Rio Grande National Forest – Assessments 1 and 3
Aquatic and Riparian Ecosystem Integrity, Systems Drivers and Stressors

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Introduction

In assessments 1 and 3, we (the USDA Forest Service, Rio Grande National Forest) evaluate available information about ecosystem integrity and stressors and threats to integrity on the Rio Grande National Forest, as outlined in Forest Service Handbook 1909.12, Chapter 10, Section 12.1 and 12.3. In this report, we will briefly assess the current condition, system drivers and stressors related to the aquatic and riparian ecosystems on the Rio Grande National Forest, with further discussion and analysis provided in appendices A and B, which are posted separately.

Riparian areas are the important ecosystems along streams, lakes, and other water bodies. Riparian areas are actually three-dimensional: they extend down into the groundwater, up above the canopy, outward across the floodplain, up the near-slopes that drain to the water, laterally into the terrestrial ecosystem, and along the water course at variable widths. On the Rio Grande National Forest, there are roughly 130,000 acres of riparian areas (about 7 percent of the Forest), associated with over 11,000 miles of streams and other wetland areas.

Plants in a riparian area, like willows and sedges, depend on the water source. This distinct water-loving vegetation makes most riparian areas easy to recognize. Riparian areas are home to a greater variety of aquatic and terrestrial wildlife than any other habitat type across the landscape. Riparian areas are also important in maintaining water quality and stream flows by capturing sediment, trapping organic matter, and regulating flooding. Riparian area *integrity* is a requirement of the new Forest Service planning rule, and the Forest Plan must ensure that no management practices cause 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.

Aquatic ecosystems are the streams, lakes, and other water bodies that support fish and other aquatic species. This topic addresses the *integrity* of aquatic ecosystems. There are over 11,000 miles of stream channels in the Rio Grande National Forest, of which 1,800 miles have perennial streams. There are also 75 lakes and reservoirs, covering about 1,200 acres on the Forest. Aquatic ecosystem *integrity* is a requirement of the Forest Service planning rule. Integrity is measured by (1) whether or not the dominant characteristics of the ecosystem (stream flow, timing, water quality, water quantity, stream temperature, sedimentation, etc.) are within the range of what would occur “naturally” (*natural range of variability*), and (2) can stay within that range as each ecosystem is influenced by stressors such as climate change, as well as developments and uses of the forest.

Information Sources and Gaps

Key Information Sources

- Rio Grande National Forest Riparian Inventory Project – USDA Forest Service Washington Office (Abood 2015)
- Assessment of Wetland Condition on the Rio Grande National Forest (Colorado Natural Heritage Program)
- Ecological Driver Classification and Analysis for Aquatic Systems – R2 Regional Office (Winters 2016)
- Fen Evaluation Report (Colorado Natural Heritage Program –expected February 2016)
- Rio Grande National Forest Monitoring Report Summary 1997-2012

Key Information Gaps

- The condition of seeps and springs on the Forest, including the number with diversions/watering troughs, and potential effects on ecological site integrity.
- Riparian habitat condition attributes involving riparian willow growth and browsing influences, particularly in lower elevation stream zones.
- Amount and distribution of pugging and hummocking in wetland-associated soils from ungulates, and ecological trend of areas where this condition exists.
- Information pertaining to how existing reservoirs on the Forest might modify preferred habitat attributes for aquatic species.
- Extent and distribution of Aquatic Nuisance Species, particularly whirling disease in tributary streams and relationship to local sedimentation issues.
- Information pertaining to aquatic invertebrate diversity, status and distribution, and effects of natural and anthropogenic disturbances to this resource.
- Information pertaining to the number of culverts possibly influencing aquatic organism passage.
- There is a lack of quantitative base and trend data on stream health, particularly physical function, i.e., stream pattern, profile, and dimension.
- There is a lack of quantitative information on groundwater resources, particularly groundwater-dependent ecosystems.

Existing Forest Plan Direction

Our current Forest Plan generally describes the types of uses allowed on the Forest for each management area. The management areas in the current plan do not follow watershed boundaries, but the functioning of each watershed was considered when the management areas were originally developed.

Our current Forest Plan incorporates direction from the Forest Service Rocky Mountain Region's Water Conservation Practices Handbook. This handbook includes guidance and direction for riparian areas in the Region. This direction is consistent with, and sometimes exceeds, state-required Best Management Practices (BMPs), and meets Clean Water Act Section 404 guidelines. Only activities that maintain or improve long-term stream health are allowed in riparian areas. There is guidance to keep heavy equipment out of these areas and only cross at designated points. Concentrated uses such as recreation sites are not allowed.

In the current Forest Plan, we established desired conditions for all stream segments near a "reference" condition (a condition based on the expected natural range of variability). Our goal is to maintain or improve fish habitat in streams, lakes, and ponds, and to protect, conserve, and restore important habitats. In the Forest Plan, we also issued guidance to return and/or maintain sufficient stream flows under appropriate authorities to minimize damage to scenic and aesthetic values, fish and wildlife habitat, and to otherwise protect the environment. While in the 1996 Forest Plan we recommended acquiring additional instream flows on the forest; in 2000, that recommendation was replaced by a negotiated settlement and related court decree from Division 3 of the Colorado Water Court between the Rio Grande National Forest, and basin water users and districts.

Direction in the current Forest Plan also restricts livestock grazing in riparian areas to specific timeframes and sets limits for the amount of grasses and shrubs that can be grazed. It contains guidance for maintaining vegetative cover, limiting soil compaction, keeping disturbed areas from connecting to each other, controlling road construction and other developments, and controlling use of chemicals or pathogenic pollutants.

In the new Forest Plan, we will incorporate some of the findings of the Forest Service's 2011 Watershed Condition Framework. This was a Nation-wide functional assessment of every 6^{th} level watershed on National Forest System lands according to 12 physical or biological indicators.

Scale of Analysis

The scales considered for the aquatic analysis include broad-scale, mid-scale, and fine-scale information. The scales vary from the Upper Rio Grande River Basin system to ecosystem-site characteristics involving specific ecological zones, stream reaches, or specific wetlands types. For analysis involving ecosystem drivers, stressors, and management activities, sixth-level subwatersheds (HUCs) were commonly used to characterize and compare conditions on the Forest.

Relevant Aquatic and Riparian Ecosystems:

Ecological riparian systems within the framework of the Southern Rocky Mountains terrestrial ecological systems that occur on the Rio Grande National Forest (Rondeau 2001, Colorado Natural Heritage Program) are included below:

- Alpine/Subalpine Wet Meadow
- Rocky Mountain Alpine-Montane Wet Meadow
- Rocky Mountain Subalpine-Montane Fen
- Rocky Mountain Subalpine-Montane Riparian Shrubland (Riparian Willow)
- Rocky Mountain Subalpine-Montane Riparian Woodland
- Lower Montane Riparian Woodland Ecological System
- Foothills Riparian Woodland and Shrubland Ecological System
- Western North America Emergent Freshwater Marsh

Additional riparian/wetlands classification from the Rio Grande National Forest Riparian Inventory Report (Abood 2015), is included below:

- Freshwater Emergent Wetlands (61.5 percent)
- Freshwater Forested/Shrub Wetlands (33.1 percent)
- Riverine (4.5 percent)
- Freshwater pond (0.85 percent)

Key Ecosystem Characteristics (12.13):

- Presence, abundance, and condition of rare and unique habitat types, such as fens, bogs, seeps and springs.
- Riparian, wetland, and groundwater- dependent habitat structure.
- Watershed and hydrology attributes, such as elevation, aspect, drainage patterns, geology, stream gradients, and distribution of perennial, intermittent, and ephemeral channels.
- Hydrologic flow regimes including time, duration, and magnitude.

- Distribution of streams and size of stream network with unimpeded aquatic organism passage.
- Stream length with adequate flow for beneficial uses and sustenance of aquatic habitats and species.
- Miles of impaired perennial streams, including stream and lake temperatures and nutrient regimes.
- Presence or absence of native species vs. non-native or invasive species.
- Species richness and distribution of macroinvertebrates.
- Lack of stressors that reduce or truncate stream connectivity and aquatic habitat quality.

Assessment of Ecosystem Integrity (12.14):

1. *Presence, abundance, and condition of rare and unique habitat types, such as fens, bogs, and seeps and springs.*

The montane fen ecological system includes extreme rich fens and iron fens, both rare within the Southern Rocky Mountains ecoregion. Since this system relies on groundwater, any disturbances that impact water quality or quantity are a threat. Fens, wetlands, seeps and springs are central to ecosystem resilience in light of uncertainties such as climate change. Based on National Wetlands Inventory mapping, there are 42,862 acres of wetlands and water bodies (lakes and reservoirs) on the Rio Grande National Forest, representing approximately 2 percent of the total land area (Lemly, Colorado Natural Heritage Program 2012). Although a final assessment of fens is forthcoming, preliminary evaluations indicate that approximately 73 percent of wetland acres on the Rio Grande National Forest are saturated hydrologic regimes, which likely consist in large part of fens.

Ecosystem integrity as assessed in the Colorado Natural Heritage Program Report indicates that most wetlands on the Rio Grande National Forest are in excellent to good condition (Lemly 2012), with the integrity of key ecosystem characteristics maintained. Floristic quality assessment indices are high for most wetlands, though they vary by both elevation and wetland type.

A query of the National Hydrography Dataset layer for Assessment 2 (Bevinger 2015) shows 125 springs/seeps across the Forest. The seeps and springs layer in the Southern Rockies Landscape Conservation Cooperative does not include site condition and integrity. Low-elevation seeps and springs frequently used for livestock and/or wildlife troughs were not included in the Colorado Natural Heritage Program assessment. Although condition data is lacking for these types of systems, we consider their ecosystem integrity low due to water diversions, trampling, and other impacts. We find this situation most often in drier areas where free-flowing water is limited.

Based on the available information, lower elevation streams and those with limited base flows should be management priorities as they are most likely to be influenced by climate change in the future. The analysis conducted for aquatic systems by Winters et al. (2016) suggests that if these conditions occur in the future, discharge would be significantly reduced, variability would be much higher, and fish and other aquatic biota on the Rio Grande National Forest would decline. To mitigate the impacts of potentially reduced flows due to climate change, we can focus management on maintaining stream structure and surrounding vegetation within their perceived natural range of variation particularly for vulnerable streams, wetlands and riparian areas (Dawson, 2011).

Pugging[1] in saturated soil areas is readily observable in some montane springs and meadow areas where livestock and native ungulates congregate. Heavy cattle use in palustrine systems can alter the hydrology

[1] Deep hoof prints left by large ungulates on moist, fine-textured soils of streams and wetlands.

by damaging soils. Soil compaction and pugging of the peat layer will change surface water flow. Heavy cattle use can also alter the successional processes within the sedge- dominated area of a fen. Cattle hoof action can lead to pugging and hummocking, creating microsites where shrubs can become established, changing the sedge-dominated meadow to carr shrubland.

2. Riparian, wetland, and groundwater- dependent habitat structure.

In addition to the two aquifers in the valley proper, there are shallow alluvial deposits along major streams and rivers in the Rio Grande basin, e.g., upper Saguache Creek, upper Rio Grande River, and upper Conejos Creek, which are hydraulically connected with the basin-fill deposits (see Assessment 2). In the mountainous areas of the Rio Grande National Forest, the alluvium is separate from underlying crystalline-rock aquifers. Groundwater recharge to the aquifers is a combination of mountain-front recharge, annual precipitation, irrigation return flow, streambed infiltration, and down-valley groundwater flow. Thus watersheds across the Rio Grande National Forest are major sources of water to the aquifers.

Groundwater supports many wetlands, springs, and seeps across the Forest. A portion of these are groundwater-dependent ecosystems, which are communities of plants, animals, and other organisms whose existence and distribution depends on access to or discharge of groundwater. There is no inventory of groundwater-dependent ecosystems on the Forest but the National Hydrography Dataset and the National Wetlands Inventory provide information about their potential extent. More specifically, springs/seeps in the National Hydrography Dataset and palustrine emergent wetland types in the National Wetlands Inventory serve as representations for groundwater-dependent ecosystems. Springs/seeps are classified as points where water issues from the ground naturally. Palustrine emergent wetlands are saturated non-tidal areas characterized by erect, rooted, herbaceous hydrophytes where the substrate is saturated to the surface for extended periods during the growing season but no surface water is typically present.

Various woody and non-woody riparian, wetland, and groundwater dependent habitat structure occurs on the Rio Grande National Forest. Approximately 55 percent of the mapped National Wetlands Inventory acres are palustrine emergent or freshwater herbaceous wetlands. When lakes and wetlands are excluded, herbaceous wetlands make up 62 percent of the wetland acres on the forest. Riparian shrub wetlands are the second-most abundant, comprising about 33 percent of the wetland acres. Lemly (2012) noted nearly 500 plant species on the Rio Grande in her wetlands assessment report, suggesting diverse riparian, wetland, and groundwater dependent habitats. Sedges (*Carex* spp.) are the most diverse non-woody plants, with 42 individual species noted. Willows (*Salix* spp.) and bluegrass (*Poa* spp.) are also diverse, with 15 species each. The Colorado Natural Heritage Program Floristic Quality Assessment conducted on 77 wetland sites during 2012 found that Rio Grande National Forest wetland conditions are good, although human influences are evident at lower elevations. The Floristic Quality Assessment suggests that we are maintaining the ecological integrity of most riparian and wetland-related vegetation.

3. Watershed and hydrology attributes, such as elevation, aspect, drainage patterns, geology, stream gradients, and distribution of perennial, intermittent, and ephemeral channels.

The Ecological Driver Cluster Analysis developed by Winters et al. (2016) provides key information regarding the physical, chemical, and hydrological attributes that describe the aquatic, riparian and wetland ecosystems on the Rio Grande National Forest. This analysis was based on the 163 subwatersheds (6th Level HUCs) that comprise the Rio Grande National Forest and found that a total of 7 statistically similar clusters with discernibly different ecological characteristics could be identified. Each of the subwatershed clusters reveal characteristics that can be translated in terms of aquatic productivity, sensitivity to management, temperature, aquatic and riparian habitat, and stream discharge characteristics.

The key ecosystem drivers evaluated in this analysis and the underlying watershed characteristics associated with them include the following:

- Geology
  - Geochemistry (calcareous or non-calcareous geology)
  - Sediment production (coarse, medium, or fine)
- Hydroclimatic regime
  - Snowfall, rain and snow, or rain dominated regimes
- Stream gradient
  - Low (less than 2 percent), medium (2 to 4 percent) or high (over 4 percent)

Geology/Geochemistry:

The Rio Grande National Forest is similar to most areas of the Rocky Mountains in that non-calcareous geology dominates the landscape, meaning there are no significant amounts of calcium carbonate. They include igneous rocks; sedimentary rocks, such as shale, sandstones, mudstones, and siltstones and metamorphic rocks derived from non-calcareous parent rocks, such as gneiss, schists, and quartzites. The properties of these geologies do not have increase aquatic productivity by themselves, and are the most abundant type of geology in the Rocky Mountains. Through the chemical actions with other compounds, igneous geology can cause slightly- to highly-acidic waters. However, non-calcareous geologies can support highly productive aquatic environments where stream gradients, hydroclimatic conditions, and other factors are associated. Non-calcareous geology can also have implications for erosion potential and sediment delivery.

In contrast, calcareous geological types tend to be highly productive for plants, benthic macroinvertebrates and fish as well. Given the inherent solubility of calcareous bedrock, its porous nature is often associated with springs, creating unique wetland habitats. Calcareous bedrock tends to raise the pH of the ambient water and by producing carbon dioxide for photosynthesis and stabilizing temperatures, often supports rare taxa and taxa outside their normal ranges (Hynes 1970). The Rocky Mountains include discrete areas of calcareous geology, creating unique and important aquatic, riparian and wetland ecosystems. Because calcareous geology is uncommon on the Rio Grande National Forest, the areas where it exists warrant consideration for future investigations involving rare taxa, springs and associated streams for native trout restoration. On the Rio Grande National Forest, these areas include the northern and western portion of the Sangre de Cristo Mountains from approximately Crestone Peak northward to the Hayden Pass area, two smaller areas in the southern Sangres near Carbonate Mountain and California Peak, and a small area of National Forest System land to the west of Villa Grove.

Geology/Sedimentation:

The geology and formative processes that have occurred on and around the Rio Grande National Forest have resulted in two distinctly different zones of sediment. These two zones were created by the “pulling apart” of the valley by the Rio Grande Rift, and abundant volcanic activity that occurred about 30 million years ago.

Sediment produced on the east side of the Forest in the Sangre de Cristo Range is mostly coarse due to an igneous intrusion through ancient sedimentary layers. This material is more difficult to move with stream and overland water flow. However, when moved by erosion processes (either natural or man-made) it settles in wetlands, pool habitats and riparian areas. Once deposited in these areas, sediment becomes extremely difficult for streams and wetlands to remove as it has settled in areas with lower stream energy, reducing aquatic habitat by filling in wetlands and riparian areas.

