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Effects of Disturbance and Management of Forest Health on Fish and Fish Habitat in Eastern Oregon and Washington

Abstract

Effects of fire, forest insects and diseases, grazing, and forest health treatments on fish populations and habitat are reviewed. Fire, insects, and disease affect fish habitat by their influence on the rate and volume of woody debris recruitment to streams, canopy cover and water temperature, stream flow, channel erosion, sedimentation, nutrients, and residual vegetation. Physical effects from fire vary greatly depending on fire severity and extent, geology, soil, topography, and orientation of the site, and subsequent precipitation. Most effects moderate within a decade. Post-fire erosion and wood recruitment are also influenced by fire lines, road construction, and timber harvest. Although some disturbances, such as severe fire and subsequent floods, appear catastrophic, and effects may last decades or centuries, natural disturbances help create and maintain diverse, productive aquatic habitats. Recolonization of fish populations following wildfires can be rapid and is related to occurrence of local refugia, life history patterns, access for migratory forms, and distribution of the species. In most livestock studies, grazing negatively affected fish habitat and populations, but results may vary depending on sites and specific grazing management. Effective approaches to grazing management similarly depend on the specific application and the commitment of operators and managers. Restoration of the structure, function, and processes of watersheds more similar to those with which native species evolved may favor those species; however, there is little documentation of the aquatic effects of those activities. Risk from vegetative treatments may be minimized by experimenting outside of critical areas (i.e., conserving key habitats and populations, focusing intensive treatments on upland sites). Use of more benign techniques (e.g., lower-impact logging systems) and pulsed treatments consistent with characteristics of natural disturbance regimes are other considerations for achieving both terrestrial and aquatic objectives.

Introduction

Native fishes, particularly salmonids listed under the Endangered Species Act (ESA), are a key issue in managing forests in eastern Oregon and Washington. This paper is part of a series of papers to provide a synthesis of literature concerning effects of selected forms of disturbance and management in response to risks from those disturbances in forested ecosystems east of the Cascade Range. Disturbance agents addressed include fire, forest insects and diseases, and ungulate grazing. This review addresses the effects of these disturbances and forest health treatments (e.g., forest fuels reduction, timber salvage, prescribed fire, grazing management) on native and invasive fishes and their habitats. This was not designed to be an exhaustive literature review but a brief summary of key relevant publications, particularly other syntheses and reviews. See Wondzell (2001) for related discussion of erosion and sedimentation.

Effects of Fire

The role of naturally occurring disturbances as a dominant influence on aquatic habitats and species has become increasingly appreciated (e.g., Reice et al. 1990, Reeves et al. 1995, Bisson et al. 1997, Benda et al. 1998). Fire, forest insects and disease, and wild ungulate grazing are integral elements of natural processes that control the development and persistence of aquatic habitat (Swantson 1991) with which native aquatic species have evolved. Most of the existing work on aquatic systems in the interior Pacific Northwest has been related to wildfire.

Physical changes in watersheds from fire that affect aquatic habitat include water yield, erosion, soils, nutrients, water temperature, and vegetative cover (Gresswell 1999). Most of the more immediate effects are linked to increased water yield and erosion (surface erosion, mass wasting, and channel erosion) and accompanying in-channel sedimentation. Soil characteristics, such as water repellency, porosity, and structure, can be altered (Klock and Helvey 1976, McNabb et al. 1989, White 1996). Brief pulses of nitrogen and phosphorus in streams adjacent to fires may occur; however, some chemical levels may be elevated for years because of reduced cycling of nutrients by vegetation (Wright 1976, Bayley et al. 1992). Water temperatures may increase during and following fires from a loss of riparian canopy and increased insolation (Helvey 1972, Helvey et al.

1976, Minshall et al. 1997). Thermal increases following fire are generally not lethal to aquatic organisms and decline with the establishment of riparian vegetation (Gresswell 1999). Revegetation of riparian zones is usually rapid unless the soil is compacted by logging (Rieman and Clayton 1997) or regrowth is suppressed by grazing. Temperature increases may be moderated by increased discharge resulting from reduced evapotranspiration (Helvey et al. 1976) and by shading from remaining standing tree boles next to streams (Rieman and Clayton 1997). Increases in temperature in smaller, headwater streams may be negligible further downstream (Gresswell 1999).

