

Appendix A – Causes for Decline of Bull Trout in the Western United States

Bull trout distribution, abundance, and habitat quality have declined range-wide (Bond 1992; McPhail and Baxter 1996; Newton and Pribyl 1994; Rieman and McIntyre 1993; Schill 1992; Thomas 1992; Ziller 1992). Several local extirpations have been documented, beginning in the 1950s (Berg and Priest 1995; Buchanan et al. 1997; Donald and Alger 1993; Goetz 1994; Light et al. 1996; Newton and Pribyl 1994; Ratliff and Howell 1992; Rode 1990; WDFW 1998). Bull trout were extirpated from the southernmost portion of their historic range, the McCloud River in California, around 1975 (Moyle 1976; Rode 1990). Bull trout have been functionally extirpated (i.e., few individuals may occur there but do not constitute a viable population) in the Coeur d'Alene River Basin in Idaho and in the Lake Chelan and Okanogan River basins in Washington (USFWS 1998b). These declines resulted from the combined effects of habitat degradation and fragmentation, blockage of migratory corridors, degradation of water quality, angler harvest and poaching, entrainment into diversion channels and dams, and introduced nonnative species. Specific land and water management activities that have depressed bull trout populations and degraded habitat include dams and other diversion structures, forest management practices, livestock grazing, agriculture, agricultural diversions, road construction and maintenance, mining, and urban and rural development (Beschta et al. 1987; Chamberlain et al. 1991; Craig and Wissmar 1993; Frissell 1993; Furniss et al. 1991; Henjum et al. 1994; Light et al. 1996; MBTSG 1995a, b, c, d, e, 1996b, c, d, e, f, h; McIntosh et al. 1994; Meehan 1991; Nehlsen et al. 1991; Sedell and Everest 1991; USDA and USDI 1995, 1996, 1997; Wissmar et al. 1994).

Dams

Dams affect bull trout by altering habitat; flow, sediment, and temperature regimes; migration corridors; and creating additional interspecific interactions, mainly between bull trout and nonnative species (Bodurtha 1995; Craig and Wissmar 1993; Rieman and McIntyre 1993; Rode 1990; USDA and USDI 1996, 1997; WDW 1992; Wissmar et al. 1994). Impassable dams have caused declines of bull trout by preventing migratory fish from reaching spawning and rearing areas in headwaters and recolonizing areas where bull trout have been extirpated (MBTSG 1998; Rieman and McIntyre 1993).

The extirpation of bull trout in the McCloud River Basin, California, has been attributed primarily to construction and operation of McCloud Dam, which began operation in 1965 (Rode 1990). The McCloud Dam flooded bull trout spawning, rearing, and migratory habitats. The dam also resulted in elevated water temperatures.

Although dams negatively affect bull trout (Gilpin 1997; Rieman and McIntyre 1993), some dams can benefit bull trout by blocking introduced nonnative species from upstream areas (MBTSG 1995d). Some dams also increase the potential forage base for bull trout by creating reservoirs that support prey species (Faler and Bair 1991; Pratt 1992).

Some of the major effects to bull trout resulting from the Federal Columbia River Power System and from operation of other hydropower, flood control, and irrigation diversion facilities include the following: (1) fish passage barriers, (2) entrainment of fish into turbine intakes and irrigation canals, (3) inundation of fish spawning and rearing habitat, (4) modification of streamflows and water temperature regimes, (5) dewatering of shallow water zones during power peaking operations, (6) reduced productivity in reservoirs, (7) periodic gas super-saturation of waters downstream of dams, (8) water level fluctuations interfering with retention of riparian vegetation along reaches affected by power peaking operations, (9) establishment of nonnative riparian vegetation along reaches affected by power peaking operations, and (10) severe reductions in reservoir levels to accommodate flood control operations.

Hungry Horse, Libby, Albeni Falls, Dworshak, Chief Joseph, Keechelus, Tieton, and Grand Coulee dams, as well as others in the Columbia River Basin and throughout the range of bull trout in the conterminous United States, were built without fish passage facilities and are barriers to bull trout migration. These barriers have contributed to the isolation of local populations of migratory bull trout. Bull trout have been observed using upstream fish passage facilities at many of the hydropower projects on the Snake and Columbia rivers. However, even dams with fish passage facilities may be a factor in isolating bull trout local populations if they are not readily passable by bull trout or if they do not provide an adult downstream migration route.

