Effects of land-use activities on lacustrine systems: literature review

William J. Liss[1], R.L. Hoffman[1], R.E. Gresswell[2], E.A. Deimling[1], and G.L. Larson[3]

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[1]Oak Creek Laboratory of Biology, Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR 97331.

[2]USDA Forest Service, Corvallis Forestry Sciences Lab, 3200 SW Jefferson Way, Corvallis, OR 97331.

[3]National Biological Service, Department of Forest Resources, Oregon State University, Corvallis, OR 97331.

Preface

The following report was prepared by University scientists through cooperative agreement, project science staff, or contractors as part of the ongoing efforts of the Interior Columbia Basin Ecosystem Management Project, co-managed by the U.S. Forest Service and the Bureau of Land Management. It was prepared for the express purpose of compiling information, reviewing available literature, researching topics related to ecosystems within the Interior Columbia Basin, or exploring relationships among biophysical and economic/social resources.

This report has been reviewed by agency scientists as part of the ongoing ecosystem project. The report may be cited within the primary products produced by the project or it may have served its purposes by furthering our understanding of complex resource issues within the Basin. This report may become the basis for scientific journal articles or technical reports by the USDA Forest Service or USDI Bureau of Land Management. The attached report has not been through all the steps appropriate to final publishing as either a scientific journal article or a technical report.

SUMMARY

ANTHROPOGENIC THREATS TO LAKES IN THE INTERIOR COLUMBIA BASIN

EXOTIC AND NON-NATIVE SPECIES

Effects of Exotic and Non-native Fishes on Native Fishes

Miller et al. (1989) documented the extinction of 27 North American fish species and cited introduction of exotic and non-native fish species as contributing to 68% of the losses. Introduced fishes may have deleterious impacts by preying on native fishes (Walters and Legner 1980; Marrin and Erman 1982; Marsh and Brooks 1989), competing with native fish faunas for limited resources which can lead to exclusion (Nelson, 1965; Larson and Moore 1985; Moore et al. 1986; Krueger and May 1991), and by hybridizing with native fishes (Nelson 1965; Cavender 1978; Leary et al. 1983; Taylor et al. 1984; Kohler and Courtenay 1986; Allendorf and Leary 1988; Krueger and May 1991).

Effects of Exotic Species on Amphibians

Introduction of exotic species is considered one of several possible causes for amphibian declines (Blaustein and Wake 1990; Wake 1991). Bullfrogs (Rana catesbiana) have been implicated in declines of native ranids in the west (Moyle 1973; Hammerson 1982). Hayes and Jennings (1986) concluded that the evidence was inadequate to distinguish between fish, bullfrogs, and other possible causes of decline.

Fish can alter behavior, inhibit growth, reduce survival, and restrict distribution of amphibians (e.g., Voris and Bacon 1966; Grubb 1972; Hayes and Jennings 1986; Petranka 1983; Stangel and Semlitsch 1987; Semlitsch 1987; 1988; Sih et al. 1988;

Bradford 1989; Figiel and Semlitsch 1990; Bradford et al., 1993). In the west, predation by introduced trout has been implicated in causing the decline of two species of frogs (Bradford 1989; Bradford et al. 1993; Fellers and Drost 1993) and two species of ambystomid salamanders (Sprules 1974; Taylor 1983; Liss et al. 1995, Torrey Tyler, in preparation, Department of Fisheries and Wildlife, Oregon State University). Introduced fish may also spread the pathenogenic fungus Saprolegnia ferax (Blaustein et al. 1993).

Effects of Exotic and Non-native Fishes on Zooplankton

Fish can selectively prey on larger, more active, more visible zooplankters and cause a shift in the size composition of the zooplankton from larger-bodied to smaller-bodied species (Zaret 1980; Northcote 1988; Gliwicz and Pijanowska 1989; Zaret 1990). Impacts on zooplankton may be particularly acute in oligotrophic lakes such as high-elevation lakes with high fish densities (e.g., Gliwcz and Prejs 1977; Dodson 1979; Kerfoot 1987; McQueen et al. 1986; Post and McQueen 1987; Zaret 1990; Liss et al. 1995, in review). Bahls (1992) estimated that about 95% of high-elevation lakes in the west now supporting fish were originally fishless. The phantom midge, Chaoborus, a major invertebrate predator in many western lakes, large calanoid copepods, and large Daphnia are particularly sensitive to predation from introduced fishes (Anderson 1972, 1974, 1980; Northcote et al. 1978; Stoddard 1987; Bahls 1990; Starkweather

1990; Chess et al. 1993; Lamontagne and Schindler 1994; Liss et al. 1995, in review).

Effects of the Opposum Shrimp (Mysis relicta)

Mysis was introduced into lakes in the western US to provide forage for sport fish, particularly kokanee salmon (Lasenby et al. 1986; Northcote 1991). It is generally conceded that Mysis has not succeeded in enhancing growth of target species and has caused major changes in the food webs of some lakes (Reiman and Bowler 1980; Lasenby et al. 1986; Martinez and Bergersen 1989; Northcote 1991; Spencer et al. 1991).

Mysis can reduce abundance and eliminate cladoceran zooplankton from lakes (e.g., Morgan et al. 1978; Goldman et al. 1979; Morgan et al. 1981; Reiman and Falter 1981; Northcote 1991; Spencer et al. 1991) and is considered to have contributed to the decline of kokanee salmon in some lakes by reducing the salmon's food supply (Reiman and Falter 1981; Lasenby et al. 1986; Martinez and Bergersen 1989; Bowles et al. 1991; Spencer et al. 1991; Northcote 1991). A moratorium on stocking Mysis has been recommended (Morgan 1982; Lasenby et al. 1981; Northcote 1991).

TIMBER HARVEST AND ROAD CONSTRUCTION

Most of our knowledge about the impacts of timber harvest and road building on aquatic ecosystems has been derived from research on streams. Very few studies have examined the impact of timber harvest on lakes. Since water within a watershed is often delivered to lakes via stream channels, impacts on streams from timber harvest and road construction eventually may be manifested within lakes. Reported effects of timber harvest and road building on lakes include increased sediment and nutrient transport into lacustrine ecosystems which can contribute to increased algal production and lake eutrophication (Leonard et al. 1979; Birch et al. 1980; Spencer 1991; Flathead Basin Forest Practices Water Quality and Fisheries Cooperative Program 1991).

GRAZING

Few studies have been conducted to examine the effects of grazing on lacustrine ecosytems. Livestock grazing can contribute to increased nutrient loading, especially of phosporus, and subsequent decreased water quality in lakes (Juul et al. 1990; Soltero et al. 1991a, 1991b). Feedlot waste runoff caused dissolved oxygen stress and high ammonia concentrations sufficient to kill essentially all the game fish in a flood control reservoir (Scalf et al. 1970; Meehan and Platts 1978).

MINING

Few studies have been conducted in the Columbia basin measuring impacts of mining on aquatic systems. In general, mining operations can pollute aquatic systems with large quantities of sediments and metal- or acid-contaminated solutions (Besser and Rabeni 1987; Davies-Colley et al. 1992) which can be deleterious to aquatic organisms (Wong et al. 1978; Leland and

Kuwabara 1985; Davies-Colley et al. 1992; Pascoe et al. 1993).

High levels of lead were reported in the sediments and some of the biota of Coeur d'Alene Lake in Idaho in the late 1980's (Henny et al. 1991; Hornig et al. 1988). In Owyhee Reservoir smallmouth bass (Micropterus dolomieu) over four years old exceeded the U.S. Food and Drug Administration's mercury limit for human consumption (Allen-Gil et al. 1995). One hundred twenty-five years of gold mining activities on the Clark Fork River in Montana have left a legacy of high concentrations of arsenic, cadmium, copper, and zinc in sediments in Milltown Reservoir and surrounding wetlands. This reservoir is a National Priority 'Superfund' List site (Pascoe et al. 1993).