The physical effects of fire on aquatic ecosystems vary greatly depending on fire severity and extent, the characteristics of the watershed (e.g., geology, soils, topography, orientation), and subsequent precipitation (Swanson 1981; Meyer et al. 1992, 1995). Most of these effects occur within 10 years of the fire. However, burned areas that experience debris flows or floods may take decades (Megahan 1991) or even centuries (Benda 1985) for reestablishment of soil and mature vegetation adjacent to channels. Changes in recruitment of large woody debris, an important component of fish habitat, can likewise be more lasting. Where fire intensity was not great enough to consume large wood, recruitment to the channel increased in most studies (Young 1994, Reeves et al. 1995, Minshall et al. 1997) and may persist until the succeeding forest is established (Swanson and Lienkaemper 1978). Wood recruitment is also dependent on pre-fire vegetative conditions (e.g., tree size and density) and post-fire timber harvest.

Most studies of the effects of fire have not considered possible distinctions between riparian and upland portions of the watershed. Recent research is beginning to help fill that void (Williamson 1999, Olson 2000). Olson (2000) found that in dry forest types with low-severity fire regimes, historically fire occurred frequently within riparian management zones and that there was little difference in the fire frequencies between riparian and upland areas.

Mortality of fishes has been reported following fires and subsequent floods in burned areas (Rieman et al. 1997; Gresswell 1999; Howell, unpublished). Mortalities have generally been associated with severe fire intensities (Minshall et al. 1989, McMahon and deCalesta 1990, Rieman et al. 1997). The exact mechanism of fish mortality is uncertain, although high stream temperatures and toxicity from smoke and ash have been implicated (Gresswell 1999). Longer-term effects can range from essentially none where recolonization is rapid (e.g., Rieman et al. 1997) to extirpation of isolated populations (Rinne 1996). Salmonid densities in defaunated reaches were similar (Rieman et al. 1997) or exceeded (Novak and White 1990) densities in unaffected reaches by 1-3 years following fire. Recolonization is related to the occurrence and proximity of local refugia (e.g., unburned or lightly burned patches); multiple life history patterns of the affected species, especially migratory forms; access for migratory fish; and overlapping generations (Rieman et al. 1997, Gresswell 1999). Effects on populations that have become fragmented, restricted to resident forms, or consist of narrowly distributed endemics may be more long lasting.

Some post-fire changes, such as erosion, fire, or flood events (and associated fish mortality) may appear catastrophic; however, these processes help create and maintain diverse and productive aquatic habitats through subsequent interactions between aquatic and riparian environments, such as increases in large wood and coarse sediments that provide hydraulic complexity (Reeves et al. 1995; Bisson et al. 1997; Rieman et al. 1997, 2000; Gresswell 1999). Many native species, particularly salmonids, are adapted to variable environments resulting from natural disturbances through diverse life histories (Rieman et al. 1997, Gresswell 1999). Fish populations can recover soon after large disturbances—within a few days (Peterson and Bayley 1993), weeks (Sheldon and Meffe 1995), or a few years (Niemi et al. 1990).

Physical effects of most fires on aquatic systems have been greatest at 10- to 100-year temporal scales and at 100- to 1,000-m spatial scales (i.e., stream reaches or segments) (Gresswell 1999). For migratory and more extensively distributed fishes, the scale of biological effects is likely to be at the individual and local population level rather than at higher levels of organization (metapopulation, evolutionarily significant unit, or species). Changes in landscape-level fire regimes (i.e., increases in fire severity and extent) (Hann et al. 1997) could increase the scale of these effects.

Anthropogenic factors are important considerations in evaluating the effects of fire and forest insects and tree disease on fishes and fish habitat. Fire suppression, pre- and post-fire logging, roads, and other management-induced changes to the watershed can greatly influence how these disturbances affect fish habitat. For example, post-fire erosion has been closely linked to fire lines, timber harvest, and road construction (Swanston 1971, Marston and Haire 1990, Gresswell 1999, McIver and Starr 2000).

Historical natural disturbances tended to be erratically distributed or "pulsed" (i.e., dispersed in time and space), whereas forest management has produced more regularly spaced, chronic "press" effects (Yount and Niemi 1990, Reeves et al. 1995, Rieman et al. 1997). For example, erosion from fires is episodic, whereas roads can be continually persistent sources of fine sediment. Anthropogenic factors not only alter the physical effects of natural disturbances but have also contributed to simplification and homogeneity of aquatic habitat. Corresponding fragmentation, isolation, and loss of life history diversity of fishes reduce their resiliency in responding to the disturbance (Bisson et al. 1992).

