Appendix C. Aquatic Ecosystems Best Available Scientific Information

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Introduction

This appendix provides a summary of the best available science (BASI) used to support the aquatic ecosystems analysis found in the body of the DEIS.

Streams and riparian areas

Stream and riparian area management

Studies in the 1960s and 1970s documented harmful effects that timber harvest and road-building, at that time, had on streams, and in response agencies began passing a series of management requirements for activities on state and federal lands near streams. These are referred to as “best management practices” (BMPs). Everest and Reeves (2007) disclosed the following regarding the development of BMPs for the Pacific Northwest: “They [BMPs] were developed through the normative process that weighed, evaluated, and incorporated many types of information. The BASI for protection of riparian and aquatic habitats was not always incorporated into forest practice rules” (p. 77). This was repeated several times even as successive monitoring efforts continued to document degraded stream conditions (Gordon H. Reeves, Olson, et al., 2016).

A crisis point was reached in the early 1990s in the western U.S. when several stocks of salmon and trout were reaching critically low numbers (Nehlsen, Williams, & Lichatowich, 1991) and ultimately were listed as threatened or endangered under the Endangered Species Act (ESA). By the mid-1990s, the FS and BLM had completed three broad-reaching documents (hereafter referred to collectively as “the strategies”) that amended forest plans across much of the public lands in the northwest to improve their conservation function. INFISH (U.S. Department of Agriculture, Forest Service, Intermountain, Northern, and Pacific Northwest Regions, 1995) addressed inland native fish habitat management including bull trout that were not covered within the geographic scope of the Pacific Anadromous Fish strategy (PACFISH). This includes parts of Idaho and Montana including the Blackfoot and Upper Clark Fork drainage.

One feature of the strategies was the extension of the distance from the stream of riparian management zones (RMZs; i.e., riparian reserves in the Northwest Forest Plan) and riparian habitat conservation areas to better protect ecological processes next to streams. Also, the precautionary principle was invoked. Reeves et al. (2016) described this principle as, “Forest managers who wanted to alter the comprehensive default prescriptions for riparian management under the [Northwest Forest Plan] in order to pursue other management goals were required to demonstrate through watershed analysis that changes would not compromise established riparian-management goals.” Not only did the burden of proof shift, these new strategies also required managers to consider ecological processes at the watershed scale. The components used in the Northwest Forest Plan, including the concept of the precautionary principle, were included in PACFISH and INFISH.

Riparian management has remained controversial, in part because of competing values and uses (Lee, Smyth, & Boutin, 2004). Strategies employed by the Northwest Forest Plan, PACFISH, and INFISH appear to have been successful at halting the loss of old growth due to harvest within riparian areas and limiting damage to aquatic systems in the Pacific Northwest (J. W. Thomas, Franklin, Gordon, & Johnson, 2006) and the intermountain region. However, some suggest a protection mindset emerged that has prevented management within riparian areas that would be desirable to sustain and/or promote ecological processes beneficial to aquatic or terrestrial ecosystems (Liquori, Martin, Coats, & Ganz, 2008; Ryan & Calhoun, 2010; J. W. Thomas et al., 2006). Speaking of the need to restore ecological conditions and make good on social, economic, and ecological commitments in the Northwest Forest Plan, Thomas et al. (2006) wrote, “Minimization of short-term risks (the modus operandi of regulatory agencies and the federal courts) has a price tag, and a very big one, related to significantly increased longer-term risks of failure to meet objectives over very long time frames. Unless the federal agencies consider the peril of inaction equal to the peril of action, the goals of the [Northwest Forest Plan] will not be reached.” Richardson, Naiman, and Bisson (2012) wrote: “In an increasingly complicated management arena, the challenge will be to find alternatives to fixed width buffers that meet the multiple objectives of providing clean water (minimizing nutrient and sediment inputs), aquatic habitat, habitat for riparian species, connectivity across landscapes, and related responses.”

Bank stability and livestock grazing

Bank stability is discussed here as an ecological process; especially as it relates to the effects of anthropogenic activities. In addition to bank degradation that could occur as a result of vegetation management, it can also occur from dispersed recreation and grazing.

Bank stability on low gradient stream reaches that support cold water fish species are of particular concern and are susceptible to livestock overuse. Riparian vegetation that stabilizes the banks has the best opportunity to slow velocity and induce deposition of materials, and recreate channel pattern, profile, and dimension appropriate for the landscape setting. Where streambank instability due to vegetation removal or changes in channel form may arise from channel widening or channel incision, vegetation along the greenline is most critical for maintaining stability. When livestock grazing is closely managed and monitored by professional land managers, assumptions are made that some degree of cattle use is compatible with riparian ecosystem management and that trends towards desired conditions can be achieved while cattle graze the area (Armour, Duff, & Wayne, 1994; Robert L. Beschta, Bilby, Brown, Holtby, & Hofstra, 1987; Bryant et al., 2004; Clary & Webster, 1990; Johnson, 1992; Platt, 1991);(Hanson, Wullschleger, Bohlman, & Todd, 1993).

Most of the literature reviewed pertains to varied conditions found in western riparian areas and is most applicable to riparian areas in sagebrush grasslands, western interior forests and prairie settings. Many of these rangelands can be affected by varying amount of grazing use. A publication discussing grazing in southwest Montana disclosed some of the history of grazing and focused attention on the stream channel response and management options (Bengeyfield, 2006). Extensive grazing by both wild and domestic ungulates can remove woody plants (Batchelor, Ripple, Wilson, & Painter, 2015), reduce the vigor of perennial forbs and grasses, and cause channel profile and function changes via bank collapse on low gradient streams (Bengeyfield, 2006; Trimble & Mendel, 1995). Widening channels, increased stream temperature, increased fine sediment, altered bank structure and loss of overhanging vegetation that may occur from excessive grazing (Kershner, Roper, Bouwes, Henderson, & Archer, 2004; Myers & Swanson, 1996) is often harmful to aquatic fauna, especially cold-water dependent species (Belsky, Matzke, & Uselman, 1999; Saunders & Fausch, 2007). Furthermore, some studies have demonstrated trout respond positively to livestock exclusion (Sievers, Hale, & Morrongiello, 2017), though mechanisms are not clearly understood.

Large wood

The fate of large wood in streams has been an important focus for aquatic scientists and managers in the western U.S. for decades (Richardson et al., 2012). Up until the 1980s, many managers were concerned about how wood in streams affected water quality and about how accumulations of wood in streams could sometimes block fish migration. These concerns led to instream wood removal programs (Mellina & Hinch, 2009). By the 1980s, scientists more fully recognized wood’s role in channel formation and maintenance (J. W. Thomas & Raphael, 1993). As with stream temperature, the precautionary principle applied by the strategies to riparian reserves and riparian habitat conservation areas also ensured that the interim widths were set wide enough to encompass any trees that could be delivered to streams, especially the two-tree width for fish-bearing streams (Everest & Reeves, 2007).

Regarding the riparian width needed to ensure streamside wood delivery to streams, debate and scientific inquiry has continued since the strategies were adopted. Studies have been completed to help identify where wood in streams comes from (L. Benda, Miller, Bigelow, & Andras, 2003; Gordon H. Reeves, Burnett, & McGarry, 2003) and the fate of wood once it is delivered above or to the stream (T. J. Beechie, Pess, Kennard, Bilby, & Bolton, 2000). In addition to streamside delivery, disturbance combined with topography can deliver a significant percentage from outside riparian management zones, especially steeper watersheds that are more dissected. Models have also been developed to help identify the likelihood of riparian trees being delivered to the stream channel (L. Benda et al., 2003; Meleason, Gregory, & Bolte, 2003; Pollock, Beechie, & Imaki, 2012; T. Spies, Pollock, Reeves, & Beechie, 2013; Welty et al., 2002). Models focused on wood delivery from the riparian areas consider distance from the stream, median tree height, and the direction that trees fall. Benda and others. (2016) also discuss how to implement tree tipping (manually falling trees into a stream) to balance the effects of thinning dense second-growth stands to accelerate large wood development. Modeling completed by Meleason and others (2003) found that > 90 percent of wood was contributed from within 30 meters of the stream edge for modeled conifer riparian stands in western Oregon and Washington. In a literature review, Spies and others (2013) found that 95 percent of wood delivered to streams from hardwood stands came from within 82 feet, and from conifer stands from within 148 feet, in forests in the western cascades of Oregon and Washington.

Livestock grazing

The primary grazing areas that have low enough precipitation and high enough evaporation rates to support grass communities, instead of coniferous stands, tend to occur in the warmer, lower elevation areas that may also include losing stream reaches. Although the losing flows in these areas tend to be principally geologically controlled, grazing related impairments can also contribute to stream-flow loss. Mechanisms include reductions in shade canopy, disruption of beaver created water storage in flood-plains, and altering width-depth entrenchment ratios. These same impairment related mechanisms often lead to an increase in water temperatures in the stream. Additional grazing related impairments are increased sediment yields and in-channel storage of fine sediments, which also impact stream channel form, function and fish habitat. Across the plan area, fine sediments are almost always darker in color than native gravels and larger sized substrates. The combination of a higher width-depth ratio and a reduced shade canopy results in higher solar radiation absorption increasing water temperatures, decreasing food production, and reducing the quality of aquatic habitat.

Wildland fire

Under natural fire regimes, fire that burned into riparian areas was influenced by a combination of factors, including weather (i.e., wind speed and direction), fuels/vegetation conditions (i.e., moisture level, downed wood, forest densities and tree species), terrain (i.e., steepness as it may affect spread of fire), climatic conditions (i.e., drought period), and just plain chance. See fire and fuels section for a description of fire history in the plan area.

Fire occurrence historically varied with vegetation type, aspect, and elevation differences. High severity fire occurrence in forested vegetation types ranged from 35 – 200 years and were typically associated with extended dry climatic periods. These types of fires would often burn at high or moderate severity through riparian areas as well, especially in the steep, deeply incised stream channels typical across much of the plan area. These fires converted forests to an early successional stage dominated by grasses, forbs, shrubs, and seedling trees. Though openings might be large and extend to the edge of streams or wetlands, in the relatively moist sites of riparian areas they typically revegetate rapidly. There is often a higher diversity and density of plants in riparian areas in this early successional stage compared to upland terrestrial sites, including broadleaved trees (such as aspen, birch and cottonwood) that benefit from the open forest conditions.