HUMAN DEVELOPMENT

Eutrophication of a body of water is indicative of deteriorating water quality due to increased plant nutrient loading, especially of nitrogen and phosphorus. Development activities that contribute to increased nutrient levels include point sources such as effluents from industries and sewage treatment plants, and nonpoint sources such as agricultural operations, residential development and septic systems, road construction, and forest practices (Ellis 1989; Tralles 1991; Dojlido and Best 1993). Cultural eutrophication has documented in several lakes in the west including lakes in relatively remote areas (Soltero et al. 1974; Brugam 1987; Brugam and Vallarino 1989; Odell Pilot Watershed Project Analysis 1994), Lake Tahoe

(Goldman 1981), and Flathead Lake in Montana, where nutrient loading to the lake was reduced by improved sewage treatment facilities in nearby towns (Stanford et al. 1994).

RECREATION

Direct recreation activities capable of influencing lakes include wilderness hiking and camping, fishing, and intensive power boat use. For lakes in heavily used areas in national parks and scenic areas, heavy foot traffic, or trampling, can have an effect on vegetation both through direct mechanical action, and indirectly through soil changes (Liddle 1975; Bowles and Maun 1982). Anglers can substantially reduce densities of desirable fish species (Willis et al. 1994; Schram et al. 1990), especially in small water bodies (Willis et al. 1994; Mosindy et al. 1987). Power boat activity can resuspend bed sediments (Garrad and Hey 1987), aid in the spread of the exotic Eurasian watermilfoil (*Myriophyllum spicatum*), and introduce toxic compounds into the aquatic environment (Maguire et al. 1986; Wachs et al. 1992; Beeker et al. 1992; Cairns and Palmer 1993).

CONCLUSIONS

1. Most public and scientific interest in aquatic ecosystems in the northwest has focused on streams and rivers. Within the interior Columbia basin region, lakes are important aquatic ecosystems valued for water quality, recreation, and as

repositories for regional biodiversity.

2. Water quality and native biota in lakes within the region are potentially threatened by an array of human activities including introduction of exotic and non-native species of fish and invertebrates, timber harvest and road building, grazing, mining, human development, and recreation. Little is known about the impacts of human activities on lakes in the region, particularly timber harvest, grazing, and mining. Very few comprehensive monitoring programs have been established to detect long-term changes in lacustine ecosystems.

3. Many of the lakes in the region are high-elevation oligotrophic lakes. These lakes are especially valued for water quality and recreation, but they are particularly sensitive to human activities especially introduction of exotic species and human actions that can increase nutrient loading such as timber harvest and cultural eutrophication. Many lakes that are otherwise pristine have been stocked with non-native fishes and Mysis which have caused significant changes in the food webs of the lakes through elimination of native species of vertebrates and invertebrates.

4. Human activities can cause significant changes in lacustrine ecosystems. Sometimes the effects of human activities have been catastrophic and virtually irreversible, with major changes in lake ecosystems occurring over relatively short time periods

(e.g., food web changes following introduction of *Mysis*), while other effects have been persistent and cumulative (e.g., increased nutrient loading). In some cases deleterious impacts have been at least partially ameliorated.

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LITERATURE REVIEW

POTENTIAL IMPACTS OF TIMBER HARVESTING AND ROAD CONSTRUCTION ON LAKES

The Columbia River Basin is composed of seven provinces. Five provinces contain forests from which timber has been or is presently being harvested (McNab and Avers 1994). These forested terrestrial ecosystems include the yellow pine forests of the Modoc Plateau, the lodgepole and ponderosa pine forests of the eastern Cascades, mixed conifer forests in the Northern and Middle Rocky Mountain provinces, and forests within the Southern Rocky Mountain Province containing at least 50% Douglas-fir.

Forested terrestrial ecosystems can be viewed as watersheds within which are embedded water resources manifested, in part, by the presence of streams and lakes. In this context, watersheds are the environment of streams and lakes (Warren 1979; Aber and Melillo 1991; Liss et al. 1995), and forests contribute to the maintenance and enhancement of biodiversity within these aquatic systems (Sharma et al. 1992). Forests also slow rates of erosion, soil loss, and sedimentation, as well as protect and enhance the quality and quantity of water within watersheds (Anderson 1988; Brooks et al. 1992; Sharma et al. 1992).

Anderson (1988), citing a 1986 report of the Montana State Water Quality Bureau, states that the single greatest threat to watersheds and aquatic life is timber harvest and accelerated road building within forests. This threat is due, in part, to the increased level of harvesting timber from steeper, more environmentally sensitive terrain (Platts and Megahan 1975; Anderson 1988). The potential capacity of a watershed to incur a given level of disruption or disturbance can be related to natural conditions within the watershed, including the geologic instability of the site, the steepness of its slopes and of its river gradients, and the shallowness of its soils (Frissell and Liss 1986; Brooks et al. 1992). The mechanical processes involved in timber harvest and road construction in conjunction with these natural conditions can trigger and augment the level of disruption or disturbance within the watershed. According to van Kesteren (1992), soil and site disturbance inevitably occur during timber harvest activities and result in changes in water quality and quantity. The Aquatic/Watershed Group (FEMAT 1993) also reported that increased rates of erosion and sedimentation in watersheds can be associated with most forest management activities.

Disturbance deforestation has been used as a label for activities associated with the extraction of timber from primary and secondary forests being utilized for sustainable timber production (Rowe et al. 1992). Numerous generalized impacts on watersheds and aquatic ecosystems have been associated with this type of timber extraction. Harvest activities can cause changes in watershed hydrologic processes, such as increases in surface runoff, frequency and intensity of flood events, and altered stream flow regimes (Brooks et al. 1992; Rowe et al. 1992; FEMAT 1993), as well as soil degradation and depletion (Brooks et al. 1992; Rowe et al. 1992; van Kesteren 1992). These perturbations can lead to changes in water quality and quantity (Brooks et al. 1992; van Kesteren 1992), disruption of nutrient cycles within aquatic ecosystems (Rowe et al. 1992), and the modification and destruction of terrestrial and aquatic habitats (van Kesteren 1986, FEMAT 1993). All of these changes can eventually culminate in the loss of biodiversity within a watershed (Rowe et al. 1992; FEMAT 1993).

Most of what is known concerning the effects of timber harvesting and road construction on aquatic ecosystems has been determined by research related to impacts to streams. Platts and Megahan (1975), Frissell and Liss (1986), Eaglin and Hubert (1993), and Havis et al. (1993) have demonstrated that timber harvesting and road construction can cause an increase in the delivery of sediments, especially fine sediments, to streams and this can be detrimental to salmonids. As the deposition of fine sediments in salmonid spawning habitat increases so too does the mortality of embryos, alevins, and fry. Frissell and Liss (1986) found that the clearcut harvesting of timber in the Euchre Creek Basin of south coastal Oregon, when associated with major storm events, resulted in the deposition of large volumes of sediment to stream channels from catastrophic and widespread mass slope failures and increased frequency of streamside slides. It has also been shown, in Pacific Northwest forests, that road networks contribute more sediment to streams than all other land management activities (FEMAT 1993). Roads modify natural hillslope drainage networks, accelerate erosion processes, and can cause or increase the frequency of landslides, surface erosion, and stream channel diversions. These impacts can be chronic, as well as catastrophic and can have serious biological consequences including decreased levels of primary productivity, invertebrate abundance, and prey availability for fish (FEMAT 1993).

Since water within a watershed is often delivered to lakes via stream channels, we can infer that impacts to streams related to timber harvest and road construction are eventually manifested within lakes. The Flathead Basin Forest Practices Water Quality and Fisheries Cooperative Program, in their final report (June, 1991) summary of recommendations, stated that stream erosion and the subsequent increase in sediment and nutrient transport due to land use activities in the Flathead Basin has contributed to lake eutrophication. The examination by Leonard et al. (1979) of the impact of anthropogenic disturbances within the Ward Valley watershed of the Lake Tahoe Basin, California, on the water quality of Lake Tahoe can be used to illustrate this relationship. Disturbances within this watershed, since 1863, included intensive logging, road building, and some recreational home construction. Perturbations within the watershed associated with these ongoing activities included accelerated surface runoff and periodic heavy main channel discharge resulting in severe stream channel and bank erosion, especially during high flow conditions. These impacts on streams within the Ward Valley watershed have resulted in an increase in the delivery of sediments to Lake Tahoe causing increases in nutrient loading levels, especially whole-lake NO_3-N concentrations in excess of natural rates. The result of these changes has been an increase in algal productivity over the entire lake and a continuing acceleration of lake eutrophication.