In contrast, the sediment produced by the volcanic activity on the west side of the valley in the San Juan Mountains is generally fine to medium in size and can move more readily either naturally or from man-made disturbance. However, it takes a considerable amount of energy to move fine sediments from low-gradient stream systems. This includes depositing fine materials onto adjacent floodplains as occurred historically, which can be constrained by roads, trails, and other human activities often associated with low-gradient stream systems.

Hydroclimatic Regime:

Annual precipitation across the Forest varies from 12 inches at lower elevations to over 50 inches at higher elevations; generally, precipitation increases 4 to 6 inches for each 1000-foot gain in elevation. In the 12 to 18 inch zone (roughly 8,000 to 9,000 feet), most, if not all, the precipitation goes to satisfying evapotranspiration demands so little to no runoff is generated; an exception can be when summer thunderstorm rainfall intensity exceeds the soil infiltration rate, resulting in overland flow potentially reaching stream channels). Above 9,000 feet the Forest is considered a snow-dominated system in that the majority of the annual precipitation falls in the form of snow, while the remainder is from late spring to early fall rains. Summer thunderstorms can produce short-duration, high-intensity precipitation.

The majority of the Rio Grande National Forest is located in the snow-driven elevation zone. Aquatic productivity is relatively low at these high elevations, although aquatic plants can thrive if food, habitat and other limiting factors are not reduced significantly. Relatively short growing seasons and low temperatures can cause unsuccessful spawning by fish. Connectivity, adequate water and timing, habitat and temperature are major limiting factors for aquatic and semi-aquatic taxa. Ensuring the important natural conditions of ecosystem elements are maintained for these taxa is critical for the continued survival of water dependent species and ecosystem services such as angling. However, a rain and snow dominated regime may exist in the lowest-elevation headwaters of subwatersheds of the Forest (Winters et al. 2016). These areas may contain plants and animals that are more representative of the valley floor. In addition, wetlands may also have different form and function if precipitation and resulting overland flow is modified. Given that the lower elevations of the valley receive little annual precipitation (approximately 8 inches per year), stream channels should be ephemeral or intermittent depending on the presence of groundwater. Wetlands would also be dependent on groundwater reaching the surface where rooting would allow wetland plants to exist.

Stream Gradient:

Stream gradient influences stream power, aquatic productivity, erosive capability, localized sediment sizes, and floodplain characteristics. As a result, stream gradient dictates the habitat for specific flora and fauna.

Steep-gradient stream reaches are considered “erosional,” and low-gradient reaches “depositional” zones. The sediment produced in the erosional zones is deposited in the depositional zones, which thus can be indicators of upstream erosion. Indeed, steep stream channels have narrow floodplains and banks that are ‘armored’ with large rocks and boulders. They are very resistant to change, but move sediment through them and are migration corridors for aquatic species. Medium-gradient streams can be considered “transitional” reaches and have some combined characteristics of the steep and low gradient reaches. Low-gradient reaches are very susceptible to upstream activities as well as local change. As a result, they typically reflect any changes in channel morphology, temperature, in-stream habitat attributes, and sediment accumulation. Their banks are comprised of typically small alluvium from upstream and are extremely susceptible to erosion if the deep rooting riparian vegetation is removed or damaged. Low-gradient reaches can act as “reservoirs” for storing water in the valley bottom, especially if there are beavers. Terrestrial and aquatic animals are often much more abundant in low-gradient valleys as habitat is diverse and abundant. Consequently, upstream activities in subwatersheds with low-gradient valleys produce excess sediment, reduce woody material and change discharge dynamics; which can have a dramatic impact on the low-gradient areas. Local impacts that remove woody vegetation, compact riparian soils and damage stream banks can also profoundly impact these important areas.

In the analysis by Winters et al. (2016), only 8 of 162 subwatersheds on the Rio Grande National Forest were dominated by low-gradient stream systems. This is not unusual for the southern Rocky Mountains considering the steep topography and erosional process associated with the landscape. The role of stream gradient reaches and their importance could be a valuable tool for future monitoring. Low-gradient reaches are sensitive to change, as are the plants and animals that inhabit them. By focusing monitoring on these important reaches, species and physical characteristics could be identified and utilized as tools for observing change.

Glaciation and Wetland Development:

Another landscape attribute that heavily influences aquatic, riparian, and wetland characteristics on the Rio Grande National Forest is past glaciation. The Rio Grande National Forest contains thousands of acres of high quality wetlands and other groundwater-dependent ecosystems that provide essential services to the Forest and lands downstream. A disproportionately large number of wetlands occur in the glaciated areas of the forest. Terminal and lateral moraines often create confined basins where impounded subsurface or surface water allows peat accumulation (Windell et al. 1996; Cooper 1990; and Cooper 2005) whereas kettle ponds promote fen formation along their margins. In addition, glaciation has created wide, relatively level mountain valleys where large wetland complexes tend to form (Rocchio 2006). Understanding how wetlands occur in relation to the geomorphology of the forest can help guide forest planning and management decisions pertaining to aquatic ecosystem sustainability, restoration, and climate change resilience. High mountain wetlands in glaciated areas are fundamentally different than wetlands occurring at lower elevations that were never glaciated (Cooper and Lee 1987).

Stream Density Analysis:

There are 23 subwatersheds within the Rio Grande National Forest that are in the highest quantile for stream density, with several others being predominantly within its boundaries. There appears to be a correlation between higher elevations and these high density subwatersheds. The San Juan and Weminuche Wilderness areas have high densities of streams that may be beneficial for native fish and amphibian recovery. In addition, these subwatersheds are generally less managed and could continue to provide colder water to lower elevations, especially if the effects of climate change increases water temperatures at lower elevations.

Stratifying the Forest by stream density presents important management considerations and options. These areas could be considered the highest priority for stream- and riparian-related management if other ecological conditions are not an issue. The subwatersheds in the western portion of the Forest may be more productive, because of the capacity for volcanic soils to hold and release important micro nutrients.

Summary of the Ecological Driver Cluster Analysis:

The analysis found that a total of 7 statistically similar clusters with discernibly different ecological characteristics could be identified on the Rio Grande National Forest. The top three cluster characteristics and others can be defined as follows:

- Approximately 33 percent of the Forest (53 subwatersheds) can be characterized by the ecological conditions dominated by non-calcareous geology, high-stream gradients, fine sediments, and a snowmelt hydroclimate.
- Approximately 32 percent of the Forest (52 subwatersheds) can be characterized by the ecological conditions associated dominated by non-calcareous geology, high-stream gradients, coarse sediments, and a snowmelt hydroclimate.
- Approximately 12 percent of the Forest (20 subwatersheds) can be characterized by the ecological conditions dominated by non-calcareous geology, high-stream gradients, coarse sediments, and a snowmelt hydroclimate. The unique characteristic of this cluster is the considerable portion of streams containing low to medium gradients.
- Approximately 9 percent of the Forest (14 subwatersheds) can be characterized by the ecological conditions dominated by non-calcareous geology, high-stream gradients, coarse sediments, and a rain and snow (72 percent) and snowmelt (30 percent) hydroclimate.
- Approximately 8 percent of the Forest (12 subwatersheds) can be characterized by the ecological conditions dominated by non-calcareous geology, high-stream gradients, fine sediments, and a snowmelt hydroclimate. The feature that separates this cluster from others is the high abundance of fine, volcanic soils that could be associated with erosion potential.
- Approximately 6 percent of the Forest (10 subwatersheds) can be characterized by the ecological conditions dominated by non-calcareous geology, high-stream gradients, coarse sediments, and a rain and snow (72 percent) and snowmelt (28 percent) hydroclimate.
- Approximately 1 percent of the Forest (2 subwatersheds) can be characterized by the ecological conditions associated dominated by a calcareous geology, high-stream gradients, coarse sediments, and a snowmelt hydroclimate.

4. Hydrologic flow regimes including time, duration, magnitude.

Information from Assessment 2 (Bevinger 2015) uses a historical USGS gage on the Alamosa River above Terrace Reservoir as a typical representation for water quantity and timing of flows, i.e., flow regime, on the Rio Grande National Forest, streamflow is predominately from annual snowmelt runoff, but there can be small increases in flow for short periods of time due to summer thunderstorm activity. The volume of water contained within these spikes typically represent only a small percentage of the thunderstorm precipitation total, roughly equivalent to the watershed area occupied by stream channels (Troendle and Bevenger 1996). Annual snowmelt flows typically start to increase in late March to early April, peak in late May to early June, and return to baseflow levels in late August to early September. There is considerable year to year variability in the shape of the annual hydrograph and the number of peak flows during the snowmelt period. The magnitude of annual peaks also exhibits considerable variability. Flow duration for perennial streams on the Forest is typical of snowmelt dominated areas where the high flow period is relatively short and gradual release of groundwater maintains a baseflow for the remainder of the water year. The flow regime described here is critical for sustaining current ecosystems across the Forest. Conversely, the current ecosystems also result in and sustain the current flow regimes across the Forest. In other words, a change in one results in a corresponding change to the other.

Streams across the Forest have a low-flow channel, a high-flow or bankfull channel, and a floodplain to handle above-bankfull flows. Flooding is a natural part of the hydrologic cycle and access to the floodplain is critical. The width of the flood prone area varies by stream and valley type and can be relatively narrow, i.e., tens of feet, to very wide, i.e., hundreds of feet. Floodplains and their associated riparian areas are very important in regulating water quality and how water is distributed over time.

Healthy stream and riparian systems dissipate flood energy and recharge alluvial aquifers. Water is then slowly released from the aquifers back to the channel during drier periods of the year.

Current trends in hydrologic flow regimes in Colorado over the past 30 years have involved increased average annual temperatures, below average snowpacks, and an earlier snowmelt and peak runoff period.

In recent years, the late-spring snowpack across the Forest has had a brownish color due to heavy deposition of desert dust. Modeling indicates that in moderately dusty year's snowmelt and the peak flow can occur about three weeks earlier and extreme dust loading can add another three weeks, or a total of six weeks of earlier melt and peak. Projected climate change effects may contribute to these issues, thereby further altering the hydrologic flow regime.

5. *Distribution of streams and size of stream network with unimpeded aquatic organism passage.*

We are in the process of collecting information regarding aquatic organism passage on the Forest. For example, information regarding the number of stream crossings and culverts is available, but details regarding the needs and consequences of passage impediments are considered an information gap. In general, impediments to aquatic organism passage are more prevalent in smaller stream networks, and where vegetation management activities have resulted in more roads and stream crossings. We want barriers to aquatic organism passage only where non-native trout species may compete with endemic species such as Rio Grande Cutthroat Trout.

The recent analysis by Winters et al. (2016) found that most culverts and stream crossings on the Rio Grande National Forest are associated with unpaved rather than paved road systems. The values associated with the subwatersheds (6^{th} Level HUCs) on the Forest ranged from 0 stream crossings per mile of stream in valley bottoms (15 subwatersheds, under 10 percent) to 1.3 crossings per stream mile in the West Fork of Pinos Creek subwatershed. A total of 20 of the 30 subwatershed within quantile 5 (the highest ranking) were located totally within the Rio Grande National Forest. These results are considerably different than the paved road results and indicate a relatively high number of unpaved roads in valley bottoms with streams and riparian areas. The concentrations and distribution of roads and stream crossings are going to correlate with levels of human activity. The highest acreage of subwatersheds with unimpeded passage for aquatic organisms is in designated wilderness areas on the west side of the Forest. However, this may not be where we identify the greatest and most important passage needs for fish species.

6. *Stream length with adequate flow for beneficial uses and sustenance of aquatic habitats and species.*

The Rio Grande National Forest is unique in that it has a federally reserved water right which was established by decree in 2000. The Decree provides the Forest 252 reserved instream flow water rights. These rights are quantified in terms of their location and amount of flow that must be satisfied at and upstream of the location, i.e., quantification point. The decree established 303 quantification points on the headwaters of streams and rivers in the eastern San Juan and La Garita mountains and the northern Sangre de Cristos. Each quantification point includes minimum high flows and maximum high flows that vary depending on the time of year. The in-stream flows, which do not involve the removal or consumption of any water, are designed to protect stream function and fish habitat. All streams on the Forest are covered under the decree with substantial benefits to aquatic habitats and species.

The decreed rights allow for improving and protecting the forests within the administrative boundary, securing favorable conditions of water flows, and furnishing a continuous supply of timber for the use and necessities of the citizens of the United States. From an aquatic species point of view the rights also allow for maintaining, improving, protecting, and minimizing damage to 1) riparian ecosystems, including stream dependent wetlands, 2) the natural physical function of stream channels, 3) viable and diverse populations of fish and wildlife, including all habitat necessary for such populations, and 4) public opportunities for outdoor recreation such as angling.

7. Miles of impaired perennial streams, including stream and lake temperatures and nutrient regimes.

Information in Assessment 2 indicates that the State of Colorado Integrated Water Quality Monitoring and Assessment Report (Colorado Department of Public Health and Environment 2012), commonly called the 303(d)/305(b) report, indicates 74 percent of the stream miles in the Rio Grande Basin either fully support all classified uses of water (54 of the 74 percent) or support at least one classified use (20 of the 74 percent). Given the above information, it is estimated that there are approximately 468 miles of impaired waterbodies (303(d) listed) within the Rio Grande River Basin, some of which are within the Rio Grande National Forest (Colorado Department of Public Health and Environment 2012). These listed waterbodies are mostly associated with historical mining and are in the Alamosa, Willow, and Kerber Creek watersheds, although the upper Hot Creek subwatershed is also noted in the aquatic report by Winters et al. (2016). Total Maximum Daily Loads (TMDLs) exist for some of these listed waterbodies. Heavy metals and pH are the primary water quality concerns. There are also multiple stream segments across the Forest on the monitoring and evaluation list maintained by the State of Colorado. The impaired waterbodies associated with mining and heavy metals have impacted aquatic habitat values and fisheries resources. The Alamosa River area around Summitville is an EPA Superfund site and considerable restoration work has taken place to improve water quality in the river and its tributaries. Restoration work has also taken place on Willow Creek near Creede (Colorado Department of Public Health and Environment 2013).

From a process category standpoint, a majority of the 152 watersheds are functioning properly relative to aquatic physical and terrestrial biological indicators. Relative to aquatic biological and terrestrial physical processes a majority of the watersheds are functioning at risk. There are no watersheds with impaired terrestrial physical and biological function. There are two watersheds with impaired aquatic physical function. The concern in both of these watersheds is water quality, flow characteristics, habitat fragmentation, and channel shape and function. There are eighteen watersheds with impaired aquatic biological function. The concern in these watersheds is with exotic and aquatic invasive species, and riparian and wetland vegetation condition. In some of the watersheds there is also a concern with life form presence.

8. Presence or absence of native species vs. non-native or invasive species.

The Rio Grande National Forest supports three native fish species endemic to the upper Rio Grande basin - these include the Rio Grande cutthroat trout, the Rio Grande sucker, and the Rio Grande chub. The exact historical distribution of Rio Grande cutthroat trout is unknown but it is likely that populations were widely distributed throughout the Upper Rio Grande watershed, occupying all streams capable of supporting trout within the drainage. Core populations (over 99 percent genetically pure) and conservation populations (over 90 percent genetically pure) of Rio Grande cutthroat trout are typically restricted to smaller 6th to 7th-level streams and currently occupy less than 220 stream miles. Colorado Parks and Wildlife has stocked Rio Grande cutthroat trout in an additional 150 miles of stream and 59 high mountain lakes for sport fishing and to maintain genetic refugia for pure historic populations. We are part of a signed interagency conservation agreement to improve and maintain habitat conditions and populations of the Rio Grande cutthroat trout. This species is currently a primary focus for restoration opportunities in the upper Rio Grande Basin.

The Rio Grande sucker is endemic to the Rio Grande drainage in Colorado and New Mexico. In Colorado, the sucker was historically found in Rio Grande, Conejos River, Hot Creek, and McIntyre Springs. The Rio Grande sucker has significantly declined from much of its historic range, and several populations have been reintroduced to lower-elevation streams on the Rio Grande National Forest. All reintroduced populations on the Forest are struggling to survive.

The Rio Grande chub is endemic to Rio Grande drainage and Closed Basin. Historically, they occurred in the Rio Grande River, Conejos River, Saguache Creek and San Luis Creek. Like the sucker, the Rio Grande chub has significantly declined from much of its historic range. Limited populations have been reintroduced to lower-elevation streams on the Rio Grande National Forest. It is currently unclear if any of these populations have survived. The forest continues to work with Colorado Parks and Wildlife to identify and prioritize locations for reintroduction of native fish.

There are approximately 680 miles of rivers and streams on the Rio Grande National Forest that sustain populations of nonnative trout. These species include rainbow trout, brown trout, brook trout, Snake River cutthroat, and splake. Other non-trout species include fathead minnow, longnose dace, and white sucker. When assessed in relationship to the watershed condition framework process, lack of native aquatic species and/or presence of exotic and/or aquatic invasive species, and vegetation condition in riparian/wetland areas are the driving concerns for most functioning at-risk or at impaired function ratings. Approximately 78 percent of the subwatersheds on the Forest received a poor watershed condition framework ranking in regards to secure presence of native species. Another 26 percent and 13 percent received fair and good rankings, respectively, indicating there is room for substantial improvement in regards to the presence of native vs. non-native species. However, non-native trout species are desired in many areas for their high recreational value as sport fish.

