Friends of the Wild Swan
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October 12, 2020

Swan Lake Ranger District
200 Ranger Station Road
Bigfork, MT 59911
Via email to: comments-northern-flathead-swan-lake@usda.gov

Superintendent Steele,

Please accept the following comments on the Mid-Swan Landscape Restoration and Wildland Urban Interface Project Draft Environmental Impact Statement (DEIS) on behalf of Friends of the Wild Swan. We incorporate by reference the comments submitted by Swan View Coalition, Native Ecosystems Council, Brian Peck Independent Consultant, and Friends of the Clearwater - Alliance for the Wild Rockies.

The DEIS lacks the specificity to adequately analyze the impacts to wildlife, fish, native plants and water quality that is required by the National Environmental Policy Act. NEPA requires that agencies take a "hard look" at the impacts to ensure they are making informed decisions prior to acting and that the public has meaningful input based on the best available science and site-specific information.

The Mid-Swan project covers 174,000 acres, Alternative B would log 197-235 mmbf of timber on 118,399 acres and Alternative C logs 101-115 mmbf on 55,478 acres, and would take at least 15 years to complete the logging. The DEIS does not disclose where units are and the Flathead has not even ground-truthed the project area -- that will be done when they are ready to log and the public has no opportunity to challenge the site-specific project elements.

The Flathead relies on photo interpretation and computer models, not on the ground inspection, to determine many important features on the landscape such as:

- Riparian management zones are narrow linear features and their composition may not be accurately represented in the photo-interpreted data (DEIS page 100)

- Roads adjacent to streams within the Inner and Outer RMZs, changes to aquatic network within the CWN, changes to unique habitat and their RMZs and changes to WCF ratings are all GIS calculated. (DEIS page 161)

- We created a model of lynx habitat in the project area based on the same three-dimensional aerial photo interpretation data. (DEIS page 201)

- Since the spatial extent of old growth is not currently known... (DEIS page 112)

• Surveys at the time of implementation would determine whether these stands qualify as old growth... (DEIS page 114)

• Photo interpreted data were also used to estimate old forest structure, a late seral stage in forest succession characterized by an abundance of larger and presumably older trees and one or multiple canopy strata (O'Hara et al. 1996). Mapped areas with old forest structure may qualify as old-growth forest following the minimum criteria in Green et al. (2011) **but we cannot determine exact stand age from aerial photography**. Verification of old growth status following Green et al. (2011) would happen during the implementation stage of this project. (DEIS page 8) [Emphasis added]

• To move existing conditions towards desired conditions, we compared the photo-interpreted existing conditions with desired conditions derived from integrated objectives and themes and assigned an initial level-of-change (LOC) category. (DEIS page 27)

• Forest structural stages were derived from photo interpreted data. (DEIS page 97)

• Vegetation characteristics available in the photo-interpreted polygon data were used to assign fuel models to the Mid Swan Project landscape. (DEIS page 133)

• Forest vegetation was classified into seven structural classes (O'Hara et al. 1996), using photo-interpreted data for tree sizes, canopy covers and number of canopy layers (strata). (DEIS page 96)

• Vegetation characteristics derived from high-resolution 3-dimensiional aerial photo interpretation were used to estimate existing habitat conditions including snow intercept cover, hiding cover and forage areas in the project area for forest ungulates. (DEIS page 267)

This failure to provide accurate baseline data also violates NEPA. The agency cannot analyze and disclose the effects nor can the public review and comment on the scope of the impacts and where those impacts may occur without site-specific information.

The Implementation Guide on Restoration (IGOR) is not a substitute for site specific analysis, it is merely a series of vague guidelines. The Mid Swan project is to be implemented in smaller geographic areas than the entire project area with logging units identified at a later time, these are not analyzed in the DEIS. Public feedback when implementation areas are proposed are informal and will not allow the public to formally comment or object based on this new information so it is essentially meaningless.