Motorized trails, travel management, and roads

Road networks have been shown to have detrimental effects on water and aquatic resources in forested landscapes when not properly maintained or built in a poor manner. Road systems can change a natural hydrologic regime by altering natural flow patterns and increasing sediment delivery to streams. Roads have been shown to destabilize side-casted material and hillsides, expand the lengths of gullies and stream channels, increase sediment delivery, and alter streamflow and channel adjustments (Furniss, Roelofs, & Yee, 1991; Quigley & Arbelbide, 1997).

Natural drainage patterns are affected long-term by the mere presence of roads. Roads intercept subsurface drainage in cut slopes, capture rainfall on hardened road surfaces, and route excess runoff into the stream channel system. These impacts increase as the road system becomes more connected, in terms of hydrology, to the natural channel network. Where a dense road network is well connected to the stream network, it can be an “extension” of the actual stream network and alter streamflow regimes. These alterations can increase the delivery of water to the mouth of a watershed during snow melts and storm events, which can increase peak flows in streams and water levels in ponds, lakes, and wetlands.

Sediment from the road system can be delivered to streams by direct erosion of cut and fill-slopes associated with stream crossings or by surface runoff from roads and ditches that carries sediment-laden water directly or indirectly to streams. In general, roads lacking surface rock, those with steep grades and steep side slopes, and those that cross streams or are in proximity to streams are the greatest contributors of sediment from surface erosion. In steep terrain, roads can increase the rate of hill slope failures and soil mass wasting. Excessive fine sediment loading can lead to changes in channel morphology and water temperature because of pool filling, widening of the channel, and making the channel shallower, which can result in water temperature increases as a result of having a shortened water column that takes less solar energy to heat. Such changes in channel morphology are typically found at road-stream crossing locations and in response to mass failures associated with road runoff. Sometimes roads capture flow out of the channel and result in the stream re-routing down the road, which typically results in road failure and more sediment delivery to streams.

Vehicular traffic also contributes to sediment delivery from roads, particularly if ruts develop in the road and if traffic is heavy during shoulder seasons when the ground is more saturated. Hauling during timber sales is typically down the same road system for weeks or months at a time, thus the quantity and repeated nature of this traffic make it a systematic, recognizable source of sediment on forest roads.

The location and design of valley bottom roads also create long-term effects on water resources. Many older roads were constructed very close to stream channel areas, often in the floodplain. Poorly placed roads can encroach on stream channel and floodplain areas. Roads can affect stream channels directly if they are located on active floodplains or directly adjacent to stream channels. Often streams were straightened to accommodate road placement. For example, a road located adjacent to a stream can be a chronic source of sediment. If the road changes the morphological characteristics of the stream, this can set forth a chain reaction of channel adjustments that can result in accelerated bed and streambank erosion, which produces excessive sediment.

Not all sediment production from roadways reaches the aquatic system. Many of the aforementioned effects of roads can be mitigated by design changes that disperse, rather than concentrate road runoff. Good design provides stable cut and fill slopes and adequate drainage that allows water to filter through vegetated strips or sediment traps before entering the stream channel. The effectiveness of vegetative strips generally increases with increased width and lower hillslope gradient. However, the effects of large-scale or chronic road impacts may still impact streams even when streams are protected by wide and intact vegetative strips.

Other design elements used to mitigate road interception and runoff are the addition of gravel surfacing and seasonal road closures. Road treatments can upgrade or remove problem culverts to allow sediment and wood to move downstream instead of accumulating upstream of roads and leading to culvert blockage and failure. However, temporary, short-term, and long-term sediment and turbidity increases can occur from project implementation, as well as from post-project stabilization.

Turbidity and sediment increases result from the construction of roads, road grading, ditch cleaning, culvert replacement, road ripping or decompaction, and the installation of water bars due to the heavy equipment excavation that these activities require. Minor amounts of fine sediment would be delivered to streams during implementation of road treatment activities and during the first substantial runoff event. Subsequent runoff events would contribute less sediment production over time but are expected to last up to one year later or until vegetation is established on bare-soil areas adjacent to streams. Design criteria and BMPs are used to minimize the amount of fine sediment entering stream channels while work is in progress and after the work is completed, including promoting vegetation establishment.

Roads that are at high risk of failure and have the potential to cause extensive resource damage are candidates for relocation or decommission. Preferred locations for roads are away from stream channels, riparian areas, steep slopes, high-erosion-hazard areas and areas of high mass movement. Realignment of roads so they traverse riparian areas and streams at perpendicular angles rather than parallel angles would improve the quality of riparian and aquatic habitats in presently impacted stream reaches by reducing chronic sediment sources. If relocation is not possible, seasonal restrictions could limit road damage and subsequent sedimentation.

The potential risk of detrimental effects exists as long as the road is retained. The continued use and existence of roadway segments that interact with stream corridors pose a risk of erosion, slope failure, and sediment delivery to receiving waters. Road obliteration reduces the long-term risk of sediment delivery to streams from roads and road-side ditches by reducing culvert failures and landslides, eliminating vehicular traffic, improving infiltration of water into the ground through decompaction of road surfaces, and reducing overland and ditch flow into streams. While some sediment is expected to be delivered to streams during culvert removal and decommissioning processes, the amount of sediment delivered to streams is expected to be significantly less than would occur if the roads were left under current maintenance. Cook and Dresser (2007) found that stream-crossings that were restored through decommissioning delivered only 3 to 5 percent of the amount of fill material that was originally located at each crossing.

Removal or closure of roads adjacent to streams can have short and long-term positive effects on soil-hydrologic function, soil productivity, and stream water temperature. Trees and other riparian vegetation can recolonize a ripped roadbed and help provide shade. The amount water or stream temperature improves depends on the existing stream shade to block solar radiation, water temperature, the stream’s size, and how much riparian road is removed or closed.

Riparian management zones

The most important change between the current 1986 plan directions for the HLC NF and the 2012 Planning Rule (36 CFR 219.8) is the requirement to establish riparian management zone (RMZ) widths. The 2012 Planning Rule directs that during plan revision efforts, RMZs shall be established in all NFS lands. The 2012 Planning Rules states that the:

> (ii) Plans must establish width(s) for RMZs around all lakes, perennial and intermittent streams, and open water wetlands, within which the plan components required by paragraph (a)(3)(i) of this section will apply, giving special attention to land and vegetation for approximately 100 feet from the edges of all perennial streams and lakes.
>
> > (A) RMZ width(s) may vary based on ecological or geomorphic factors or type of water body; and will apply unless replaced by a site-specific delineation of the riparian area.
> >
> > (B) Plan components must ensure that no management practices causing detrimental changes in water temperature or chemical composition, blockages of water courses, or deposits of sediment that seriously and adversely affect water conditions or fish habitat shall be permitted within the RMZs or the site-specific delineated riparian areas.

The Helena NF Plan had riparian areas designated west of the Continental Divide by Amendment Number 14 in 1996, i.e. the 1995 Inland Native Fish Strategy (INFISH) decision. The rest of the HLC NF does not have riparian management areas currently designated.

This section provides background information and a summary of the best available science (BASI) that was used to identify the appropriate widths for designating riparian management areas on the HLC NF.

Ecological functions and width

Regarding the widths of management areas next to streams, the interim minimum distances listed in INFISH for fish-bearing streams west of the continental divide (300 feet) and permanently flowing non-fish-bearing streams (150 feet) arguably remain the most controversial components of the existing strategies. Numerous studies that have been completed since the strategies were first published investigate how management affects the different ecological processes that are a function of riparian management zones. Though most studies were conducted for riparian habitats west of the continental divide, they were applied throughout the plan area. The ecological processes that function within riparian zones are first discussed individually below and then in combination, as they affect both aquatic and riparian conditions and biota.

After considering new science, Reeves and others (2016) proposed two options to direct management in riparian management zones in the Northwest Forest Plan area. The first option the authors considered was a “one-size-fits-all-approach” that retains the fixed buffer width where the inner 75 feet next to the stream is managed strictly to conserve aquatic function and the outer 75 feet allows ecological forestry to meet other resource objectives, including commercial harvest. The use of the term “ecological forestry” is referring to Franklin and Johnson (2012) and means that harvest retains structural and compositional elements of the pre-harvest stands, follows natural stand development principles, and applies return intervals that are consistent with disturbance regimes and that all management activities and applications are informed by landscape considerations.

The second option, described as a “context- dependent approach” by Reeves and others (2016), does not have a fixed inner width; instead, the inner width is variable and context dependent based on characteristics of the stream reach: “susceptibility to surface erosion, debris flows, thermal loading, and habitat potential for target fish species." The second option allows for natural variation and will require more analysis to inform decision-maker choices to benefit all resources. The context-dependent approach depends on landscape considerations that are expected to occur through watershed analysis. Unlike the past, when earlier attempts at watershed analysis struggled because of lack of analytical tools (Gordon H. Reeves, Williams, Burnett, & Gallo, 2006), better tools and data are now readily available (Burnett et al., 2007; Irvine et al., 2015; Isaak, Young, Nagel, Horan, & Groce, 2015; K. S. McKelvey et al., 2016). Although the options were developed for the Northwest Forest Plan Area and therefore are influenced by the conditions in that region, the underlying concepts of both options can be applied to the USFS Northern Region, including the HLC NF.

Debate remains among scientists and the public as to whether active vegetation management should occur anywhere in RMZs, even when large percentages of those zones in many areas across the West were previously managed for strictly economic purposes and no longer match distributions of conditions that would have occurred naturally. Although, the magnitude of commercial harvest in riparian zones is minor on the HLC NF (refer to the Timber section of the EIS), other activities such as mining and road building have had impacts on riparian areas in the plan area. The differing opinions between scientists makes it difficult for managers to design and implement restoration actions in RMZs (Gordon H. Reeves, Olson, et al., 2016). Pollock and Beechie (2014) urge caution when considering vegetation treatments near streams because there are many trade-offs to consider, especially for some terrestrial vertebrate species that depend on large dead wood. Their study shows that emphasizing the development of large-diameter trees via thinning to create key pieces available for streams can have negative consequences for terrestrial vertebrate species. Reeves and others (2016) discuss how tree tipping can be used to offset short-term deficiencies of woody debris in small streams and adjacent riparian areas. Rieman and others (2015) suggest that it is not clear whether the considerable funding expended to date on habitat restoration treatments has been successful. Going forward, they recommend, "(1) a scientific foundation from landscape ecology and the concept of resilience, (2) broad public support, (3) governance for collaboration and integration, and (4) a capacity for learning and adaptation" (p. 124). Monitoring and adaptive management will be essential to continually learn from and refine riparian management, including on sites where only passive management occurs.