Other aquatic species that occur on the Rio Grande National Forest include the boreal toad and the northern leopard frog. Once historically common, the boreal toad now exists in four or five habitat areas confined to one district. Historic information suggests the northern leopard frog was never common on Forest lands, and primarily restricted to mid and lower elevation sites along sites such as the Conejos River. Both amphibian species are now uncommon and declining; succumbing to stressors associated with chytrid fungus, and perhaps non-native trout species in some locations.

A draft aquatic nuisance species plan has been developed for the Rio Grande National Forest March 2011). Based on current information, whirling disease is the only aquatic nuisance species affecting the fisheries resource on the Forest. Whirling disease is currently assumed present in all 4^{th} order and larger streams with an unknown status in smaller streams. Exotic aquatic plant species such as Eurasian water milfoil and water hyacinth have been located in the San Luis Valley but are not currently known to occur on the Forest. The Colorado Parks and Wildlife currently monitors equipment at select lakes and reservoirs for exotic mollusk species such as quagga and zebra mussels. No invasive mollusks have been found to date locally. Chytrid fungus remains a serious issue for local amphibian populations and we are recommending it as a topic of concern in our plan revision effort.

9. Species richness and distribution of macroinvertebrates.

Information pertaining to the diversity and distribution of macroinvertebrate on the Rio Grande National Forest is one of our information gaps. However, one previous study found that six families of macroinvertebrates are common in the upper Rio Grande Basin and that median values for total abundance, taxa richness, and mayfly and stonefly abundance are reduced at sites influenced by mining. Depressed abundance is also been found where trace elements have been elevated (Deacon et al. 2001). Based on current habitat, water quality, and angling information, we suspect that we are at desired condition for macroinvertebrates in most subwatersheds on the Forest. However, as with riparian condition the available information indicates there are specific areas and stream reaches that are impacting macroinvertebrates.

10. Lack of stressors that reduce or truncate stream connectivity and aquatic habitat quality.

A thorough evaluation of watershed condition framework rankings for all subwatersheds on the Forest was conducted in Assessment 2 (Bevinger 2015). Although primarily subjective, the watershed condition framework process does offer an opportunity to compare condition rankings across the Forest. When looking at overall watershed condition across the Rio Grande Forest one-half of the 152 watersheds rated in 2011 are functioning properly, 49 percent are functioning at-risk, and only one watershed has impaired function. Attribute and indicator ratings for three of the four process categories - aquatic physical, and in particular, aquatic biological and terrestrial physical - are the driving influences behind the overall condition scores. In regards to aquatic habitat quality, 66 percent of the subwatersheds were ranked as good, 28 percent as fair, and 6 percent as poor. There are no fragmentation concerns on 78 percent of the subwatersheds, while another 15 and 7 percent were ranked as fair to poor, respectively.

Sixteen of the twenty-four attributes in the watershed condition framework process are the principle influences on overall watershed condition. These are water quality, channel shape and function, native species, exotic and/or aquatic invasive species, vegetation condition, open road density, road and trail maintenance, proximity to water, soil productivity, soil erosion, soil contamination, fire regime condition class, rangeland vegetation condition, extent and ratio of spread, and insects and disease. The primary human stressors associated with these rankings and the greatest current and potential threats to watershed and water resource integrity across the Forest involve surface disturbing activities such as timber harvest, grazing, mining, recreation developments, road construction, and off-road motorized use. These effects can be direct, indirect, or cumulative. The scientific literature that supports this is voluminous but can be summarized by stating the effects are due primarily to soil erosion and compaction in uplands and riparian areas, changes in the amount of water and sediment delivery to streams and wetlands; and associated adjustments then made by the stream systems. The literature also clearly shows that applying best management practices (BMPs), can mitigate the effects of land use, allowing for watershed protection.

In terms of natural stressors, the current outbreak of spruce bark beetle has now influenced over 588,000 acres of high-elevation spruce-fir forest (2015 data) and incurs the most debate regarding potential influences to water and aquatic habitat attributes. As noted in Assessment 2, because spruce stands grow in areas of the Forest with the greatest precipitation and because evapotranspiration rates are higher in spruce than other conifer types, there is potential for increased stream flow from bark beetle activity. Historical watershed studies have shown that manipulating conifer vegetation, e.g., timber harvest, can lead to measurable and detectable increases in streamflow at the sub-watershed scale (MacDonald and Stednick 2003). However, timber harvest results in the removal of biomass so comparing beetle kill to timber harvest is not entirely appropriate. In other words, both types of disturbance change the evapotranspiration component of the water cycle but interception and evaporation processes change less under beetle kill.

More recent watershed studies, conducted after the start of the current beetle infestations, have detected no consistent changes in streamflow (Gordon and Ojima 2015). However, much of this work has been in lodgepole pine stands, which grow in areas of lower precipitation; additionally evapotranspiration demands of this forest type are less. Modeling efforts of bark beetle infestation indicate there may be a 5 to 10 percent increase in runoff. Such a small change may be real but watershed studies have shown that an increase of at least 15 percent is necessary to detect the change at the sub-watershed scale (MacDonald and Stednick 2003). Without specific studies on the Forest, it is unknown if recent insect activity has changed evapotranspiration rates that will affect stream flows. Once they fall, however, the dead spruce should contribute significantly to the amount of downed wood in the systems.

Description of Natural Range of Variation (or Alternative Approach) (12.14a or b):

The reference models or conditions we use for describing the natural range of variation for the various riparian and aquatic ecosystems on the Rio Grande National Forest and elsewhere are not well defined. In our 1996 Forest Plan, we used reference streams to monitor and inform stream health, but did not address many of the other various aquatic systems on the Rio Grande National Forest, nor the ecological components that maintain them. However, the reference stream information we have suggests that most streams and riparian systems on the Rio Grande National Forest are in good to excellent condition.

The multi-metric ecological integrity assessment utilized by the Colorado Natural Heritage Program during the 2012 wetlands assessment provides a sound alternative approach to assessing whether the integrity of an ecosystem is being maintained (Lemly 2012). Using this method, A ranks indicate reference conditions (no or minimal human impact), B ranks indicate slight deviation from reference, C ranks indicate moderate deviation from reference, and D ranks indicate significant or severe deviation from reference. Nowhere on the Rio Grande National Forest, has an ecological integrity assessment rank of D. Of the 77 various wetlands surveyed, 41 were A-ranked, 32 were B-ranked, and 4 were C-ranked. A-ranked systems occurred primarily in the alpine and subalpine zones, with lower elevation sites more likely to receive B-ranks. However, a few wetlands are ranked C due to stressors including grazing, hydrologic modifications, and surrounding land use activities. Riparian shrublands, wet meadows, and fens were most of the A- and B-ranks, with riparian woodlands and marsh slightly lower ranks. It is also important to note that some riparian ecosystem types, such as those associated with man-made reservoirs or low-elevation riparian woodlands, are not well-represented in the sites sampled for the ecological integrity assessment index.

Other available information, such as the Watershed Condition Framework, portray a different scenario. For example, almost one-half of the watersheds on the forest are functioning at-risk due to rangeland vegetation condition concerns and four watersheds have impaired function due to rangeland vegetation condition concerns. (See Assessment 2). This suggests that there are deviations from the stream reference conditions we used to assess natural range of variation in the 1996 Forest Plan.

Assessment of Status and Trends (12.14c):

The quantitative information we have regarding the status and trends of riparian and aquatic systems on the Rio Grande National Forest is limited, with data gaps evident at this time in regards to some desired attributes. However, the current available information suggests these types of ecological systems, and the key ecological characteristics that maintain them, are for the most part healthy and functioning. The ecological integrity assessment method we used to assess a sample of mapped wetlands and aquatic habitats across the forest serves as an adequate reference model for some, but not all, ecosystem integrity attributes. Information we have regarding the status and trends of key ecosystem characteristics on the Rio Grande National Forest includes the following:

- The dominant key ecosystem characteristics and associated physical, chemical, and biological processes appear to be functioning and retain the ability to contribute to the long-term integrity of existing riparian and aquatic habitats. Although not all systems are represented, the highest integrity rankings are associated with high-elevation areas most likely to be associated with low-human use impacts. Integrity attributes for some aquatic systems are reduced but remain in good to fair condition in most lower-elevation areas where human-related activities are more evident. Additional information regarding underlying geological, physical, and geochemical processes have been developed by Winters et al. (2016) that further describe the expected function of each watershed, and the unique sensitivities and opportunities of particular areas.

- Some key ecosystem characteristics have been altered from reference conditions, particularly in lower elevation areas where human-related activities are more evident. The two major rivers on the forest – the upper Rio Grande and Conejos – both have been dammed which influences natural water flows, sediment transport, and key ecosystem processes such as pulse flood events important to lower elevation vegetation systems such as cottonwood bosques. Some aquatic nuisance species, such as chytrid, appear to be spreading, affecting the persistence of native amphibian species. While most key ecosystem characteristics appear to be stable and functioning, specific information on potential changes is lacking.
- Existing barriers to ecological connectivity for aquatic organisms do exist and have been noted in other sections of this report. These primarily involve road crossing and culverts, although some natural barriers also exist. Some barriers are desirable to discourage interactions with non-native fish species. In some cases, human impacts are contributing to stream impacts.
- The current information suggests that key ecosystem characteristics in high-elevations are secure, even if rare, while some low-elevation characteristics may be vulnerable to environmental changes, particularly in combination with certain management influences. For example, low-elevation riparian cottonwood is rare and also vulnerable given the activities associated with low-elevation stream systems.
- Restoration activities are desirable in mid- to lower elevations. Focused management of some natural and anthropogenic stressors is warranted and has been on-going.
- The Rio Grande National Forest includes the very headwaters of the Rio Grande River system, which is important to supplying many downstream ecosystem services. Broader landscape perspectives involving certain stressors and drivers are applicable.
- Trends are an information gap for certain riparian and aquatic ecosystem characteristics.

Assessment of Riparian Areas and Groundwater-dependent Ecosystems (12.14d)

The Rio Grande National Forest covers 1.83 million acres in south central Colorado and contains the very headwaters of the Rio Grande River. The Forest’s diverse geography creates a template for equally diverse wetlands, which provide ecological services to both the Rio Grande National Forest and lands downstream. Though now recognized as a vital component of the landscape, many wetlands have been altered by a range of human land uses since European settlement. Across the Rio Grande National Forest, mining, logging, reservoir construction, water diversions, livestock grazing, and recreation have all impacted wetlands. In order to adequately manage and protect wetland resources on the Rio Grande National Forest, we need reliable data on their location, extent and current condition.

Between 2008 and 2011, Colorado Natural Heritage Program (CNHP) partnered with Colorado Parks and Wildlife on a U.S. Environmental Protection Agency (EPA) funded effort to map and assess the condition of wetlands throughout the Rio Grande Headwaters River Basin which includes the Rio Grande National Forest. Existing paper maps of wetlands created by the U.S. Fish and Wildlife Service’s National Wetland Inventory Program were converted to digital data. In addition to the mapping, 137 wetlands were surveyed across the Rio Grande Headwaters basin using condition assessment methods developed by the Colorado Natural Heritage Program over the past decade. Of the wetlands surveyed, 52 were located on the Rio Grande National Forest in 10 different watersheds. To supplement the EPA-funded study, the U.S. Forest Service provided funding through a challenge cost share agreement for additional wetland sampling on the Rio Grande National Forest to develop more comprehensive information about types, abundances, distribution, and condition of the forest's wetlands. Through this agreement, 25 additional wetlands on the forest were surveyed and all data from the Rio Grande National Forest were summarized.

Based on digitized National Wetland Inventory mapping, there are 42, 862 acres of wetlands and water bodies found within the Rio Grande National Forest, of which lakes and rivers comprise 4,687 acres or 11 percent. This estimate for wetlands and water bodies represents approximately 2 percent of the total area in the Rio Grande National Forest. Slightly over half (55 percent) of the national wetland inventory mapped acres are freshwater herbaceous wetlands. Shrub wetlands make up another 30 percent. When broken down by hydrologic regime, saturated wetlands, saturated wetlands are the most common, comprising 73 percent of national wetland inventory acres. Within the forest, 82 percent of all lakes are mapped with a dammed/impounded modifier, indicating that most lakes are reservoirs of one kind or another. Beavers influence only 4 percent of all wetland acres, but 23 percent of ponds are mapped as beaver ponds and 6 percent of shrub wetlands are mapped with beaver influence. Sixty five percent of all national wetland inventory acres occur in the subalpine ecoregions, which make up roughly the same proportion of the Forest's land area. Another 29 percent of national wetland inventory acres occur in the alpine zone. Lower elevation zones contain very few wetland acres.

In total, 77 wetland sites were surveyed across the Rio Grande National Forest, including 30 riparian shrublands, 27 wet meadows, 17 fens, 2 riparian woodlands, and 1 marsh. Nearly 500 plant taxa were encountered during the surveys, including 445 identified to species level. Of the 445 species identified, 420 (94 percent) were native species and 25 (6 percent) non-native species. Noxious weeds, an aggressive subset of non-natives, were present in only four plots.

Wetland condition measures indicate that wetlands on the Rio Grande National Forest are in excellent to good condition. Floristic quality assessment indices were high for most wetlands surveyed, though did vary by both elevation and wetland type. Ecological integrity assessment scores rated most wetlands with an A- or B-rank, indicating that wetlands were either in reference condition or deviated only slightly from reference condition. A handful of wetlands received C-ranks, due to stressors including grazing, hydrologic modifications, and surrounding land use.

System Drivers (12.31)

The ecological driver assessment of Winters et al. creates a multi-scale approach that assists with prioritizing ecological land units including watersheds, vegetative communities, and geomorphic settings. Identifying the major ecological drivers important for assessing aquatic, riparian, and wetland resources within a geographic area can help us determine the spatial distribution and levels of productivity for these various systems, and aid in our future planning and management decisions.

We chose five ecological drivers to describe the setting for aquatic, riparian and wetland resources on the Rio Grande National Forest (Wohl et al. 1981). These include:

1. Geochemistry: (calcareous, non-calcareous, or salt bearing)
2. Sediment production based on geology: (fine, moderate, or coarse)
3. Dominant hydroclimatic regime: (rain and snow, or snowfall)
4. Stream gradient: high (over 4 percent), medium (2 to 4 percent), or low (under 2 percent)
5. Glaciation: (glaciated, or non-glaciated).

Geology/Geochemistry:

The Rio Grande National Forest is similar to most areas of the Rocky Mountains in that non-calcareous geology dominates the landscape. Non-calcareous geologies do not contain significant amounts of calcium carbonate. They include igneous rocks; sedimentary rocks, such as shale, sandstones, mudstones, and siltstones; and metamorphic rocks derived from non-calcareous parent rocks, such as gneiss, schists, and quartzites. These geologies do not have properties that increase aquatic productivity by themselves, and are the most abundant type of geology in the Rocky Mountains. Through the chemical actions with other compounds, igneous geology can result naturally in slightly to highly acidic waters. However, non-calcareous geologies can support highly productive aquatic environments where stream gradients, hydroclimatic conditions, and other factors are associated. Non-calcareous geology can also have implications for erosion potential and sediment delivery.

In contrast, calcareous geology tends to be highly productive for plants, benthic macroinvertebrates and fish as well. Given the inherent solubility of calcareous bedrock, its porous nature is often associated with springs, creating areas of unique wetland habitats. Calcareous bedrock tends to raise the pH of the ambient water and produce carbon dioxide for photosynthesis and stabilizing temperatures, it also often supports rare taxa and taxa outside their normal ranges (Hynes, 1970). The Rocky Mountains contain a limited amount of calcareous geology, making its presence unique and important for aquatic, riparian and wetland ecosystems. Calcareous geology is also uncommon on the Rio Grande National Forest, and the few areas that do contain it should be investigated to identify rare taxa, springs and associated streams for native trout restoration. On the Rio Grande National Forest, these areas include the northern and western portion of the Sangre de Cristo Mountains from approximately Crestone Peak northward to the Hayden Pass area, and two smaller areas in the southern Sangres near Carbonate Mountain and California Peak. The other area of identified calcareous geology on the Rio Grande National Forest is a small area of National Forest System land to the west of Villa Grove.

Geology/Sedimentation:

The geology and related processes that have occurred on and around the Rio Grande National Forest have resulted in two distinct zones of sediment size. This distinction is due in large part to the “pulling apart” of the valley by the Rio Grande Rift, and abundant volcanic activity that occurred about 30 million years ago. Sediment produced on the east side of the forest in the Sangre de Cristo Range is coarse due to a mix of sedimentary conglomerates, shales, and igneous intrusions. This material is more difficult to move with stream and overland water flow. However, when moved by erosion processes it will settle in wetlands, pool habitats and riparian areas. Once excess sediment is deposited in these low-energy areas, it becomes extremely difficult for streams and wetlands to remove it, so it can modify aquatic habitat and fill in wetlands and riparian areas. In contrast, the sediment produced by the volcanic activity on the west side of the valley in the San Juan Mountains is generally fine to medium in size and consequently easier to move with water. However, it still takes some energy to move fine sediments from low-gradient stream systems where they settle, and deposit them on floodplains as occurred historically. This process can be constrained by roads, trails, and other human activities that disrupt or impede the movement of water in low-gradient stream systems.