Aquatics

• The aquatic condition indicators rely on GIS computer models and aerial photo derived road and stream locations. There is no reference in the DEIS as to whether the road locations, culverts, stream crossings have been ground-truthed. Furthermore, the DEIS does not contain or utilize objective and measurable habitat criteria such as temperature, pool frequency and other parameters that gauge stream baseline condition because neither the Forest Plan nor the Bull

Trout Recovery Plan contain these criteria. Yet this is essential to determining whether habitat conditions are improving or declining. Attached are the comments by Dr. Christopher Frissell on the Bull Trout Recovery Plan that are relevant to this project.

Montana Dept. of Fish, Wildlife and Parks has regularly used McNeil core samples to determine the median percentage of streambed material smaller than 6.35mm since 1987 on 10 bull trout spawning streams in the Swan watershed. (Attached) Research shows a negative relationship between fine sediment and embryo survival to emergence -- as the percentage of fine material increased, habitat quality decreased.

In the Flathead basin when the percentage of fine materials in spawning gravels in any given year is greater than 35% the stream is considered threatened as a bull trout spawning and/or rearing stream. When the percentage of fine materials in spawning gravels in any given year is greater than 40% the stream is considered impaired. In 2018 five bull trout spawning streams exceeded 35% fine sediment: Squeezer (40%), Lion (36.1%), Jim (39.1%), Soup (36.2%) and Woodward (40%).

The DEIS discloses that higher levels of fine-grained sediment are in managed stream reaches than within reference reaches. It pinpoints the extensive road network in most watersheds as the cause of this increased sediment. Yet, this project does little to reduce the 567 miles of Forest Service roads (which is in addition to the 578 miles of roads on other ownerships).

In addition, temporary roads were not modeled for sediment delivery because they are not additions to the NFS road system and are typically rehabilitated five years after construction. The IGOR (INF20) states that a temporary road shall not exist for more than 5 years after completion of project activities. If the project takes four or five years to complete than those temporary roads can have impacts for as many as 10 years, and then if a watershed has multiple entries over the 15 year period more temporary roads can be constructed. This is an impact!

The impacts of building temporary roads are the same as building permanent roads such as:

-The greatest surface erosion from roads occurs during the construction phase and first year after.
-Soil erosion and compaction (as always occurs with roads) causes long-term loss of soil productivity.
-The loss of topsoil and attendant loss of soil productivity is permanent.
-Road obliteration does not immediately stop severely elevated soil erosion from roads.
-Temporary roads have enduring impacts on aquatic resources.
-Roads and increased sedimentation cause long-term negative impacts on a variety of aquatic biota.

Yet the DEIS does not disclose the landtypes in the project area, soil compaction levels nor does it analyze the impacts to soils.

• The USFWS recognized sediment as one of the key factors impacting water quality and fish habitat. [*See* USFWS 2010]

The introduction of sediment in excess of natural amounts can have multiple adverse effects on bull trout and their habitat (Rhodes et al. 1994, pp. 16-21; Berry, Rubinstein, Melzian, and Hill 2003, p. 7). The effect of sediment beyond natural background conditions can be fatal at high levels. Embryo survival and subsequent fry emergence success have been highly correlated to percentage of fine material within the streambed (Shepard et al. 1984, pp. 146, 152). Low levels of sediment may result in sublethal and behavioral effects such as increased activity, stress, and emigration rates; loss or reduction of foraging capability; reduced growth and resistance to disease; physical abrasion; clogging of gills; and interference with orientation in homing and migration (McLeay et al. 1987a, p. 671; Newcombe and MacDonald 1991, pp. 72, 76, 77; Barrett, Grossman, and Rosenfeld 1992, p. 437;Lake and Hinch 1999, p. 865; Bash et al. 2001n, p. 9; Watts et al. 2003, p. 551; Vondracek et al. 2003, p. 1005; Berry, Rubinstein, Melzian, and Hill 2003, p. 33). The effects of increased suspended sediments can cause changes in the abundance and/or type of food organisms, alterations in fish habitat, and long-term impacts to fish populations (Anderson et al. 1996, pp. 1, 9, 12, 14, 15; Reid and Anderson 1999, pp. 1, 7-15). No threshold has been determined in which fine-sediment addition to a stream is harmless (Suttle et al. 2004, p. 973). Even at low concentrations, fine-sediment deposition can decrease growth and survival of juvenile salmonids.