Effects of Insects and Disease

There have been few studies of the direct effects of forest insects and disease on aquatic systems. Some effects may be comparable to those of fire. For example, all of those disturbances may influence the rate and volume of recruitment of woody debris to stream channels, canopy cover in riparian areas, snow accumulation, evapotranspiration, and stream flow (Swanston 1991). The amount of woody debris in the channel would likely increase in the short term from trees killed in the riparian area and then decline until replacement trees are large enough to provide that function. However, recruitment of dead wood would probably occur more slowly following an insect outbreak than a stand-replacing fire (Youngblood and Wickman, *In press*). Reduction of canopy cover and litter fall in the riparian area could increase solar insolation and water temperatures in smaller streams and reduce allocthonous nutrient input after an initial pulse. Increase in stream flow following mortality has also been documented (Love 1955; Bethlahmy 1974; 1975). The recurrence of these effects is on the order of 10-100 years (Swanston 1991). Like fire, the magnitude of these effects and potential changes to fish habitat depend on the extent, patchiness, and severity of the disturbance and characteristics of the site. Effects would be most evident where the mortality from the disturbance is extensive and severe. Other effects of insect and disease outbreaks, such as the pattern of disturbance (e.g., patch size, frequency) and amounts of erosion and residual vegetation, may differ from the effects of wildfire. Forest insects and disease could also indirectly affect aquatic habitat through increases in fuels and accompanying changes in fire regimes.

Effects of Grazing

General effects of grazing by wild and domestic ungulates are discussed in this issue by Kie and Lehmkuhl (2001). Platts (1991) in a review of aquatic effects of livestock grazing concluded that "improper livestock grazing degrades riparian and aquatic habitats, resulting in decreased production of salmonids." Much debate concerning this issue revolves around what constitutes "improper" grazing. As McInnis and McIver (*In press*) and Ehrhart and Hansen (1997) point out, there is abundant evidence that heavy livestock grazing is detrimental to riparian habitat, but there is little information on effects of lower stocking levels or which grazing strategies are most effective at improving riparian habitat. Quantifying livestock use is not simply a measure of livestock numbers, but also distribution, duration, and timing (Holecheck et al. 1989, Larsen et al. 1998).

Several recent comprehensive literature reviews have also examined the effects of grazing on aquatic ecosystems (Hawkins 1998, Larsen et al. 1998, Belsky et al. 1999, Rinne 1999). Most of the available information addresses primarily effects of livestock grazing on vegetation and, to a lesser extent, other aspects of aquatic and riparian habitat (e.g., geomorphology, stream banks, water quality) rather than direct effects on fishes (e.g., diversity, density, biomass, population trends). Specific effects on aquatic habitat include reduction and changes in the composition of riparian vegetation, shallowing and widening of the stream channel, increases in fine sediment, reduction of pools, increases in water temperature, lowering of the water table, changes in the timing and volume of stream flow, bank trampling, and accelerated bank erosion (Platts 1991). Many of these effects are interrelated. In a meta-analysis of grazing studies by Hawkins (1998), riparian vegetation and channel characteristics, such as pools, width:depth ratios, and levels of fine sediments generally associated with favorable salmonid habitat declined in most studies. However, differences between ungrazed and grazed treatments that were greater than 1.5 times, which was considered by the author to be ecologically significant, occurred in less than half (48%) of the vegetation studies and 20% of the studies examining channel characteristics. Most of studies that consider effects on fishes deal principally with salmonids (Rinne 1999). Seven of the nine studies of effects of grazing on fishes reviewed by Hawkins (1998) reported reductions in the biomass or density of fishes, and in 6 of those, the differences between grazed and ungrazed treatments were 1.5-fold or greater. Belsky et al. (1999) concluded that although some studies showed no statistical differences among grazed and ungrazed areas, no studies they reviewed indicated positive effects of livestock grazing on riparian areas. All of these reviews emphasize the difficulty of drawing generalizations from extant studies because of inadequacies in study design, replication, controls, and statistical rigor and variability of sites and grazing management.

Evaluating the effects of grazing on aquatic systems can often be difficult because streams are inherently dynamic and variable (Platts 1991; Swanston 1991). Natural events can produce some effects that appear similar to grazing. Isolating effects of grazing treatments from the cumulative watershed effects of management can also be difficult. The legacy of long-term effects of historical livestock grazing can confound results.

Large wild ungulates, such as deer and elk, can produce some effects on aquatic habitat similar to those of livestock primarily through bank trampling and consumption of riparian vegetation, particularly shrubs (Swanton 1991; Hanley and Taber 1980 and Hanley 1987 cited in Swantson 1991). Effects of wild ungulates on aquatic habitat could also be influenced by current and past livestock grazing.

