and some physiographic implications. Geological Survey Professional Paper 252. U.S. Department of the Interior, Geological Survey, Washington, D.C.

Levins, R. 1966. The strategy of model building in population biology. American Scientist 54:421-431.

Lewis, W.M. 1967. Predation as a factor in fish populations. Pages 386-390 in Reservoir fishery resources symposium. American Fisheries Society, Bethesda, MD.

Li, H.W. and P.B. Moyle. 1981. Ecological analysis of species introductions into aquatic systems. Transactions of the American Fisheries Society 110:772-782.

Likens, G.E. 1984. Beyond the shoreline: a watershed-ecosystem approach. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie 22:1-22.

Likens, G.E., F.H. Bormann, R.S. Pierce, J.S. Eaton and N.M. Johnson. 1977. Biogeochemistry of a forested ecosystem. Springer-Verlag, New York, NY.

Loar, J.M. and N.O. Sale. 1981. Analysis of environmental issues related to small scale hydroelectric developments. V: Instream flow needs for fisheries resources. ORNL/TM-7861, Oak Ridge National Laboratory, Oak Ridge, TN.

Lorenzen, M.W. and A.W. Fast. 1977. A guide to aeration/circulation techniques for lake management. U.S. Environmental Protection Agency, EPA-600/3-77-004, Washington, DC.

Marcus, M.D. 1980. Periphyton community response to chronic nutrient enrichment by a reservoir discharge. Ecology 61:387-399.

Marcus, M.D. 1987a. Limnological properties of a Rocky Mountain headwater reservoir. Canadian Journal of Fisheries and Aquatic Sciences. Chapter 2 in M.D. Marcus. Ecology and management of headwater reservoirs. Ph.D. dissertation, University of Wyoming, Laramie, WY.

Marcus, M.D. 1987b. Exploring robust relationships in multivariate matrices of environmental and community variables. Chapter 3 in M.D. Marcus. Ecology and management of headwater reservoirs. Ph.D. dissertation, University of Wyoming, Laramie, WY

Marcus, M.D., J.E. Schillinger and D.G. Stuart. 1978. Limnological studies in Montana: Hyalite Reservoir and responses of lotic periphyton to deep-water discharges, grazing and logging. Report 92, Montana University Joint Water Resources Reseach Center, Montana State Univeristy, Bozeman, MT.

Marcus, M.D., S.W. Wolff, M.S. Kinsey and B.R. Parkhurst. 1985. Fishing in Wyoming: Results of the 1984 fishing pressure survey. A report to the Wyoming Game and Fish Department. Western Aquatics, Inc., Laramie, Wyoming.

Martin, D.B. 1967. Limnological studies on Hebgen Lake, Montana. Ph.D. thesis. Department of Biology, Montana State University, Bozeman, Montana.

Martin, D.B. and R.D. Arneson. 1978. Comparative limnology of deep-discharge reservoir and a surface-discharge lake on the Madison River, Montana. Freshwater Biology 8:33-42.

Mathias, J.A. and J. Barica. 1980. Factors controlling oxygen depletion in ice covered lakes. Canadian Journal of Fisheries and Aquatic Sciences 37(2):185-194.

McAfee, M.E. 1980. Effects of water drawdown on the fauna in small cold water reservoirs. Water Resources Bulletin 16:690-696.

McAfee, M.E. 1984. Small accessible coldwater reservoir studies. Federal aid study F-59-R, Work plan no. II, Job 1. Rainbow trout stocking evaluations. Colorado Division of Wildlife, Fort Collins, CO.

McConnell, W.J. and W.F. Sigler. 1959. Chlorophyll and productivity in a mountain river. Linmology and Oceanography 4:335-351.

Meehan, W.R., F.J. Swanson and J.R. Sedell. 1977. Influences of riparian vegetation on aquatic ecosystems with particular reference to salmonid fishes and their food supply. Pages 137-145 in Importance, preservation and management of riparian habitat: A symposium. USDA Forest Service General Technical Report RM-43. USDA Forest Service, Ft. Collins, CO.

Messer, J.J., E.K. Israelsen and J.D. Adams. 1981. Natural salinity removal processes in reservoirs. Utah Water Research Laboratory, Utah State University, Logan, UT.

Milhous, R., D. Wegner and T. Waddle. 1981. User's guide to the physical habitat simulation system. Instream flow information paper 11, FWS/OBS-81/43. Cooperative Instream Flow Service Group, U.S. Fish and Wildlife Service, Ft. Collins, CO.