Entrainment of bull trout may also occur at various projects in the Columbia River Basin including Libby, Hungry Horse, Albeni Falls, Rocky Reach, Rock Island, Wells, Dworshak, Bonneville, Ice Harbor, Lower Monumental, Little Goose, and Lower Granite dams. Fish can be killed or injured when passing the dams. Potential passage routes include spill through the turbines or the juvenile bypass systems, but the relative passage success of these routes for adult salmonids has not been thoroughly investigated. However, one study of fish facilities at Foster and Green Peter dams on the South Santiam River, Oregon, conducted in the early 1970s revealed that passage through turbines resulted in a 22 to 41 percent mortality rate for adult steelhead (Wagner 1973). Additionally, a 40 to 50 percent injury rate for adult salmonids passing through the juvenile fish bypass system at McNary Dam has been noted (Wagner and Hinson 1993; Wagner 1991). Adult bull trout may experience similar mortality rates. In addition, those adult fish that survive downstream passage at dams without upstream passage facilities are isolated in downstream reaches away from their natal (native) streams. As indicated above, the loss of these larger, more fecund migratory fish is detrimental to their natal populations.

The creation of mainstem Columbia and Snake river reservoirs (i.e., the areas of slow moving water behind the dams) combined with introductions of piscivorous species [e.g., bass (*Micropterus spp*), walleye (*Stizostedion vitreum*)] have also affected the habitat of bull trout and other salmonids. An increase in predator populations, both native [e.g., northern pikeminnow (*Ptychocheilus oregonensis*)] and nonnative, as a result of creating artificial habitat and concentrating prey may be a factor in the decline of listed Snake River salmon species (NMFS 1991a, b, c). Ideal predator foraging environments have been created in these pools, particularly for warmwater species in the summer. Smolts that pass through the projects are subjected to turbines, bypasses, and spillways that may result in disorientation and increased stress, conditions that reduce their ability to avoid predators below the dams. Creation of the pools above the dams has resulted in low water velocities that increase smolt travel time and increase predation opportunity. Increased water temperatures, also a result of the impoundment of the river, have also been shown to increase predation rates on salmonid smolts (Viggs and Burley 1991). Because bull trout are apex (top) predators of other fish, negative effects to the salmonid smolt prey base, and the resulting decline in adult returns, are likely to affect bull trout negatively as well. Additionally, increased water temperatures, influenced by the presence of dams, also decreases the suitability of the lower Snake and Columbia river pools for bull trout in the late spring through early fall.

Uncontrolled spill over dams, or even high levels of managed spill, at hydropower projects can produce extremely high levels of total dissolved gas that may impact bull trout and other species. These high levels of gas super-saturation can cause gas bubble disease trauma in fish. Gas bubble disease is caused by gas being absorbed into the bloodstream of fish during respiration. Effects can range from temporary debilitation to mortality, and super-saturation can persist for several miles below dams where spill occurs. The states of Oregon and Washington have established a 111 percent total dissolved gas level as State water quality standards. However, total dissolved gas levels of up to 120 percent have been experienced during recent years of managed spill in the Federal Columbia River Power System, with involuntary spill episodes resulting in total dissolved gas levels of as high as 140 percent at some sites (NMFS 2000). At levels near 140 percent, gas bubble disease may occur in over three percent of fish exposed. At levels of up to 120 percent the incidence of gas bubble disease decreases to a maximum of 0.7 percent of fish exposed (NMFS 2000).

Manipulated flow releases from storage projects alter the natural flow regime, affect water temperature, have the potential to destabilize downstream streambanks, alter the natural sediment and nutrient loads, and cause repeated and prolonged changes to the downstream wetted perimeter (MBTSG 1998). Power peaking operations, which change the downstream flow of the river on a frequent basis, cause large areas of the river margins to become alternately wet and then dry, adversely affecting aquatic insect survival and production. Changes in water depth and velocity as a result of rapid flow fluctuations, and physical loss or gain of wetted habitat, can cause juvenile trout to be displaced, thus increasing their vulnerability to predation. Additionally, rapid flow reductions can strand young fish if they are unable to escape over and through draining or dewatered substrate. These effects also indirectly adversely affect bull trout by degrading the habitat of their prey (small fish) and the food upon which they depend (aquatic insects).