Several researchers have been able to identify specific effects of timber harvest and road construction on lakes. Birch et al. (1980) studied four lakes in western Washington. They found that timber harvest activities within the watersheds of three of the lakes caused increases in lake sedimentation rate and lake productivity, potentially accelerating changes in the trophic status of each lake. Spencer (1991) examined the historical deposition of sediment in three lakes of the Flathead Basin associated with land use within the lake watersheds. Sediment core evidence indicated that timber harvest activities and road construction, including railroad construction, increased sedimentation rates above natural levels. Road construction appeared to be the greatest cause of disturbance resulting in enhanced fine sediment deposition in lakes downstream from the construction areas. Cohen et al. (1993) investigated the potential effect of three classes of sediment disturbance caused by deforestation along the perimeter of Lake Tanganyika, Africa, on the species richness of three groups of aquatic organisms.

They found that shallow-water areas with soft and hard substrates that are located near terrestrial sites that have incurred high levels of sediment disturbance have significantly lower levels of Ostracod species richness. In addition, the species richness of fish and diatoms was also lowest in these lake locations. Cohen et al. (1993) hypothesize that this difference is due to increased sediment loads entering the lake caused by disturbance associated with deforestation activities.

Although timber harvest activities and road construction have been identified as negatively affecting aquatic ecosystems, there are several caveats associated with these impacts. According to Platts and Megahan (1975) and Spencer (1991) surface erosion and increased levels of sedimentation can often decrease after initial disturbance. Negative impacts tend to increase when activities occur on less accessible environmentally sensitive terrain with steep slopes composed of highly erodible soils, and subject to high climatic stresses (Platts and Megahan 1975; Anderson 1988). Also, according to Eaglin and Hubert (1993), the extent and intensity of harvest and construction activities mitigate the intensity of their impact. A method developed by van Kesteren (1986) to assess the sensitivity of land to timber harvesting activities identifies vulnerable watersheds as those having high to extreme slope gradients, high levels of potential soil erodibility, soils having moderate to very poor drainage, and soil moisture contents in excess of field capacity for long periods of the year. According to van Kesteren

(1986) areas judged to be potentially sensitive to negative impacts should be intensively ground surveyed to determine their suitability for timber harvest and road construction, as well as to ensure that these land use activities, if undertaken, are carried out based on guidelines designed to maximize environmental protection.

POTENTIAL IMPACTS OF LIVESTOCK GRAZING ON LAKES

The grazing of domestic livestock, especially cattle and sheep, in the western United States intensified during the mid-nineteenth century associated with the settling of the west by homesteaders (Jacobs 1991). According to Horning (1994), over 270 million acres of public land has been allotted by the Forest Service and the BLM for livestock grazing in 11 western states. In the Columbia River Basin, grazing is considered a primary land use activity of concern in three provinces (i.e., Palouse, Intermountain Semi-Desert, and Sierran Steppe/Mixed Forest) (McNab and Avers 1994). In the three Rocky Mountain provinces the potential impact of grazing has been estimated as moderate and the impact of grazing in the Eastern Cascades Province is considered slight or highly localized (i.e., see Table 2. Potential effects of land-use activities...).

The potential for livestock grazing activities to negatively impact an area is related, in part, to the evolutionary history of the site being grazed (Archer and Smeins 1991), including its geology, climate, geomorphology, soils, vegetation, water runoff patterns, etc. (Meehan and Platts 1978). Level of grazing pressure also influences the intensity of any potential impact related to grazing activities (Meehan and Platts 1978; Archer and Smeins 1991).

Investigators have determined numerous impacts of grazing activities on the terrestrial ecosystems of watersheds. The presence of livestock in an area can lead to a reduction of soil structure and eventual soil compaction, as well as damage to and loss of vegetative cover, which contribute to an increase in the rate and erosive force of surface runoff (Meehan and Platts 1978; Thurow 1991). The rate of soil erosion eventually increases, leading to a loss of stored nutrients in the soil and a decrease in the level of site productivity (Thurow 1991). The degree of soil erosion associated with livestock grazing is mitigated by the slope gradient and aspect of the site being grazed, the condition of the soil, type and density of vegetation, and the accessibility of the site to livestock (Meehan and Platts 1978).

Livestock grazing activities can affect water quantity and quality in aquatic ecosystems (Meehan and Platts 1978; Jacobs 1991). According to Horning (1994), the single most important factor in the destabilization of riparian/aquatic ecosystems is grazing and trampling of riparian vegetation. Particulate pollution of aquatic systems associated with livestock grazing activities is due, especially, to an increase in the amount of sediment delivered to and transported by the system (Meehan and

Platts 1978; Jacobs 1991; Thurow 1991; Horning 1994). Any chemical pollution caused by grazing activities is due, in part, to increased levels of nutrients delivered to aquatic systems which have been transported from storage in terrestrial soils and leeched from livestock wastes (Jacobs 1991).

What is known about the impact of livestock grazing activities on aquatic ecosystems has been derived primarily from stream-related research. Meehan and Platts (1978) state that unstable stream conditions exist as part of the natural conditions of streams, and grazing activities can augment these unstable conditions, as well as create additional instability within a stream system. Grazing affects four components of stream ecosystems including streambank soil structure, streamside vegetation, stream channel morphology, and water quality (Platts 1978). Livestock trampling of a stream's riparian area can cause a breakdown of soil structure leading to localized and accelerated soil erosion (Thurow 1991; Williamson et al. 1992) manifested as bank sloughing, slumping, and shearing (Platts 1978; Vollmer and Kozel 1993). Livestock grazing can alter the species composition of streamside vegetation (Platts 1978; Stebbins 1981; Archer and Smeins 1991; Thurow 1991; Vollmer and Kozel 1993), as well as diminish the productivity of a site by depleting and eradicating vegetation (Meehan and Platts 1978; Platts 1978; Archer and Smeins 1991; Thurow 1991; Vollmer and Kozel 1993; Horning 1994). The impact on stream channel morphology is manifested by deterioration of channel form

(Williamson et al. 1992) related to channel widening and filling (Platts and Nelson 1985; Horning 1994), loss of undercut banks (Horning 1994), changes in channel substrate composition (Platts 1978; Platts and Nelson 1985), and disruption of pool-riffle relationships (Platts 1978). Stream water quality is affected by an increase in the rate of sediment deposition and amount of suspended sediments (Meehan and Platts 1978; Platts 1978; Vollmer and Kozel 1993; Horning 1994), increased nutrient loading (Platts 1978; Taylor et al. 1989; Hall and Amy 1990; Jacobs 1991), and an increase in the presence of potentially harmful bacteria (e.g., *Pseudomonas aeruginosa* and *Aeromona hydrophila*) due, in part, to runoff contaminated by livestock wastes (Taylor at al. 1989; Hall and Amy 1990; Thurow 1991).

Fish populations and the community composition and structure of aquatic insects are affected by the impacts of livestock grazing on streams. Salmonids are negatively affected by increased deposition of fine sediments on spawning sites, as well as the in-stream trampling of these sites by livestock (Meehan and Platts 1978; Williams et al. 1990; Vollmer and Kozel 1993; Horning 1994). Increased ammonia and nitrate levels and a concomitant increase in the abundance of bacteria harmful to fish have also been shown to increase the mortality and morbidity of fish in streams (Taylor et al. 1989; Hall and Amy 1990). Rinne (1988) found that the communities of aquatic insects in reaches along grazed portions of a stream's riparian area were different than in reaches associated with areas excluded from livestock grazing. Communities in reaches associated with grazing activities were composed of organisms more tolerant of increased silt levels, increased levels of total alkalinity and mean conductivity, as well as elevated water temperatures.