Hydroclimatic Regime:

Annual precipitation across the forest varies from 12 inches at lower elevations to over 50 inches at higher elevations. Generally, precipitation increases 4 to 6 inches for each 1000-foot gain in elevation. In the 12 to 18 inch zone (roughly 8,000 to 9,000 feet), most, if not all, the precipitation is taken up satisfying evapotranspiration demands so little to no runoff is generated; an exception can be when intense summer thunderstorms cause overland flow. Above 9,000 feet the forest is considered a snow-dominated system where the majority of the annual precipitation falls as snow, while the remainder is from late spring to early fall rains. Summer thunderstorms can produce short-duration, high-intensity precipitation.

The majority of the Rio Grande National Forest is located in the snow-dominated elevation zone. Aquatic productivity is relatively low at these high elevations, although aquatic species of plants can thrive if food, habitat and other limiting factors are not reduced significantly. Relatively short growing seasons and low temperatures for fish can result in failed spawning efforts. Connectivity, adequate water and timing, habitat and temperature are major limiting factors for aquatic and semi-aquatic taxa. Ensuring that important natural conditions of ecosystem elements maintaining the conditions for these taxa are critical for the continued survival of water-dependent species and ecosystem services such as angling.

There may be a rain and snow dominated zone at the headwaters of subwatersheds that originate at the lowest elevations of the Forest (Winters et al. 2016). These areas may contain plants and animals that are more representative of the valley floor. In addition, wetlands may also have different form and function if precipitation and resulting overland flow is modified. Given that the lower elevations of the valley receive little annual precipitation (approximately 8 inches per year), stream channels would be expected to be ephemeral or intermittent depending on the presence of groundwater. Wetlands would also be dependent on groundwater reaching the surface where rooting would allow wetland plants to exist.

Stream Gradient:

Stream gradient influences stream power, aquatic productivity, erosive capability, localized sediment sizes, and floodplain characteristics. As a result, stream gradient dictates the habitat for specific flora and fauna.

Steep-gradient stream reaches are often called "erosional" reaches while low-gradient reaches are called "depositional" zones. The sediment that is produced in the erosional zones is deposited in the depositional zones, making them indicators of upstream erosion. Indeed, steep stream channels have narrow floodplains and banks that are 'armored' with large rocks and boulders. They are very resistant to change, but move sediment through them, and are corridors for migration for aquatic species. The medium-gradient streams can be considered "transitional" reaches and have some of the characteristics of the steep- and low-gradient reaches. Low-gradient reaches are sensitive and responsive to upstream activities as well as local change. As a result, they typically exhibit changes in channel morphology, temperature, in-stream habitat attributes, and sediment accumulation. Their banks are typically comprised of small alluvium from upstream and are extremely susceptible to erosion if the deep rooting riparian vegetation is removed or damaged. Low-gradient reaches can act as "reservoirs" for storing water in the valley bottom, especially if beavers have colonized there. Terrestrial and aquatic animals are often much more abundant in low-gradient valleys, as habitat is diverse and abundant. Consequently, activities in the subwatershed with low-gradient valleys can have a dramatic impact on those valleys if they produce excess sediment, reduce woody material and change discharge dynamics. In addition, local impacts that remove woody vegetation, compact riparian soils and damage stream banks can have a profound impact on these important areas.

In the analysis by Winters et al. (2016), 8 of 162 subwatersheds on the Rio Grande National Forest are dominated by low-gradient stream systems. This is not unusual for the southern Rocky Mountains considering the steep topography and erosional process associated with the landscape. The role of stream gradient reaches and their importance could be a valuable tool for future monitoring. Low-gradient reaches are sensitive to change, as are the plants and animals that inhabit them. By focusing monitoring on these important reaches, species and physical characteristics could be identified and utilized as tools for observing change.

Glaciation and Wetland Development:

Glaciation has heavily influenced the aquatic, riparian, and wetland development on the Rio Grande National Forest. The Rio Grande National Forest contains thousands of acres of high quality wetlands and other groundwater-dependent ecosystems that provide essential services to the forest and lands downstream. A disproportionately large number of wetlands occur in the glaciated areas of the forest. Terminal or lateral moraines often create confined basins where impounded subsurface or surface water allow peat accumulation (Windell et al. 1996; Cooper 1990; and Cooper 2005), and kettle ponds promote fen formation along the fringes. In addition, glaciation has created wide, relatively level mountain valleys where large wetland complexes tend to form (Rocchio 2006). Understanding wetland occurrence in relation to the geomorphology of the forest can help guide our forest planning and management decisions pertaining to aquatic ecosystem sustainability, restoration, and climate change resilience. High mountain wetlands in glaciated areas are fundamentally different from wetlands occurring at lower elevations that were never glaciated (Cooper and Lee 1987).

Stream Density Analysis:

There are 23 subwatersheds completely the Rio Grande National Forest that are in the highest quantile for stream density, and several others mostly inside forest boundaries. There appears to be a high correlation between elevation and the high stream-density subwatersheds. The San Juan and Weminuche Wilderness areas have high densities of streams that may be beneficial for native fish and amphibian recovery. In addition, these subwatersheds are less managed and could continue to provide colder water to lower elevations, especially if the effects of climate change increases water temperatures at lower elevations. Stratifying the forest by stream density presents us with several important management considerations and options. First, these areas could be considered the highest priority for stream and riparian related management if other ecological conditions are not an issue. The relatively high number of these subwatersheds in the western part of the forest should be more productive, because of the characteristics of volcanic soils that make them highly productive.

Summary of the Ecological Driver Cluster Analysis:

Our analysis found that a total of 7 statistically similar clusters with discernibly different ecological characteristics could be identified on the Rio Grande National Forest. The top three cluster characteristics and others can be defined as follows:

- Approximately 33 percent of the forest (53 subwatersheds) can be characterized by the ecological conditions dominated by non-calcareous geology, high-gradient streams, fine sediments, and a snowmelt hydroclimate.
- Approximately 32 percent of the forest (52 subwatersheds) can be characterized by the ecological conditions associated dominated by non-calcareous geology, high-gradient streams, coarse sediments, and a snowmelt hydroclimate.
- Approximately 12 percent of the forest (20 subwatersheds) can be characterized by the ecological conditions dominated by non-calcareous geology, high-gradient streams, coarse sediments, and a snowmelt hydroclimate. The unique characteristic of this cluster is the considerable portion of streams containing low to medium gradients.
- Approximately 9 percent of the forest (14 subwatersheds) can be characterized by the ecological conditions dominated by non-calcareous geology, high-gradient streams, coarse sediments, and a rain and snow (72 percent) and snowmelt (30 percent) hydroclimate.
- Approximately 8 percent of the forest (12 subwatersheds) can be characterized by the ecological conditions dominated by non-calcareous geology, high-gradient streams, fine sediments, and a

snowmelt hydroclimate. The feature that separates this cluster from others is the high abundance of fine, volcanic soils that could be associated with erosion potential.

- Approximately 6 percent of the Forest (10 subwatersheds) can be characterized by the ecological conditions dominated by non-calcareous geology, high-gradient streams, coarse sediments, and a rain and snow (72 percent) and snowmelt (28 percent) hydroclimate.
- Approximately 1 percent of the Forest (2 subwatersheds) can be characterized by the ecological conditions associated dominated by a calcareous geology, high-gradient streams, coarse sediments, and a snowmelt hydroclimate.

Man-made Stressors

Man-made stressors can directly or indirectly degrade or impair key ecosystem characteristics important to aquatic habitat values. Several of the stressors listed and described below are also included in other assessments that may contain additional information.

Stressors Associated with Water Use:

Water originating from the Rio Grande National Forest is nationally recognized for its importance to uses and values to downstream users in the San Luis Valley and beyond. Both surface and ground water are highly valued on and off the forest and support numerous uses. Important man-made stressors associated with water use include:

Stream Diversions –

Stream diversions can seriously reduce instream habitat and migration for aquatic biota, nutrient transport, sediment movement and water quality, including temperature (Poff et al. 1997, Wohl, 2001). Water diversions during natural low flow periods can completely dewater streams, directly influencing the distribution and abundance of aquatic life (Auerbach, et al, 2012). Indirectly, stream diversions can influence channel geometry downstream. Peak flows during the annual snowmelt runoff maintain stream channels by moving sediment and scouring the streambed (Wohl, 2000). Reducing peak flows in low gradient, sinuous stream channels can cause channel narrowing, reduced channel capacity, and less overall instream habitat. Physical changes from diverting high flows also affect the organisms that evolved under the historical conditions. While the size of a diversion structure may be relatively small on the landscape, the influence to aquatic, riparian, and wetland resources may be very large if the appropriate issues are not addressed.

The results of the diversion density analysis conducted by Winters (2016) found that stream diversions are common in most subwatersheds on the forest. However, most of the diversions are also adjacent to the San Luis Valley floor, where considerable agriculture and subsequent water management occurs. These results indicate that a higher percentage of diversions at this scale are either on private land or relatively close to the public land boundary. The results indicate that only 7 (4 percent) of the subwatersheds located totally within the Rio Grande National Forest or the Great Sand Dunes National Park are without diversions. The diversion densities range from 19 to 0 diversions per permanent stream mile, with most of the quantile 5 (highest density) being on both public and private lands (32 total). The numbers of subwatersheds in quantiles 2 through 5 were almost equally divided. Diversion density in Quantile 5 is generally quite high, with 4 subwatersheds having more than 9 structures per mile of stream. The abundance of these structures could limit movement and habitat of fish adapted to the lower elevation such as the Rio Grande Sucker and chub. Quantile 4 subwatersheds average less than 2 diversions per stream mile, and are probably not evenly distributed; leaving connected habitat sufficient to support populations of fish and other aquatic organisms, if stream flows and associated water quality are adequate.

The lowest density of diversions is associated with the South San Juan, Weminuche and La Garita Wilderness areas. Diversions associated with these wilderness areas most likely predate the Wilderness Act (1964), and/or are on private land outside the National Forest boundary. The two subwatersheds in quantile 5 located in the southern portion of the study area are associated with the Conejos River, and agricultural land nearby.

There are 3 subwatersheds in quantile 5 that are located entirely within the Rio Grande National Forest boundary. Two of these are associated with the Conejos River, and agricultural land nearby. Another is located just southwest of the town of Creede, and contains a large flat area called Seven Parks. There is also considerable private land adjacent to the Rio Grande River where diversions associated with haying and other uses are occurring. The last subwatershed in quantile 5 is located in the northern portion of the Forest. There is little private land in this area, with only small headwater streams present. The area does not appear to be used for agriculture, but there are numerous unpaved roads. We need additional information to understand the uses associated with the high density of diversions in this area.

Water Transmission Ditches –

Water transmission ditches are used primarily to move water from one stream to another, to move water to some form of impoundment for later use, or to move water directly to the area where it is to be used (i.e. agricultural field). These ditches are important components of infrastructure that permitted Euro-American settlers to mine, ranch, and farm in the National Forests and Grasslands of the western United States (Wohl 2001).

There are about thirty other ditches and pipelines that divert water from forest streams for irrigation, recreation, and domestic purposes. These uses are not considered major.

Transbasin Water Diversions –

Transbasin diversions typically move water through tunnels under the basin divide or through ditches intercepting tributaries on one side of the divide and transporting it around mountain peaks into another basin. Transbasin diversions can have a whole host of impacts including transporting “new water” to the receiving basin. This may facilitate the invasion of non-native fish, plants and animals that may result in the loss of native species. Diverting water from one watershed to another can also result in physical, chemical, and biological changes to the receiving body of water. Some examples would be channel erosion, increased sediment transport, and increased sediment deposition in area reservoirs and stream channels. All of these impacts may lead to changes in channel geometry (Wohl 2001). Affected streams often exhibit elevated bank erosion, increased width-to-depth ratios, downcutting, and loss of instream aquatic organism habitat.

There are nine trans-basin diversions on the Rio Grande National Forest. Seven of the diversions add water to forest streams while the other two divert water from forest streams. All of the diversions are in the upper Rio Grande River basin. As a result, species that have evolved in other basins could not be inadvertently transferred. The uppermost subwatershed, West Willow Creek is diverted into the Farmers Creek subbasin. While Willow Creek exhibits modified stream flows, the receiving portion of Farmers Creek should be influenced from increased stream flows, as well as reduced flows at diversions into the Blue Creek subwatershed. The Blue Creek subwatershed also receives and diverts flows into the outlet of the South Fork of the Rio Grande River.

The diversion systems on the Rio Grande National Forest comprise a relatively small percentage of the permanent stream systems within their boundaries (the largest being 23 percent of the total permanent stream length). However, these systems may be moving significant amounts of sediment downstream as a result of erosion. Currently, there are no transbasin diversions from other river basins that could transfer unwanted species (including pathogens) and very different water chemistry and discharge into the Rio Grande system. In the future, transbasin projects identified within, as well as outside, the Rio Grande River system should consider the major biological and physical changes that can occur when they are implemented. It is extremely expensive to mitigate the effects of these types of projects, especially changes within receiving streams.

Reservoirs/Dams –

Reservoirs have been constructed in the Rocky Mountains for recreation, power generation, snow making, and flood control. The predominant use for reservoirs has been for agriculture and municipal consumption (Wohl 2000). Reservoirs were constructed as early as the mid-1800s to help facilitate the timing of downstream flows to coordinate with the agricultural growing season in the Rio Grande Valley.

Much of the surface water in the Rocky Mountains comes from annual snowmelt runoff from mountain areas. Consequently, large volumes of water are produced in the late spring and summer months, whereas flows are minimal (e.g., baseflow) during the remainder of the year (Wohl 2000). There is a relatively short period of time that ‘excess’ water is available in the stream system, while demand for excess water extends throughout the year. Reservoirs and water storage allows for a degree of control over where water can be stored to meet the demands of downstream users. Reservoirs were constructed west of the Continental Divide to meet local municipal and agricultural needs, but many reservoirs have been built to store water that is subsequently transferred east of the Divide via transbasin diversion structures. This has not been the case in the Rio Grande Valley, where agreements and policies have kept water in the Upper Basin.

Reservoirs can have both beneficial and detrimental effects on aquatic, riparian, and wetland resources. Dams clearly have an immediate influence on local conditions, and can influence resources many miles in either direction. The most noticeable influence of dams is the artificial lakes they create which inundate historic riparian or wetland areas and create a new and different environment. Animal diversity can be increased or decreased, depending on the system. For example, while some native species of plants and animals may not be able to tolerate the conditions created by reservoirs, some non-native species may thrive under those conditions, resulting in competition or replacement of native species. However, they are typically replaced with popular non-native recreationally valuable species. Many reservoirs in the Rocky Mountains are now dominated by non-native species of fish. While many of these species were planted to increase recreational opportunities, several, including the common carp (*Cyprinus carpio*) are undesirable in many areas (Baxter and Stone 1995). The influence of other non-native species, such as crayfish (*Decapoda*), mollusks (*Mollusca*), and bullfrogs (*Rana catesbeiana*) may also be significant.

Dams create barriers to migration of aquatic organisms (Ward and Stanford 1982). Populations of fish and other aquatic biota can be isolated upstream of reservoirs, limiting gene flow and potentially affecting population viability. In addition, the inability of fish to migrate upstream of a dam can limit their ability to reach critical spawning habitat. Overall, dams can severely fragment populations of mobile aquatic organisms, effectively creating a series of isolated communities.

The negative effects of reservoirs on downstream water quality are well documented (Wohl 2001; Allen 1995; Ward and Stanford 1982; Hynes 1970). In addition, the disruption of the nutrient, sediment, and woody debris transport can influence conditions downstream. For example, reservoirs act as nutrient and sediment ‘sinks’, limiting the downstream transport of these materials (Ward and Stanford 1982). When sediment is sequestered in reservoirs, downstream stream reaches may begin to erode their banks and scour their channels. Reservoirs also trap large woody debris, which keeps them from reaching downstream stream segments where they would normally influence channel and riparian form and function (Maser and Sedell 1994).

Dam structure and operating practices also affect downstream processes. The temperature and nutrient content of water exiting reservoirs may depend on whether water is released from the bottom or top of the dam. These effects can be dramatic; and algae, benthic macroinvertebrate, and fish communities can be altered significantly (Maser and Sedell 1994). Bottom release reservoirs can increase trout production in reaches downstream from reservoir dams because temperature fluctuations are stabilized and food supply increases. However erratic flows, gas supersaturation, and reduced spawning success due to low water temperatures can sometimes counteract the positive influences.

There are 21 major reservoirs across the Rio Grande National Forest. The largest of these are the Rio Grande Reservoir on the upper Rio Grande River and the Platoro Reservoir on the Conejos River. The inundated stream length under reservoirs in the upper Rio Grande basin is a relatively minor part of the total permanent stream length. The range of densities was 0 to 0.2 miles of stream inundated for the total stream miles within each subwatershed identified on the forest.

Current reservoir locations and abundance do not appear to be having widespread effects across the Rio Grande National Forest. Individual small reservoirs impact localized areas, while dams, both large and small, can serve as barriers for recovering native fish upstream. The presence of a dam acts as a barrier for non-native fish movement upstream, and the reservoir pool can maintain a “stable” environment for native fish.