Minckley , W.L. and J.E. Deacon. 1968. Southwestern fishes and the enigma of "endangered species." Science 159:1424-1432.

Minshall, G.W., K.W. Cummins, R.C. Petersen, C.E. Cushing, D.A. Burns, J.R. Sedell and R.L. Vannote. 1985. Developments in stream ecosystem theory. Canadian Journal of Fisheries and Aquatic Sciences 42:1045-1055.

Motten, A.F. and C.A.S. Hall. 1972. Edaphic factors override a possible gradient of ecological maturity indices in a small stream. Limnology and Oceanography 17:922-926.

Mulholland, P.J., J.D. Newbold and J.W. Elwood. 1985. Phosphorus spiralling in a woodland stream: seasonal variations. Ecology 66:1012-1023.

Neel, J.K. 1963. Impacts of reservoirs. Pages 575-593 in D.G. Frey (editor). Limnology in North America. University of Wisconsin Press, Madison, WI.

Neel, J.K. 1967. Reservoir eutrophication and dystrophication following impoundment. Pages 322-332 in Reservoir fishery resources symposium. American Fisheries Society, Bethesda, MD.

Nelson, R.W., G.C. Horak and J.E. Olson. 1978. Western reservoir and stream habitat improvements handbook. FWS/OBS-78/56. Office of Biological Services, U.S. Department of the Interior, Fish and Wildlife Service, Washington, DC.

Nelson, W.C. 1985. Alkalinities of high altitude lakes in Colorado. Part I. Research studies. Colorado Division of Wildlife, Aquatic Wildlife Section, Ft. Collins, CO.

Newbold, J.D. 1985. Applicability of the nutrient spiralling concept to river-reservoir systems. Page 53 in Abstracts, Third International Symposium on Regulated Streams. University of Alberta, Edmonton, Alberta, Canada.

Newbold, J.D., J.W. Elwood, R.V. O'Neill and A.L. Sheldon. 1983. Phosphorus dynamics in a woodland stream ecosystem: a study of nutrient spiralling. Ecology 64:1249-1265.

Newbury, R.W. 1985. The effects of stream regulation on watersheds. Pages 21-23 in Abstracts, Third International Symposium on Regulated Streams. University of Alberta, Edmonton, Alberta, Canada.

Noble, R.L. 1986. Preditor-prey interactions in reservoir communities. Pages 137-143 in G.E. Hall and M.J. Van Den Avyle (editors). Reservoir fisheries management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, MD.

Oglesby, R.T. 1982. The MEI Symposium--overview and observations. Transactions of the American Fisheries Society 111:171-175.

O'Neill, R.V., J.W. Elwood and S.G. Hildebrand. 1979. Theroretical implications of spatial heterogeneity in stream ecosystems. Pages 79-101 in G.S. Innis and R.V. O'Neill (editors). Systems analysis of ecosystems. International Cooperative Publishing House, Fairland, MD.

Osterkamp, W.R., L.J. Lane and G.R. Foster. 1983. An analytical treatment of channel-morphology relations. U.S. Geological Survey Professional Paper 1288.

Parker, M. 1986. Beaver, water quality, and riparian systems. Pages 88-94 in Wyoming Water 1986 and Streamside Zone Conference, Casper, WY. Wyoming Water Research Center, University of Wyoming, Laramie, WY.

Parrett, C. and J.A. Hull. 1985. Streamflow characteristics of mountain streams in western Montana. Water-Supply Paper 2260, United States Geological Survey, Alexandria, VA.

Peters, J.C. 1986. Enhancing tailwater fisheries. Pages 278-285 in G.E. Hall and M.J. Van Den Avyle (editors). Reservoir fisheries management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, MD.

Petts, G.E. 1984. Impounded rivers, perspectives for ecological management. John Wiley and Sons, New York, NY.

Pfitzer, D.W. 1954. Investigations of water below storage reservoirs in Tennessee. Transaction of the 19th North American Wildlife Conference 19:271-282.

Platts, W.S. 1986. Managing riparian stream habitats. Pages 59-62 in Wyoming Water 1986 and Streamside Zone Conference, Casper, WY. Wyoming Water Research Center, University of Wyoming, Laramie, WY.

Platts, W.S., W.F. Megahan and G.W. Minshall. 1983. Methods for evaluating stream, riparian, and biotic conditions. General Technical Report INT-138, USDA Forest Service, Ogden UT.