Perturbations similar to those related to impacts on streams and stream riparian areas due to livestock grazing activities may also occur when livestock are present in the riparian areas and along the shorelines of lakes. Although no specific studies that we know of have examined direct impacts on lake riparian areas and shorelines due to grazing and trampling by livestock, several studies have documented the connection between streams and lakes and livestock grazing activities. Juul et al. (1990) and Soltero et al. (1991a, 1991b) examined potential causes of decreased water quality and increased nutrient levels in two lakes (i.e., Deer Lake and Sacheen Lake) in eastern Washington. Heavy grazing activity occurred on marshlands and meadows at the north end of each lake and these areas were drained by the primary tributaries entering the lakes. Investigators determined that increases in nutrient loading levels, especially phosphorus, could, in part, be attributed to these livestock grazing activities, and contributed to decreasing water quality and changing lake trophic status. Soltero et al. (1991b) stated that the extent of impact by livestock is related to animal density, the proximity of livestock to streams, the accessibility of streams to livestock, and the concentration of animal waste. Finally, Meehan and Platts (1978) reported that feedlot waste runoff entering a 45-

acre flood control reservoir (Scalf et al. 1970) caused dissolved oxygen stress and high ammonia concentrations sufficient to kill essentially all the game fish (i.e., largemouth bass, white crappie, and channel catfish) in the reservoir.

Several investigators (i.e., Platts and Nelson 1985; Rinne 1988; Taylor et al. 1989; Williams et al. 1990; Vollmer and Kozel 1993) have demonstrated that better management and control of livestock movement and land use can positively affect the rehabilitation and restoration of aquatic ecosystems impacted by intensive grazing activities. Vollmer and Kozel (1993) identify grazing site rotation and restriction of access to critical sites as two important management alternatives. They anticipate that the implementation of alternatives that reduce streambank and stream channel deterioration will positively enhance chinook salmon populations within the Bear Valley Basin of Idaho. Taylor et al. (1989) found that the exclusion of cattle from the Ash Springs system in Nevada reduced ammonia, nitrite, and pathogenic bacteria levels to concentrations which allowed fish populations to recover. Recommendations for improving the water quality of Sacheen Lake, Washington, included reducing the surface tributary total phosphorus load transported to the lake by a creek draining a cattle grazing area (Soltero et al. 1991b). Livestock management alternatives such as these can positively affect damaged and deteriorated aquatic ecosystems, yet, recovery of heavily impacted sites can proceed quite slowly, often taking years after livestock have been physically excluded from an area

(Platts 1978).

POTENTIAL IMPACTS OF MINING ON LAKES

The interior Columbia River basin is an area rich in minerals. Minerals mined throughout the region include metals (antimony, copper, gold, iron, lead, mercury, molybdenum, silver, zinc) industrial minerals (asbestos, clay, diatomite, dolomite, feldspar, fluorine, gravel, limestone, perlite, phosphate, pumice, sand, silica, talc, zeolite), gem stones/abrasives (garnet), and fuel minerals (uranium; Bryant et al. 1980; Jackson and Kimerling 1993). There are also potential reserves of barite, vanadium, and cobalt (Bryant et al. 1980). Aluminum is processed in the region, though most of the barite and alumina is imported from other countries (Bryant et al. 1980).

The primary geological processes involved with the building of mountain ranges within the interior Columbia basin have resulted in diverse metallic mineral deposits (Jackson and Kimerling 1993; King et al. 1995). Consequently, a large variety of metal mining operations occur throughout the mountainous areas in region (Bryant et al. 1990), especially the Rocky Mountain provinces. It is noteworthy that the mining industry still operates under laws developed in the last quarter of the 1800's (Audubon, Jan 1994).

Few studies have been conducted in the Columbia basin measuring impacts of mining on aquatic systems. However, based on studies in a variety of areas, some generalizations can be made. Mining operations can pollute aquatic systems with large quantities of sediments and metals- or acid-contaminated solutions. Mining activities can increase loads of sediment to watersheds through erosion of mine tailings (Besser and Rabeni 1987); by direct discharge of mining wastes to streams or rivers and their connected lakes or reservoirs (REF); through movement of ground water streamlines which intersect the surface watershed (Davies-Colley et al. 1992). Coarse particles which enter watersheds are likely to settle relatively rapidly (Davies-Colley et al. 1992), creating a greater impact on the aquatic systems nearest the mining activities (frequently streams and rivers). However, fine inorganic particles (like clays) settle slowly and may therefore travel long distances from the point of their introduction. These fine suspended solids (SS) may then have a relatively greater impact on lakes which are often further from mining activities. Fine SS reduce light available for benthic algae and plants, thereby reducing biomass and primary production. Fine SS may reduce the quantity and quality of epilithon (stone surface biofilm) which serves as food for benthic invertebrates. Accumulation of SS may also damage respiratory structures of benthic invertebrates (Davies-Colley et al. 1992), thereby reducing their abundance.

Anthropogenic activities have increased global background levels of many toxic metals, and areas with reserves of metals can naturally have higher background levels than other areas (Allen-Gil et al. 1995). Nevertheless, effects of metals can be much more severe near mining and smelting operations (King 1995). Many of these metals can have severe environmental effects. Specific ecosystem responses to contaminants are dependent on the chemical, physical, biological, and geological processes at each site (Pascoe et al. 1993). Trace metal toxicity to aquatic organisms produces effects from reduction in growth, to reduced reproduction, to death, depending on concentration (Leland and Kuwabara 1985). Mollusks and fish generally survive higher concentrations of metals than do other phyla (Leland and Kuwabara 1985), but only at adult stages. Embryonic and larval stages are generally the most sensitive stages of the life cycle to heavy metals (Leland and Kuwabara 1985). Some metals in combination can more strongly inhibit primary production when present together than when present individually (Wong et al. 1978). Therefore, water quality criteria for single metals are insufficient for protecting aquatic life if several metals are present (Borgmann 1980). Dissolved metals and other minerals can travel through the same pathways as fine sediments.

Coeur D'Alene River Valley

Among the areas in the Columbia basin with proven reserves and high potential for mining, the Coeur d'Alene (CDA) mining district of northern Idaho - in the Northern Rocky Mountain province - is among the most developed (Bryant et al. 1990; Henny et al. 1991). This district is the nation's leading producer of silver, but like many other metals, this occurs in complex ores which also contain lead, zinc, copper, and antimony (Bryant et al. 1980). This district also supplies about 8% of the country's lead production (Eisler 1988). Mining and smelting activities have occurred at Kellogg-Smelterville on the Coeur d'Alene River since the last century (Henny et al. 1991). In that time, millions of tons of contaminated tailings have been washed into the South Fork of the Coeur d'Alene River and Lake Coeur d'Alene (Henny et al. 1991). Airborne smelter emissions averaged 8.3 - 11.7 mt of lead per month for an 18-year period (Henny et al. 1991). In the early 1970's, high levels of metal contamination were found throughout the northern two-thirds of the Coeur d'Alene river and lake system (Hornig et al. 1988), and levels of lead in sediments had not declined by the late 1980's (Henny et al. 1991; Hornig et al. 1988). The CDA system includes a series of lateral lakes along the CDA River which have become contaminated through sediment carried by winter floodwaters (Henny et al. 1991).

Of all the metals mined in the CDA valley, lead is one of most extensively studied for toxic effects. All measured effects of lead on living organisms are adverse (Eisler 1988). Lead does not biomagnify up the food chain: species at lower trophic levels are most vulnerable to lead poisoning (Henny et al 1991). While tundra swans (*Cygnus columbianus*) have continued to die from environmental lead in the CDA River Valley, ospreys (*Pandion haliaetus*) have not, though blood lead levels in the latter were high enough to show some physiological effect (Henny et al 1991).