In the future, there may be proposals to increase the size of existing reservoirs on National Forest System lands. While surface area of modified reservoirs could become larger and inundate more public land, the increase in volume could provide opportunities for modifying reservoir releases to improve downstream conditions.

Stressors Associated with Transportation

Roads:

Roads are widespread and permanent features of National Forest System lands. Roads and road networks under Forest Service jurisdiction can be managed to reduce or eliminate (in some cases) negative impacts to aquatic, riparian, and wetland ecosystems. The benefits that accrue to aquatic and wetland ecosystems by correctly designing, constructing, and maintaining roads in the National Forests are many. Properly designed, constructed, and maintained roads offer the following benefits to streams, riparian areas, and wetlands:

- reducing soil erosion and sedimentation;
- reducing soil compaction;
- reducing chemical contaminations of soil and water;
- maintaining water quality conditions conducive to supporting aquatic life;
- maintaining nutrient cycling in lotic and lentic environments;
- supporting the abundance of riparian and wetland plant communities; and
- promoting many additional benefits to stream, riparian, and wetland environments.

Appropriate road and travel management will be necessary on Forest lands, and include managing motorized use and removing any class of unwanted road.

On the Rio Grande National Forest, there are far more subwatersheds with unpaved roads identified in valley bottoms (86) than all the other quantile values combined (77) (Winters 2016). Paved roads are restricted to highways bisecting the Rio Grande Valley, both north to south and east to west. While there are paved state and county roads on the San Luis Valley floor, there are relatively few on the Rio Grande National Forest. The highest density of paved roads was 0.17 miles per stream mile within the Blue Creek subwatershed.

Road crossings can fragment water and riparian ecosystems and isolate populations of plants and animals. The result can be population loss if combined with other impacts such as fire or contaminants. In addition, if too few individuals are isolated inbreeding can occur, resulting in low reproductive success and ultimate elimination. Many fish and wildlife species will not pass through a poorly designed road crossing. This isolates fish, and results in road fatalities for terrestrial animals and some amphibians. Fish, amphibians, and other wildlife will use properly designed crossings. Fish, aquatic macroinvertebrates and some amphibians need stream crossings designed so they can swim upstream or downstream through them. Stream velocity, depth, habitat, etc., all need to be considered for proper movement.

Road crossings on the Rio Grande National Forest are associated with the main travel corridors in the basin. The higher values on the east side of the basin are mostly to the west of the Sangre de Cristo Mountains, except the Cuchara Pass road. Subwatersheds on the western side of the drainage are mostly associated with highway 160, over Wolf Creek Pass and highway 149 thru Creede.

Unpaved Roads:

There are far more unpaved than roads in the valley bottoms on the Rio Grande National Forest. While 31 subwatersheds are in the highest quantile for road density in the valley bottoms (app. 0.4 miles per stream miles within the valley bottom), only 12 have no unpaved roads. The lowest densities were associated with the wilderness areas in the western part of the forest and the steep Sangre de Cristo Mountains on the eastern side. The highest values were in the northern portion of the basin, although unpaved roads appear to be common throughout the forest associated with the valley bottoms. While the highest density of unpaved roads may appear relatively limited (0.4 miles per stream mile in the subwatershed), it is important to note that the valley bottom represents a limited amount of a subwatershed and the overall total density is most likely considerably higher if the uplands are included. The density of unpaved roads is definitely skewed towards more roads within the basin than less. As mentioned previously, unpaved roads in these areas can have significant direct and indirect influences on riparian ecosystems due to their position, the need to remove trees that would fall naturally into the stream channel and riparian areas, and the way roads restrict lateral stream movement. Instream habitat is compromised over time as a result of all of these factors.

Understanding the influence, both directly and indirectly on aquatic and semiaquatic systems, of unpaved roads is paramount to healthy ecosystems. Those subwatersheds that are located in quantiles 4 and 5 could be considered priorities for evaluation and subsequent treatment if high-value aquatic resources are identified. Low-gradient reaches previously identified as ecological drivers could be considered the highest priority due to their relatively large riparian areas, sensitivity to temperature changes, natural water storage capacity in the alluvium and abundant instream habitat. By focusing on low-gradient reaches as a high priority, more than 50 percent of the stream reaches could be considered lower priorities until low-gradient areas are more resilient.

Unpaved Stream Crossings:

Stream crossings are associated with most unpaved roads in the valley bottoms. Historically these crossings were fairly primitive, being constructed of corrugated culverts, stream fords or small bridges. The results of this analysis display values ranged from 0 (15 subwatersheds, under 10 percent) to 1.3 crossings per stream mile in the West Fork of Pinos Creek subwatershed. A total of 20 of the 30 subwatershed within quantile 5 are located totally within the Rio Grande National Forest. These results are considerable different than the paved road results and indicate a relatively high number of unpaved roads associated with valley bottoms and associated streams and riparian areas. While not equally distributed, there are likely areas that are considerably more concentrated than others.

Road systems have been developed on National Forest administered lands for a number of management related activities. As a “multi-purpose management” agency, densities could be higher than on adjacent lands where management is not as diverse. Historic and current vegetation management, grazing, water development and access for recreation all require transportation routes to access specific areas. Often these roads are maintained, with other roads eventually built to access new sources of interest. Past maintenance and construction techniques reflect the prevailing lack of knowledge regarding the ecological importance of connectivity within riparian and aquatic ecosystems at the time.

Due to many limitations, we would need to prioritize important subwatersheds for passage issues using the results of the ecological driver analysis, and proposed fish and other riparian reintroductions. In many areas we have relocated road systems where crossings are unneeded. Due to our limited funds, it is extremely important that we focus on the most important areas first. In addition, over time we can reduce the funds needed to rebuild damaged crossings as maintenance costs would go down significantly. There could be new areas for reintroducing native fish that would benefit greatly by removing barriers to movement and help restore populations.

Proposed new roads should be engineered in accordance with current science and applicable management values for the area related to erosion, fragmentation and restoration. We also need to identify high priority areas where construction could be more sensitive to valley floor ecology and species needs as well. There are currently building specifications that include sensitivity to important values on public lands. The incorporation of new design criteria to ensure proper stream function and biological processes are now available and should be considered in these designs.

Trails:

The results of our forest trail analysis indicate that most identified trails are located in designated wilderness areas with the highest density located in the South Fork Saguache Creek and North Fork Conejos River. There are 79 subwatersheds without trails, which is by far the highest number of any quantile (0.16 miles per stream mile in the valley bottom per subwatershed). These results imply that there are few trails in the valley bottom. However, in analyzing the GIS, we see that a large percentage of the few trails there are, are located adjacent to streams. An example would be the Ute Creek system of trails in the Weminuche Wilderness (USDA Forest Service Recreation Map). Trail systems exist along all major tributaries and the mainstem from its confluence with Rio Grande Reservoir to the Continental Divide and high elevation lakes almost 9 miles to the west. The large number of tributaries is reflected in the low density value. The area around the community of Crestone, and the popular “Crestone Needles” located on the Pike San Isabel National Forest also contains subwatersheds in the highest 2 quantiles, due to the steep topography and lack of roads. This is a popular area for recreationists during the summer months.

There are benefits that accrue to aquatic, riparian, and wetland ecosystems if land managers properly design and maintain (e.g., best management practices) trail systems. The primary benefits to aquatic ecosystems are reduced local degradation to these ecosystems due to trail construction, trail use, and trail maintenance; and increased difficulty in introducing invasive species. Riparian and wetland vegetation is effective in reducing erosion and sedimentation in streams and in retaining soil moisture. Plant-root networks provide soil stability and soil cohesion.

OHV Use:

Of the 163 subwatersheds on the Rio Grande National Forest, 113 (about 70 percent) do not have trails in the valley bottoms, and most are at lower elevations. There are 11 (about 7 percent) in the highest quantile, one of which is the headwaters of La Garita Creek in the south west portion of the forest. All the other trail systems have relatively few miles of trail in the valley bottoms. These results are based on identified trails identified by the Rio Grande National Forest and do not include user developed trails that could be present. As in all of the transportation, user created roads and trails are typically present, and enforcement is limited. However, the limited amount of trails in the valley bottom indicates that this use is probably limited and isolated.

Based on our assessment results there are few trails in the valley bottoms. That does not mean that there are not isolated areas where OHV use could be causing resource damage. As with all forest uses, OHV use should be monitored to ensure streams and wetlands are not compromised. Discussions of opening new trails to OHV use should consider the monitoring, maintenance and ecosystems in these areas. OHV technology continues to improve; with increased size, horsepower, speed, and aggressive tread designs; increasing the potential for resource damage. However, not all OHV use is damaging. OHV groups maintain many areas where they enjoy riding, and like many other activities, a limited few unethical users can potentially impact aquatic resources.

Additive Influences of the Transportation Group:

One subwatershed had no transportation activities identified within its boundaries. There is a trend toward less motorized activity associated with wilderness areas in the western and a portion of the northern part of the forest. Most of the highest additive levels (quantiles 4 and 5) are found associated with areas of higher populations such as Creede, South Fork and Del Norte. These results could be artifacts of early development to access natural resources needed to maintain infrastructure; and in the case of Creede; to access mining areas.

The results of the transportation category show that transportation routes are located throughout the valley bottoms of the Rio Grande National Forest. These activities have occurred for the last 200 plus years, beginning with primitive trails to wagon trails, to a wide range of transportation routes. Our results indicate that the cumulative ranks of transportation is highly skewed towards more than less activities, showing that the conditions within valley bottoms are conducive to transportation development. Future considerations could be given to limiting routes in these sensitive areas, as well as ensuring that the processes for maintaining healthy riparian areas and streams, and natural storage in low gradient reaches are maintained. In addition, as discharge levels should become more variable, maintaining the quality of water related habitats, and reducing the effects of flooding on infrastructure is going to be important.

Recreation: Developed Recreation Sites

Recreation camp sites are often located in valley bottoms to afford visitors direct access to water related activities as well as the aesthetic value of streams. Often, however, these sites impinge on riparian areas, can cause erosion and damage stream function. The severity of effects is relative to the amount of use and the location of the sites. Out of a total of 163 subwatersheds evaluated, 118 or 73 percent of the subwatersheds evaluated have no developed recreation sites in the valley bottoms. Most of the higher densities (albeit very few) are in subwatersheds that include the Rio Grande and South Fork of the Rio Grande Rivers, mainly due to the high recreation use such as fishing.

The relatively “flat” nature of valley bottoms as well as close proximity to water have made building developed recreation a high priority. However, sites are increasingly being located away from direct contact to riparian areas and streams. One of the problems is that valley bottoms are often relatively small, and roads bisect the area. Crossing roads to enjoy the activities associated with the streams leads to unsafe conditions that are often difficult to mitigate. Enclosed toilets that do not seep into streams and associated water tables are a major advancement in design. Locations away from active riparian areas still within the valley bottom allow streams to migrate laterally and discourage trailing. While flooding can still be an issue with large events, more thought and planning is being given to location. In the valley bottoms, the closer developed recreation sites are to active riparian and streams, the more likely that some resource damage could occur. Areas identified outside of valley bottoms would be easier to maintain, and have less risk to aquatic ecosystems and from flooding.

Ski Areas –

Only one active ski area is currently located within the Rio Grande National Forest, Wolf Creek Ski Area at the top of the South Fork of the Rio Grande River.

Ski areas are generally considered a culmination of human activities. Sediment from large parking lots often enter stream channels, snow making requires water at times when very limited water is available in streams, nutrients and chemicals enter streams from deicing practices, treated sewage and other sources, etc. While recreational skiing is one of the highest use activities in the state of Colorado, careful consideration should be taken to ensure that impacts to high elevation wetlands and other aquatic ecosystems are not encouraged. Glaciated valleys, identified in the ecological driver section as having high densities of wetlands are a major concern as they are often targeted for development. Sediment runoff can also be routed to settling ponds before they reach stream channels where they settle and can considerable ecological damage.

Stressors Associated with Biological Considerations

Aquatic Nuisance Species:

Weed free hay is one technique that has been used to minimize invasive plants from entering wilderness areas. Using a similar idea for stocking trout in wilderness areas and other water bodies within Forest boundaries is also an important topic to be discussed with partners.

Because aquatic invasives are typically transferred by recreational equipment such as boats, wading boats and other items that carry water from one body of water to another, education on decontamination and inspections at high volume reservoirs are crucial. Other equipment that has the potential to transfer water from one infected water body to another, such as firefighting equipment, need to be identified and techniques used to ensure invasives are not transferred.

Influence of Beaver:

Beaver (*Castor canadensis*) are habitat-modifying keystone species in lotic, riparian, and wetland habitats (Butler 1995; McKinstry et al. 2001; Collen and Gibson 2001), and their dam-building activities can influence a large proportion of a given watershed and landscape (Naiman et al. 1988). Beaver are an integral part of most headwater stream ecosystems in the Intermountain region (Wohl 2001), where their activities can alter stream channels and riparian zones and affect basic processes such as nutrient cycling (Naiman et al. 1988; Naiman et al. 1994). Beaver impoundments generally have a positive effect on aquatic, riparian, and wetland habitats because they:

- elevate water tables and enhance riparian vegetation (Olson and Hubert 1994),
- create deeper water habitats with velocity and thermal refugia (Hagglund and Sjoberg 1999),
- decrease water velocities and promote sediment retention (Naiman et al. 1988),

- improve water quality by facilitating riparian habitats that intercept nutrient and chemical contaminants (Olson and Hubert 1994), and
- increase upstream water storage to buffer against floods and summer droughts (Olson and Hubert 1994).

These beaver-induced hydrological and biogeochemical changes also affect aquatic, riparian, and wetland biota. For example, beaver ponds exert strong influences on aquatic invertebrates (McDowell and Naiman 1986; Clifford et al. 1993; Margolis et al. 2001), fishes (Snodgrass and Meffe 1998; Hagglund and Sjoberg 1999), herpetofauna (e.g., amphibians and reptiles: Russell et al. 1999; Metts et al. 2001), and birds (Brown et al. 1996; McKinstry et al. 2001).

Beaver impoundments often facilitate sediment deposition and storage (Wohl 2001), but in their absence, sediment might be transported further downstream and be deposited in crucial fish spawning habitats. Rocky Mountain region hydrographs are likely affected by loss of beaver impoundments because water storage capacity in first through fourth order streams is reduced. Thus, water is delivered more quickly in the absence of beaver dams, rather than being slowly released to buffer against periods of low flow.

Researchers at the University of Wyoming identified habitat suitability characteristics necessary for the colonizing of beavers in mountain streams. They modeled suitable beaver habitat using GIS (Olson and Hubert 1994). Model parameters used existing vegetation information, and National Hydrography Dataset stream layers included:

1. Gradient up to 4 percent
2. Within 200 meters of aspen or willow vegetation type
3. Stream order less than 4

Stream reaches that had high correlation with these variables were mapped on a 1:24,000 scale and densities were calculated on a total stream mile bases for each subwatershed. Beavers have been trapped in irrigation systems on the valley floor by the Colorado Parks and Wildlife and periodically transplanted into Rio Grande National Forest streams (Randy Ghormley, Forest Fish and Wildlife Program Manager, pers. comm.). Approximately 22 subwatersheds are located within the National Forest and National Park boundaries, several relatively far from private lands or the valley floor. Further evaluation at the site or reach level would ensure survival, low migration and reduced conflicts if the beavers migrated from the area.

Only one subwatershed on the Rio Grande National Forest has no adequate beaver habitat, and is located in a steep portion of the very southern edge of the National Forest and beyond. Based on these analysis results, there is an opportunity for beaver to play a major role in achieving watershed health objectives on the Rio Grande National Forest.

Grazing:

The reference condition prior to European settlement included effective populations of ungulate predators. Human manipulation of ungulate and predator populations is a significant stressor on watershed, riparian, and stream channel function. Ungulates without effective predators are known to overgraze the riparian vegetation necessary to provide bank stabilization and a food source for beavers. Willows (*Salix* spp.), alders (*Alnus* spp.), and quaking aspen (*Populus tremuloides*) are often browsed to an extent that recruitment levels fail to sustain a resilient system. Deciduous components are preferentially consumed allowing conifer encroachment (Roger and Mittanck 2014). This results in a cascading effect that reduces soil organic carbon, which reduces the soil’s water holding capacity (Shepperd et al. 2006; Woldeselassiea et al. 2012), which promotes warm season bunchgrasses over cool season bunchgrasses. Eventually, overgrazing removes bank stabilizing vegetation, creating channel downcutting and a dysfunctional floodplain (Beschta and Ripple 2006).

Livestock grazing has the potential to significantly influence riparian, wetland, and aquatic resources if it is not managed properly (Binkley and Brown, 1993). There are numerous references to document these potential influences, although most discuss impacts from improper rather than proper grazing management (Kauffman and Krueger 1984; Buckhouse 1981; Meehan and Platts 1978; Binkley and Brown 1993; Larson et al. 1998). Livestock grazing be a tool for positive effect when managed properly and when focused on meeting well defined interdisciplinary objectives. Grazing can:

- stimulate new growth of both herbaceous and woody species;
- increase total production;
- provide increased palatability and nutrient quality to other animal grazers;
- increase herbaceous plant density, and
- alter habitat structure and composition to meet specific species objectives (such as managing for specific threatened or endangered species habitats or alteration of habitat relationships to favor certain species) (Krueger and Anderson 1985).