Aquatic systems are complex interactive systems, and isolating the effects of sediment to fish is difficult (Castro and Reckendorf 1995d, pp. 2-3). The effects of sediment on receiving water ecosystems are complex and multi-dimensional, and further compounded by the fact that sediment flux is a natural and vital process for aquatic systems (Berry, Rubinstein, Melzian, and Hill 2003, p. 4). Environmental factors that affect the magnitude of sediment impacts on salmonids include duration of exposure, frequency of exposure, toxicity, temperature, life stage of fish, angularity and size of particle, severity/magnitude of pulse, time of occurrence, general condition of biota, and availability of and access to refugia (Bash et al. 2001m, p. 11). Potential impacts caused by excessive suspended sediments are varied and complex and are often masked by other concurrent activities (Newcombe 2003, p. 530). The difficulty in determining which environmental variables act as limiting factors has made it difficult to establish the specific effects of sediment impacts on fish (Chapman 1988, p. 2). For example, excess fines in spawning gravels may not lead to smaller populations of adults if the amount of juvenile winter habitat limits the number of juveniles that reach adulthood. Often there are multiple independent variables with complex inter-relationships that can influence population size.

The ecological dominance of a given species is often determined by environmental variables. A chronic input of sediment could tip the ecological balance in favor of one species in mixed salmonid populations or in species communities composed of salmonids and nonsalmonids (Everest et al. 1987, p. 120). Bull trout have more spatially restrictive biological requirements at the individual and population levels than other salmonids (USFWS (U.S. Fish and Wildlife Service) 1998, p. 5). Therefore, they are especially vulnerable to environmental changes such as sediment deposition.

Aquatic Impacts

• Classify and analyze the level of impacts to bull trout and westslope cutthroat trout in streams, rivers and lakes from sediment and other habitat alterations:

Lethal: Direct mortality to any life stage, reduction in egg-to-fry survival, and loss of spawning or rearing habitat. These effects damage the capacity of the bull trout to produce fish and sustain populations.

Sublethal: Reduction in feeding and growth rates, decrease in habitat quality, reduced tolerance to disease and toxicants, respiratory impairment, and physiological stress. While not leading to immediate death, may produce mortalities and population decline over time.

Behavioral: Avoidance and distribution, homing and migration, and foraging and predation. Behavioral effects change the activity patterns or alter the kinds of activity usually associated with an unperturbed environment. Behavior effects may lead to immediate death or population decline or mortality over time.

Direct effects:

Gill Trauma - High levels of suspended sediment and turbidity can result in direct mortality of fish by damaging and clogging gills (Curry and MacNeill 2004, p. 140).

Spawning, redds, eggs - The effects of suspended sediment, deposited in a redd and potentially reducing water flow and smothering eggs or alevins or impeding fry emergence, are related to sediment particle sizes of the spawning habitat (Bjornn and Reiser 1991, p. 98).

Indirect effects:

Macroinvertebrates - Sedimentation can have an effect on bull trout and fish populations through impacts or alterations to the macroinvertebrate communities or populations (Anderson, Taylor, and Balch 1996, pp. 14-15).

Feeding behavior - Increased turbidity and suspended sediment can affect a number of factors related to feeding for salmonids, including feeding rates, reaction distance, prey selection, and prey abundance (Barrett, Grossman, and Rosenfeld 1992, pp. 437, 440; Henley, Patterson, Neves, and Lemly 2000, p. 133; Bash et al. 2001d, p. 21).

Habitat effects - All life history stages are associated with complex forms of cover including large woody debris, undercut banks, boulders, and pools. Other habitat characteristic important to bull trout include channel and hydrologic stability, substrate composition, temperature, and the presence of migration corridors (Rieman and McIntyre 1993, p. 5).

Physiological effects - Sublethal levels of suspended sediment may cause undue physiological stress on fish, which may reduce the ability of the fish to perform vital functions (Cederholm and Reid 1987, p. 388, 390).