Most lead entering natural waters is precipitated to the sediments by common anions, especially in lakes of high pH, but there it remains a source of contamination through uptake by aquatic macrophytes and benthic organisms (Eisler 1988). Species associated with the sediments are very vulnerable to lead poisoning. In Lake CDA, lead concentrations were higher in brown bullhead (*Ictalurus nebulosus*), a bottom feeder, than in other fish species higher up the food chain (Henny et al. 1991). Though lead does not biomagnify, it does bioaccumulate. It is toxic to all phyla of aquatic biota, though responses of individual species to lead contamination differ markedly (Eisler 1988). As with other metals, toxic effects are strongly correlated with dissolved concentrations (Besser and Rabeni 1987). Growth and survival of midge larvae (*Chironomus riparius*) and crawfish (*Orconectes nais*) have been shown to be negatively affected by leachates from lead mine tailings (which contained lead, zinc, cadmium, copper, and nickel). Adverse effects have been noted on Daphnid reproduction (1.0 ug Pb2+/liter), on rainbow trout (*Oncorhynchus mykiss*) survival (3.5 ug tetraethyllead/l), and on marine algal growth (5.1 ug Pb2+/l; Eisler 1988).

Owyhee Reservoir

Smallmouth bass (*Micropterus dolomieu*) in the Owyhee Reservoir in Oregon were evaluated for mercury concentration. Fish over four years old exceeded the U.S. Food and Drug Administration's mercury limit for human consumption (Allen-Gil et al. 1995). Though background levels of mercury may be relatively high due to natural sources in the area, it is thought that these high mercury levels could have been derived from the use of mercury in gold and silver extraction. Extensive gold and silver mining took place in the Jordan Creek region of the Owyhee basin between 1860 and 1920 (Allen-Gil et al. 1995). Unlike lead, mercury accumulates faster in species at higher trophic levels than in those at lower levels (Phillips et al. 1987). Though mercury can be held to sediments and particulate matter by sorption (Lepple 1973), it is also methylated by microbes in sediment and water (Allen-Gil et al. 1995). Methylmercury is one of the most toxic forms of mercury because of its "propensity for the nervous system, long retention time in the body, and effect on developing tissue" (Lepple 1973). Fish and shellfish can tolerate levels of mercury that are toxic to humans if eaten (Lepple 1973). Mercury exists if fish mainly as methylmercury (Lepple 1973). Methylmercury increases with age in fish (Lepple 1973; Allen-Gil et al. 1995). It has a more pronounced effect on algal growth than does copper, lead, or cadmium (Lepple 1973) perhaps because it inhibits synthesis of chlorophyll and lipids (Lepple 1973).

Yellowstone and Milltown Reservoirs

Commercial gold mines are located throughout the Rocky Mountain provinces (Bryant et al. 1980). A current controversy surrounds the location of a gold mine three miles from Yellowstone National Park (NY Times, Jan 15, 1995; O'Connell

1994). 125 years of gold mining activities on the Clark Fork River in Montana have left a legacy of high concentrations of arsenic, cadmium, copper, and zinc in sediments in Milltown Reservoir and surrounding wetlands. This reservoir, located 100 miles (240 km) downstream of the source of mining wastes, is a National Priority `Superfund' List site (Pascoe et al. 1993). In some gold-mining activities, mercury is used to separate gold from sediment resulting in potential mercury contamination of surface waters and land (Lacerda et al. 1989). The relatively new "heap leach" technique used in gold mining results in large quantities of cyanide and metal-contaminated water which can leak from holding reservoirs or be abandoned by a bankrupt mining company (NY Times, Aug 14, 1994). Even placer mining can have an impact by generating high inorganic sediment loads, even when there is no connection to surface waters in the watershed (Davies-Colley et al. 1992).

Examples of well-studied mining sites in the interior Columbia basin are rare. However, given the vast mineral reserves in the area (Bryant et al. 1980; Jackson and Kimerling 1993), the potential for impacts is great. Because of the presence of very valuable (and highly toxic) metals in ore deposits of the Rocky Mountain provinces, impacts of mining in this area could be greatest.

POTENTIAL IMPACTS OF DEVELOPMENT

The development of land for human habitation and associated economic and recreational activities has altered the natural environment throughout North America. This development began as early as the late-17th century in conjunction with homesteading and the agricultural activities of European immigrants. Brugam (1978), using sediment cores from Linsley Pond, Connecticut, determined that minor changes in lake biota occurred in conjunction with the beginning of farming in the area in 1700. Rapid eutrophication of the pond, beginning approximately 1915, was related to the increased development of agricultural activities and eventual suburban development of the area. Brugam and Vallarino (1989) used surface sediments and core bottom assemblages of fossil diatoms in determining the historical effects of human settlement near and around four western Washington lakes. Disturbances within watersheds included the construction of dwellings, deforestation and land clearing, the draining of bogs and wetlands, and, in some basins, urbanization. They found that these activities affected lakes by causing increases in concentrations of dissolved ions, changes in lake trophic status, and an increase in the abundance of the diatom, *Asterionella formosa*, an indicator of human disturbance in the Pacific Northwest. Spencer (1991) found that habitation of the Flathead Basin, Montana, and subsequent economic development (e.g., timber harvest and timber products production, road and railroad construction, etc.) near three lakes he examined increased rates of sediment deposition in the lakes.

The level and intensity of land development and land use activities have consistently increased into the present. The environmental disturbances of simple agriculturally-based settlements have evolved into perturbations associated with urban and suburban development, industrialization, large-scale intensive and mechanized agricultural practices, as well as areas developed solely for recreational use. Aquatic ecosystem perturbations related to these activities include: 1) thermal and oil pollution; 2) toxicity due to the presence of synthetic organic compounds and increases in heavy metal ions; 3) introduction of pathogenic organisms; 4) addition of biodegradable organic material resulting in potentially catastrophic changes in dissolved oxygen levels; 5) acidification; 6) increased sedimentation rates; and 7) enhanced eutrophication (Ellis 1989).

The eutrophication of a body of water is indicative of deteriorating water quality due to a buildup of nutrients, especially nitrogen and phosphorus. Increased levels of nutrient loading can be related to changes and/or disturbances within a watershed (Stauffer 1991; Dojlido and Best 1993), and according to Dojlido and Best (1993), the eutrophication of surface waters has increased rapidly due to increased levels of human land development activities and practices. Development activities that contribute to increased nutrient levels include point sources such as industrial effluents and water-borne sewage systems, and nonpoint sources such as agricultural operations, residential development and septic systems, road construction, forest practices, etc. (Ellis 1989; Tralles 1991; Dojlido and

Best (1993). The advanced eutrophication of Long Lake, Washington, due to a substantial increase in that impoundments nutrient load caused by significant inputs from the city of Spokane's primary sewage treatment plant is an example of point source pollution (Soltero et al. 1974). Brugam (1978) found that suburban housing development during the 1960's acted as a nonpoint source of increased mineral and organic inputs into Linsley Pond, Connecticut, causing hypereutrophication, major changes in the zooplankton community, and an increase in the abundance of *Asterionella formosa*.

Nonpoint source pollution of water seems to be the most problematic cause of deteriorating water quality because the origin of perturbation is often difficult to identify. According to the Montana Water Quality Bureau 1990 Report (Tralles 1991), nonpoint source pollution due to human land use is the major cause of deterioration of aquatic ecosystems in Montana, and contamination occurs in surface waters and groundwater. Power and Schepers (1989) determined that in North America nitrate pollution of groundwater originates, in part, from septic tanks, fertilizers, and improper use of animal manures. Gold et al. (1990) discovered that shallow domestic wells in southern Rhode Island were contaminated by effluents from septic tanks, and that cropland activities, home lawn fertilizing, unsewered residential areas, and animal waste disposal areas were sources of nitrate-nitrogen contamination of groundwater. The water supply of a rural Texas residential area was also found to be contaminated by septic tank effluent leeching into groundwater (Hayes et al. 1990). Yates and Yates (1988) state that septic tanks contribute substantial amounts (i.e., > one trillion gallons) of waste to subsurface water annually. They identify this waste as a major source of water-borne disease, causing as much as 50% of all viral outbreaks in the United States. Since the 1950's, nonpoint sources associated with recreational, housing, and commercial development within Lake Tahoe tributary watersheds have been implicated as contributing to increasing levels of nutrients and eutrophication in the lake (Leonard et al. 1979).