Reservoirs created by dams have also inundated bull trout habitat. For example, reservoirs created by the construction of Libby and Hungry Horse dams have inundated miles of mainstem river and tributary habitat previously used by many local populations of bull trout. Reservoir water level manipulations can create migration barriers at the confluence of tributaries entering the reservoir, as well as negatively affecting littoral rearing habitats for prey species of bull trout. Reservoir levels are often drawn down substantially during drought years, or annually as operators evacuate flood control reservoirs to make room for spring snow melt runoff. Reduced volumes of water in reservoirs can affect their overall productivity that may ultimately reduce the food base of predators such as bull trout. Other reservoirs are unproductive and provide poor habitat for bull trout compared to natural riverine habitats (e.g., Noxon and Cabinet Gorge). However, reservoirs such as Libby, Hungry Horse, and Dworshak now provide suitable habitat for adfluvial populations of bull trout that was not available prior to dam construction.

Forest Management Practices

Forest management activities, including timber extraction and road construction, affect stream habitats by altering recruitment of large wood, erosion and sedimentation rates, runoff patterns, the magnitude of peak and low flows, water temperature, and annual water yield (Cacek 1989; Furniss et al. 1991; Spence et al. 1996; Spencer and Schelske 1998; Swanson et al. 1998; Wissmar et al. 1994). Activities that promote excessive substrate movement reduce bull trout production by increasing egg and juvenile mortality, and reducing or eliminating habitat (e.g., pools filled with substrate) important to later life-history stages (Brown 1992a; Fraley and Shepard 1989). The length and timing of bull trout egg incubation and juvenile development (typically more than 200 days during winter and spring) and the strong association of juvenile fish with stream substrate make bull trout vulnerable to changes in peak flow alterations or disturbances to channels and substrates (Goetz 1989; MBTSG 1998; McPhail and Baxter 1996; Pratt 1992).

Roads constructed for forest management are a prevalent feature on managed forested and rangeland landscapes. Roads have the potential to adversely affect several habitat features, (e.g., water temperature, substrate composition and stability, sediment delivery, habitat complexity, and connectivity) (Baxter et al. 1999; Trombulak and Frissell 2000). Roads may also isolate streams from riparian areas, causing a loss in floodplain and riparian function. The aquatic assessment portion of the Interior Columbia Basin Ecosystem Management Project provided a detailed analysis of the relationship between road densities and bull trout status and distribution (Quigley and Arbelbide 1997). The assessment found that bull trout are less likely to use streams in highly roaded areas for spawning and rearing, and do not typically occur where average road densities exceed 1.1 kilometers per square kilometer (1.7 miles per square mile).

Although bull trout occur in watersheds where timber has been harvested, bull trout strongholds primarily occur in watersheds with little or no past timber harvest, such as the wilderness areas of central Idaho and the South Fork Flathead River drainage in Montana (Henjum et al. 1994; MBTSG 1995d; Rieman et al. 1997; USDA and USDI 1997). However, the Swan River Basin, Montana, has had extensive timber harvest and road construction, and is a bull trout stronghold (Watson and Hillman 1997). The overall effects of forestry practices on bull trout in parts of the Swan River Basin are difficult to assess because of the complex geomorphology and geology of the drainage (MBTSG 1996f).

Roads may affect aquatic habitats considerable distances away. For example, increases in sedimentation, debris flows, and peak flows affect streams longitudinally so that the area occupied by a road can be small compared to the entire downstream area subjected to its effects (Jones et al. 2000; Trombulak and Frissell 2000). Upstream from road crossings, large areas of suitable habitat may become inaccessible to bull trout due to fish passage barriers (e.g., culverts).