Ploskey, G.R. 1983. A review of the effects of water level changes on reservoir fisheries and recommendations for improved management. Technical Report E-83-3, prepared by the U.S. Fish and Wildlife Service, U.S. Department of the Interior, for the U.S. Army Engineer Waterways Experiment Station, C.E., Vicksburg, MS.

Ploskey, G.R. 1986. Mangement of the physical and chemical environment: Effects of water-level changes on reservoir ecosystems, with implications for fisheries management. Pages 86-97 in G.E. Hall and M.J. Van Den Avyle (editors). Reservoir fisheries management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, MD.

Rawson, D.S. 1942. A comparison of some large alpine lakes in western Canada. Ecology 23:143-161.

Rawson, D.S. 1952. Mean depth and the fish production of large lakes. Ecology 33:513-521.

Redmond, L.C. 1986. The history and development of warmwater fish harvest regulations. Pages 186-195 in G.E. Hall and M.J. Van Den Avyle (editors). Reservoir fisheries management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, MD.

Reiser, D.W., M.P. Ramey and T.R. Lambert. 1985. Review of flushing flow requirements in regulated streams. Pacific Gas and Electric Company, Department of Engineering Research, San Ramon, CA.

Riggs, H.C. 1969. Mean streamflow from discharge measurements. International Association Scientific Hydrology Bulletin 14:95-110.

Rimmer, D.M. 1985. Effects of reduced discharge on production and distribution of age-0 rainbow trout in seminatural channels. Transactions of the American Fisheries Society 114:388-396.

Rollins, Brown and Gunnell, Inc., and Forsgren-Perkins Engineering. 1985. West Fork Dam and Reservoir, Level III, Phase I, Executive summary. Wyoming Water Development Commission, Cheyenne, WY.

Ruane, R.J., C.E. Bohac, W.M. Seawell and R.M. Shane. 1986. Improving the downstream environment by reservoir release modifications. Pages 270-277 in G.E. Hall and M.J. Van Den Avyle (editors). Reservoir fisheries management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, MD.

Ryder, R.A. 1965. A method for estimating the potential fish production of north-temperate lakes. Transactions of the American Fisheries Society 94:214-218.

Ryder R.A. 1982. The morphoedaphic index--use, abuse, and fundamental concepts. Transactions of the American Fisheries Society 111:154-164.

Ryder, R.A., S.R. Kerr, K.H. Loftus and H.A. Reigier. 1974. The morphoedaphic index, a fish yield estimator--review and evaluation. Jounal of the Fisheries Research Board of Canada 31:663-688.

Schillinger, J.E. and D.G. Stuart. 1978. Quantification of non-point water pollution from four land use activities: logging, cattle grazing, mining, and subdivisions. Report 93, Montana University Joint Water Resources Reseach Center, Montana State Univeristy, Bozeman, MT.

Simons, D.B. 1979. Effects of stream regulation on channel morphology. Pages 95-111 in J.V. Ward and J.A. Stanford (editors). The ecology of regulated streams. Plenum Press, New York, NY.

Simons, D.B. and R.T. Milhous (editors). 1981. Proceedings: Workshop on downstream river channel changes resulting from diversions or reservoir construction. FWS/OBS-81/48. Office of Biological Services, U.S. Department of the Interior, Fish and Wildlife Service, Washington, DC.

Sokal, R.R. and F.J. Rohlf. 1981. Biometry, the principles and practice of statistics in biological research. W.H. Freeman and Company, San Francisco, CA.

Soltero, R.A., J.C. Wright, and A.A. Horpsted. 1973. Effects of impoundments on the water quality of the Bighorn River. Water Research 7:343-354.

Spencer, S.L. 1986. Needs of reservoir user-groups. Pages 301-305 in G.E. Hall and M.J. Van Den Avyle (editors). Reservoir fisheries management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, MD.

Statzner, B. and B. Higler. 1985. Questions and comments on the River Continuum Concept. Canadian Journal of Fisheries and Aquatic Sciences 42:1038-1044.

Stuber, R.J. 1985. Trout habitat, abundance, and fishing opportunities in fenced vs. unfenced riparian habitat along Sheep Creek, Colorado. Pages 310-314 in Riparian ecosystems and their management--Reconciling conflicting uses. First North American Riparian Conference, April 16-18, 1985, Tucson, AZ. USDA Forest Service General Technical Report RM-120. USDA Forest Service, Ft. Collins, CO.

Summerfelt, R.C. 1986. Summarization of the symposium. Pages 314-327 in G.E. Hall and M.J. Van Den Avyle (editors). Reservoir fisheries management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, MD.