Forest Health and Productivity Treatments

Grazing Strategies

A number of grazing strategies have been developed to meet riparian and livestock production objectives. Except for exclusion (e.g., fenced riparian exclosures), no single technique has consistently improved degraded riparian areas (Elmore and Kauffman 1994). Livestock exclusion has produced the most dramatic and rapid rates of riparian recovery (Elmore and Kauffman 1994); however, it may not be required for improvement or optimum in all cases (Ehrhart and Hansen 1997). Fencing to exclude both livestock and big game has also been used to promote recovery of aspen in riparian areas. Riparian exclosures or any approach to managing livestock use of riparian areas needs to incorporate considerations for upland range management as well so that problems are not merely shifted from one area to another (Elmore and Kauffman 1994).

The "best" grazing strategy depends on the specific application (e.g., site, livestock, grazing patterns, operator), objectives, management feasibility, and cost-effectiveness. Riparian areas differ widely with respect to their sensitivity to grazing, potentially suitable grazing alternatives, vegetation, and condition. For example, adjustments to the timing of use (e.g., early or late season) may be very limited or not possible in higher-elevation pastures. Objectives can be the most difficult of these elements to define. What kinds and degree of grazing effects on aquatic habitat of fishes or other species dependent on streams and riparian areas are ecologically significant? What rate of riparian recovery is needed, especially for depressed species including ESA listed species? As previously discussed, empirical data on the degree of ecological effects of grazing treatments, much less the rates of change and the biological response of fishes, are very limited. Although much of the focus of grazing management may be on the techniques or prescriptions, the commitment and the necessary resources (e.g., funding, staff) of operators and managers are essential (Ehrhart and Hansen 1997). This may be the greatest challenge to successful grazing management.

Fire, Forest Insects, and Tree Disease

In a broadscale analysis of the interior Columbia Basin, Rieman et al. (2000) found that declines in native fish populations and habitat frequently paralleled changes in the structure, composition, and disturbance patterns of adjacent forests and that restoration of those forests more consistent with natural disturbances could also benefit aquatic ecosystems. The effectiveness of such restoration will depend on the design of the projects, the techniques used, and minimizing the risks to aquatic species.

Efforts to address forest health through active management of vegetative structure and composition and fire need to be coupled with consideration of forested watershed function and needs for restoration of aquatic ecosystems (Bilby and Bisson 1991, Rieman and Clayton 1997, Rieman et al. 2000). Benefits to aquatic species are less likely to accrue if aquatic communities continue to be constrained by other factors. For example, focusing projects in watersheds that are heavily roaded would often correspond with areas that have degraded aquatic conditions and the highest departure from historical vegetative and disturbance patterns (Hann et al. 1997, Hessburg et al. 1999, Rieman et al. 2000). Existing roads, rather than new roads, could be used to facilitate vegetative management; unnecessary and particularly damaging roads could be obliterated or modified to reduce their effects on aquatic habitats.

Adverse effects of past forest management practices on aquatic habitat are well documented (e.g., Salo and Cundy 1987, Meehan 1991, Lee et al. 1997). Effects of roads, in particular, have recently been emphasized (Lee et al. 1997, Gucinski et al. 2000, Trombulak and Frissel 2000). Future forest health treatments that differ from traditional management approaches by employing potentially lower-impact techniques, such as prescribed fire and logging systems that reduce ground disturbance, compaction, and road requirements may help avoid unintended long-term damage to aquatic ecosystems.

Forest health treatments can potentially influence the composition of aquatic species assemblages, including exotic species primarily through accompanying alteration of instream and riparian habitats. For example, reduction of forest canopy from logging or grazing in riparian areas can result in increases in stream temperature, that can in turn favor changes in aquatic species composition through interspecific competition, differential disease resistance, and predation (Li et al. 1987, Strange et al. 1992, Li and Moyle 1999). Introduced fishes, such as brown trout (*Salmo trutta*), brook trout (*Salvelinus fontinalis*), and warmwater species like smallmouth bass (*Micropterus dolomieu*), tolerate or prefer higher water temperatures than coldwater native fishes, such as bull trout (*Salvelinus confluentus*), cutthroat trout (*Oncorhynchus clarki*), and other salmonids (Bjornn and Reiser 1991, Howell and Buchanan 1992, Li et al. 1994, Buchanan and Gregory 1997). Other effects associated with vegetation management, particularly in riparian areas, such as reduction of large woody debris, increased fine sediment, and changes in trophic inputs (Gregory et al. 1987, Tait et al. 1994) could also favor non-native species. Changes in natural disturbance regimes (e.g., timing, frequency, intensity) can promote invasion of non-native fishes (Reeves et al. 1995, Reeves et al. 1998) and riparian plants that affect interactions between aquatic and terrestrial environments (DeFerrari and Naiman 1994, Planty-Tabacchi et al. 1996, Heckman 1999). Roads constructed to facilitate logging and other forest management activities can provide access to streams for intentional stocking of non-native hatchery fish and illegal fish introductions (Lee et al. 1997, Trombulak and Frissel 2000).