See the Multiple Uses Assessment for more information on range, including allotments and stocking densities.

When evaluating the influences of livestock grazing at the reach/site scale, the following data collection methods should be considered:

1. Residual stubble height over time.
2. Green-line cross-section and woody retention on selected reference areas of interest.
3. Invasive species inventory relative to livestock grazing influences.
4. EPA stream bank stability rating relative to livestock influences.
5. Proper functioning condition assessment as the findings relate to livestock management practices.
6. Channel morphology changes relative to livestock influences.
7. Water quality and/or macroinvertebrate indexes as the findings relate to livestock management practices.
8. Degree to which terms and conditions from the grazing permit (and associated plans or instructions - such as allowable use, pasture timing requirements, etc.) are met.

Historically, improperly managed livestock grazing impacted many landscapes. In some instances, these impacts have persisted through time and can be seen in down cut stream channels and altered plant communities, especially on low gradient riparian hardwood communities. For the most part, current influences of livestock grazing are localized and are limited on a landscape basis. However, certain aquatic, riparian, and wetland areas continue to show impacts. Frequently these are the lower gradient areas both with historical carry-over effects and/or current influences from other human activities (recreation, forest management, road management, big game, invasive species, and so forth).

Seeps and Spring Developments:

See information in section 12.14. Also see climate change section, below.

Timber and Vegetation Management:

Timber management and associated activities (such as roads associated for timber management) can have an influential effect on aquatic habitats and species. Sedimentation is a key concern associated with timber harvest and road use. Project design criteria associated with the Water Influence Zone are intended to eliminate or minimize sedimentation risks. However, the application of streamside buffers and other design criteria needs to be flexible to incorporate various stream functions depending upon the type of stream and stream gradient involved. For example, lower-elevation stream gradients can be more sensitive to man-made stressors and sedimentation risks than higher-gradient streams, and may warrant additional considerations to maintain floodplain function. Timber management can also directly influence recruitment of downed logs and woody material into streams and along floodplains, which is critical for stream function and aquatic habitat integrity. We need to ensure that wood recruitment remains a key ecological process of both perennial and ephemeral systems. We should address the importance of healthy stream conditions associated with side channel stream refugia given potential climate change.

Wildfire:

The dynamics of aquatic systems are largely driven by topography, climate, and the patterns of disturbance events such as wildfires and large storms (Bisson et al. 2003). Although wildfires can exhibit both short- and long-term influences on aquatic systems, they are also critically important to their maintenance, resiliency, and productivity (Rieman et al. 2003, Bisson et al. 2003). In a review of the current knowledge concerning fire and aquatic ecosystems in the Western United States, Bisson et al. (2003) suggest that wildfire disturbances are inevitable in most landscapes and that the primary means of reducing their short-term negative effects is to conserve and restore the physical and biological processes and patterns that create and maintain a diverse network of aquatic habitats and populations.

Aquatic systems and the terrestrial landscapes that encompass them are recognized as being intricately linked and structured by the disturbance processes associated with particular fire regimes (Bisson et al. 2003). In Fire Regime V systems (200 years and over return interval), these processes primarily involve periodic bark beetle epidemics and windthrow events as well as infrequent, large-scale, stand-replacement fires (Veblen et al. 1991b, Veblen et al. 1994, Schmidt et al. 2002). Climatic factors had and will continue to have a profound influence on terrestrial and aquatic systems and their interactions with disturbance processes such as wildfire (Bisson et al. 2003). However, one of the primary factors that influence the significance of a wildfire on aquatic systems is the pre-fire condition of the watershed. Watersheds that have been highly roaded and influenced by past management activities present the highest risk to maintaining diverse and connected aquatic populations in the post-fire environment (Bisson et al. 2003. Minshall 2003, Benda et al. 2003). For example, some of the impacts of the Hayman Fire in Colorado are attributed to the pre-fire condition of the watershed, where the timing and recovery of the aquatic system is expected take longer in drainages that had been subject to chronic human disturbances (Kershner et al. 2003). A well-distributed beaver populations can also help minimize the impacts of an initial fire event as well as alleviate potential post-fire effects associated with flooding and soil erosion (Decker et al. 2003).

Post-fire erosion can also occur chronically and episodically. Chronic erosion is characteristic of roaded environments, revegetation difficulties, or ground-disturbing fire suppression activities and tends to deliver fine sediments over long periods of time. Chronic pre- and post-fire disturbances that continually deliver sediment will extend the time period needed for recovery (Bisson et al. 2003, Kershner et al. 2003). In contrast, pulse events deliver sediment and large coarse woody debris through landslides or debris flows and are generally short lived (Kershner et al. 2003). The coarse woody debris are gradually depleted as they decay, break up, and are transported downstream where they contribute to habitat values for aquatic invertebrates and fish species (Bisson et al. 2003, Minshall 2003). Pulse events involving sediment and coarse woody debris through fluvial transport are typical of mountain drainage basins in

North America and important due to their contribution to the physical and biological diversity of aquatic systems. These events do result in temporary impacts to the aquatic environment but are also considered fundamental to the development and restoration of stream systems (Benda et al. 2003). This was exemplified in a post-fire landscape in Idaho where stream tributary confluences influenced the receiving channels by rejuvenating alluvial fans, boulder accumulations, coarse woody debris supplies, terraces, side channels, and floodplains (Benda et al. 2003).

Based on outcomes from past local wildfires such as the 2002 Million Fire, it is possible that hydrologic pulse events could be associated with individual fire catchments in some areas of the forest if a wildfire occurred under the severe climatic conditions such as those that occurred at that time. The significance of these events could also be further influenced if they occurred in areas with sensitive soils or high amounts of roads. If hydrologic pulse events do occur, local populations of fish could be extirpated and the macroinvertebrate communities in first to fourth-order streams may be reduced by as much as 85 to 90 percent in highly affected drainages. The post-fire macroinvertebrate communities may also shift from consumers to disturbance-adapted strategists and habitat generalists due to a decrease in leaf litter and other food resources (Spencer et al. 2003, Minshall 2003). However, the macroinvertebrate community usually responds quickly and returns to pre-fire conditions within a year or two, although wide variations may continue for 5 to 10 years post-fire (Minshall 2003). Some studies have indicated that burned streams will actually result in a greater abundance, species richness, and diversity of macroinvertebrates for up to 45 years post-fire (Albin 1979 *in* Menshall 2003). However, most studies indicate a convergence with unburned reference streams within a 10 to 15 year timeframe. The recovery of fish populations also generally track the initial increase in primary and secondary production and usually show a positive response within the first five years if connectivity with key refugia occur within the watershed (Kershner et al. 2003). Fish will generally recolonize fire-affected areas rapidly when barriers are not a problem. New fish colonists generally come from upstream of the fire area, however, they will also move from surrounding watersheds and mainstem systems if migration and aquatic passage is not limited (Kershner et al. 2003). The effect of pulse events may vary, as evidenced by the West Fork Fire Complex of 2013 which at 88,817 acres exceeded the Million Fire several-fold. This most recent wildfire burned approximately 88,800 acres on the Rio Grande National Forest during dry and windy climatic conditions, most as stand replacement fire at high to moderate intensity. Although hydrologic pulse events involving sedimentation and fish kills occurred in at least one tributary during the year after the fire, effects to the main channel of the river were negligible and fish populations were higher than they had been in nine years.

Thus, although temporarily dramatic, the overall effect of wildfire on macroinvertebrates and fish in intact, unfragmented stream ecosystems is that they usually experience recovery in the short to early mid-term (Minshall 2003). In the long term, the general consensus in the literature is that within intact systems the disturbances associated with fire-related flooding, sedimentation, and woody debris inputs are important to the integrity of aquatic systems and at times represent the restoration activity itself (Bisson et al. 2003, Minshall 2003). In fragmented populations such as native cutthroat trout on the Rio Grande National Forest, however, protection of small core populations may be necessary when high-intensity wildfire occurs.

Places at Risk

Assessment 2 notes that in general most watershed and stream health concerns on the Rio Grande National Forest are associated with existing system roads and trails, uncontrolled motorized use on unauthorized roads and trails, and livestock grazing in riparian areas. These concerns relate to accelerated soil erosion and compaction, sediment delivered to streams and wetlands, stream channel aggradation and degradation, and the direct, indirect, and cumulative effects on the water cycle and water quality. These concerns are primarily related to proper implementation and/or administration of existing forest plan direction rather than a significant need to change existing direction.

This aquatic assessment provides additional information important to understanding the ecological drivers of the aquatic habitats on the forest, as well as the natural and human stressors important to the continued ecological integrity of the aquatic and riparian systems. The results include additional information pertaining to the importance of certain geologic features on the landscape, such as past glaciation for wetlands and fens and potential differences in stream productivity based on the underlying geology. This assessment also offers additional insight regarding the vulnerabilities of various stream types and aquatic habitats, with low (under 2 percent) and perhaps medium (2 to 4 percent) stream gradients warranting additional considerations from a management perspective, particularly where stream channels have been constricted or human stressors influence floodplain conditions.

This aquatic assessment also offers additional information regarding the unique risks to aquatic integrity as associated with potential climate change. These risks should be viewed in conjunction with the ecosystem drivers that maintain aquatic systems and habitats. Although all ecosystem processes that maintain aquatic integrity warrant review during plan revision, low-elevation stream types and seeps and springs within drier vegetation communities are uniquely vulnerable to climate change and human stressors associated with them.

Need for Change

Based on information in this section of the assessment, there is a need to revise and update the aquatic habitat section of the Forest Plan. For the purposes of forest planning, considerations for aquatic habitats and species are combined with soil and water resources. However, the information in this assessment for riparian and aquatic ecosystem integrity, highlights some specific needs with aquatic habitats and species on the forest, especially in regards to the unique aquatic endemics to the upper Rio Grande Basin. Examples include the need for additional management direction addressing the importance of stream connectivity and aquatic organism passage programs and their relationship to aquatic habitat resiliency, and incorporating the Aquatic Nuisance Species Plan for the Rio Grande National Forest into plan revision components. Potential mitigation strategies for chytrid fungus and local amphibian species are part of this recommendation. We need additional updates regarding restoration opportunities for native aquatic species and for managing low-elevation riparian systems, seeps and springs; including hummocking and pugging guidelines. Updates should also increase awareness regarding the ecological drivers for local aquatic habitats including glaciation, stream gradient, and geological factors associated with aquatic habitat productivity.

We need to update the existing Forest Plan with current information related to aquatic habitats and potential vulnerabilities associated with climate change. Central to this update is the need to maintain cold water systems and side channel refugia for aquatic species. We need to acknowledge the connection between current human stressors and a potential increase in vulnerability to aquatic integrity as related to climate change. Thus, while our standards and guidelines for aquatic systems should be adaptable and flexible they should also be well defined, firm and measurable.

Stressors Associated with Potential Climate Change

Aquatic systems and water dependent taxa will be significantly affected if the trend for changing temporal and spatial scales in air temperature and patterns of precipitation continue (Bryson et al, 2008). Potential effects of climate change in the southern Rocky Mountains include:

- reduced precipitation,

- reduced period of snowmelt runoff,
- attenuated periods of base flows in streams,
- increased water needs by the predicted increase in Colorado's population, and
- increased stream temperatures (Reiman and Isaak, 2010).

There are several other indirect impacts, such as an increase in non-native species (Rahel 2010) and reduced overall adequate habitat for cold water species. Some scientists speculate that increased water temperatures in higher elevations will be advantageous for cold water native fish. However, instream habitats are more limited in the smaller streams found there; food animals such as benthic macroinvertebrates may not tolerate even a limited change in water temperature, and reduced precipitation and more sporadic precipitation events could lead to drought.

We used a model recently developed by the USDA Rocky Mountain Research Station (http://www.fs.fed.us/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml) that incorporates current and projected variables to estimate changes in mean annual discharge in the Rio Grande National Forest for the future (See appendix B, posted separately).

It is also the intent of this analysis to understand and potentially prepared for "short-term drought situations" that occur in the Southern Rocky Mountains (McKee et al. 2000). While these events are measured in years, and followed by periods of "normal to high" discharge, the effects on aquatic ecosystems could mimic to a limited degree the effects of future long-term climate change (figure 6). The resiliency of aquatic ecosystems in the future may be reflected to a limited degree in the effects of short-term drought conditions. If the components that affect the resiliency of aquatic ecosystems are not in place or at a much lower level of condition than would be expected, even short-term droughts could have major impacts on aquatic systems. If we are able to substantially increase resiliency, the onset of climate change impacts may not be realized until later in the future, and the effects may not be as intense.

Climatic variables influence hydrological processes; so any change in precipitation, evapotranspiration, snow accumulation and snow melt will influence recharge to groundwater systems. Future climate change will affect recharge rates and, in turn, groundwater levels and the amount of groundwater available to support springs and wetlands (Ludwig and Moench 2009). Land use changes can also alter watershed conditions and generate responses in biological communities and processes. Land use changes may even override hydrologic modifications caused by large-scale climate shifts.

Lemly reports that 65 percent of all mapped national wetland inventory acres occur in the subalpine ecoregions, and another 29 percent of national wetland inventory acres occur in the alpine zone. These ecoregions also correspond to glaciated areas. There are few wetland acres in lower elevation zones.

In the western United States, winter and spring air temperatures have increased (Folland et al. 2001), the onset of snowmelt is earlier, and the snow water equivalent of snowpack has decreased. These and other indicators are especially important for groundwater dependent ecosystems, and have implications for groundwater recharge; which maintains springs, wetlands, soil moisture, and stream baseflows. In mountainous terrain, snowpack is the main source of groundwater recharge (Winograd et al. 1998). Higher minimum temperature can reduce the longevity of the snowpack, and decrease the length of time aquifer recharge can occur in various lithologies, potentially leading to faster runoff and less groundwater recharge. Decreased groundwater recharge can negatively affect aquatic biota and reduce wetland habitat. Unfortunately, we lack comprehensive data on groundwater-surface water interactions which limits our ability to model current groundwater resources and predict changes due to climate (Drexler et al. 2013).

The hydrogeologic setting is important to consider when assessing the climate-induced changes to groundwater systems. Geologic units respond differently to changes in precipitation due to their hydrogeologic properties (hydraulic conductivity, primary vs secondary porosity, fractures vs porous media, etc.). On the Rio Grande National Forest we can delineate several different hydrogeologic settings (igneous/metamorphic, volcanic aquifers, and unconsolidated glacial/alluvial aquifers).

The properties of aquifers are essential to consider; small, shallow unconfined aquifers respond more rapidly to climate change, whereas larger and confined systems have a slower response. Alpine glacial deposits generally support small, unconfined and shallow aquifers. These are more likely to have renewable groundwater on shorter time scales and will be particularly sensitive to changes in climate (Winter 1999, Sophocleous 2002).

The recharge characteristics of different hydrogeologic settings also influence vulnerability to climate change. For example, igneous and metamorphic rocks with low permeability and porosity, low volume groundwater discharges to groundwater dependent ecosystems, and are recharged only during large infrequent precipitation or snowmelt events; may be less influenced by climate change than unconsolidated or basalt aquifers that have high permeability and porosity, larger volume discharges to groundwater dependent ecosystems, and are recharged more frequently.

The effects of climate change on groundwater dependent ecosystems differ as a function of local geology and specific climatic variables. In a study that combined aerial photodocumentation (over 50-80 years) and climate analysis, Drexler et al. (2013) showed that five fens in the Sierra Nevada (California) decreased 10 to 16 percent in area. This decrease in area occurred over decades with increased annual mean minimum air temperature and decreased snow water equivalent and snowpack longevity. However, two fens in the southern Cascade Range, underlain by different geology than the Sierra Nevada, did not change in area. The volcanic lithologies of the Cascade sites, with high groundwater storage capacity, likely affected groundwater recharge.

Changes to groundwater levels can alter the interaction between groundwater and surface water (Hanson et al. 2012). Groundwater storage acts as a moderator of surface water response (Maxwell and Kollet 2008). Climate-induced changes in groundwater/surface water interactions will directly affect stream baseflows and associated springs and wetlands (Earman and Dettinger 2011; Kløve et al. 2012; Candela et al. 2012; Tujchneider et al. 2012). Changes in groundwater can change wetland water balances, leading to lowered water level and reduced groundwater inflow. For terrestrial and riparian vegetation, a shift in location, as well as in species composition, can occur.

A reduced average groundwater level tends to promote soil aeration and organic matter oxidation. Generating and maintaining peat soils over time depends on stable hydrological conditions, and in recent studies of peatlands exposed to groundwater lowering, soil cracking, subsidence and secondary changes in water flow and storage patterns are noted (Kværner and Snilsberg 2008, 2011).