Monitoring and research reports over the past 20 years have documented the efficacy of RMZs and their ability to protect the functional attributes for riparian and aquatic resources and water quality. Using stream temperature as a response variable, a study in Oregon found no differences before and after project using a no-cut buffer as small as 25 feet (Groom, Dent, Madsen, & Fleuret, 2011). Similarly, a comprehensive study in Oregon and Washington that evaluated various buffer widths found no increases in stream temperature using a 50 foot buffer (P. D. Anderson & Poage, 2014). The study did point out that the efficacy depended on the adjacent disturbance and contrast in forest canopy. Many researchers suggest that a 30-meter buffer next to fish-bearing and perennial streams is generally likely to be sufficient to protect against temperature increase (P. D. Anderson & Poage, 2014; Gordon H. Reeves, Pickard, et al., 2016; Sweeney & Newbold, 2014; Witt, Barton, Stringer, Kolka, & Cherry, 2016). Even so, considerations of context and geography are also appropriate. In a discussion of fixed-width riparian buffers, Richardson and others (2012) state that although these types of protections are administratively simple to implement at a reach scale, watershed considerations and location within the catchment provide additional important context.

Water temperature

Among the more commonly studied management concerns, as they relate to ecological processes near streams and determining the appropriate widths for designating riparian management areas on the HLC NF, are the effects of nearby harvest on stream temperature. Initial studies completed by Chen, Franklin, and Spies (1993) and the Forest Ecosystem Management Assessment Team (1993) found that streamside buffers of approximately 125 meters were needed to protect ecological processes such as wind speed and humidity near streams, which at the time were thought to be able to increase stream temperature. In the Pacific Northwest where Chen and others completed their studies, average site potential tree height next to streams was identified as approximately 50 meters. This finding was partially responsible for the second tree height applied to riparian reserve and riparian habitat conservation area widths in the existing strategies (Everest & Reeves, 2007; Gordon H. Reeves, Pickard, & Johnson 2013).

A study that modeled the effects of riparian reserves on stream temperature in Washington found that the first 10 meters were most important in protecting stream temperature and buffers greater than 30 meters did not appreciably lower stream temperatures (Sridhar, Sansone, LaMarche, Dubin, & Lettenmaier, 2004). A study on headwater stream microclimate by Anderson and others (2007) found that the first 10 meters had the most effect on microclimate above the stream and that temperatures in the streambed increased only when streamside vegetation closer than 50 feet was removed (P. D. Anderson & Poage, 2014). A review of studies by Moore and Wondzell (2005) suggested that a riparian reserve that was the width of one tree height was likely large enough to protect the ecological processes that control stream temperature. A subsequent study (Rykken, Chan, & Moldenke, 2007) found that stream effects helped to offset edge effects documented by Chen and others (1993). While Pollock and others (2009) did not find a correlation between recent (greater than 20 year old) streamside harvest 600 feet upstream of a monitoring site and increased stream temperature, they did find a significant relationship between basins that had greater than 25 percent harvest in the last 40 years and increased stream temperature. While the increased temperature reported (Pollock et al., 2009) was significant, it is unclear if there is a corresponding biological effect on native salmonids in the region where the studies were conducted (Gordon H. Reeves et al., 2013). For example, if a substantial rise in water temperature does not become a limiting factor in a stream reach, it may not have an effect. However, if the rise exceeds the thermal limits or growth optima of a species, negative effects would be realized.

For the past generation, many researchers suggest that a 30 meter buffer next to fish bearing and perennial streams is generally likely to be sufficient to protect against temperature increase (Gordon H. Reeves, Olson, et al., 2016; Sweeney & Newbold, 2014; Witt et al., 2016). Even so, considerations of context and geography are also appropriate. In a discussion of fixed width riparian buffers, Richardson and others (2012) stated that while these types of protections are administratively simple to implement at a reach scale, watershed considerations and location within the catchment provide additional important context. While the best available science indicates there could be some flexibility for management, this strategy does not recommend changing the widths of inner and outer riparian reserves.

Water quality: sediment and nutrients

Forest management practices such as road building and harvest have long been a concern regarding their potential to generate fine sediment and subsequent effect on water quality (Robert L. Beschta, 1978). Grazing impacts are also major sediment source as cattle can effect streambank stability and streambank vegetative protection (Clary & Kinney, 2002; Clary & Webster, 1990). Altered sediment rates have also been linked to changes in stream condition and ultimately trout and salmon survival in cold water streams (Clary & Webster, 1990; Jensen, Steel, Fullerton, & Pess, 2009). Some activities that led to degraded stream conditions and water quality, i.e. clearcutting next to streams and aggressive forest road building, are highly unlikely to occur present day on NFS lands in the Northern region. Reductions in sediment and nutrient delivery have resulted from sequentially improving BMPs (Everest & Reeves, 2007) and regional strategies that have offered greater protection (USDA, 1995a). In recent decades, researchers interested in forest management and water quality have investigated the effectiveness of management policy and law (T. C. Brown, Brown, & Binkley, 1993; Cristan, Aust, Bolding, Barrett, & Munsell, 2016; Rashin, Clishe, Loch, & Bell, 2006). In general, the latest Forest BMP reviews have found very little unnatural introductions of total suspended sediments and nutrients when BMPs are properly installed before activities begin and maintained throughout management efforts (Cristan et al., 2016; Sugden et al., 2012). Increased nitrogen levels may be an exception and may still present as elevated outside of natural conditions (Gravelle, Ice, Link, & Cook, 2009). Directions carried forward from existing strategies combined with conservation and improvement strategies discussed elsewhere in this document should help to continue improving trends.

For the water resource and quality, BASI was used to inform this FEIS. The data and reports provide background information on the current and historic water quality conditions across the HLC NF. Across the plan area, water quality monitoring in conjunction with forest project activities have been occurring since the last forest plan was developed. The HLC NF has extensive watershed monitoring programs. For more than three decades, data have been collected at monitoring sites in timber sales and other major projects. The number of years of data collection at each site has varied based on project needs. The forest used other data including various total maximum daily loads inventory and monitoring programs, the Youth Forest Monitoring Program, and monitoring done by other governmental agencies (e.g. MT DEQ, US EPA).

Vegetation management

Managing vegetation on forest lands can impair water quality by routing runoff and sediment onto bottomland stream areas. Over the last planning period, management addressed these impacts by regulating the extent of upland timber harvest, applied BMPs to limit connection from impervious surfaces, and minimized entries into riparian habitat conservation areas to provide protection from upslope activities and filter runoff. The use of these BMPs were instituted in the 1980s to control non-point source pollution (Binkley & Brown, 1993), and the riparian habitat conservation areas were established with the INFISH amendment in 1995. Using results from State of Montana audits, the FS BMPs were effective 96 percent of the time (Ziesak, 2015). Using a similar audit scheme, the FS was 100 percent effective in establishing the correct buffer to meet the State of Montana design standards for streamside management zones (SMZs).

Forest management disturbs uplands through removal of tree canopy and the yarding of the material to a central processing facility. Site preparation historically reduced groundcover by broadcast burning remaining vegetation to bare soil for planting and clear remaining fuels. The practice in the 1980s produced higher severity fire because the purposeful clearing of vegetation also removed protective groundcover. The HLC NF has largely moved away from this practice with either mechanical piling/burning or prescribed fire as primary methods for reducing hazardous fuels. A change in contemporary timber practices to whole tree yarding has further reduced remaining vegetation while preserving protective groundcover covering at least 85 percent of the area based on soil monitoring data.

Studies have documented increased sediment erosion associated with timber harvest, but the primary agent is sediment from roads (Charles H. Luce & Black, 1999; Sugden & Woods, 2007). Management controls non-point delivery of sediment in harvest areas through the use of water and soil conservation practices and BMPs (FSH 2509.22.10, R1/R4 Amendment 1) (USDA, 2012), oriented on the stabilization of log skidding and landing networks where erosion is most probable. Otherwise, forests generally have very low erosion rates with chronic erosion after disturbance lasting typically one to three years (William J. Elliot, Hall, & Scheele, 2000). After timber harvest and site preparation, regrowth of vegetation covers the soil surface with plant litter, soils armor, and potential erosion hazard becomes low (ibid).

Where prescribed fire is applied and blackens the area, the runoff can increase from reduced infiltration. Blackened soil areas can accelerate runoff due to soil sealing from ash that lowers the infiltration capacity of soils (Doerr et al., 2006). These conditions vary spatially and decrease over the first year as products of burning in the soil degrade (ibid). Natural forest conditions have hydrophobic conditions that resist infiltration when soils dry and from plant litter waxes, but the main difference is that burned areas burned areas lack surface roughness to dissipate rain splash energy and interrupt runoff. Other factors that increase runoff from harvest and burn areas are steep slopes, low groundcover, and long slope lengths (W. J. Elliot, 2013). Runoff transports loose soil particles and deposits sediment down the slope proportional to runoff energy. One reason sedimentation decreases over time is that the sediment supply decreases after bare surfaces armor, lacking a ready sediment supply. Over the past planning period, management has mitigated prescribed fire by not lighting fire within stream buffer areas and burning during cool and moist conditions that results in low and moderate severity fire.

The loss of forest canopy on harvest sites changes the water balance, and studies in the Pacific Northwest have documented cases where excess water from harvest areas influence peak and timing of stream flows (S. C. Anderson, Moore, McClure, Dulvy, & Cooper, 2015; Keppeler & Ziemer, 1990; Moore & Wondzell, 2005; Stednick, 1996). In reviews, these cases depended largely on the extent of harvest and climatic regime (Grant, Lewis, Swanson, Cissel, & McDonnell, 2008). The effect diminishes in time as vegetation re-establishes. Peak flow increases were raised as a concern from the potential to alter stream morphology and degrade water quality. The altering of streamflow can also influence stream temperature (Swanston, 1991), although the principle factor in affecting stream temperature is changes to riparian cover that shades streams (Robert L. Beschta et al., 1987; Gomi, Moore, & Dhakal, 2006; L. H. MacDonald & Stednick, 2003).

Watershed yield studies specifically targeted timber harvest activities that would generate a response and may not necessarily mimic current forest practices. Beschta and others (2000) found a weak relationship between forest harvest and increased peak flows, and reported “mixed messages” about the relationship between forest harvest and peak flow responses. Numerous studies documented the effects of forest canopy removal on peak flows in the Pacific Northwest (R. L. Beschta et al., 2000; Hubbart, Link, Gravelle, & Elliot, 2007; Jones & Grant, 1996; Kuras, Alila, & Weiler, 2012; R. F. Thomas & F., 1998; Tonina et al., 2008), but surprisingly, very few demonstrated a direct link between water yield/peak flow changes and measured channel impacts in forested environments. In the latest review for Pacific Northwest studies, Grant and others (2008) suggested that if degradation were to occur, channels most sensitive to peak flow changes are low gradient (less than 2%) with gravel bed and sand bed substrates.