Forest management activities have also altered the frequency and duration of floods or high flows (USDA and USDI 1997). Roads and clear-cutting of forested areas tend to magnify the effects of floods, leading to higher flows, erosion, and bedload that scour channels (McIntosh et al. 1994; Spencer and Schelske 1998; Swanson et al. 1998; USDA and USDI 1997), and degrade bull trout habitat (Henjum et al. 1994). Erosion from road landslides increases bedload to stream flows (Furniss et al. 1991). Increased bedload increases the scouring effect of high stream flows, increasing channel width and instability and loss of habitat diversity, especially pools (Henjum et al. 1994; McIntosh et al. 1994). Bull trout eggs and fry in the gravels during scouring likely survive at low rates (Henjum et al. 1994), as do those with large sediment loading. For instance, hundreds of landslides associated with roads on the Clearwater and Panhandle national forests resulted from high flow events in 1995 (Patten and Penzkover 1996), likely reducing survival of bull trout eggs and fry. Habitat degradation has also reduced the number and size of bull trout spawning areas (USDA and USDI 1997).

Livestock Grazing

Improperly managed livestock grazing degrades bull trout habitat by removing riparian vegetation, destabilizing streambanks, widening stream channels, promoting incised channels and lowering water tables, reducing pool frequency, increasing soil erosion, and altering water quality (Henjum et al. 1994; Howell and Buchanan. 1992; MBTSG 1995a, b, e; Mullan et al. 1992; Overton et al. 1993; Platts et al. 1993; Uberuaga 1993; USDA and USDI 1996, 1997). These effects reduce cover, increase summer water temperatures, cause habitat degradation, and promote formation of anchor ice (e.g., ice attached to the bottom of an otherwise unfrozen stream, often covering stones, etc.) in winter, and increase sediment in spawning and rearing habitats.

Negative effects of livestock grazing on bull trout habitat may be minimized if grazing is managed appropriately for conditions at a specific site. Practices generally compatible with the preservation and restoration of bull trout habitat include fences to exclude livestock from riparian areas, rotation schemes, relocation of water and salting facilities away from riparian areas, and use of herders.

Agricultural Practices

Agricultural practices, such as cultivation, irrigation diversions, and chemical application, contribute to non-point source pollution (i.e., water quality impairment) and loss of instream flows in some areas within the range of bull trout (IDHW 1991; MDHES 1994; WDE 1992). These practices can release sediment, nutrients, pesticides, and herbicides into streams; increase water temperature; reduce riparian vegetation; and alter hydrologic regimes, typically by reducing flows in spring and summer. Irrigation diversions also affect bull trout by altering stream flow, dewatering streams, and entrainment. The effects of the myriad of small irrigation diversion projects throughout the range of bull trout may be an even greater significance than the large hydropower and flood control projects. Many of these diversions are located high in the watershed and either physically block fish passage by means of a structure (i.e., a dam), or effectively block passage by periodically dewatering a downstream reach (e.g., diversion of flows through a penstock to a powerhouse; diversion of flows for the purposes of irrigation). Reduced stream flows can also result in structural and thermal passage barriers. Additional effects include water quality degradation resulting from irrigation return flows and runoff from fields and entrainment of bull trout into canals and fields (MBTSG 1998). Some irrigation diversion structures are reconstituted annually with a bulldozer as “push up” dams and not only affect passage, but also significantly degrade the stream channel. Even though these “push up” dams are not legal, there is a prevalence of these structures throughout the range of bull trout which has resulted in the isolation of bull trout populations in the upper watersheds in many areas. Bull trout may enter unscreened irrigation diversions and become stranded in ditches and agricultural fields. Diversion dams without proper passage facilities prevent bull trout from migrating and may isolate groups of fish (Dorratcaque 1986; Light et al. 1996). Other effects of agricultural practices on aquatic habitat include stream channelization and large wood removal (Spence et al. 1996).

Transportation Networks

Roads degrade bull trout habitat by creating flow constraints in ephemeral, intermittent, and perennial channels; increasing erosion and sedimentation; creating passage barriers; channelizing stream reaches; and reducing riparian vegetation (Furniss et al. 1991; Ketcheson and Megahan 1996; Trombulak and Frissell 2000). In the Clearwater River Basin of Idaho, for example, Highway 12 is adjacent to much of the Clearwater River, and crosses the river at eight different bridge sites. The highway has constrained the river in some areas and highway maintenance may negatively affect bull trout and their habitats (CBBTTAT 1998). Moreover, the proximity of the highway to the Clearwater River increases the likelihood of hazardous materials or fuel spills entering the river. Similar situations exist along primary and secondary highways across the range of bull trout.