Freshwater springs are dependent on a continuous discharge of groundwater, and form subsurface to surface water and aquatic to terrestrial ecotones, which are important components of aquatic biodiversity (Ward and Tockner 2001). Springs and spring brooks are physically stable environments that support stable biological communities (Barquin and Death 2006). Biota with low dispersal abilities and long generation times are more common in permanently flowing springs, whereas biota with strong dispersal ability are favored in non-permanent habitats (e.g. Erman and Erman, 1995; Smith and Wood, 2002). Climate change-induced modifications to recharge may have a profound impact on spring communities. Such changes may be reflected in reduced summertime flows with possible drying, and increased winter flow and associated flooding, which can affect biological communities (Green et al. 2011).

Management Considerations for Climate Change

Managing aquatic systems begins with inventory of the types and condition of existing resources. The Forest Service has published several guides to inventory of aquatic systems including the Groundwater Dependent Ecosystem Inventory Field Guides (USDA Forest Service 2012). These field guides provide protocols for inventorying palustrine wetlands and springs.

Land uses surrounding palustrine wetlands and springs can potentially alter the hydrology and nutrient inputs of these systems, thus changing their underlying processes. Draining, water developments, heavy cattle use, ORV use and irrigation practices can alter hydrology, reducing species diversity. Increased land use within 100 meters has been found to be correlated with increased nutrient levels in peatlands, suggesting that setbacks should be 100 meters or more for adequate protection (Jones 2003).

Heavy cattle use in palustrine systems can alter the hydrology by damaging soils. Soil compaction and pugging within the peat layer will change surface water flow. Heavy cattle use can also alter the successional processes within the sedge- dominated area of a fen. Cattle hoof action can lead to pugging and hummocking, creating microsites where shrubs can become established, changing the sedge-dominated meadow to carr shrubland.

Projected Future Change in Mean Annual Flow for Streams within the Rio Grande National Forest and surrounding ecosystem: Focus on Climate Change

Climate change analysts predict future changes in streamflow through altered temperatures, precipitation patterns and frequency of catastrophic weather events. This will in turn impact physical processes and organisms not only in these streams but throughout the ecosystems these streams feed (Reiman and Isaak 2010). Fish and other aquatic organisms would be most directly and dramatically affected by changing hydrologic flows, particularly those species adapted to specific flow regimes (Wenger et al. 2010). To estimate future change in flow metrics for the Rio Grande National Forest, we used modeled mean annual flow metric data (Wenger et al. 2010) which predict flows in 2040 and 2080 (figure 1 and figure 2). By predicting where flows will change most dramatically across the Rio Grande National Forest due to climate change, we can identify areas for management focus.

The most dramatic reduction in projected flows is forecast for the Sangre de Cristo Mountains (figure 1, figure 2 and figure 3) and the portions of the forest adjacent to the San Luis Valley (figure 1 and figure 2). These patterns are consistent between the 2040 and 2080 projections with relatively greater decrease estimated for 2080. At the extreme of the predictions, we estimate as much as a 20 percent decline in flow levels for some streams by 2040 and as much as a 30 percent decrease by 2080. Conversely, flows are forecast to increase for the Cochetopa Hills in 2040 and 2080 (figure 1, figure 2 and figure 4), with flows on some streams predicted to be declining by 2080. The central portion (moving from south to north) of the Rio Grande National Forest is expected to experience more variability and relatively larger declines in mean annual flows (figure 4). Lower elevation streams should be more variable in discharge, have more flow reduction relative to historic levels (figure 5), becoming more pronounced in 2080 than 2040. Relative to original, historic flow levels, we expect larger relative declines and more variability in streams with lower overall flows (figure 6).

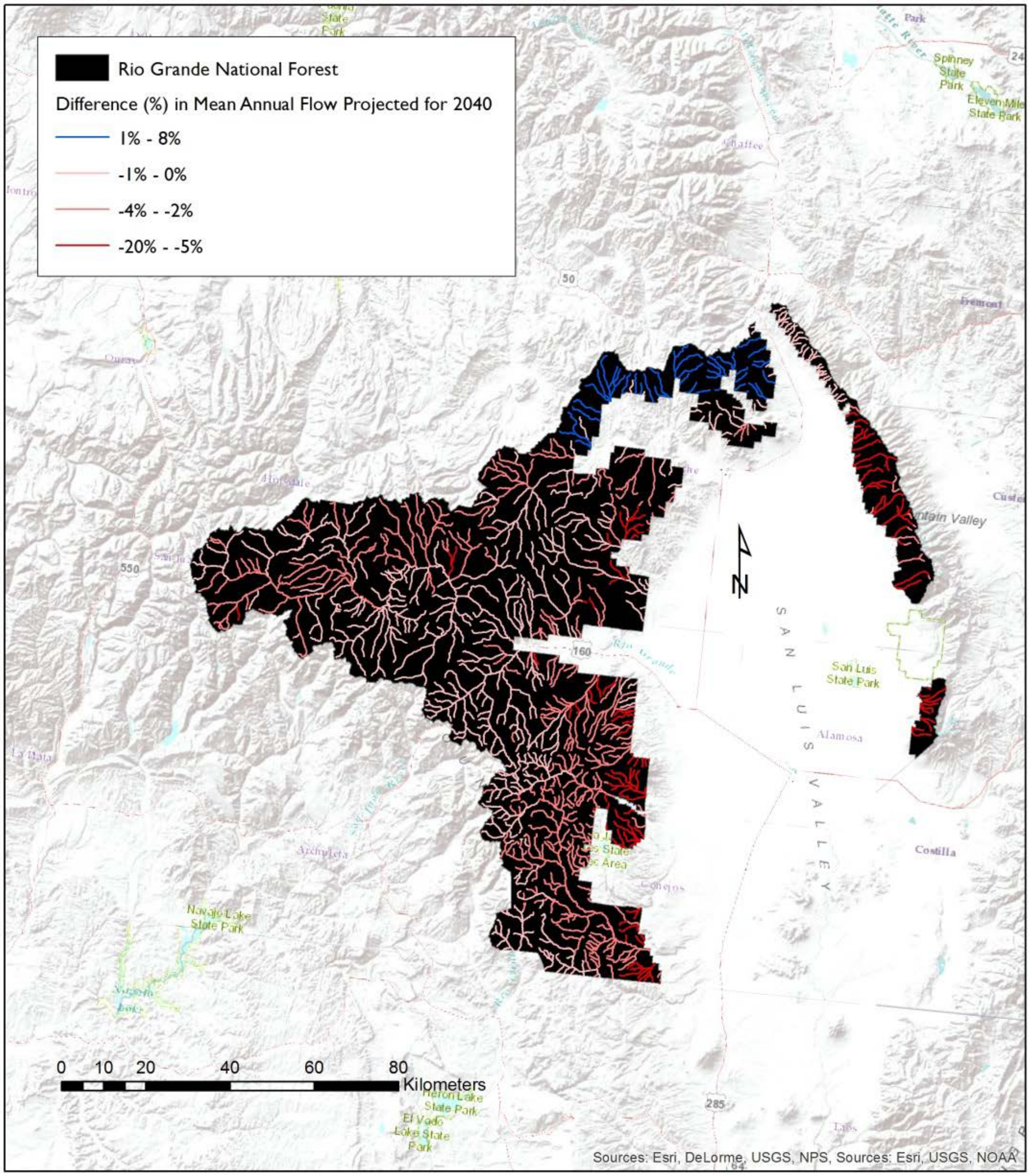

Figure 1. Percent difference in historic mean annual flows relative to flows projected for 2040 on the Rio Grande National Forest. The southern Sangre de Cristo mountain range and streams at the lower elevations are predicted to have the largest change under unregulated conditions.

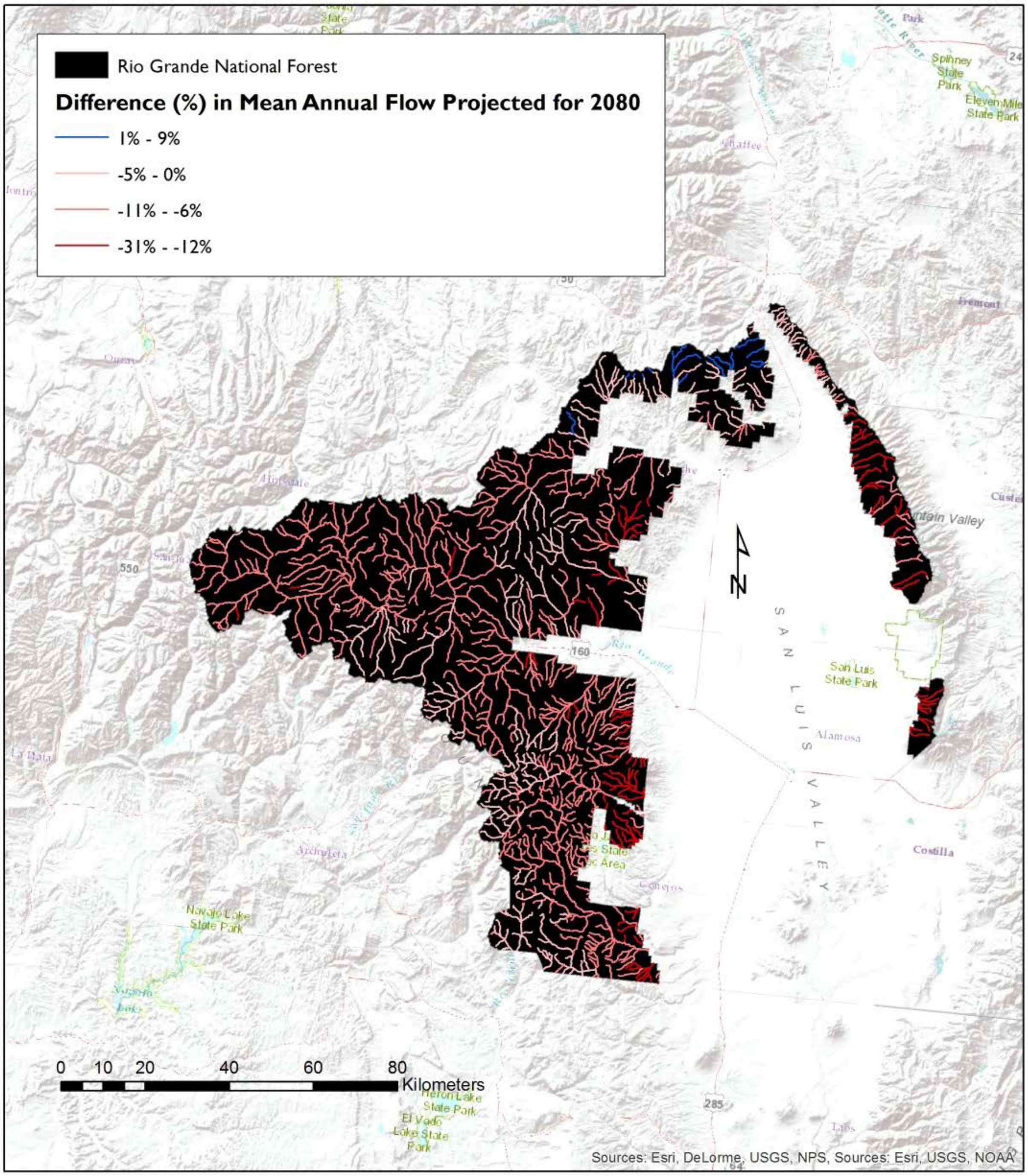

Figure 2. Percent difference in historic mean annual flows relative to flows projected for 2080 on the Rio Grande National Forest. Mean annual discharge will decline for all; most streams are expected to be reduced by more than 5 percent in 2080.

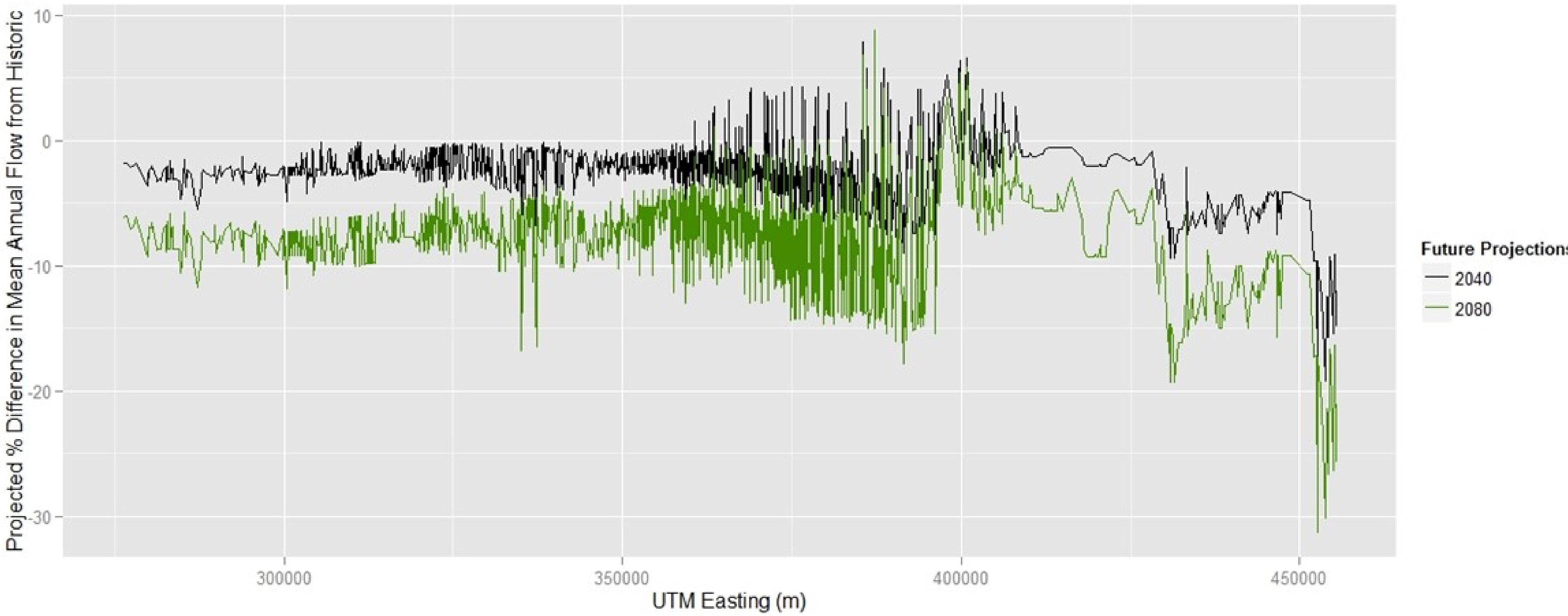

Figure 3. Changes in predicted difference in mean annual flow (for 2040 and 2080) shown from west to east (top) across the Rio Grande National Forest. The relatively high percentage change in the east boundary can be explained by the influence of the Sangre de Cristo mountain range where mean annual flows are expected to be reduced as much as 30 percent in some years by 2080.

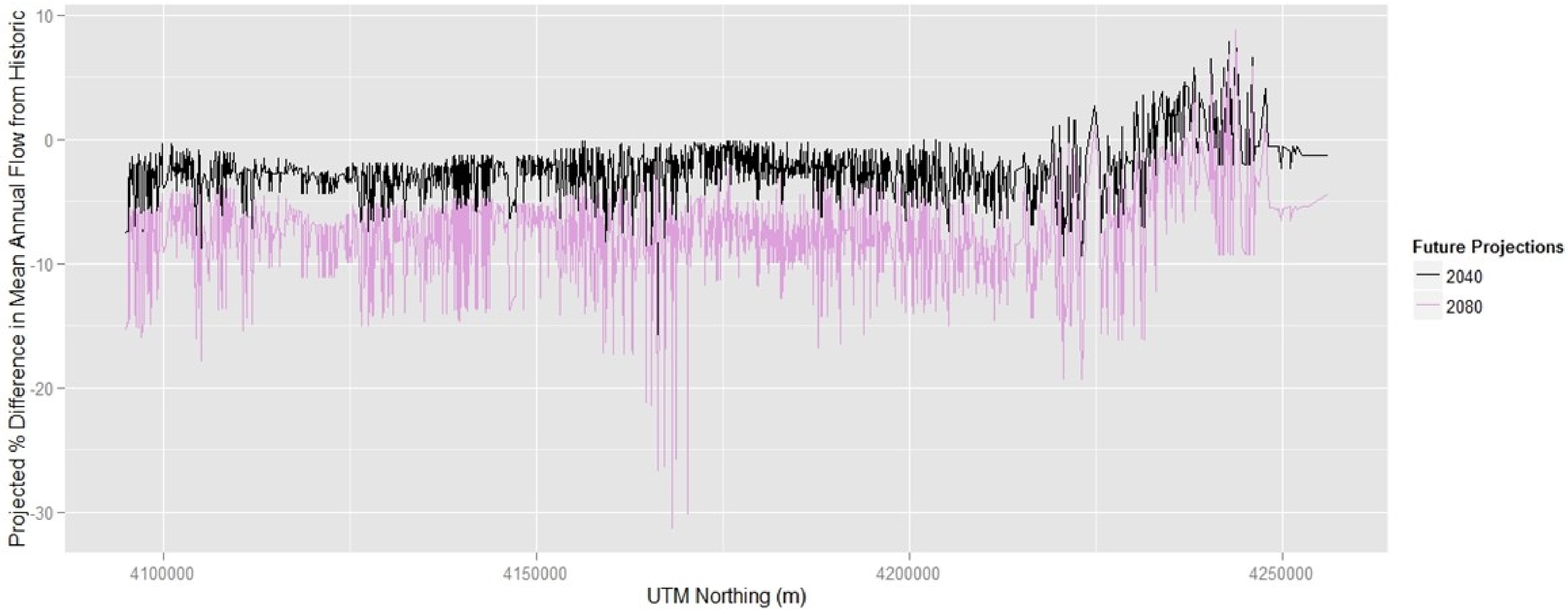

Figure 4. Changes in predicted difference in mean annual flow (for 2040 and 2080) shown from south to north (bottom) across the Rio Grande National Forest. The estimated increase in mean annual flows in the north (bottom) can be explained by the high elevation Continental Divide.