Swingle, H.S. and W.W. Swingle. 1967. Problems in dynamics of fish populations in reservoirs. Pages 229-243 in Reservoir fishery resources symposium. American Fisheries Society, Bethesda, MD.

Sylvester, R.O. 1965. The influence of reservoir soils on overlying water quality. Pages 327-335 in O. Jaag (editor). Advances in water pollution research. Pergamon Press, New York, NY.

Taylor, W.W. and D.L. King. 1983. Fisheries monitoring and management in freshwater lakes, reservoirs, and ponds. Transactions of the 48th North American Wildlife and Natural Resources Conference 342-348.

Tennant, D.L. 1975. Instream flow regimens for fish, wildlife, recreation and related environmental resources. U.S. Fish and Wildlife Service Department, Billings, Montana. 18 pp.

Toetz, D.W. 1976. Mineral cycling in reservoirs. Annals of the Oklahoma Academy of Sceience 5:21-28.

Turk, J.T. and D.B. Adams. 1983. Sensitivity to acidification of lakes in the Flat Tops Wilderness Area. Water Resources Research 19:346-350.

United States Department of Agiculture (USDA) Soil Conservations Service. 1980. Upper North Laramie River watershed, Albany County, Wyoming, Final environmental impact statement. Soil Conservations Service, United States Department of Agiculture, Casper, WY.

United States Environmental Protection Agency (US EPA). 1983. Technical support manual: Waterbody surveys and assessments for conducting use attainability analyses. Office of Water Regulations and Standards, U.S. Environmental Protection Agency, Washington, DC.

United States Fish and Wildlife Service (US FWS). 1983. Listing endangered and threatened species and designating critical habitat; amended procedures to comply with the 1982 Amendments to the Endangered Species Act. Federal Register 48(153), U.S. Printing Office, Washington, DC.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell and C.E. Cushing. 1980. The River Continuum Concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Varley, J.D., A.F. Regenthal and R.W. Wiley. 1971. Growth of rainbow trout in Flaming Gorge Reservoir during the first six years of impoundment. Pages 121-136 in G.E. Hall (editor). Reservoir fisheries and limnology. Special Publication Number 8, American Fisheries Society, Bethesda, MD.

Ward, J.V. 1974. A temperature-stressed stream ecosystem below a hypolimnial release mountain reservoir. Arch. Hydrobiol. 74:247-275.

Ward, J.V. 1976. Comparative limnology of differentially regulated sections of a Colorado mountain river. Arch. Hydrobiol. 78:319-342.

Ward, J.V. and J.A. Stanford (editors). 1979a. The ecology of regulated streams. Plenum Press, New York, NY.

Ward, J.V. and J.A. Stanford. 1979b. Ecological factors controlling stream zoobenthos with emphasis on thermal modification of regulated streams. Pages 35-55 in J.V. Ward and J.A. Stanford (editors). The ecology of regulated streams. Plenum Press, New York, NY.

Ward, J.V. and J.A. Stanford. 1983. The serial discontinuity concept of lotic ecosystems. Pages 29-42 in T.D. Fontaine III and S.M. Bartell (editors). Dynamics of lotic ecosystems. Ann Arbor Sciences, Ann Arbor, MI.

Weber, C.I. (editor). 1973. Biological field and laboratory methods for measuring the quality of surface waters and effluents. EPA-670/4-73-001. Office of Research and Development, U.S. Environmental Protection Agency, Cincinnati, OH.

Webster, J.R. and B.C. Patten. 1979. Effects of watershed perturbations on stream potassium and calcium dynamics. Ecological Monographs 49:51-72.

Wells, J.D. 1976. The fishery of Hyalite Reservoir during 1974 and 1975. M.S. thesis. Department of Biology, Montana State University, Bozeman, MT.

Wesche, T.A. 1980. The WWRI trout cover rating method--development and application. Water Resources Research Institute, University of Wyoming, Laramie, WY.

Wesche, T.A. 1983. Overview of instream habitat evaluation methods currently applied in the western region. Proceedings of the Western Game and Fish Commissioners Annual Meeting 63:191-201.

Wesche, T.A. 1985. Stream channel modifications and reclamation structures to enhance fish habitat. Pages 103-163 in J.A. Gore (editor). The restoration of rivers and streams, theories and experience. Butterworth Publishers, Boston, MA.