Behavioral effects - These behavioral changes include avoidance of habitat, reduction in feeding, increased activity, redistribution and migration to other habitats and locations, disruption of territoriality, and altered homing (Anderson, Taylor, and Balch 1996, p. 6; Bash et al. 2001t, pp. 19-25; Suttle, Power, Levine, and McNeely 2004, p. 971).

We presented this information during scoping yet none of these effects were analyzed for bull trout or westslope cutthroat trout.

• The DEIS acknowledges that there would be between 49,250-58,750 trips on the haul route for alternative B and between 25,250-28,750 trips for alternative C, most existing Forest Service

roads could be used as a haul route, and impacts from dust and road use can cause pulse effects to water quality. Yet there is no disclosure of the impacts to aquatic life and water quality.

It also states that a well-maintained road system is less likely to fail, which can potentially result in large pulses of sediment delivery yet the Flathead receives less than 1/6th of the funding necessary to maintain the road system. Typically roads are maintained if they are a main route or when there is a timber sale. It is obvious that there is no well-maintained road system on the Flathead.

The DEIS also does not disclose what the cost to taxpayers are for additional road maintenance on Highway 83 from 25,250 to 58,750 log trucks on the highway.

• Alternative B proposes 3.7 miles of new road construction crossing through RMZs. The DEIS assumes that it's OK to build new permanent and temporary roads in RMZs because it's offset by decommissioning RMZ roads somewhere else. However, this does not account for soil types, slope, or aspect and this information is not disclosed in the DEIS.

The DEIS also discloses that Alternative B would result in short-term and long-term negative effects to aquatic ecosystem during project implementation and beyond. What are those short and long term negative impacts? How will aquatic life be impacted? If, for example, there is too much fine sediment in a stream from project actions and bull trout are unable to successfully spawn how does that affect the population? For how many years? Will they be extirpated from that stream due to reduced spawning success from fine sediment increases?

The DEIS does not contain an analysis of bull trout critical habitat. How does the project impact the Primary Constituent Elements? It does disclose that it is likely that bull trout would be adversely affected resulting in the loss of some individuals but long term the restoration work is expected to have beneficial effects that far offset the initial loss. How will the project adversely modify critical habitat? What time frame is considered "long term"? How many individuals of a threatened species can be lost without extirpating a spawning population?

• The IGOR Figure 9 on page A-43 allows 10% of existing culverts on all proposed ISS roads to be retained. The DEIS contains no analysis of the impacts this has on aquatic ecosystems.

• What monitoring has been done for macroinvertebrates to determine a baseline? What monitoring will be done for macroinvertebrates during the project?

• If beavers are relocated into the Swan Valley will trapping them be banned?

• The DEIS states: "Effects from the proposed actions are not expected to be discernible from existing conditions more than 300 meters into Swan Lake, as the large volume of water present would quickly mix any transported fine-grained sediment and rapidly dilute this effect to zero."

Swan Lake is a WQLS and a Water Quality Protection Plan and TMDLs were prepared for the Swan Lake Watershed in 2004. Low dissolved oxygen (DO) concentrations in the deeper basins

led to Swan Lake being placed on the state's 303(d) list in 1996. The 2004 Beneficial Use Determination is:

"Data indicate that beneficial uses are being supported, however there is a documented adverse pollution trend as evidenced by the Spencer (1991b) sediment-core study. This study clearly shows that the sedimentation in Swan Lake has increased >3 times its historic (late 1800s) rates, and that much of the increase occurred concurrently with large-scale timber harvest in the watershed since the 1960's. This increased sediment/nutrient/carbon load to the sediments *may* be responsible for the oxygen depletions noted in the deeper basins. A more recent work (Ellis et al, (1999a) has failed to make a clear connection between land use and water quality, but that study indicated that the complexities of this flood plain riverine system make such a correlation difficult. An increase in the noted oxygen depletions is to be avoided in order to maintain the lake in its current oligotrophic state."

The TMDL further states: "The remaining siltation listing is associated with increased accumulation of inorganic and organic material (specifically organic carbon) to the lake bottom/sediments... The increased organic material in lake sediments can lead to DO reductions and subsequent depletion (anoxic conditions). A reduction in DO can directly limit aquatic life and cold-water fish habitat, and can also lead to conditions where phosphorus is released from the bottom sediment layer. This phosphorus could then enter the water column, leading to additional negative impacts to aquatic life and cold-water fish due to nutrient enrichment conditions in Swan Lake, as well as increasing downstream nutrient loading to Flathead Lake."