Rural-residential and suburban development around or near lakes, reservoirs, and wetlands have been shown to have negative impacts on these lentic systems. These developments are often initiated by or are a response to the aquatic recreational potential of the area. One of the primary problems associated with such developments is the increase of nitrogen and phosphorus in surface waters due to the leeching of effluents from septic systems (Lewis et al. 1984; Ehrenfeld and Schneider 1991; Scott 1991; Stauffer 1991; Sorrie 1994), the over use of fertilizers (Lewis et al. 1984), and runoff from highways and roads (Lewis et al. 1984; Ehrenfeld and Schneider 1991). Developments affect riparian areas by changing the vegetation (Lewis et al. 1984), altering plant community composition and structure (Ehrenfeld and Schneider 1991), removing or eradicating vegetation (Ehrenfeld and Schneider 1991; Scott 1991; Sorrie 1994), and changing riparian soils through disturbance and loss of topsoil (Lewis et al. 1984). Shoreline encroachment and the building of structures (e.g., boat docks) over submerged areas also can eradicate, damage, and fragment shoreline and submerged nearshore habitat (Scott 1991; Sorrie 1994).

The potential for water quality deterioration also exists in isolated, minimally populated areas where land development has been restricted to resorts, vacation homes/cabins, and campgrounds. According to the Odell Pilot Watershed Analysis (1994), the decline of water quality in Odell Lake, Oregon is due , in part, to inputs of nutrients from septic systems associated with a resort, nearby campgrounds, and summer homes built along the lake shoreline. Improvements to these systems have stabilized nutrient levels in the lake, although they remain above historic levels. Septic systems associated with three vacation home sites near a coastal lagoon in Rhode Island were found to be responsible for bacteria levels exceeding acceptable groundwater standards and nitrate-nitrogen levels above standards for drinking water (Postma et al. 1992). These conditions were caused by intermittent use of the systems creating uneven effluent loading in selected portions of the absorption field. Brugam (1987) and Brugam and Vallarino (1989) found that the moderate eutrophication of Pine and Otter Lakes in western Washington appeared to coincide with the construction of recreational cabins and suburban homes around the lakes.

Stauffer (1991) states that processes and activities within the catchment of a lake influence the lake's abiotic and biotic composition, and that many natural, as well as anthropogenic disturbances contribute to lake condition. Activities and practices associated with residential and commercial land development have been shown to have a widespread impact on lentic ecosystems and the level of disturbance appears related to the degree and intensity of development. Many disturbances are associated with nonpoint sources of pollution and contribute to the decline of water quality and ecosystem deterioration in populated, as well as in isolated areas of seasonal and limited use. Sorrie (1994), in examining the once pristine coastal plain ponds of New England, has demonstrated that lentic ecosystems can be populated by unique assemblages of plants and animals. He also reports that due to land development and recreation activities between 1970-1990, associated perturbations have left few pristine ponds and threaten the integrity and continued existence of many unique biological assemblages. The potential general consequences to lentic ecosystems of residential/suburban, commercial, and recreational development include the deterioration of natural baseline environmental conditions, alteration of the dynamic relationships of plants and animals within lentic systems, replacement of natural communities of organisms with anthropogenic-influenced and tolerant communities, and the deterioration of water quality. Therefore, the development of land around or near lentic ecosystems, if carried out at all, needs to be performed with an appreciation for and understanding of the potential perturbations and changes to natural systems that could result from this type of activity.

POTENTIAL IMPACTS OF RECREATION

Recreation relating to lakes and their watersheds ranges from wilderness hiking and fishing to intensive power boat use. Road building, in order to improve access to lakes, residential development for seasonal housing, and the inclusion of septic systems in the watershed all have impacts on lakes and watersheds which are discussed in another section.

Mountain lakes, especially those in national parks and scenic forested areas, may be among the most susceptible to impacts from recreation. Susceptibility of lakes to water-quality degradation due to recreational use depends on: pollutant-loading likelihood; and lake sensitivity (Gilliom et al. 1980). In mountain lakes, sensitivity depends on lake type. Large, deep lakes with a large inflow may be least susceptible to water quality degradation in these lakes. Pollutants would be diluted by large volumes of water and could settle along with particulate matter (Gilliom et al. 1980). Lakes that are small and shallow, or that have a small inflow, are likely be more sensitive to pollutants (Gilliom et al. 1980). Gilliom et al. (1980) describe wilderness mountain lakes as having a high pollutant-loading likelihood if soil, geologic, or hydrologic characteristics of its watershed favor the transport of pollutants to the lake. For mountain lakes not in wilderness areas, we might also include cultural characteristics of the lake's watershed as influencing its pollutant-loading likelihood.

Wilderness Recreation and Foot Traffic

Pollutant loading likelihood of wilderness is greatest for lakes on steep, soil-covered slopes, on exposed bedrock, or where a large portion of the shoreline is wet area. Appropriate siting of privies, campsites, and trails can minimize impacts to lakes with a high pollutant-loading likelihood (Gilliom et al. 1980). Therefore, in areas subject only to wilderness recreation, the most serious pollutants are likely to be pathogens such as *Giardia* (Gilliom et al. 1980; Suk et al. 1987).

For lakes in heavily used areas in national parks and scenic areas, heavy foot traffic, or trampling, can have an effect on vegetation both through direct mechanical action, and indirectly through soil changes (Liddle 1975). Resistance to trampling depends on plant life form, with large and broadleaved plants suffering most, and grasses generally being most resistant (Burden and Randerson 1972). Tolerance of plant communities to trampling also depends on trampling intensity, nature of substrate, and environmental conditions (Bowles and Maun 1982; Harrison 1980). Loss of vegetation from shorelines, wetlands, or steep slopes can cause erosion and pollution problems (Burden and Randerson 1972; Gilliom et al. 1980). Alteration of vegetation due to trampling effects may be undesirable, especially in national parks. However, these problems can be averted or minimized with proper management practices (Burden and Randerson

1972; Gilliom et al. 1980). Construction of access roads and visitor facilities with septic systems are also likely to have effects (discussed in another section).

Sportfishing

Anglers can substantially reduce densities of desirable fish species (Willis et al. 1994; Schram et al. 1990), especially in small water bodies (Willis et al. 1994; Mosindy et al. 1987). However, the greatest threat from sportfishing to rare native fishes is likely to be the introduction of non-native fishes (US Fish and Wildlife Service 1987). Effects of these introductions are discussed in another section.

Power Boating and Off-Road Vehicle Use

Power boats can have a range of impacts on lakes. Resuspension of bed sediments can occur with passage of a single moving boat where the concentration of sediment resuspended is related to boat speed and hull shape (Garrad and Hey 1987). Patterns of suspended sediment concentrations have been found to correlate with frequency of boat movements. Concomitant high levels of turbidity and reduced light penetration may be a major factor in declining populations of submerged macrophytes. Loss of macrophytes may, in turn, make shorelines more susceptible to erosion from boat wash (Garrad and Hey 1987). Power boats can also aid in the spread of the exotic Eurasian watermilfoil (*Myriophyllum spicatum*). Because it reproduces from fragmented stems as well as seeds and rhizomes, it is easily transported between water bodies when pieces of it become entangled on boat propellers and gear (Reed 1977).

Outboard engines introduce hydrocarbon emissions to the aquatic environment. Though several hundred kinds of bacteria and lower fungi can degrade some hydrocarbons (Wachs et al. 1992), at least in part, persistence and rate of degradation depend on their particular isomeric structure (Bayona et al. 1986). Motor boat emissions also have a high phenol content which is quite toxic to aquatic organisms (Wachs et al. 1992). Increased lead levels in reservoirs may be attributed to recreational boating and gasoline spills (Cairns and Palmer 1993). Boats, especially larger ones, may also introduce tributyltin (TBT) to the aquatic environment. TBT is a biocide agent used in anti-fouling paints (Beeker et al. 1992), and is one of the most toxic compounds to aquatic organisms ever introduced deliberately to water (Maguire 1987). It has adverse effects on freshwater mollusks at concentrations as low as 0.1 ug per liter (Hall and Pinkney 1985).