A dirt road is adjacent to much of the West Fork of the Jarbidge River in Nevada and Idaho. McNeill et al. (1997) determined that construction and maintenance of the Jarbidge Canyon Road has influenced the morphology and function of the river. Within a single 4.8 kilometer (3 mile) reach, there are seven bridge crossings, and the largest bridge spans only 62 percent of the average width of the river (McNeill et al. 1997). Maintenance of the road and bridges requires frequent channel and floodplain modifications that affect bull trout habitat, such as channelization; removal of riparian trees and beaver dams; and placement of rock, sediment, and concrete (Jay Frederick, U. S. Forest Service, personal communication, February 6, 1998; McNeill et al. 1997).

Transportation networks also affect bull trout habitat in protected areas such as National Parks. Roads have been constructed to provide access to the Hoh River and Quinault River basins, including areas within Olympic National Park. These roads were typically built following river valleys and often constrain the floodplains. As a result, these roads have been subjected to high flow events and shifts in river channels, forcing extensive streambank armoring to maintain them (Chad 1997; USNPS 2000). Bank armoring impairs bull trout habitat through reduced habitat complexity, stream channelization, reduced riparian vegetation, and bank erosion downstream. Within the Olympic National Park, about 1,770 meters (5,476feet) of rip-rap were documented along the Hoh River in 1997 (Chad 1997), and additional bank stabilization projects have occurred since then.

Mining

Mining degrades aquatic habitat used by bull trout by altering water chemistry (e.g., pH); altering stream morphology and flow; disturbing channel substrates; initiating channel incision and headcuts; and causing sediment, fuel, and heavy metals to enter streams (Martin and Platts 1981; Spence et al. 1996). The types of mining that occur within the range of bull trout include extraction of hard rock minerals, coal, gas, oil, and sand and gravel. Past and present mining activities have adversely affected bull trout and their habitat in Idaho, Oregon, Montana, Nevada, and Washington (Johnson and Schmidt 1988; MBTSG 1995b, e, 1996b, d; McNeill et al. 1997; Moore et al. 1991; Platts et al. 1993; Ramsey 1997; WDW 1992).

For example, it is thought that bull trout were widely distributed in the Coeur d'Alene River Drainage, Idaho (Maclay 1940). However, extensive mining and associated operations have modified and degraded stream channels and floodplains, created barriers to fish movement, and released toxic substances, especially in the South Fork Coeur d'Alene River (PBTTAT 1998). Portions of the system were essentially devoid of aquatic life during surveys conducted in the 1940s, and bull trout have been functionally extirpated in the Coeur d'Alene River Basin since 1992 (USFWS 1998b).

Residential Development and Urbanization

Residential development is rapidly increasing within many portions of the range of bull trout. Residential development alters stream and riparian habitats through building next to streams, contaminant inputs, and increased stormwater runoff, resulting in changes in flow regimes, streambank modification and destabilization, increased nutrient loads, and increased water temperatures (MBTSG 1995a). Indirectly, urbanization within floodplains alters groundwater recharge by rapidly routing water into streams through drains rather than through more gradual subsurface flow (Booth 1991).

Urbanization negatively affects the lower reaches of many of the large rivers and their associated side channels, wetlands, estuaries, and near-shore areas. Activities such as dredging; removing large wood (e.g., snags, log jams, drift wood); installing revetments, bulkheads, and dikes; and filling side channels, estuarine marshes, and mud flats have led to the reduction, simplification, and degradation of habitats (PSWQAT 2000; Spence et al. 1996; Thom et al. 1994). Pollutants associated with urban environments such as heavy metals, pesticides, fertilizers, bacteria, and organics (oil, grease) have contributed to the degradation of water quality in streams, lakes, and estuaries (NRC 1996; Spence et al. 1996).