To the extent that forest health treatments promote long-term changes in the structure, function, and processes of watersheds that are more in keeping with the patterns that native species evolved with, those treatments ultimately may favor native aquatic species (Beechie and Bolton 1999), unless their short-term effects on species composition cannot be reversed. However, there is little documented evidence to date to support the hypothesis that forest health treatments to manage overstory vegetation (e.g., thinning, timber harvest, prescribed fire) promote native aquatic species and discourage invasions by non-native forms. The effects of those treatments will depend on what specific activities are undertaken and the success of the mitigation.

Risks and potential benefits of forest health and aquatic restoration treatments need to be carefully assessed (Rieman et al. 1997). In some cases, the ecological risks of active management may be greater than the risks of large, severe fires (Lee at al. 1997, Rieman and Clayton 1997), insects, and disease. Unfortunately, there is little information concerning the effectiveness of forest health treatments on reducing risks to aquatic habitats and species or the aquatic risks of treatments using current techniques. For example, a recent review of the use of prescribed fire found no literature on the effects of prescribed burning on aquatic systems (S. Wollrab, USDA Forest Service, Rocky Mountain Research Station, Boise, ID. 1999. Can prescribed fire be used to restore habitat for salmonids? http://www.fs.fed.us/rm/boise/fish/revisedfish/projects/fish%20projects/tech%20tran%20projects/firepage.htm [April 2001]).

Given the major uncertainties associated with ecosystem analysis and management (Baker 1994, Stanley 1995, Noss and Scott 1997), Reeves et al. (1995) and Rieman et al. (2000) recommend minimizing aquatic risks by conserving remaining key terrestrial and aquatic habitats and populations and by experimenting with restoration outside of critical areas. The "healthiest" remaining fish populations and habitat are frequently found where forests are also healthy and, consequently, the need for forest health treatments is lowest (Rieman et al. 2000). Focusing treatments in upland areas, particularly dry forest types, which have the greatest risk of surface fire ignition and spread, may reduce potentials for uncharacteristic high-intensity fires and for adversely affecting downslope riparian processes (Williamson 1999). Occasional high-intensity fires in riparian areas may be exceptional cases resulting from unique disturbance histories or unusual weather conditions (Williamson 1999). Olson's (2000) findings concerning fire frequency in riparian areas suggest that relations between riparian vegetation types and their characteristic fire regimes should influence management of riparian areas. That is, reintroduction of fire and management of vegetation consistent with the vegetation type and corresponding fire frequencies and severities should be incorporated into riparian management if the objective is to maintain historical disturbance processes. However, the methods used to achieve that objective may differ in riparian areas given their potential effects on habitats of aquatic and riparian-associated terrestrial species (see Wales 2001).

To make aspects of forest management practices more consistent with natural disturbance regimes and the resulting creation and maintenance of aquatic habitat, Reeves et al. (1995) suggest extending riparian management areas to increase potential wood delivery to the channel, particularly along smaller first- and second-order streams, increasing intervals of harvest rotations to approximate natural disturbance intervals, and concentrating rather than dispersing timber management. These recommendations, however, are based on disturbance regimes of the central Oregon Coast Range, which differ in some respects from those east of the Cascades. For example, in the Coast Range historical fires tended to be infrequent, concentrated, lethal, stand-replacement events (Benda 1994), whereas dry forests east of the Cascades, which have been the focus of much of the concerns about forest health, experienced fires historically that were frequent, extensive, and low severity (Hann et al. 1997, Heyerdahl 1997). We can apply similar general principles to designing forest health treatments that emulate the characteristics of eastside natural disturbances and aquatic habitats. This would minimally entail: (1) protection and restoration of riparian areas to provide a continued source of large wood and other biologically influential processes and functions (e.g., shade, sediment supply), and (2) intervals and distribution of treatments that are similar to natural disturbance intervals and distribution. Natural disturbance agents such as wildfire, insects, and forest pathogens produce a very complex mosaic of effects (Aber et al. 2000). Attempting to conduct management activities that mimic this complexity will be challenging (Rieman and Clayton 1997) and will require well-designed studies and monitoring. The experimental status of forest health treatments suggests an important role for research and truly adaptive management in developing approaches that achieve objectives for both terrestrial and aquatic ecosystems.

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Note

This special issue of *Northwest Science* is a set of papers reviewing the state of knowledge about disturbance processes in eastern Oregon and Washington, related management practices, and effects on key management issues.