Impacts of off-road recreational vehicle (ORV) use are documented only for a few types of natural systems. On sand dunes and shorelines, off-road vehicles can result in significant reductions of vegetation (Wisheu and Keddy 1991; Anders and Leatherman 1987). Impacts in dune areas can occur even at the rate of one vehicle pass per week (Anders and Leatherman 1987). ORV disturbance can alter plant community composition, or create openings in cover vegetation, on shorelines (Wisheu and Keddy

1991). Partial loss of vegetation from shorelines can cause further erosion that continues until those shorelines are devoid of vegetation (Wisheu and Keddy 1991). Seed `banks,' or buried seed, in shoreline wetlands may be particularly sensitive to intense disturbance since they tend not to be deeply buried (Wisheu and Keddy 1991), so recovery of disturbed shorelines may be very slow.

Because of their combined potential effects, motorized vehicles (boats and off-road vehicles) may have significant effects on scenic mountain lakes.

EXOTIC AND NON-NATIVE FISH INTRODUCTIONS

The introduction of exotic and non-native fish species into aquatic ecosystems of the United States and Canada can be traced back to the 1680's (Courtenay et al. 1986). Introductions whether intentional or accidental have contributed to increased rates of dispersal of exotic and non-native species (Taylor et al. 1984; Townsend and Winterbourn 1992). As of 1989, 45 species of exotic fish have been identified as maintaining established (i.e., breeding) populations in North America and the status of 56+ species is presently unknown (Taylor et al. 1984; Courtenay et al. 1986; Crossman 1991).

Although many introduced species have been successful in adapting to aquatic systems in new locations, the introduction of new species, in general, has caused numerous problems (Li and Moyle 1981). Successful species have broad physiological tolerances and opportunistic habitat and resource requirements, as well as reproductive traits typically consisting of extended or continuous breeding with short generation times (Taylor et al. 1984). Five broad categories of negative impacts associated with the introduction of exotic and non-native species include: 1) habitat alterations; 2) change in community trophic organization and structure; 3) changes in spatial relationships leading to competitive interactions and displacement of native species; 4) hybridization culminating in gene pool deterioration; and 5) introduction of parasites and disease (Taylor et al. 1984; Crossman 1991). According to Li and Moyle (1981), the most sensitive systems appear to be nutrient-poor (e.g., oligotrophic lakes) and introductions destabilize these systems by disrupting communities that have evolved the capacity to function under low nutrient conditions. Miller et al. (1989) has documented the extinction of 27 North American fish species. Habitat alteration or loss has been identified as contributing to 73% of these extinctions while the introduction of exotic and non-native fish species has been cited as contributing to 68% of the losses.

The Effects of Exotic and Non-Native Fish on Native Fish

Changes to aquatic communities due to the introduction of exotic and non-native fish species can occur as a result of predation, interspecific competition, hybridization, and changes in trophic organization and structure. Yet, according to Fausch (1988), negative effects associated with introductions are not usually explained by simple direct relationships, especially when attempting to examine changes due to competition. Disequilibrium within a system may have less to do with interspecific interactions than with factors such as habitat alteration or loss (Miller et al. 1989), changes in water temperature (Nelson 1965), or different species requirements for physical conditions such as water temperature and stream gradient (Larson and Moore 1985).

Numerous investigators have examined the effects of predation by introduced fish species on native fish. Marrin and Erman (1982) found that brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*) introduced into a California Reservoir preyed upon native tui chub (*Gila bicolor*) and Tahoe suckers (*Catostomus tahoensis*). Introduced channel catfish (*Ictalurus punctatus*) and flathead catfish (*Pylodictis olivaris*) were found to prey on and limit the survival of native juvenile razorback suckers (*Xyrauchen texanus*) in Gila River, Arizona (Marsh and Brooks 1989), and Townsend and Winterbourn (1992) expressed their concerns about the effects of predation by introduced channel catfish on native New Zealand fish species. Also in New Zealand, McDowall (1990) determined that predation by introduced brown trout, rainbow trout, and chinook salmon (*Oncorhynchus tshawytscha*) was harmful to several native galaxiid species. In the Pacific Northwest, introduced smallmouth bass (*Micropterus dolomieu*) have been shown to prey upon migrating salmon smolts (Pflug and Pauley 1984; Bennett et al. 1991) and stocked northern pike (*Esox lucius*) and walleye (*Stizostedion *vitreum*) eat native trout (Wydoski and Bennett 1981). Finally, Walters and Legner (1980) cite several investigators who have found that *Gambusia affinis*, usually introduced for the purpose of controlling mosquitos in lentic systems, can be detrimental because of their tendency to also prey on juvenile native mosquito-eating fishes.

The potential for competition between native and introduced species appears high, although Fausch (1988) also concluded that many experiments do not well elucidate the important mechanisms associated with competition. Nelson (1965) discovered that the introduction of brown trout in reservoirs of the Kananaskis River System, Alberta, displaced Dolly Varden (*Salvelinus malma*), cutthroat trout (*Oncorhynchus clarki*), brook trout (*Salvelinus fontinalis*), and rainbow trout. He also determined that Dolly Varden were negatively impacted by competition with the introduced white sucker (*Catostomus commersoni*). According to Larson and Moore (1985) the distribution of native brook trout in streams of the Great Smoky Mountains National Park declined concurrent with the introduction and expansion of rainbow trout distributions. Their data was consistent with the hypothesis that brook trout were excluded from preferred habitat due to competition with rainbow trout. Moore et al. (1986), following up on this research, found that increased brook trout densities coincided with decreases in the densities of rainbow trout.

Krueger and May (1991), examining the history of salmonid introductions in North America, found evidence suggesting that introduced salmonids may often compete with native fish faunas for limited resources which can lead to competitive interference and eventual exclusion. The ultimate negative impact of this type of competition is extinction of one or more species. For instance, Rode (1988) found that large scale stocking of brook trout in McCloud Reservoir contributed to the elimination of the California bull trout (*Salvelinus confluentus*), and that brown trout and bull trout competed for resources in the lower reaches of the McCloud River. Donald and Alger (1993) determined that in certain low-elevation lakes located in northern Montana, southwestern Alberta, and east-central British Columbia, competition for resources between lake trout (*Salvelinus namaycush*) and bull trout culminates in the displacement of bull trout. According to McDowall (1990), competitive exclusion of native New Zealand galaxiids by introduced salmonids can be chronic, as well as catastrophic.

Shifts in trophic relationships due to introductions of exotic and non-native fish species can indirectly impact native species. Alewife (*Alosa pseudoharengus*) and rainbow smelt (*Osmerus mordax*) introduced into Lake Michigan eventually became the dominant planktivores, and through competition reduced or exterminated populations of native planktivorous species (Stewart et al. 1981). Alewife and rainbow smelt also became the dominant prey for native predators (e.g., lake trout), and this shift in prey, as well as fluctuations in their abundance caused destabilization of the existing predator-prey system. Matuszek et al. (1990) reported that the introduction of cisco (*Coregonus artedii*) into Lake Opeongo, Ontario, as prey for lake trout was initially positive since the cisco contributed to increased lake trout growth rates. Yet, as the growth rates increased so to did lake trout abundance which subsequently led to a decrease in the size of the cisco and a concomitant decrease in piscivorous lake trout growth rates. Growth rates of non-piscivorous lake trout were also negatively affected since the availability of their insect prey base was diminished due to the initial increase in lake trout abundance. Finally, Wydoski and Bennett (1981) reported that the introduction of northern pike into Carbody Reservoir, Colorado, contributed to a decrease in growth rates of surviving white crappie (*Pomoxis annularis*) due, perhaps, to concomitant changes in other parts of the food chain culminating in a diminished prey base.