The concern over changes to peak flow from timber harvest was raised when timber was harvested on a larger scale than current. The HLC NF no longer harvests timber at a rate seen in the 1970s to 1990s. Average annual harvest rates were 15 to 30 million board feet (23,525 acres) in the 80s compared to roughly 6 to 7 million board feet (4,397 acres) in 2012 and 2013 (see Timber section). In addition, many of the classic watershed studies could not disentangle the effects from roads where at least 2 percent of the study areas had roads and skidding network (Grant et al., 2008). Forest management has somewhat alleviated these effects by establishing streamside buffer zones (riparian habitat conservation areas with INFISH), reducing road construction and implementing best management practices.

Wetlands and groundwater

A key factor that determines wetland type and function is water regime. Water regime pertains to the depth, duration (hydro period), frequency, diurnal fluctuation, and seasonal timing of groundwater and surface water. A large suite of variables – not just water yield, peak flow, and base flow - have been used as “indicators” to describe hydrologic change in watersheds, streams, and rivers (Gao, Vogel, Kroll, Poff, & Olden, 2009; Konrad, Booth, & Burges, 2005; Merritt, Scott, Poff, Auble, & Lytle, 2010; N. L. Poff, 2009; N. Leroy Poff, Bledsoe, & Cuhaciyan, 2006; N. L. Poff & Zimmerman, 2010). A similarly large number could be used to characterize changes in wetlands.

In general terms, some indicator variables that apply to estimating the hydrologic effects of vegetation management on wetlands include:

- volume of water inputting to wetland (i.e., water yield of contributing area) and its timing
- peak water level or flow within the wetland: magnitude (depth or rate) and timing
- minimum water level or flow: magnitude (depth or rate) and timing
- percentage of days annually with surface water or measurable flow (both continuous and total)
- fluctuation (variance) in water level or flow: daily or annual
- percent of wetland water budget derived from groundwater vs. surface runoff vs. direct precipitation (and snow vs. rain)

Small isolated headwater wetlands are perhaps most at risk from hydrologic changes occurring in their catchments because their hydrologic inputs are usually the least. In glaciated landscapes, some wetlands that comprise only one-third of their catchment area can produce 50-70% of the annual streamflow, because wetlands often occur where groundwater intercepts the land surface (Verry, Brooks, Nichols, Ferris, & Sebestyen, 2011).

Vegetation management

Many but not all studies have shown that removal of trees near a stream or in a wetland causes a mean annual rise in the local water table (A. E. Brown, Zhang, McMahon, Western, & Vertessy, 2005; Grant et al., 2008; Guillemette, Plamondon, Prevost, & Levesque, 2005; Mallik & Teichert, 2009; L. B. Miller, McQueen, & Chapman, 1997; Moore & Wondzell, 2005; Scherer & Pike, 2003; Smerdon, Redding, & Beckers, 2009; Stednick, 1996, 2008; Charles A. Troendle, MacDonald, Luce, & Larsen, 2010; Winkler et al., 2010). As regeneration occurs in cutover areas, the previous rates and amounts of water transfer between uplands and wetlands return. This usually begins within 3-7 years post-harvest (R. L. Beschta et al., 2000) - less if the area has not been clearcut (R. B. Thomas & Megahan, 1998). Hydrologic recovery to preharvest conditions takes 10 to 20 years in some coastal watersheds but may take many decades longer in mountainous, snow-dominated catchments (Moore & Wondzell, 2005; Whitaker, Alila, Beckers, & Toews, 2002).

The probability of a harvest operation having an effect on a wetland's water regime is greatest if trees are removed directly from a wetland or, if removed from outside the wetland, the removal occurs close to and upslope from the wetland. Several other factors influence the degree to which tree removal causes water tables to rise. Especially on windy south-facing forest edges during the summer, tree roots can transfer large amounts of soil moisture to foliage and then to the atmosphere via transpiration and evaporation (Keim & Skaugset, 2003). This effectively removes some of the water before it can reach wetlands and streams. Trees also intercept significant volumes of rain and especially snow, allowing some of that retained water to evaporate before it can reach wetlands and streams located farther downslope (C. A. Troendle & King, 1987; Winkler, Spittlehouse, & Golding, 2005). Thus, when trees are removed from within or above a wetland that potential source of liquid water becomes available, the water table often rises, and the wetland may receive more water.

This has been suggested by the data from many studies of streams and watersheds in the Pacific Northwest (R. L. Beschta et al., 2000; Hetherington, 1987; Hudson, 2001; Jones & Grant, 1996; J. S. Macdonald, Beaudry, MacIsaac, & Herunter, 2003; McFarlane, 2001; R. B. Thomas & Megahan, 1998;

C. A. Troendle & Reuss, 1997). If resulting increases in peak flows are great, the morphology of channels can be affected (Grant et al., 2008). This can create, expand, or shrink wetlands. Depending on the soils and topography, the slash burning and soil compaction components of some harvest operations provide additional surface runoff to wetlands, at least during a few years post-harvest (Lamontagne, Schiff, & Elgood, 2000). In addition, in snow-affected areas, clear-cuts have sometimes been shown to cause greater runoff during rain-on-snow events (Berris & Harr, 1987) and earlier peaking of streamflow (or wetland water levels).

On the other hand, harvest might measurably reduce runoff to streams and wetlands in some parts of the Pacific Northwest during low runoff periods, partly by temporarily eliminating trees that otherwise contribute water by intercepting fog (R. D. Harr, 1982; R. Dennis Harr, 1983). During the autumn, streams in clearcut watersheds in the Pacific Northwest tend to have lower flows than in uncut watersheds (R. D. Harr, Harper, Krygier, & Hsieh, 1975). Also, cutting or windthrow of trees in or near wetlands can increase open-water evaporation sufficiently to reduce water persistence in late summer (Petrone, Silins, & Devito, 2007), especially in larger wetlands and/or in drier parts of the Pacific Northwest.

Livestock grazing

In some cases, grazing has the potential to damage springs and other types of groundwater dependent wetland habitats. These off-channel aquatic features have incredibly high biodiversity and serve important ecosystem functions. They are also attractive to livestock as they offer palatable browse and flat, cool resting spots. If not properly managed, this conflict can lead to water quality issues, damaged organic soils, and reduced wildlife habitat. Impacts to these areas are commonly noticeable earlier in the grazing season than most other types of sites within pastures. Actions in response to this use pattern has typically been to fence-off these features when damage has been repeatedly noted. This is effective as long as fences can be consistently maintained. Maintenance failure can result in higher levels of damage as cattle may remain there longer as they move further away from the point of entry, limiting access to outside the exclosure. Adaptive management has been implemented in some areas to resolve localized issues.

Riparian dependent terrestrial species

BASI, since the Strategies were published, has sharpened focus on aquatic/riparian interactions. One review found that buffers wider than 30 meters are large enough to protect water quality and aquatic biota in small streams (Sweeney & Newbold, 2014). In some circumstances, such as a narrow band of riparian dependent vegetation alongside an intermittent stream that has low connectivity, these characteristics could lead to a reduced width for the RMZ if only aquatic functions are being considered. However, RMZs have had increasing focus applied to their ability to support terrestrial organisms and processes. Starting as far back as the Forest Ecosystem Management Team (1993), "Protection of riparian-associated terrestrial organisms has become an explicit conservation objective associated with protection of streams (Richardson et al., 2012)." Numerous studies have published research on riparian use by species from invertebrates (Bunnell & Houde, 2010) to amphibians (Olson & Burton, 2014) and from mammals (Kevin S. McKelvey & Buotte, in press; Wilk, Raphael, Nations, & Ricklefs, 2010) to avifauna (Lehmkuhl et al., 2007) (T. A. Spies, Stine, Gravenmier, Long, & Reilly, 2018).

Science published on wildlife use of the riparian area is more varied and subsequently more complicated. In a literature review considering appropriate widths for RMZs, Wenger (1999) found that buffer distances reported to protect terrestrial wildlife ranged from as little a few feet to over 1000 feet. A distance of 300 feet was recommended for most wildlife acknowledging that the distance might be difficult to implement in all management applications. Lee and others (2004) completed a literature review of management prescriptions next to water bodies in both Canada and the U.S. They found that while prescriptions for buffer widths varied by water type such as wetlands, intermittent streams, and fish bearing streams, they were generally wide enough to protect many of the important riparian processes that support aquatic biota. However, buffers were generally less than recommended widths to protect terrestrial fauna. Marczak and others (2010) found that for buffers less than 50m wide, responses by different taxa became more variable as compared to untreated riparian areas. They also found that taxa did not respond similarly to riparian treatments; edge related species increased in abundance or diversity while some interior associated species declined. Some species presence and abundance remained unchanged. Ultimately, they found that current buffers do not retain terrestrial fauna at levels comparable to unmanaged sites for all taxa. They offered that sometimes upland terrestrial vegetation might need to be combined with the protections that come with RMZs for some sensitive terrestrial species (Semlitsch and Bodie, 2003). They concluded that increases in protections in some locations should be balanced with some riparian areas allowing partial resource extraction (Marczak et al., 2010; Gordon H. Reeves, Pickard, et al., 2016).

Riparian ecosystems are equally important habitat to wildlife for feeding, drinking, cover, breeding seasonal habitat, and habitat connectivity. They are often rich in bear foods such as skunk cabbage and other herbaceous plants with nutritious bulbs. Many wildlife species are associated with riparian ecological systems, including beaver, Canada lynx, grizzly bear, harlequin ducks, and mink. Upland vegetation within riparian areas in combination with the riparian vegetation create zones that provide important wildlife habitat and connectivity values. Most wildlife use RMZs and/or aquatic habitats for at least some of their daily or seasonal needs. Due to their widespread distribution and linear or clustered pattern, they provide extensive and important habitat connectivity areas for numerous species of wildlife. Refer to wildlife section for information on riparian associated wildlife species and habitat connectivity.

During the past few decades, land managers have recognized the importance of riparian ecosystems in maintaining water quality, terrestrial, and aquatic habitat. As a result, riparian conservation measures have been developed for federal, state, and private lands – helping to preserve and protect the integrity of the riparian and wetland habitats, as well as the water quality of associated waterbodies.