Applicable Source Load Allocations for Swan Lake are:

- Road Erosion: Nutrient and particulate organic carbon loading associated with sediment delivery from road erosion.
Allocation: 40% total reduction in modeled sediment loading from road stream crossings based on FRS method.

- Riparian and Streambank Protection: Nutrient and POC loading associated with eroding banks, loss of woody debris and riparian vegetation impacts.
Allocation: 10% decrease in total loading throughout the Swan Lake Watershed. Canopy density is used as a surrogate to measure progress.

- Other Timber Harvest Impacts: Nutrient and POC loading from timber harvest (other than road erosion and riparian harvest covered above); this also includes road culvert failures.
Allocation: No loading increase.

The DEIS disregards Swan Lake's threatened status and minimizes the impacts that cumulative sediment pollution has on the lake that is critical habitat for bull trout. One of the deeper basins is directly impacted by sediment transported to the lake from the river. That deep basin has been precariously close to 0 DO which would lead to impacts to fish and other aquatic life. The DEIS must quantify and analyze the impacts to Swan Lake rather than flippantly blowing them off.

Old-Growth Forest Habitat

The project proposes logging in old growth forest habitat and claims it will still function as old growth after it is logged yet cites no science to support that theory. "Should any stands qualify as old growth under Green et al. (2011) prior to treatment, they will remain old growth after treatment." (DEIS page 110)

The DEIS also discloses: "Proposed treatments may include even-aged regeneration on up to 702 acres and regeneration openings on up to 7,998 acres. Regeneration treatments would generally only be used when there are few desirable trees on site. Many large trees would be retained." (DEIS page 111) And "Commercial harvest is proposed in 15.6% of all old forest structure. Treatment options in old forest structure include even-aged regeneration harvest on up to 473 acres within WUI." (DEIS page 112) Clearcutting old growth habitat is not restoration nor will it remain old growth after being chainsawed.

Alternative B proposes the construction of up to 4.1 miles of roads through stands classified as old forest structure. Of these, 0.5 miles would be temporary roads. Surveys at the time of implementation would determine whether these stands qualify as old growth and whether alternative routes are available to prevent the loss of old growth forest. (DEIS page 114)

Regeneration opening adjacent to old growth could increase windthrow and stem breakage and further increase the amount of large wood on the ground. (DEIS page 114)

In essence, the Flathead is proposing to log, road and clearcut old growth or potential old growth forest without disclosing to the public where this might take place because the Flathead does not know where old growth habitat is located because they haven't bothered to survey for it.
See:
Since the spatial extent of old growth is not currently known, maximum treatment acres are shown for late seral stands with old forest structure. (DEIS page 112)

Surveys at the time of implementation would determine whether these stands qualify as old growth... (DEIS page 114)

Photo interpreted data were also used to estimate old forest structure, a late seral stage in forest succession characterized by an abundance of larger and presumably older trees and one or multiple canopy strata (O'Hara et al. 1996). Mapped areas with old forest structure may qualify as old-growth forest following the minimum criteria in Green et al. (2011) but we cannot determine exact stand age from aerial photography. Verification of old growth status following Green et al. (2011) would happen during the implementation stage of this project. (DEIS page 8)

The DEIS claims that: "Commercial harvest or other mechanized treatments under alternative B would not reduce the quantity of old growth." (page 113) The Flathead doesn't understand that old growth is more than just big trees, it is a host of attributes including snags, down woody material, multiple canopy layers, etc. that are essential for wildlife. The DEIS talks about retaining "desirable" tree species, but desirable to humans, not necessarily wildlife who need those other attributes for survival.

Old growth manipulation was addressed in the 2000 Contract Review of Old Growth Management on School Trust Lands by R.D. Pfister, W.L. Baker, C.E. Fiedler and J.W. Thomas.