Fisheries Management

Introductions of nonnative species by the Federal government, State fish and game departments, and private parties, across the range of bull trout have contributed to declines in abundance, local extirpations, and hybridization of bull trout (Bond 1992; Donald and Alger 1993; Howell and Buchanan. 1992; Leary et al. 1993; MBTSG 1995a, c, 1996a, g; Palmisano and Kaczynski 1997; Platts et al. 1995; Pratt and Huston 1993).

Introduced brook trout (*Salvelinus fontinalis*) threaten bull trout through hybridization, competition, and possibly predation (Clancy 1993; Leary et al. 1993; MBTSG 1996a; Rieman and McIntyre 1993; Thomas 1992; WDW 1992). Hybridization between brook trout and bull trout has been reported in Montana (Hansen and DosSantos. 1997; MBTSG 1995a, e, 1996d, e, f), Oregon (Markle 1992; Ratliff and Howell 1992), Washington (WDFW 1998), and Idaho (Adams 1996; Tim Burton, U. S. Forest Service, personal communication, July 1, 1997). Hybridization results in offspring that are frequently sterile (Leary et al. 1993), although recent genetic work has shown that reproduction by hybrid fish is occurring at a higher level than previously suspected (Kanda 1998). Hybrids may be competitors. Brook trout mature at an earlier age and have a higher reproductive rate than bull trout. This difference may favor brook trout over bull trout when they occur together, often leading to replacement of bull trout with brook trout (Clancy 1993; Leary et al. 1993; MBTSG 1995a). The magnitude of threats from nonnative fishes is highest for resident bull trout because they are typically isolated and exist in low abundance.

Brook trout apparently adapt better to degraded habitats than bull trout (Clancy 1993; Rich 1996), and brook trout also tend to occur in streams with higher water temperatures (Adams 1994; MBTSG 1996g). Because elevated water temperatures and sediments are often indicative of degraded habitat conditions, bull trout may be subject to stresses from both interactions with brook trout and degraded habitat (MBTSG 1996a). In laboratory tests, growth rates of brook trout were significantly greater than those for bull trout at higher water temperatures when the two species were tested alone, and growth rates of brook trout were greater than those for bull trout at all water temperatures when the species were tested together (McMahon et al. 1998, 1999).

Nonnative lake trout (*Salvelinus namaycush*) (i.e., west of the Continental Divide) also negatively affect bull trout (Donald and Alger 1993; Fredenberg 2000; MBTSG 1996a). A study of 34 lakes in Montana, Alberta, and British Columbia, Canada, found that lake trout likely limit foraging opportunities and reduce the distribution and abundance of migratory bull trout in mountain lakes (Donald and Alger 1993). Over 250 introductions of lake trout and other nonnative species have occurred in nearly 150 western Montana waters within the range of bull trout (Vashro 2000). The potential for introduction of lake trout into the Swan River Basin and Hungry Horse Reservoir on the South Fork Flathead River, both in Montana, is considered a threat to bull trout (MBTSG 1995d, 1996f). The presence of several lake trout has been recently documented in Swan Lake (MFWP 1999). In Idaho, lake trout and habitat degradation were factors in the decline of bull trout from Priest Lake (Mauser et al. 1988; Pratt and Huston 1993). Lake trout have invaded Upper Priest Lake and are a threat to the bull trout there (Fredericks 1999). Juvenile lake trout are also using some riverine habitats in Montana, possibly competing with bull trout (MBTSG 1996a).

Introduced brown trout (*Salmo trutta*) are established in several areas within the range of bull trout and likely compete for food and space and prey on bull trout (Platts et al. 1993; Pratt and Huston 1993; Ratliff and Howell 1992). In the Klamath River Basin, for example, brown trout occur with bull trout in three streams and have been observed preying on bull trout in one (Light et al. 1996). Brown trout may compete for spawning and rearing areas and superimpose redds on bull trout redds (Light et al. 1996; MBTSG 1996a; Pratt and Huston 1993). Elevated water temperatures may favor brown trout over bull trout in competitive interactions (MBTSG 1996a). Brown trout may have been a contributing factor in the decline and eventual extirpation of bull trout in the McCloud River, California, after dam construction altered bull trout habitat (Rode 1990).