Fisheries and other aquatic species

Around the beginning of the 20th century, the influx of human populations began along with the development of the land and resources to support those populations. This has resulted in many new human-caused disturbances to the watershed systems, and the pattern of many of those disturbances has tended to be a more sustained or “press” disturbance regime. A press disturbance forces an ecosystem to a different domain or set of conditions (G. H. Reeves, Benda, Burnett, Bisson, & Sedell, 1995). Many of those disturbances tend to mimic historic “natural” processes (i.e. livestock grazing and American Bison), but the frequency and intensity has been greatly amplified. In some cases, the watershed systems have begun to adjust to those press disturbances; or have become altered by them, resulting in an inability to support aquatic dependent resources.

Stream habitat degradation in the western U.S. became a great concern in the early 1990s, as well as the potential the loss of salmon, trout, and char populations (Nehlsen et al., 1991; Rieman & McIntyre, 1993). By the mid 1990’s, The FS and BLM completed three broad reaching documents that amended forest plans across much of public lands in the west to improve their conservation function. Two of those documents were: *Record of Decision for Amendments to Forest Service and Bureau of Land Management Land Planning Documents Within the Range of the Northern Spotted Owl* (often referred to as Northwest Forest Plan Record of Decision, 1994); and the *Decision Notice/Decision Record for Interim Strategies for Managing Anadromous Fish-Producing Watersheds on Federal Lands in Eastern Oregon and Washington, Idaho and Portions of California* (USDA, 1995a). Both documents greatly improved protection for migratory salmon and steelhead. These documents influenced the development of the last of the three broad strategies developed, which was the *Inland Native Fish Strategy-Interim Strategies* (INFISH) *for Managing Fish-Producing Watersheds in Eastern Oregon and Washington, Idaho, Western Montana and Portions of Nevada* (USDA, 1995b). While INFISH was originally expected to last 18 months to three years while an effort similar to the Northwest Forest Plan, the Interior Columbia Basin Ecosystem Management Project (Frissell et al., 2014), was completed for the Interior Columbia River Basin. That strategy was never completed, but science from that effort has been retained in the form of guidance for plan revisions occurring in areas covered by INFISH and PACFISH. Interior Columbia Basin Ecosystem Management Project science and guidance is followed in the 2020 Forest Plan.

INFISH was designed to maintain options for inland native fish by reducing negative impacts to aquatic habitat. Riparian management objectives, standards and guides, and monitoring requirements were implemented beginning in 1995 to avoid causing further damage and begin recovery of aquatic habitats. The 1986 Helena NF Plan was amended by INFISH in 1996. This strategy is still in effect west of the continental divide on portions of the Divide and all of the Upper Blackfoot GAs of the HLC NF. The INFISH strategy does not apply to those GAs of the HLC NF east of the continental divide.

Since INFISH was implemented, there have been numerous changes to policy, BASI, and the condition of listed species. There have been tremendous advances in knowledge regarding physical habitat and ecological interactions at many scales and across scientific disciplines, as well as advances in spatial database management. Scientists findings disclosed in BASI urge managers and biologists working to maintain and improve aquatic habitat to look beyond just the stream reach when considering how best to plan and implement project activities. Climate change science has also emerged as an important aspect of forest and river management since INFISH was adopted.

When instituted, riparian management objectives were considered by many to be an important component of INFISH. Riparian management objectives were developed from PACFISH objectives measured in habitats across the range of anadromous fish in Washington, Oregon, and Idaho. The objectives selected were considered good indicators of ecosystem health, and were thought to be, "a good starting point to describe the desired condition for fish habitat." (INFISH, p. E-3, 1995- emphasis added). INFISH guidance recommended that riparian management objective values should "be refined to better represent conditions that are attainable in a specific watershed or stream reach based upon local geology, topography, climate and potential vegetation" (USDA, 1995b). Since INFISH was adopted on the Helena NF west of the continental divide, data has been collected and used for comparison purposes in project design, consultation, and monitoring. As indicated in INFISH, the riparian management objectives of pool frequency, width/depth ratio, and supporting feature water temperature categories are applicable to all systems, and large woody debris to forested systems, while bank stability and lower bank angle may apply more to nonforested habitat areas of the HLC NF where specific land uses may affect these habitat features. INFISH did not provide any sediment indicators as riparian management objectives.

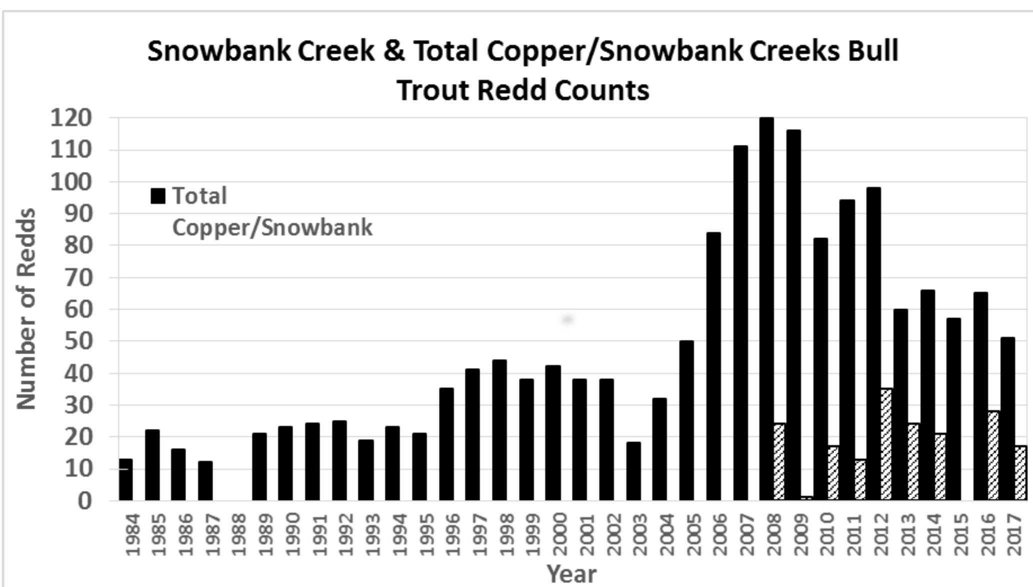
Several factors have contributed to the decline of bull trout. Habitat degradation, interaction with exotic species, over harvesting, and fragmentation of habitat by dams and diversions, are all factors contributing to the decline (Rieman & McIntyre, 1995). Historically, bull trout populations were distributed throughout the core areas and in larger tributaries and were in higher densities than they are today.

Substantial impacts to the population were likely related to water rights and water diversions, overgrazing, and clearing stream riparian areas. Use of surface waters required diversions, which were not usually screened, led to the entrainment of various age classes of aquatic species. In addition to unscreened diversions, the withdrawal of water from the stream diminished the ability to provide adequate habitat for aquatic species. Clearing of riparian shrubs and damage to streambanks by overgrazing also caused impacts to streams geomorphology making them wider and warmer. Eroding banks introduced higher amounts of sediment than could be transported by the streams, which exacerbated stream morphology problems and reduced fish spawning success. Significant timber harvest and road building was also taking place. These activities led to additional increases in fish barriers following the installation of undersized culverts. These barriers resulted in increasing sediment delivery and stream temperatures, as well as other water quality impacts. Many of these impacts have been reduced or eliminated, decreasing some stressors on the population so they no longer play as large a role as they historically did. For instance, fish barriers have been identified as a significant impact and multiple agencies and partners have engaged in removing or upgrading culverts. Also, connectivity has been improved by removing or mitigating local barriers such as irrigation diversion structures on and off forest lands to provide fish passage.

Bull trout spawning occurs in the fall, and the eggs incubate in the stream gravel until hatching in January (Fraley & Shepard, 1989). The alevins remain in the gravel for several more months and emerge as fry in early spring. Unlike many anadromous salmonids, which spawn once and die, bull trout are capable of multi-year spawning (ibid). The historic range of bull trout stretched from California, where the species is now extinct, to the Yukon Territory of Canada (Haas & McPhail, 1991).

Bull trout populations in the Little Blackfoot drainage currently appear to be resident populations. The population trend for spawning bull trout in the primary spawning tributaries (Blackfoot Core Area, Copper and Snowbank Creeks) on the HLC NF shows an overall increasing trend between 1984 and 2017. Total redd counts in the two streams showed the highest numbers counts in 2008 and have decreased since then but still remain above 1984-2005 levels. A critical issue with this monitoring data is the short time scale. In addition to the Index Section initiated in 1984, observations of substantial spawning occurring upstream led to adding the Upper Section on Copper Creek in 1996. Following work that reconnected Snowbank and Copper creeks, a redd count section was established on Snowbank Creek in 2008. Furthermore, the additional downstream section of Copper Creek on NFS lands has been examined to determine if spawning habitat is present. The trends in the number of redds over the past 32 years shows an increase following the 2003 Snow-Talon Fire in the Copper/Snowbank Creeks drainage (Figure 1). As funding allows, the Forest expects to continue to collaborate with MTDFWP and USFWS on completing bull trout redd count surveys.

Figure 1. Total Snowbank and Copper Creeks bull trout redd counts 1984-2017



Historically, bull trout numbers were likely at much higher levels and we assume that the population is still well below its potential. Currently, the main factor limiting recovery of bull trout in the Blackfoot is thought to be the lack of high quality tributaries throughout the watershed (USDA-USFWS, 2013). However, it is unlikely that this impact is entirely responsible for the overall decline. Numerous other impacts have contributed to the decline, including inadvertent angling-related mortality, warming water temperatures, anthropogenic sediment delivery, non-native fish competition and hybridization. Future concerns are anticipated to be associated with the protection of instream flows in an era of increasing human consumption of surface and groundwater since these factors can have profound effects on the habitat requirements of bull trout and connectivity for the migratory fluvial form.

Aquatic invasive species

Nonnative invasive species are a serious threat to all aquatic habitats in the U.S. The severity of this threat is difficult to assess or predict in this plan area, or in any other specific locality. Virtually every biological lifeform has been a documented agent in disruptive outbreaks in North America. These lifeforms cover the range from viruses to mammals. Included in documented losses to ecologic integrity and beneficial uses are vegetative lifeforms that range from single-cell algae to vascular plants such as nonnative trees.

The ecological and economic impacts of invasions vary greatly in scale. Effects from invasive species also vary with local environmental dynamics and complexity of the ecosystem. For example, whirling disease (*Myxobolus cerebralis*) appears to have produced major changes in the assemblage of fish species in some Montana rivers but not in others. Where ecological disruptions have been noted, there were lost recreational opportunities and revenues for the tourism, outfitting, and related industries. The intensity of effects has been less pervasive, suggesting the conditions and habitat to complete the life cycle requiring two hosts have proven to date to be less suitable than in other waters and/or resistance exists in the salmonid populations.