Wesche, T.A., C.M. Goertler and C.B. Frye. 1985a. Importance and evaluation of instream and riparian cover in smaller trout streams. Pages 325-328 in Riparian Ecosystems and Their Management--Reconciling Conflicting Uses. First North American Riparian Conference, April 16-18, 1985, Tucson, AZ. USDA Forest Service General Technical Report RM-120. USDA Forest Service, Ft. Collins, CO.

Wesche, T.A., V.R. Hasfurther, W.A. Hubert and Q.D. Skinner. 1985b. Assessment of flushing flow recommendations in a steep, rough, regulated tributary. Wyoming Water Research Center, University of Wyoming, Laramie, WY.

Wesche, T.A., V. Hasfurther, Q. Skinner and W. Hubert. 1983. Development of a methodology to determine flushing flow requirements for channel maintenance purposes. Research proposal submitted to Wyoming Water Research Center, University of Wyoming, Laramie, WY.

Wesche, T.A. and P.A. Rechard. 1980. A summary of instream flow methods for fisheries and related research needs. Eisenhower Consortium Bulletin number 9. 122 pages.

Wesche, T.A., D.W. Reiser, W.F. Wichers and D.J. Wichers. 1977. Fishery resources and instream flow recommendations for streams to be impacted by Cheyenne's proposed Phase II developments. Completion Rept. submitted to Wyoming Game and Fish Department. Wyoming Water Resources Research Institute, University of Wyoming, Laramie, WY.

Western Division of the American Fisheries Society. 1982. The best management practices for the management and protection of western riparian stream ecosystems. American Fisheries Society, Bethesda, MD.

Wetzel, R.G. 1975. Limnology. W.B. Saunder Company, Philadelphia, PA.

Whalen S.C., S.A. Leathe, R.W. Gregory and J.C. Wright. 1982. Physico-chemical limnology of the Tongue River Reservoir, Montana. Hydobiologia 89:161-176.

Whitworth, W.E. 1984. Reservoir quality index models: regression models of trout productivity for small reservoirs in Wyoming. Completion report submitted to U.S. Fish and Wildlife Service, U.S. Department of the Interior. University of Wyoming, Laramie, WY.

Whitworth, W.E. 1985. Development of a reservoir quality index model for Wyoming. Pages 51-56 in J. Sartoris (editor). Collected papers from the 20th annual meeting. Colorado-Wyoming Chapter of the American Fisheries Society, March 20-21, 1985, Laramie, WY.

Willis, D.W. 1986. Review of water level management on Kansas reservoirs. Pages 110-114 in G.E. Hall and M.J. Van Den Avyle (editors). Reservoir fisheries management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, MD.

Wright, J.C. 1967. Effect of impoundment on productivity, water chemistry, and heat budgets of rivers. Pages 188-199 in Reservoir Fishery Resources Symposium. American Fisheries Society, Bethesda, MD.

Wunderlich, W.O. 1971. The dynamics of density-stratified reservoirs. Pages 219-231 in G.E. Hall (editor). Reservoir Fisheries and Limnology. Special Publication Number 8, American Fisheries Society, Bethesda, MD.

Wydoski, R.S. and D.H. Bennett. 1981. Forage species in lakes and reservoirs of the western United States. Transactions of the American Fisheries Society 110:764-771.

Wyoming Game and Fish Commission. 1985. Mitigation policy. Wyoming Game and Fish Department, Cheyenne, WY.

Wyoming Game and Fish Department. 1977. Wyoming stream fishery classification. Wyoming Game and Fish Department, Cheyenne, WY.

Wyoming Game and Fish Department. 1984. Upper Bear River report, Level III fish and wildlife impact report. Prepared for the Wyoming Water Development Commission. Wyoming Game and Fish Department, Cheyenne, WY.

Wyoming Game and Fish Department. 1985. Little Snake River Water Managemant Project, Level III interim wildlife impact report. Prepared for the Wyoming Water Development Commission. Wyoming Game and Fish Department, Cheyenne, WY.

Wyoming Water Development Commission. 1984. Water development potential in the Tongue River Basin. Wyoming Water Development Commission, Cheyenne, WY.

Wyoming Water Development Commission. 1986. Legislative report, Wyoming Water Development Program. Wyoming Water Development Commission, Cheyenne, WY. 106 pp.

Zubik, R.J., Jr. 1983. The fishery of Hyalite Reservoir, Montana, with an evaluation of cutthroat trout reproduction in its tributaries. M.S. thesis. Department of Biology, Montana State University, Bozeman, MT.