Nonnative northern pike (*Esox lucius*) have the potential to negatively affect bull trout. Northern pike were introduced into Swan Lake in the 1970s (MFWP 1997), and predation on juvenile bull trout has been documented (MBTSG 1996f), but the bull trout population has not declined. Northern pike were also introduced into Salmon, Inez, Seeley, and Alva lakes in the Clearwater River Basin, and a tributary to the Blackfoot River, Montana (MBTSG 1996f). Northern pike numbers have increased in Salmon Lake and Lake Inez, having a negative effect on bull trout. Northern pike in Seeley Lake and Lake Alva are also expected to increase in numbers (Rod Berg; Montana Fish, Wildlife, and Parks; personal communication; November 13, 1997).

Introduced bass (*Micropterus spp.*) may negatively affect bull trout (MFWP 1997). In the Clark Fork River, Montana, Noxon Rapids Reservoir supports fisheries for both smallmouth bass (*M. dolomieu*) and largemouth bass (*M. salmoides*). Both have been high priority sport fish species in management of Noxon Rapids Reservoir. The Montana fishery management objective for Cabinet Gorge Reservoir, downstream of Noxon Rapids Reservoir, is to enhance bull trout while managing the existing bass fishery (MFWP 1997). However, a 1999 Federal Energy Regulatory Commission settlement with the Avista Corporation for dam relicensing makes recovery of bull trout a management priority (Kleinschmidt Associates and Pratt 1998).

Managers are now attempting to balance these potentially conflicting objectives. In the North Fork Skokomish River, Washington, Cushman Reservoir supports largemouth bass, that may prey on juvenile bull trout rearing in the reservoir and lower river above the reservoir (WDFW 1998).

Opossum shrimp (*Mysis relicta*), a crustacean native to the Canadian Shield, was widely introduced in the 1970s as supplemental forage for kokanee salmon (*Oncorhynchus nerka*) and other salmonids in several lakes and reservoirs across the northwest (Nesler and Bergersen 1991). The introduction of opossum shrimp in Flathead Lake changed the lake's trophic dynamics resulting in expanding lake trout populations and causing increased competition and predation on bull trout (MBTSG 1995c; Weaver 1993). Conversely, in Swan Lake, Montana, introduced opossum shrimp and kokanee increased the availability of forage for bull trout, contributing to the significant increase in bull trout numbers in the Swan River Basin (MBTSG 1996f).

Nonnative fish threaten bull trout in relatively secure, unaltered habitats, including roadless areas, wildernesses, and national parks. For instance, brook trout occur in tributaries of the Middle Fork Salmon River within the Frank Church-River of No Return Wilderness, including Elk, Camas, Loon, and Big creeks (Thurow 1985) and Sun Creek in Crater Lake National Park (Light et al. 1996). Glacier National Park has self-sustaining populations of introduced nonnative species, including lake trout, brook trout, rainbow trout (*Oncorhynchus mykiss*), Yellowstone cutthroat trout (*Oncorhynchus clarki bouvieri*), lake whitefish (*Coregonus clupeaformis*), and northern pike (MBTSG 1995c). Although stocking in Glacier National Park was terminated in 1971, only a few headwater lakes contain exclusively native species, including bull trout. The introduction and expansion of lake trout into the relatively pristine habitats of Kintla Lake, Bowman Lake, Logging Lake, and Lake McDonald in Glacier National Park has nearly extirpated the bull trout due to predation and competition (Fredenberg 2000; Marnell 1995; MBTSG 1995c).

Some introduced species, such as rainbow trout and kokanee, may benefit large adult bull trout by providing supplemental forage (Faler and Bair 1991; Pratt 1992; Vidergar 2000). However, introductions of nonnative game fish can be detrimental due to increased angling and subsequent incidental catch and harvest of bull trout (Bond 1992; MBTSG 1995c; Rode 1990; WDW 1992).

Isolation and Habitat Fragmentation

Although bull trout are widely distributed over a large geographic area, the effects of human activities over the past century have reduced their overall distribution and abundance, as well as fragmented their habitat. This fragmentation reduces the amount of available habitat and increases isolation from other populations of the same species (Saunders et al. 1991). Burkey (1989) concluded that when species are isolated by fragmented habitats, low rates of population growth are typical in local populations and their probability of extinction is directly related to the degree of isolation and fragmentation. Without sufficient immigration, growth for local populations may be low and probability of extinction high (Burkey 1989, 1995).