When a new aquatic invasive species invasion occurs in a locality, it often requires research and observation time before reliable inferences can be made regarding spread patterns, specific effects, and potential containment strategies. A baseline often is lacking to predict how an invasive species from another region or continent will respond when introduced into a new environment. Since a local environment contains a unique assemblage of thousands of interconnected components and processes, the results in one area can vary slightly or significantly from previously infected areas.

Prevention of invasions is of paramount importance in land and natural resource management. If an aquatic invasive species becomes established, elimination may be nearly impossible and efforts for containment can be very difficult, time consuming, and expensive. This involves recognizing the vectors for infection and spread and implementing safeguards, or resource protection measures, to minimize and prevent the transmission of invasive organisms through these pathways. An example of a transmission vector would be heavy equipment, pumps or mineral exploration equipment that come into contact with water. This equipment can be highly mobile from drainage to drainage with some exploration equipment even transported globally between projects, allowing microbes, spores, planktonic larval and adult stages, and plant materials to be easily be spread. Effective sanitation and inspection measures are essential resource protection measures.

Spread and introduction vectors are inherent to most projects and types of forest use. Thus, components of the 2020 Forest Plan require mechanisms for addressing aquatic invasive species. More general or universal objectives and procedures, such as using current BMPs for equipment washing before and after entering an area, are recommended for inclusion. High risk activities within individual resource areas are likely best addressed in resource-specific sections. This better assures that these components are included as resource protection measures at the project level. These activities would include but aren’t limited to;

transporting water across drainage boundaries for fire suppression, constructing stream fords, operating equipment in a riparian area and near a water course, and the use of pumps and sumps for mineral exploration, fire suppression, or construction related dewatering activities.

Conservation watershed network

The BASI indicates the HLC NF is and will be important for conservation of native bull and westslope cutthroat trout within their range. The plan area is located along both sides of the continental divide and it is predicted the plan area would provide cold water refugia into the future due to the effects of climate change being slower in high elevation mountain streams.

Climate shield and temperature models across level 6th hydrologic unit codes in the plan area looked closely at where cold water is predicted to persist into the future in the face of climate change. The models identified that cold water is predicted to persist in the sub-watershed in the Blackfoot GA that was identified as a priority watershed under INFISH. HLC NF priority bull trout and westslope cutthroat trout occupied watersheds and designated critical habitat by the USFWS are included in the HLC NF conservation watershed network. Isaak and others (2015) identified bull trout and westslope cutthroat trout probabilities of persistence into the future under different climate warming scenarios as well as cold water refugia. The Climate Shield Model (Isaak et al., 2015) was used as a starting point to identify watershed with cold water that may persist into the future.

At the broadest of scale considerations, information in USFWS bull trout recovery plan was reviewed to help place habitat and core populations located within the HLC NF in context with recovery needs of the species across its range in the western U.S. For recovery units like the Columbia Headwaters, the plan strategy states, “A viable recovery unit should demonstrate that the three primary principles of biodiversity have been met: representation (conserving the breadth of the genetic makeup of the species to conserve its adaptive capabilities); resilience (ensuring that each population is sufficiently large to withstand stochastic events); and redundancy (ensuring a sufficient number of populations to provide a margin of safety for the species to withstand catastrophic events).”

Soils

Overview

Soil monitoring data on the HLC NF have demonstrated that allowing ground-based equipment only on slopes below 35 % maintains the level of detrimental soil disturbance below the regional threshold (U.S. Department of Agriculture, Forest Service,, 2017).

Coarse wood debris in the form of slash can provide a practical and effective mitigation for reducing harvest impacts on soil physical function and processes ((R. T. Graham et al., 1994; Harvey, Jurgensen, & Graham, 1989). Leaving harvest slash along skid trails can prevent compaction (Han, Han, Page-Dumroese, & Johnson, 2009) and enhance soil recovery (D. S. Page-Dumroese, Jurgensen, & Terry, 2010). Coarse wood debris contains very little nutrient value (Laiho & Prescott, 1999), but its function as groundcover and tempering soil climate promotes soil biologic activity. Target coarse wood levels balance needs for fuels reduction, soil production and wildlife. Optimal ranges for Montana and Idaho forests were reported as 5 to 20 tons per acre for warm sites and 10 to 30 tons per acre for cooler sites (J. K. Brown, Reinhardt, & Kramer, 2003). Any benefits from road decommissioning will depend largely on site potential for recovery (Switalski, Bissonette, DeLuca, Luce, & Madej, 2004). Road treatments will stabilize the surface from erosion, while soil biology, soil chemical and hydrologic properties slowly recover as plants recolonize. Lloyd and others (2013) quantified road recovery on the Nez Perce –

Clearwater NF, showing faster soil recovery for treated roads where the road prism was outsloped along with some level of revegetation versus abandoned roads. It was found that topsoil developed on treated roads more readily than topsoil on roads abandoned for nearly thirty years.

Adequate canopy and groundcover are the best protections against soil erosion. Overland flow and surface erosion are rare in Rocky Mountain forests (Wondzell & King, 2003). Based on the disturbed Water Erosion Prediction Project model, a soil erosion model amended for forested environments, soil erosion rarely occurs if groundcover exceeds 85% cover (William J. Elliot, Page-Dumroese, & Robichaud, 1999).

The steep topography of the HLC NF naturally predisposes slopes to erosion after wildfires. Erosion caused by intense rainfall following these fires will continue as a natural geomorphology agent as it has occurred episodically in Rocky Mountain forests for millennium (Kirchner et al., 2001; D. Miller, Luce, & Benda, 2003). When taking a closer look over a century scale, fire incidence coincides with warm phases of the Pacific Decadal Oscillation (Morgan, Heyerdahl, & Gibson, 2008). This latest warm cycle has continued with periods of dry springs and hot summers. These conditions align with large scale fire patterns based on tree-ring research (Gray & McCabe, 2010; Madany, Swetnam, & West, 1982). Climate change predictions suggest a continued increase in monthly temperatures along with longer periods of drought that increase the wildfire hazard.

In regards to prescribed burning and wildfires, across blackened areas, the net effect of the burn residue and surface sealing of soil pores can exacerbate erosion potential by slowing infiltration (Larsen et al., 2009; Wondzell & King, 2003). This post burn condition is highly variable spatially and decreases over time (Doerr et al., 2006). One benefit of fuels treatments is that it re-introduces fire into the system. Burning creates a net increase in available nutrients, both in terms of the products of fire contained in ash residue and the higher decomposition rates after the fire. Almost immediately, burning increases the amount of available nitrogen for plants and soil biota (Choromanska & DeLuca, 2002; Hart, DeLuca, Newman, MacKenzie, & Boyle, 2005). In drier habitats, this increase can be detected as much as 50 years after a fire event (McKenzie, Gedalof, Peterson, & Mote, 2004). The burning also produces charcoal production that enhances conditions soil by increasing water holding capacity and providing exchange sites (DeLuca & Aplet, 2008).

Ability of soil to maintain ecological functions

FSM Chapter 2550 Soil Management identifies six soil functions: soil biology, soil hydrology, nutrient cycling, carbon storage, soil stability and support, and filtering and buffering. Soil is the foundation of the ecosystem; in order to provide multiple uses and ecosystem services in perpetuity, these soil functions need to be active.

The soil biology attributes of note on the Forest are roots and aeration, plant community potential, and thermodynamics. Little information currently exists on the trends of soil biology. It is likely that severe or frequent burns (natural or prescribed) reduce the diversity of the soil biota by reducing the soil organic matter required to support the biota. Similarly, erosion may reduce soil biota diversity. Climate change will likely change the soil biota due to increased accumulation and decomposition of organic matter and changes in soil temperature and moisture. The climate change effects are site specific. Invasive species cover may also reduce soil biota diversity.

Soil hydrology is the ability of the soil to absorb, store, and transmit water both vertically and horizontally. Soil hydrology is extremely important on the Forests because the ecosystem productivity is typically limited by water. Soil can regulate the drainage, flow, and storage of water and solutes, including nitrogen, phosphorus, pesticides and other nutrients and compounds dissolved in water. When properly functioning, soil partitions water for groundwater recharge and use by plants and animals. Changes in soil bulk density, soil chemistry, soil structure, soil pores, and ground cover can alter soil hydrology. The main impacts to soil hydrology on the Forest are compaction, erosion, loss of vegetation cover, and hydrophobicity from severe burns. Interception by roads also affects soil hydrology. The historic soil impacts from past activities have affected soil hydrology especially in areas where road densities are high.

Nutrient cycling is the movement and exchange of organic and inorganic matter back into the production of living matter. Soil stores, moderates the release of, and cycles nutrients and other elements. During these biogeochemical processes, analogous to the water cycle, nutrients can be transformed into plant available forms, held in the soil, or even lost to the atmosphere or water bodies. Soil is the major 'switching yard' for the global cycles or carbon, water, and nutrients. Carbon, nitrogen, phosphorus, and many other nutrients are stored, transformed, and cycled through the soil. Decomposition by soil organisms is at the center of the transformation and cycling of nutrients through the environment. Decomposition liberates carbon and nutrients from the complex material making up life forms and puts them back into biological circulation, so they are available to plants and other organisms. Decomposition also degrades compounds in soil that would be pollutants if they entered ground or surface water. Nutrient cycling can be assessed by considering organic matter composition on a site and the nutrient availability. The major impacts to nutrient cycling are compaction and loss of organic matter and topsoil.

Nearly all nitrogen in forest systems is bound to organic matter. Very little of the total pool of nitrogen is available to plants; only about 2.5 percent of total organic nitrogen is released annually (Grigal & Vance, 2000). The rate of nitrogen release from organic matter (a process called mineralization) is controlled by microbial decomposition, which in turn is controlled by environmental factors as well as the amount and chemical composition of organic matter (Drury, Voroney, & Beauchamp, 1991; Grigal & Vance, 2000). Rates of mineralization are highly spatially variable within stands (Campbell & Gower, 2000). The availability of nitrogen from organic matter has been said to 'most often limit the productivity of temperate forests' (Hassett & Zak, 2005). Logging residues are a source of nitrogen during early periods of stand growth after harvest (Hyvonen, Olsson, Lundkvist, & Staaf, 2000; Malkonen, 1976). Dead woody material left after logging provides carbon-rich material for microbes to feed upon; and typically microbial populations increase after forest harvests due to the input of logging residues. When logging residue is removed for fuels management and/or site prep microbial populations may decrease.