Hybridization of native species with introduced species is also a potential impact of introductions. Although considered rare in open waters (Taylor et al. 1984) hybridization does occur and can cause deterioration of native faunal gene pools. Nelson (1965) found that the decrease in the cutthroat trout population of the Kananaskis River System was due, in part, to hybridization with introduced rainbow trout. Krueger and May (1991) identified infertile interspecific hybrids as a negative outcome associated with the interbreeding of native species with introduced salmonids.

Hybridization between native bull trout and introduced brook trout has been categorized as extensive (Cavender 1978; Leary et al. 1983), and brown trout have been identified as interbreeding with native salmonids throughout North America (Kohler and Courtenay 1986). Thirty-eight percent of the fish extinctions in North America are due, in part, to hybridization (Miller et al. 1989).

According to Li and Moyle (1981), part of the difficulty associated with introductions of new species has been our limited ability to predict potential impacts. Exotic and non-native species are usually introduced into systems with which they didnot evolve and which do not contain natural controls for them. Therefore, when they escape from locations of introduction they have the potential of rapidly establishing and expanding new populations (Wydoski and Bennett 1981; Townsend and Winterbourn 1992), as well as directly and indirectly altering entire ecosystems. Eventually it becomes quite difficult for systems to provide for the requirements of introduced and native species (Wydoski and Bennett 1981), often to the detriment of indigenous fauna.

The Effects of Exotic and Non-Native Fish on Amphibians and Salamanders

The introduction of various exotic species into non-indigenous locations has been identified as one of several potential factors causing declines in amphibian populations (Blaustein et al. 1994), and Blaustein and Wake (1995) have suggested that fish introductions into aquatic systems may be harmful to amphibians. Fish are known to prey upon eggs, tadpoles and larvae, as well as adult amphibians. They may also block migration between amphibian communities and thereby complicate the reconstitution of endangered populations (Blaustein and Wake 1995).

In their review of ranid frog species declines in western North America, Hayes and Jennings (1986) surmised that fish introductions and the access of fish to early ranid life stages supports the hypothesis that fish predation has contributed to declines in anuran populations. Heyer et al. (1975) identified fish as a dominant aquatic predator capable of extirpating anurans from lentic systems. The introduction of salmonids into streams and lakes of the Sequoia and Kings Canyon National Parks, California, led to the elimination of the native mountain yellow-legged frog (*Rana muscosa*) from all aquatic systems with fish within the boundaries of the parks (Bradford 1989; Bradford 1991; Bradford et al. 1993). It has also been speculated that fragmentation of *Rana muscosa* populations due, in part, to fish predation has contributed to the disappearance of *Rana muscosa* from sites without fish, as well as extinction of this species in some drainage systems within the parks (Bradford et al. 1993).

Fish introductions into aquatic systems have also been implicated as negatively affecting salamander populations. Olenick and Gee (1981) found that rainbow trout, stocked in Manitoba potholes, and native tiger salamanders (*Ambystoma tigrinum*) had considerable overlap in diet and were only partially segregated spatially and temporally. They speculated that these conditions could lead to some level of interspecific competition. The presence of bluegill (*Lepomis macrochirus*) in experimental ponds significantly decreased the growth and activity of larval *Ambystoma maculatum*, and the reduction of larval body size was attributed to a combination of predator avoidance and exploitative competition (Figiel and Semlitsch 1990). Several investigators (e.g., Nussbaum et al. 1983; Semlitsch 1988; Sih et al. 1988) have reported that fish will consume embryos and larvae of several species of ambystomatid salamanders. Research conducted in lakes of the North Cascades National Park Service Complex, Washington, indicates that densities of larval *Ambystoma macrodactylum* are significantly lower in lakes containing introduced trout (Liss et al. 1995). In a lake from which trout were removed (ca. 1990), and containing a remnant population of *Ambystoma gracile*, densities of salamander embryo masses and larvae increased considerably from 1990 to 1993: twice as many embryo masses and five times as many larvae were observed since fish removal (Liss et al. 1995).

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FLATHEAD LAKE

AN EXAMPLE

Flathead Lake

Flathead Lake is one of the largest and most pristine natural lakes in the western US and is valued for the clarity and purity of its water (Spencer et al. 1991; Stanford et al. 1994, 1995). Water quality is outstanding because nearly 70% of the lake's watershed occurs in Glacier National Park and surrounding wilderness areas. Nevertheless, human activities continue to threaten the quality of the lake's waters.

Cultural eutrophication

Both point and non-point sources of pollution have affected water quality in Flathead Lake. The increase in primary production of algae from 1977 to 1989, a possible indicator of deteriorating water quality, was associated with high levels of phosphorous and nitrogen loading from sewage treatment plants located on tributaries to the lake (Stanford et al. 1994, 1995). Water clarity, as measured by secchi depth, was negatively related to nutrient loading (Stanford et al. 1995). A program to reduce phosphorous input to the lake was instituted by the Montana Water Quality Department. The program included technological improvements in sewage treatment plants to remove phosphorous from effluent and a ban on the sale of phosphorous-containing detergents. This program has resulted in a decline in the contribution of sewage treatment plants from near 20% to less than 5% of the total phosphorous load delivered to Flathead Lake. The decline in nutrients has been accompanied by a decline in primary productivity (Stanford et al. 1994, 1995).

Although nutrient inputs from sewage treatment plants have been reduced, investigators have documented high concentrations of phosphorous and nitrogen stored in the sediments of a tributary stream of Flathead Lake receiving effluent from a sewage treatment facility. They suggest that these nutrient can be released under anoxic condition and so may partially offset reductions in nutrient discharge achieved by technological improvements in the treatment plants (Stanford et al. 1994).

Although input of nutrients from sources along the lake shoreline such as domestic sewage drain fields probably do not contribute extensively to the total nutrient load of the lake (Stanford et al. 1994), localized shoreline sources can increase algal production along the lake margin, creating a "ring around the lake" (Stanford et al. 1994, 1995).

There is concern about increased nutrient loading from non-point sources. In part this increase may be related to population expansion which is occurring in the Flathead watershed at a rate of 2% per year. Input from bulk precipitation is also a significant source of nutrients, accounting for 10-38% of the total phosphorous load over the time period of record (Stanford et al. 1995). Smoke and dust in the atmosphere are thought to be responsible for the high levels of phosphorous in precipitation (Stanford et al. 1995).

Exotic Species

At least seventeen species of fish and invertebrates have been introduced into Flathead Lake (Spencer et al. 1991). The appearance of the opossum shrimp, Mysis relicta, apparently triggered major shifts in the food web of the Flathead Lake ecosystem. In addition to Mysis, a complex of introduced species including kokanee salmon (Oncorhychus nerka), lake trout (Salvelinus namaycush), and lake whitefish (Coregonus clupeaformis) may have been involved in the food web changes. Mysis prey upon zooplankton. Following the appearance of the shrimp in Flathead Lake, cladoceran zooplankton and a diaptomid copepod decreased in abundance (Spencer et al. 1991; Beattie and Clancey 1991). Kokanee salmon, a valuable sport fish, also declined dramatically as Mysis abundance peaked. Kokanee may feed extensively on zooplankton, especially cladocerans, (Morgan et al. 1978; Reiman and Bowler 1980; Beattie and Clancy 1991). Mysis are thought to have contributed to the decline in salmon through reduction in the salmon's food resources. Since Mysis and kokanee are able to coexist in some lakes (Beattie and Clancy 1991), the decline in kokanee in Flathead Lake may have been related not only to interspecific competition with Mysis but also to the presence of kokanee predators such as lake trout (Spencer et al. 1991; Bowler et al. 1991) and other planktivorous competitors such as juvenile lake whitefish (Beattie and Clancy 1991). The collapse of the kokanee population in Flathead Lake following Mysis introduction reduced the number migrating bald eagles and other wildlife that congregated to feed on kokanee during their spawning migration (Spencer et al. 1991).

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