Metapopulation concepts of conservation biology theory have been applied to the distribution and characteristics of bull trout (Dunham and Rieman 1999; Rieman and McIntyre 1993). A metapopulation is an interacting network of local populations with varying frequencies of migration and gene flow among them (Meffe and Carroll 1997). Local populations may be extirpated, but can be reestablished by individuals from other local populations. Thus, multiple local populations distributed throughout a watershed provide a mechanism for spreading risk because the simultaneous loss of all local populations is unlikely. Habitat alteration, primarily through the construction of impoundments, dams, and water diversions, has fragmented habitats, eliminated migratory corridors, and isolated bull trout in the headwaters of tributaries (Dunham and Rieman 1999; Rieman and Dunham 2000; Rieman et al. 1997; Spruell et al. 1999). Based on population genetics, there is more divergence among bull trout than among salmon (Leary and Allendorf 1997), indicating less genetic exchange among bull trout populations. The recolonization rate for bull trout is very low and recolonization may require a very long time, especially in light of the man-made isolation of various bull trout populations.

Migratory corridors allow individuals access to unoccupied but suitable habitats, foraging areas, and refuges from disturbances (Saunders et al. 1991). Maintenance of migratory corridors for bull trout is essential to provide connectivity among local populations, and enables the reestablishment of extinct populations. Where migratory bull trout are not present, isolated populations cannot be replenished when a disturbance makes local habitats unsuitable (Rieman and McIntyre 1993; USDA and USDI 1997). Moreover, limited downstream movement was observed for resident bull trout in the Bitterroot River Basin (Nelson 1999) suggesting that reestablishment of migratory fish and potential re-founding of extinct bull trout populations may be a slow process, if it occurs at all.

Because isolation and habitat fragmentation resulting from migratory barriers have negatively affected bull trout by: (1) reducing geographical distribution; (2) increasing the probability of losing individual local populations (Rieman and McIntyre 1993); (3) increasing the probability of hybridization with introduced brook trout (Rieman and McIntyre 1993); (4) reducing the potential for movements in response to developmental, foraging, and seasonal habitat requirements (MBTSG 1998); and (5) reducing reproductive capability by eliminating the larger, more fecund migratory form from many local populations (MBTSG 1998; Rieman and McIntyre 1993), restoring connectivity and restoring the frequency of occurrence of the migratory form will be an important factor in providing for the recovery of bull trout. The manner and degree to which individual dams and diversions affect specific bull trout local populations is likely to vary depending on the specific physical factors at play and the demographic attributes of the local population in question.

Evidence suggests that landscape disturbances, such as floods and fires, have increased in frequency and magnitude within the range of bull trout (Henjum et al. 1994; USDA and USDI 1997). Passage barriers and unsuitable habitat that prevent recolonization, have resulted in bull trout extirpation through these landscape disturbances (USDA and USDI 1997). Also, isolated populations are typically small, and more likely to be extirpated by local events than larger populations (Rieman and McIntyre 1995), and can exhibit negative genetic effects.

Inadequacy of Existing Water Quality Standards

Temperature regime is one of the most important water quality factors affecting bull trout distribution (Adams and Bjornn 1997; Rieman and McIntyre 1995). Given the temperature requirements of bull trout (Buchanan and Gregory 1997), existing water quality criteria developed by the States under sections 303 and 304 of the Clean Water Act may not adequately support spawning, incubation, rearing, migration, or combinations of these life-history stages (62 FR 41162) (NDEP 1998; Oregon 1996; Washington 1997).

Elevated levels of contaminants may result in either lethal (e.g., mortality) or sublethal effects to bull trout. Sublethal impacts may include reduced egg production, reduced survival of any life stage, changes in behavior, reduced growth, impaired osmoregulation, and many subtle endocrine, immune, and cellular changes. Contaminants may also affect the food chain and indirectly harm bull trout by reducing prey availability due to reduced habitat suitability for prey species. Lethal impacts from contaminant inputs are most likely from spills, whereas sublethal impacts may occur from such land uses as agriculture, residential/urban, mining, grazing, and forestry.