Carbon storage is the ability of the soil to store carbon. The carbon cycle illustrates the role of soil in cycling nutrients through the environment. More carbon is stored in soil than in the atmosphere and above-ground biomass combined. Compaction and loss of organic matter and topsoil can be assumed to affect carbon storage.

Soil structure and support gives soil the ability to maintain its porous structure to allow passage of air and water, withstand erosive forces, and provide a medium for plant roots. Soils also provide anchoring support for human structures and protect archeological treasures. Soil support is necessary to anchor plants and buildings. Both flexible (it can be dug) and stable (it can withstand wind and water erosion), soil also provides valuable long-term storage options including protecting archeological treasures and land-filling human garbage. The need for structural support can conflict with other soil uses. For example, soil compaction may be desirable under roads and houses, but can be devastating for the plants growing nearby.

Soil acts as a filter to protect the quality of water, air, and other resources. Toxic compounds or excess nutrients can be degraded or otherwise made unavailable to plants and animals. The minerals and microbes in soil are responsible for filtering, buffering, degrading, immobilizing, and detoxifying organic and inorganic materials, including industrial and municipal by-products and atmospheric deposits. Soil absorbs contaminants from both water and air. Some of these compounds are degraded by microorganisms in the soil. Others are held safely in place in the soil, preventing contamination of air and water. When the soil system is overloaded, such as with the excess application of fertilizer or manure, or when the soil is unstable, some contaminants will be released back to the air and water through erosion or leaching.

Soil impairments and disturbances

Land-use and forest practices have affected soil functions, and these functions are intertwined, making it difficult to discuss them separately. Management action such as timber activities, livestock grazing, road management, fuels management, and recreation can all have effects such as compaction, erosion, and loss of organic matter, and can impair the majority of soil functions. While these effects have not been eliminated in current practices, the FS has decreased these types of effects. This reduction, coupled with soil restoration activities, should result in a sustainable or possibly even increased capacity of the soils to support multiple uses and ecosystem services.

Harvesting timber requires machinery to cut and yard trees to landings sites that can compact and displace soils (Cambi, Certini, Neri, & Marchi, 2015; D. S. Page-Dumroese, M. F. Jurgensen, et al., 2010). Intensity and extent of impacts are managed by project mitigation and best management practices. Using soil monitoring, the FS evaluates the efficacy of forest treatments by comparing disturbance extent against soil quality thresholds. When soil disturbance surpasses these thresholds, long-term impairment could occur and the disturbance is considered detrimental to soil quality. Soil surveys have found ground-based harvest and skidding methods have resulted in the highest disturbance levels (U.S. Department of Agriculture, Forest Service,, 2017). Contemporary methods have reduced impacts with lower pressure, wider track or tread equipment, although economics and advances in mechanization have driven operators to favor ground-based equipment. Forest monitoring has not found forest treatment intensity to equate to disturbance, because skid trails are a far greater disturbance factor than the degree of tree removal. Soil compaction largely occurs after only three passes by equipment and most pronounced on skid trails (Han et al., 2009; Williamson & Neilsen, 2000). Because the same skid trail networks are used for both thinning and regeneration type harvests they have near equal rates of soil disturbance (Milner, 2015).

Fire impacts soils by consuming organic matter and producing surface conditions prone to soil erosion. The impact is described qualitatively as soil burn severity which conveys the magnitude of energy released from the consumption of fuels and the duration of heating. When fires burn all the above ground biomass and forest floor, a large portion of the nutrient supply is volatized into the atmosphere (Erickson & White, 2008; Neary, Klopatek, DeBano, & Ffolliott, 1999). The inherent soil quality may remain intact after wildfire since wind driven fire rarely heats deep into soil (Hartford & Frandsen, 1992). However, after the wildfire, the lack of forest canopy and bare soil creates conditions suitable for erosion. Water and wind erosion transport and deposit soil material incrementally downslope until slopes stabilize. Erosion is highest where fires burn severely on steep hillsides. Though natural, recovery in these areas depends on available moisture and recolonization from neighboring vegetation and soil patches.

The potential impacts of anthropogenic climate change on the forest soil resource are not well known at this time. Warmer, more moist winters may result in large areas of reduced capability for winter harvest operations; a common soil protection practice on the HLC NF. Increased frequency and severity of summer droughts could threaten effective vegetation cover through increased wildfire, and pathogen and insect activity. Literature suggests that opportunities may exist to manage the soil carbon pool (Harmon & Marks, 2002; Johnson & Curtis, 2001). However, predicted soil carbon response to anthropogenic climate change is extremely uncertain at this time (Friedlingstein et al., 2006; Todd-Brown et al., 2013).

Soil has the ability to either store or release greenhouse gases; thereby, potentially influencing climate change. More carbon is stored in soil than in the atmosphere and above-ground biomass combined (Yanai, Currie, & Goodale, 2003). Soil carbon is in the form of organic compounds created through photosynthesis in which plants convert atmospheric carbon dioxide into organic carbon compounds. The organic compounds enter the soil system when plants and animals die. Immediately, soil organisms begin consuming the organic matter, releasing water, heat, and carbon dioxide back to the atmosphere. Thus, if no new plant residue is added to the soil, soil organic matter will gradually disappear. If plant residue is added to the soil at a faster rate than soil organisms convert it to carbon dioxide, carbon will gradually be removed from the atmosphere and stored (sequestered) in the soil. Some forms of soil carbon are very stable and will persist for long periods. It is unknown at this time as to how forest practices affect soil carbon storage, although research is on-going.

Current findings from the FS Long Term Soil Productivity study suggest that the extent of impacts to soil relate to texture and organic matter (D. Page-Dumroese, D. Neary, & C. Trettin, 2010; Powers et al., 2005) but often as confounding variables. For example coarse textured soils appear resistant to compaction (Gomez, Powers, Singer, & Horwath, 2002), but also nutrient poor and at risk to the nominally least risky treatments that remove forest floor (D. Page-Dumroese et al., 2006; D. S. Page-Dumroese, D. Neary, & C. Trettin, Eds., 2010). Forestry research has underscored the importance of organic matter documenting the soil benefits of downed wood (Russell T. Graham et al., 1994; Harvey et al., 1989), forest floor and soil organic matter (Jurgensen et al., 1997). The Rocky Mountain Research Station has responded by initiating studies to establish minimal necessary amounts of organic matter by habitat type. The forest floor can act as a mulch and buffers the soil microclimate to hold water on warmer, less moist sites for soil and plant processing in addition to providing a nutrient cache. Colder, moist sites would not have the same water issues and thus adequate forest floor can be less constraining for growth.

Future climate and fire influences on aquatic ecosystems

Over the last 50 years, average spring snowpack (April 1 snow water equivalent) has declined and average snowmelt runoff is occurring on average 15 days earlier in the spring where expected future changes could be as much as 20 to 40 days earlier in many streams (Stewart, Cayan, & Dettinger, 2004). These trends are observed for northwestern Montana, the entire Pacific Northwest, and much of the western U.S. Several recent studies of the same trends across the entire western U.S. have concluded that natural variability explains some, but not all, of the west-wide trend in decreasing spring snowpack and earlier snowmelt runoff.

Shifts in climate could play out mostly in mid elevation forests where winter moisture comes as rain rather than snow, and where a decrease in snowpack could result in prolonged periods of soil moisture deficit. It is likely this would continue the trend of earlier spring, as much as two months over the next century (Charles H. Luce, in press).

A decrease of snowpack could extend soil drought to the mid elevations that is now common to low elevation ponderosa pine forests. The seasonal water deficits could stress mesic species such as lodgepole and sub-alpine fir that make up the mixed conifer forests. It is possible drought stress would affect mid elevation forests even more because forest species shift will occur according to aspect in this zone. Concave slope areas would grow mesic species since these areas have moist deep soils from converging slope water. The upper extent of the timber line would likely move up in elevation as the growing season extends in these normally cold limited environments.

Impacts to streams and riparian areas

Fire and changing conditions on the landscape that result from a warming climate must be kept in mind when considering riparian management needs (Dwire, Meyer, Riegel, & Burton, 2016; Joyce, Talbert, Sharp, Morissette, & Stevenson, in press; C. Luce et al., 2012; Gordon H. Reeves, Olson, et al., 2016). When considered by subregion, model runs in the Northern Region show that averaged temperatures would continue to become warmer during the first half of the 21st century (Joyce et al., in press). Some locations in the region are expected to become drier and have more periods of drought; while overall, precipitation is expected to range from 5% less to an increase of up to 25%, with a mean increase expected to be 6 to 8% (ibid). Climate is expected to reduce stream flows (C. H. Luce & Holden, 2009), reduce the storage capacity associated with snowpack (C. Luce et al., 2014b), and shift the timing of run-off in some locations (C. Luce et al., 2012; C. Luce et al., 2014a).

Climatic changes are expected to differentially affect tree species and their distribution on the landscape, as well as some of the pathogens that act upon them (Keane et al., in press). There is also significant concern that climate change effects combined with altered disturbance regimes caused by fire suppression would change ecosystems (Hessburg, Agee, & Franklin, 2005; C. Luce et al., 2012). Finally, climate change may create conditions heretofore not observed and cause ecosystems to shift in novel ways (C. Luce et al., 2012; Gordon H. Reeves, Olson, et al., 2016). These changes include how riparian areas respond to potentially novel disturbance regimes (Dwire et al., 2016; Hessburg et al., 2015; Gordon H. Reeves, Pickard, et al., 2016). How land managers prepare and respond becomes ever more crucial.

The relation of fire behavior between riparian areas and adjacent uplands is influenced by a variety of factors, contributing to high spatial variation in fire effects to riparian areas. Landform features, including broad valley bottoms and headwalls, appear to act as fire refugia (Camp, Oliver, Hessburg, & Everett, 1997). Biophysical processes within a riparian area, such as climate regime, vegetation composition, and fuel accumulation are often distinct from upland conditions (Dwire & Kauffman, 2003). This can be especially true for understory conditions (Halofsky & Hibbs, 2008). Riparian areas experiencing moderate annual climate conditions can have higher humidity and can act as a buffer against fire and therefore as a refuge for fire-sensitive species (ibid). Some studies have found fire typically occurs less frequently in riparian areas (Dwire et al., 2016; Russell & Mcbride, 2001).