"In addition, there is the question of the appropriateness of management manipulation of old-growth stands - both those extant and those in process of development toward old-growth condition. Opinions of well-qualified experts vary in this regard. **As long term results from active management lie in the future - likely quite far in the future - considering such manipulation as appropriate and relatively certain to yield anticipated results is an informed guess at best and, therefore, encompasses some unknown level of risk. In other words, producing "old-growth habitats through active management is an untested hypothesis**. Scientists vary in their degree of faith in such manipulations being successful in mimicking natural processes. Adherence to the precautionary principle (and the probability of losing in court when dealing with threatened or endangered species) have produced the more common approach of "reserve strategies" considering the above mentioned variables of numbers of old-growth patches, stand size, juxtaposition with other stands, and connectivity."

The Flathead has no plan for retaining old-growth forest habitat on the landscape and allowing stands to develop into old growth. Instead this project will fragment old growth, create abrupt edges from roads and cutting units adjacent to it, and degrade the habitat quality for old growth associated birds and wildlife.

Pfister, Baker, Fiedler and Thomas lay out considerations for designing old growth networks:

"Interior" old growth habitat (>100 meters from edge of an opening or stand of lesser age or a road) is the most important component of old-growth habitat (Baker and Knight 200). I.e., in general, larger stands are more effective as habitat than smaller stands. Fragmentation of existing patches of old growth by roads, timber harvesting, or other created openings will decrease effectiveness of the patch as habitat due to the reduction in amount of interior old-growth conditions (Baker and Knight 200). I.e, in general, non-fragmented stands are more effective as habitat than smaller stands. To decrease or avoid fragmentation, roads can be designed to avoid large patches, harvest units can be placed on or near existing roads, and roads can be closed where no longer needed.

Stands of old-growth forests will function best as habitat when they are connected to other stands. Connectivity can thus be achieved by corridors of actual old growth or by suitable closed-canopy or mature condition of the matrix between old-growth stands (Thomas and others 1990, Bennett 1999). Stands designated as future old growth that are presently mature may be suitable. Linkages should, when possible, contain a large fraction of interior forest (i.e., >100 meters from high contrast edge) (Bennett 1999).

When designating old-growth patches (whether extant or planned) it is important to span a representative cross-section of sites, rather than to concentrate them in streamside areas or on poorer sites.

Place longer-rotation or less intensive uses adjacent to designated old growth, so that a lower intensity managed zone serves as a buffer for old-growth system (Noss and Cooperrider 1994). Avoid placing high intensity land uses (e.g., clearcuts, roads) next to designated old growth.

Don't worry about the appropriate mix of stand structures and compositions within old growth, because the science has not yet provided that kind of resolution.

Other things being equal, big old-growth reserves are better than small ones, unfragmented reserves are better than fragmented, reserves closer together are better than reserves far apart, reserves connected to others are better than those not connected.

Integrate future replacement old growth into the network. Where otherwise equivalent replacement stands exist, choose those adjacent to designated old growth as future old growth.

Designate the existing old growth and future old growth and place them on maps.

Furthermore, the DEIS doesn't even analyze impacts to old growth associated wildlife from clearcut logging and species conversion to human "desirable" tree species. These forests and wildlife evolved with fire and Forest Service "undesirable" tree species. What monitoring has the Flathead done to determine that reducing old growth forest characteristics to the Green, et al. minimum won't impact old growth associated wildlife? By not looking at impacts to the wildlife who need these forest conditions the DEIS violates NEPA.

Wildlife

• The DEIS identifies five cross-valley wildlife corridors based on more mobile, cover-dependent species crossing the valley without detection or disturbance. In addition to a combination of riparian networks and multistory forest connectivity for less mobile species.

Have these cross valley corridors been verified for use by wildlife? For example, in the early 1990s the USFWS identified grizzly bear linkage zones in the Swan Valley based on cover and land use, however years later when GPS collars were available it was found that bears used the valley differently than was assumed with these linkage zones.

Have these cross valley corridors been verified on the ground or is this once again a computer exercise?

The DEIS discloses that wildlife will be displaced by logging, roading and burning activities during at least the 15 year life of this project. How will this impact the cross valley corridors?

Why weren't John Squires lynx connectivity corridors identified?