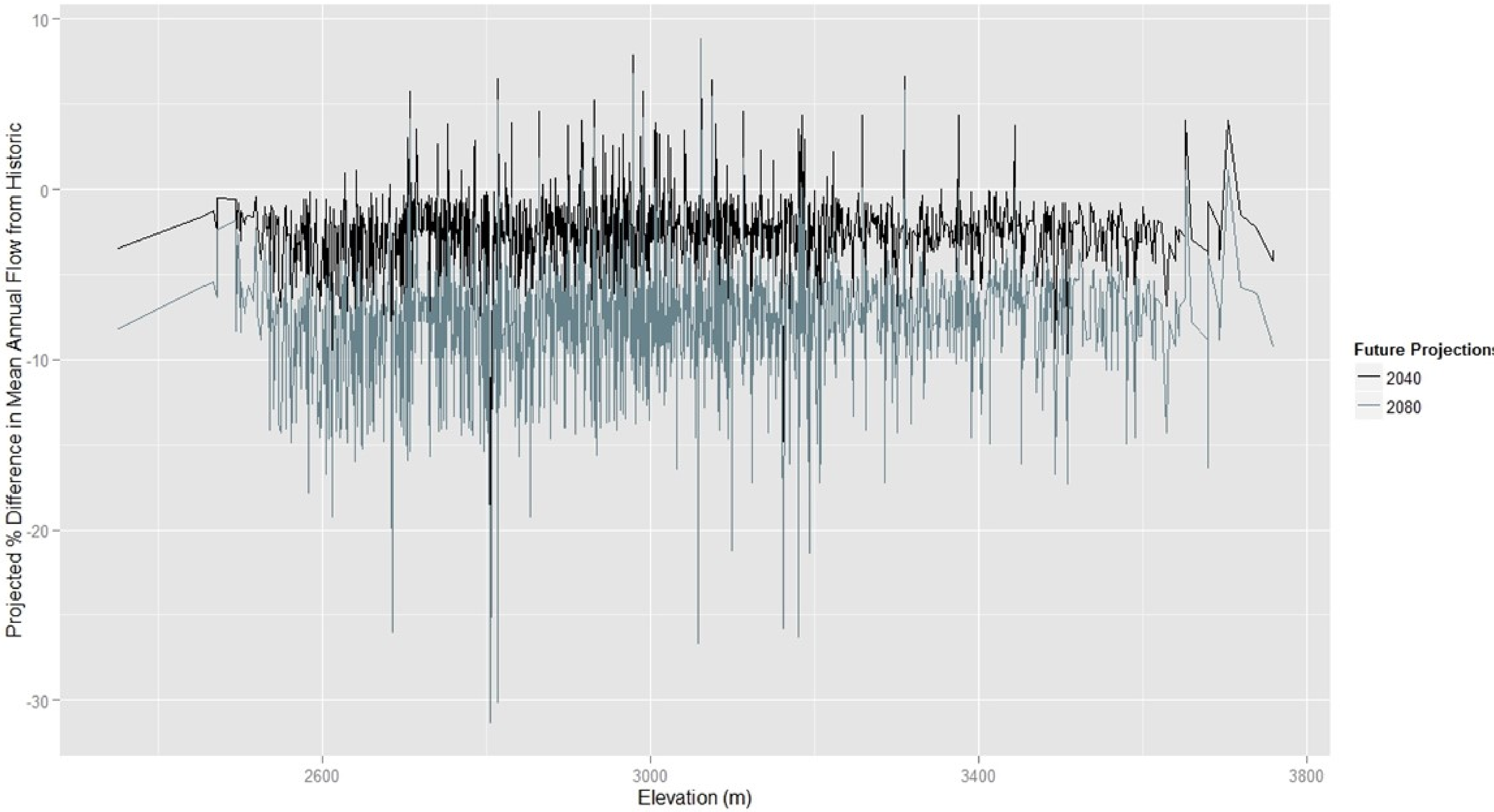

Figure 5. Percent difference in mean annual flows in 2040 and 2080 relative to elevation of stream reach midpoint.

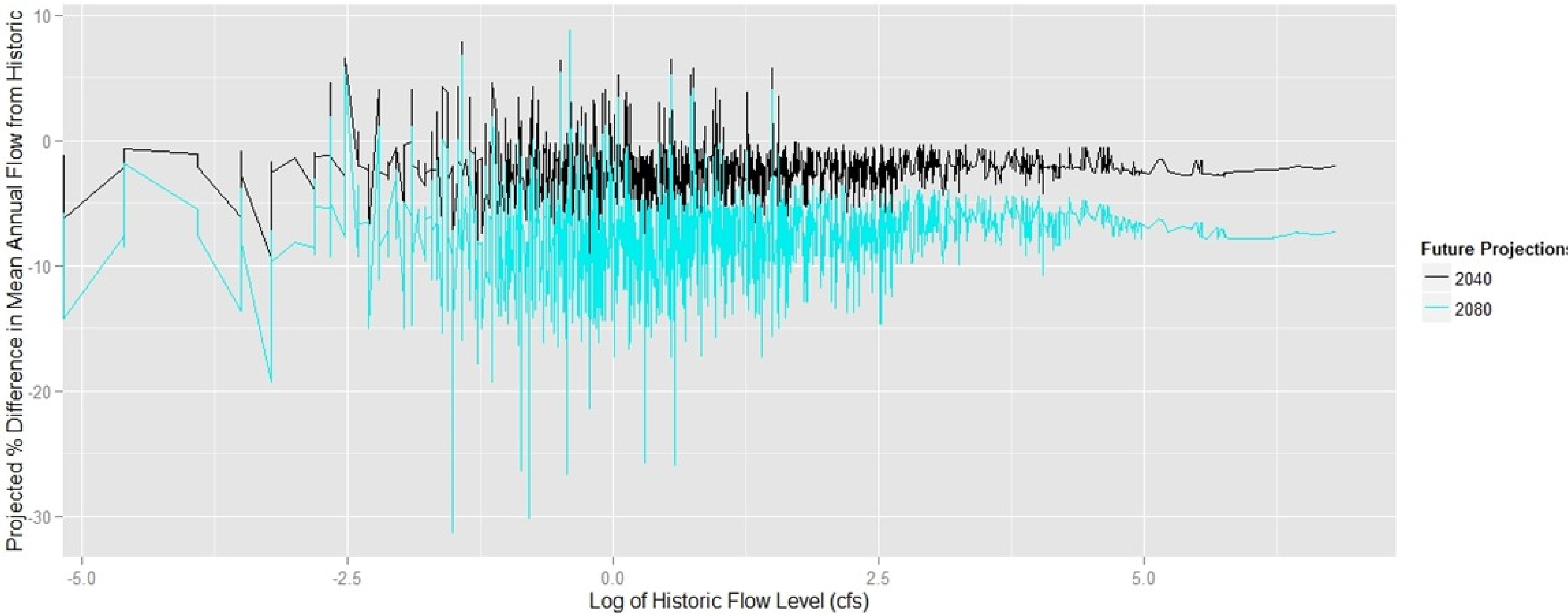

Figure 6. Percent difference in mean annual flows in 2040 and 2080 relative to historic flow levels on the Rio Grande National Forest.

Conclusions

The influence of long-term stream flow modification from climate change research are limited (Reiman and Isaak, 2010). However, available information indicates that we should prioritize lower elevation streams and those with limited base flows for management as they will most likely be influenced in the future. The analysis we conducted shows that discharge could be significantly reduced, variability could be much higher, and we should expect declining fish and other aquatic biota if these conditions occurred (Wenger et al. 2011). To mitigate the impacts of projected reduced flows, our management should focus on maintaining stream structure and surrounding vegetation within the natural range of variation; particularly at streams, wetlands and riparian areas thought to be more vulnerable to climate change (Dawson 2011). We should manage keystone species such as beaver, which restore streams to more natural flow rates and water table levels, to maintain stream function and store water naturally on the landscape. At the project level, we need to address activities that are influencing the natural hydrology, physical and chemical parameters, and habitat of these water-dependent ecosystems (see in large part Meehan, 1991). While it is important we understand that there is variability in this modelling effort, we should periodically collect and review monitoring data on stream flow rates and temperatures where on the Rio Grande National Forest streams and their associated ecosystems are threatened by climate change.

Literature Cited

Beschta, R.L.; Ripple, W.J. 2006. River channel dynamics following extirpation of wolves in northwestern Yellowstone National Park. Earth Surface Processes and Landforms. 31: 1525-1539.

Binkley, D., and T.C. Brown, 1993. Management impacts on water quality of forests and rangelands. USDA General Technical Report RM-239.

Bisson, P.A., B.E. Rieman, C. Luce, P.F. Hessburg, D.C. Lee, J.L. Kershner, G.H. Reeves, and R.E. Gresswell. 2003. Fire and aquatic ecosystems of the Western USA: current knowledge and key questions. Forest Ecology and Management.

Buckhouse, J.C. 1981. Riparian response to certain grazing management. Paper presented at Society for Range Management 34th Annual Meeting, Tulsa, OK.

Center of the American West (CAW). University of Colorado, Boulder. 2003. Online: Population Statistics: http://www.centerwest.org/futures/people/population. html; Change in Land Use Categories: http://www.centerwest.org/futures/development

Chamberlin, T.W., R.D. Harr, and F.H. Everest. 1991. Timber harvesting, silviculture, and watershed processes. In: Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19:181-205.

Clark, R.N.; Gibbons. D.R. 1991. Recreation. In: Meehan, W.R. ed. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19, Bethesda, MD:459-481.

Collen, P.; Gibson, R.J. 2001. The general ecology of beavers (*Castor* spp.), as related to their influence on stream ecosystems and riparian habitats, and the subsequent effects of fish – A review. Reviews in Fish Biology and Fisheries. 10:439-461.

Colorado Department of Public Health and Environment. 2012. Integrated Water Quality Monitoring and Assessment Report, State of Colorado. 2012 update to the 2010 305(b) Report.

Colorado Department of Public Health and Environment. 2013. Integrated Water Quality Monitoring and Assessment Report, State of Colorado. 2012 update to the 2010 305(b) Report.

Deacon, J.R., N.E. Spahr, S.V. Mize, and R.W. Boulger. 2001. Using water, bryophytes and macroinvertebrates to assess trace element concentrations in the upper Colorado River Basin, Hydrobiologia, Volume 455, Issue 1, pp. 29-39.

Dunham, J.B., M. Young, and R.E. Gresswell. 2003. Effects of fire on fish populations: landscape perspectives on persistence of native fishes and non-native fish invasions. Forest Ecology and Management.

Hammerson, G.A. 1986. Amphibians and reptiles of Colorado. State of Colorado, Division of Wildlife, Denver, CO.

Havlick, D.G. 2002. No place distant: Roads and recreation on America's public lands. Island Press. Washington. D.C.

Hagglund, A.; Sjoberg, G. 1999. Effects of beaver dams on fish fauna of forest streams. Forest Ecology and Management. 115:259-266.

Hoffman. C. 1998. Let it snow. Smithsonian. 29:50-53.

Kauffman, J. B., and W.C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management Implications: A review. Journal of Range Management 37(6):683-685.

Krueger, H.O., and S.H. Anderson. 1985. The use of cattle as a management tool for wildlife in shrubwillow riparian systems. *In*: Riparian ecosystems and their management: reconciling conflicting uses. *Edited by:* R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Folliott, and R.H. Hamre.16-18 April 1985.

Tucson, AZ. USDA Forest Service. General Technical Report RM-120.

Larson, R.E., W.C. Krueger, M.R. George, M.R. Barrington, K.C. Buckhouse, and D.E. Johnson. 1998. Viewpoint: Livestock influences on riparian zones and fish habitat: Literature classification. Journal of Range Management 51(6):661-664.

Lemly, Joanna. 2012. Assessment of Wetland Condition on the Rio Grande National Forest. Colorado Natural Heritage Program, Colorado State University, Fort Collins, CO 80523.

McIntosh, B.A., J.R. Sedell, R.F. Thurow, S.E. Clarke, and G.L. Chandler. 2000. Historical changes in pool habitats in the Columbia River basin. Ecological Applications 10:1478-1496.

Meehan, W.R. and W.S. Platts. 1978. Livestock grazing and the aquatic environment. Journal of Soil and Water Conservation 33(6):274-278.

Meyer, C.B.; Knight, D.H. 2001. Historic variability for upland vegetation in the Bighorn National Forest, Wyoming. Draft unpublished report sent to Bighorn National Forest, Sheridan, Wyoming and the Division of Renewable Resource, Rocky Mountain Region, U.S. Forest Service, Lakewood, CO. USFS Agreement No. 1102-0003-98-043.

Michael, J.L. 2000. Pesticides. In: Dissmeyer, G.E. ed. Drinking water from forests and grasslands: A synthesis of the scientific literature. U.S. Department of Agriculture, Forest Service. General Technical Report SRS-39.

Naiman, R.J.; Johnson, C.A.; Kelley, J.C. 1988. Alteration of North American streams by beaver. Bioscience. 38:753-762.

Naiman, R.J.; Pinay, G.; Johnson, C.A., [et al.]. 1994. Ecosystem alteration of boreal forest streams by beaver (*Castor Canadensis*). Ecology. 67: 1254-1269.

Pilliod, D.S., R.B. Bury, E.J. Hyde, C.A. Pearl, and P.S. Corn. 2003. Potential effects of fire and fuel reduction practices on aquatic amphibians in the United States. Forest Ecology and Management.

Olson, R.; Hubert, W.A. 1994. Beaver: Water resources and riparian habitat manager. University of Wyoming, Laramie.

Ralph, S.C.; Poole, G.C.; Conquest, L.L. [et al.]. 1994. Stream channel morphology and woody debris in logged and unlogged basins of western Washington. Canadian Journal of Fisheries and Aquatic Sciences. 51:37-51.

Rahel, F.J.; Hubert, W.A. 1991. Fish assemblages and habitat gradients in a Rocky Mountain-Great Plains stream: Biotic zonation and additive patterns of community change. Transactions of the American Fisheries Society. 120:319-332.

Ries, J.B. 1996. Landscape damage by skiing at the Schauinsland in the Black Forest, Germany. Mountain Research and Development. 16:27-40.

Roger, P.C.; Mittanck, C.M. 2014. Herbivory strains resilience in drought-prone aspen landscapes of the Western United States. Journal of Vegetation Science. 25: 457-469.

Shepperd, W.D.; Rogers, P.C.; Burton, D.; Bartos, D.L. 2006. Ecology, biodiversity, management, and restoration of aspen in the Sierra Nevada. RMRS-GTR-178. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station.

Stafford, M.P. 1994. Control of purple loosestrife with herbicides and the effect and subsequent response of wetland plant communities. In: Proceedings of the Idaho Weed Control Association 49th Annual Meeting, February 1994. Boise, ID.

Winter, T. C.; Harvey, J.W.; Franke, O.L., [et al.]. 1998. Ground water and surface water, a single resource. U.S. Department of the Interior, Geological Survey, Circular 1139. Washington, D.C.

Winters, D.S. and Tonya, T. 2016. Assessment 1: Aquatic condition, relationship to ecological drivers and management influences and identification of potential direction for the future. USDA Forest Service, Rocky Mountain Region, Lakewood, CO.

Winters, David S., A. Wuenschel, J. Gurrieri, T. Tenyah, D. Cleary, and S. Synowiec. 2016. Assessment 3: Aquatic and Semi-Aquatic Ecological Drivers. Chapter 1: Ecological Driver Classification and Analysis. USDA Forest Service, Rocky Mountain Region, Lakewood, CO. 61 pp.

Wireman, M. 2000. Hardrock mining. In: Dissmeyer, G.E. ed. Drinking water from forests and grasslands: A synthesis of the scientific literature. U.S. Department of Agriculture, Forest Service. General Technical Report SRS–39.

Wohl, E. 2000. Mountain rivers. Water Resources Monograph 14. American Geophysical Union. Washington DC.

Wohl, E.E. 2001. Virtual rivers: Lessons from the Mountain Rivers of the Colorado Front Range. Yale University Press, New Haven, CT. 210 p.

Walcott, C.D. 1899. Bighorn Forest Reserve. Government Printing Office. Washington, DC.

Woldeselassiea, M.; Van Miegroet, H.; Gruselle, M.; Hambly, N. 2012. Storage and stability of soil organic carbon in aspen and conifer forest soils of Northern Utah. Soil Science Society of America Journal. 76(6): 2230-2240.