Depending on geologic and topographic features, riparian conditions and response to fire vary (Halofsky & Hibbs, 2008). A study in mixed severity conifer stands in the Sierra Nevada found that riparian and upland conditions are similar and consequently fire effects are similar (Van de Water & North, 2010). Under severe fire weather conditions and high fuel accumulation, riparian zones may become corridors for fire movement (Pettit & Naiman, 2007). Fire effects occurring upstream will likely influence downstream conditions (Wipfli, Richardson, & Naiman, 2007), as well as future fire behavior (Pettit & Naiman, 2007). Effects of high severity fire on aquatic systems will likely have short term negative affects at the reach scale but beneficial effects over time at that same scale as recolonization naturally occurs (Gresswell, 1999). At a watershed scale, fire effects for one life history phase can be negative, while in the same watershed, the fire effects will be beneficial for another life history phase (Flitcroft et al., 2016). Considering these varied conditions that occur from the stream edge to upslope and from river mouth to mountaintop, riparian response to fire is complex and heterogeneous and therefore requires considerate effort to design treatment plans that maximize benefits for both terrestrial and aquatic dependent species.

Restoration treatments in riparian areas

In the face of larger fires and disease outbreaks, the challenge of how to integrate management of aquatic and terrestrial resources has now confronted the agency for over a generation, including the Northern

Region. Rieman and others spoke directly to this perception and identified opportunities for convergence. (Rieman, Lee, Thurow, Hessburg, & Sedell, 2000), as have many others since (Hessburg et al., 2015; Gordon H. Reeves, Olson, et al., 2016; Gordon H. Reeves, Pickard, et al., 2016; Rieman, Hessburg, Luce, & Dare, 2010). Current habitat has been degraded in many dry and mesic forests, and treatments (such as road improvement or relocation, culvert replacement, thinning, prescribed fire and wildfire use to restore old forest structure) could create more suitable aquatic habitat in the long term. Rieman and others stated, "By working strategically it may be possible to establish mosaics of fuel and forest conditions that reduce the landscape risk of extremely large or simultaneous fires without intensive treatment of every subwatershed (Rieman et al., 2000)." Further, they suggested recovery of function in some watersheds may not be possible without some human intervention.

Dry forest treatments, while still controversial (Williams & Baker, 2012), are broadly supported by current scientific literature (Hessburg et al., 2016) and have continued to gain acceptance from the public and greater use by managers. In the Northern Region of the FS, restoring mixed severity fire regimes also remains controversial and complicated for numerous reasons such as the habitat needs of ESA listed species like steelhead, bull trout, lynx and grizzly bear. Treating riparian areas in mixed severity forests can be especially controversial and complicated. In locations where up-slopes and riparian forests have qualitatively similar fire effects, treatments guided by scientific findings are likely to restore ecological function of fire regimes at the landscape level (Finney et al., 2007). Position in the landscape relative to elevation, location within the stream network, and climate regime should be carefully considered to ensure understanding of riparian function (Gordon H. Reeves, Olson, et al., 2016; Gordon H. Reeves, Pickard, et al., 2016) (Pettit & Naiman, 2007). Because the effects of restoration treatments on departed riparian habitats are poorly understood, focused research in an adaptive management framework will be necessary.

In addition to vegetation treatments in riparian areas, stream channel restoration treatments will likely be considered to help aquatic ecosystems adapt to climate change. In a paper titled "*Restoring Salmon Habitat for a Changing Climate*" (T. Beechie et al., 2013), the authors recommend actions that connect streams to floodplains, restore flow, and help degraded channels aggrade as actions most likely to improve water temperatures. They also disclose that instream channel actions are unlikely to ameliorate climate change effects entirely.

Impacts to fisheries and other aquatic species

Expected climatic and hydrologic trends, combined with climate-related trends in wildfires and forest mortality from insects and diseases, can significantly affect aquatic ecosystems and species (J. Dunham, Rieman, & Chandler, 2003; J. B. Dunham, Rosenberger, Luce, & Rieman, 2007; Isaak et al., 2010). A growing body of literature has linked these hydrologic trends with impacts to aquatic ecosystems and species in western North America, often as a result of climate-related factors affecting stream temperatures and the distribution of thermally suitable habitat (Bartholow, 2005; Isaak et al., 2010; Kaushal et al., 2010; Morrison, Quick, & Foreman, 2002; Petersen & Kitchell, 2001). Lower summer stream flows and higher air temperatures, as observed over recent decades in Montana, are generally expected to result in increased stream temperatures. However, stream temperatures are controlled by a complex set of site-specific variables; including shading from riparian vegetation, wind velocity, relative humidity, geomorphic factors, groundwater inflow, and hyporheic flow (Caissie, 2006).

Potential changes in streamflow and rising stream temperatures are likely to increase risks to maintaining existing populations of native, cold-water aquatic species. Over the last century, most native fish and amphibians have declined in abundance and distribution throughout the western U.S., including western and central Montana and on the HLC NF. It is unknown whether, or to what degree, these changes are attributable to climate trends. Potential climate-induced trends of altered streamflow timing, lower summer flows, and increased water temperature would likely reduce the amount, quality, and distribution of habitat suitable for native trout and contribute to fragmentation of existing populations. Climate-related impacts are likely to add cumulatively to other stressors on native fish and amphibian species. Non-native trout and other aquatic species better adapted to warm water temperatures may increase in abundance and expand their existing ranges.

Westslope cutthroat trout and bull trout populations are sensitive to increased water temperatures (Bear, McMahon, & Zale, 2007; J. Dunham et al., 2003; Selong, McMahon, Zale, & Barrows, 2001). The latest science and modeling results (Isaak et al., 2015) for predicting localized climatic changes were reviewed to assess possible changes in summer water temperatures. Outputs from models which accurately back-predict historical temperatures were used for analyzing climatic effects on aquatic wildlife populations. It appears that these are relatively consistent in predicting that local, average summer air temperatures are predicted to increase between 2 to 4 degrees Celsius by 2050 (Dare et al., 2007);(Barsugli & Anderson, 2009).

A warming climate would decrease biodiversity through a number of potential pathways; including invasive hybridization (Muhlfeld et al., 2014). Isaak and others (2010; 2012) concluded a warming climate has already increased stream temperature and the volume of available habitat is shrinking resulting in a bottle neck to key species.

Water temperatures in montane stream systems do not respond directly in magnitude to changes in maximum and average air temperatures. For instance, a one degree increase in average air temperature parameters will almost universally result in less than a one degree increase in either average or maximum water temperatures. Buffering influences from factors such as groundwater discharge, and the role of direct solar radiation in heating stream water, prevent this from occurring. In the plan's geographical area, for every degrees Celsius increase in air temperature, a 0.44 degrees Celsius increase in average water temperature is predicted (Isaak et al., 2010; Mohseni, Erickson, & Stefan, 1999; Mohseni, Stefan, & Eaton, 2003). This would indicate that under constant catchment basin characteristics, an increase in summer water temperatures ranging from 0.88 to 1.76 degrees Celsius could be expected between now and 2050. This extrapolated prediction is consistent with trends measured in recent decades of approximately 0.24 degrees Celsius per decade (Isaak et al., 2012). Extending this rate out to 2050 would match the low-end of this range without considering or adjusting for rate changes due to emission patterns or other influencing trends. This extrapolated range is also consistent with predictions found in a recently published paper (Isaak & Rieman, 2013). This article provides more of an accuracy check than an independent collaboration of the results brought forward in this assessment. Both efforts use similar citations and are primarily based on the same source data and modeling runs.

Decade-long averages of summer temperatures naturally vary across the North American continent. There is a pattern of warmer and cooler decades. Any future decade could fall at the margins of the historic variation before modeled increases are put into consideration.

One of the climate change related viability concerns for aquatic wildlife populations in this plan area is whether adding the predicted 0.88 to 1.76 degree Celsius increase to current maximum summer temperatures would lead to mortality concerns. The term "mortality concerns" in this context addresses temperature related fish-kill events that could reasonably be expected to occur during prolonged, extreme heat/drought events in the warmer sections of a stream. A fish-kill does not necessarily occur when temperatures exceed the critical thermal maximum for a species. The magnitude, duration, frequency of these events as well as the local microhabitat conditions are important factors. A weather event in which water temperatures slightly exceed a "reduced survivability threshold" for a few-minutes on only one day of the summer would be much less likely to create a fish-kill than a heat/drought event in which temperatures exceed the same threshold by a higher magnitude, across multiple hours each day and persisting over the span of several days.

There are climatic factors in addition to maximum summer water temperatures that affect survival and lifecycle completion for fish and mussel species. Thermal regimes in other seasons can affect the timing of spawning and the success of egg incubation. Earlier snowmelt run-off could increase scour during critical time periods in the lifecycles of trout, char, and mussels (Isaak et al., 2012). Earlier loss of snowpack also leads to lower summer flows which have been correlated within this plan area with decreased densities of westslope cutthroat trout (Nelson et al., 2011). Receding summer flows can lead to lower winter flows depending on fall precipitation events and effects of drought cycles on groundwater levels. Low winter flows are a concern as the critical over-wintering habitat is restricted.

Groundwater influence and discharges into surface water has been shown to both moderate temperature and be positively correlated with salmonid abundance (Ebersole, Liss, & Frissell, 2001). Perennial stream reaches in higher-elevation areas that have well-timbered valley bottoms and ground-water entry would be most resilient to warming conditions and changing weather patterns promoting earlier run-off. Lower elevation stream reaches, lacking riparian shade, containing high sediment loads, with impaired width-depth ratios, and losing flows to groundwater would be the least resilient reaches to changing conditions. This class of impacts often correlates spatially across the plan area with the stream reaches identified in the previous sentence as being least resilient to changing climatic conditions.

Future climate impacts to soils

Any future changes to length of growing season based on climate would affect soil and plant respiration. Typically, soils become active where temperatures exceed 44 degrees Fahrenheit and decrease in activity when soil moisture declines below 10 percent moisture (Davidson, Belk, & Boone, 1998). The combination of adequate temperature for growth is expressed as growing degree days. Using a 30 year compilation of mean annual data (Holden et al., 2015) growing degrees vary according to topographic gradient, aspect and valley form for the HLC NF. Bottomlands can have up to a 220 day growing season except where cold air drainage constrains growth. Middle elevations have from 160 to 200 day growing season varying mostly by aspect. In upper elevations, the cold air temperatures restrict the growing season down to 100 days with the greatest limitations above 7,500 feet. On areas that could experience longer seasonal drought, the effective growing degree days for soil respiration would decrease while upper elevations might have a longer growing season. As warming occurs, available soil moisture will be the primary control at mid to lower elevations. In Colorado, a study found that in complex terrain available water was the limiting factor to soil respiration for ponderosa and lodgepole (Berryman, Battaglia, & Hoffman, 2015). On finer scales the outcome becomes complicated by the interaction of the forest canopy and topographic position. Soil water can be maintained by the shading of forest canopy which reduces evaporative losses from wind and sun.

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