• The DEIS discloses that logging, roads and helicopters will displace wildlife yet there is no analysis of how this impacts feeding, breeding and shelter or whether there is sufficient habitat to support them over the long term. It also does not analyze the impacts on an animal's home range and whether that displacement will put them in conflict because they are displaced to another animal's home range.

• The DEIS does not analyze the impacts from reduced thermal cover for ungulates. Dense forest canopy limits snow depth allowing animals to move through deep snow in the winter. In winter

and summer dense forest canopy provides thermal regulation, it is warmer under the canopy in the winter and cooler under the canopy in the summer. Mule deer, elk and moose populations are in decline yet the additional stress of logging off thermal cover was not evaluated in the DEIS. Nor does the Forest Plan contain standards for ungulate cover.

• The DEIS does not analyze the impacts to Region 1 Sensitive Species such as bald eagle, black-backed woodpecker, common loon, harlequin duck, bighorn sheep, fisher, wolf, bats and toads. Ironically the black swift was analyzed but according to Region 1 it doesn't even occur on the Flathead. [See attached Region 1 Sensitive Species list] Since sensitive species are designated because population viability is a concern on National Forest lands it is imperative that impacts to them be analyzed to ensure that numbers and population distribution are adequate so they are not extirpated on Forest Service land.

• Lynx connectivity will be fragmented in 11 watersheds under Alternative B and 4 under Alternative C. How does this impact lynx travel through the valley?

• The EA is confusing because it uses the term multi story which is not the NRLMD/Forest Plan's "multi story mature or late successional forest" in VEG-S6. There is no definition of multi story and how it differs from multi story mature habitat. Furthermore, the DEIS essentially ignores the large openings that will be created. There is no analysis of lynx avoiding these large openings, there is no analysis of roads running through or adjacent to old-growth forest habitat, there is no analysis of logging in or adjacent to late successional forest and there is no analysis of lynx being displaced from key habitat.

• The DEIS discloses that 16,028 acres of multistory habitat in 11 of the 12 LAUs will be impacted in the project area. The DEIS does not disclose where this multi story habitat is located, whether it is mature multi story habitat, or the quality of the habitat. It assumes that all habitat is equal and lynx can be displaced -- except for Woodward Creek where there is the highest quality habitat that will be logged over 7 years.

• The DEIS's assumptions about regeneration fail to consider climate change and the possibility that the logged forest will not regrow in the future as it may have in the past. It also fails to factor in the inevitable fires that still will occur and are additive to the logging that is proposed.

• The project results in openings greater than 40 acres in size but the DEIS does not disclose where those will be. Regional Forester policy (FSM 2471.1) directs the size of harvest openings created by even-aged silvicultural practices (e.g. seed tree, shelterwood, and clearcut harvest prescriptions) would be normally 40 acres or less. The National Forest Management Act at 219.27(d)(2)(ii): Size limits exceeding those established in paragraphs (d)(2) and (d)(2)(i) of this section are permitted on an individual timber sale basis after 60 days public notice and review by the Regional Forester.

The EIS does not analyze the impacts to wildlife of exceeding the opening size, it instead tiers to a generic Forest Plan standard that states that exceptions "may" occur. The impacts to wildlife must be analyzed on a site specific level and not rely on *aspirational* desired conditions.

• The baseline habitat conditions for grizzly bears in the Swan Valley is below what is scientifically necessary for secure core and road densities are above thresholds necessary for grizzly bears to safely live in the Swan Valley.

The Mid Swan project will reduce hiding cover and displace bears. In the Meadow Smith and Cold Jim subunits there will be near constant activity. Meadow Smith will have five years of activity, one year of rest then another four years of activity. Cold Jim will have four years of activity, one year of rest and another four years of activity. This will especially impact females with cubs.

• The DEIS fails to include logging on the Swan River State Forest in the grizzly bear cumulative effects analysis.

Climate Change

• The Forest Service did not take a hard look at how climate change affects and is affected by this project. Published scientific reports indicate that climate change will be exacerbated by logging, and that climate change will lead to increased wildfire severity (including drier and warmer conditions that may render obsolete the proposed effects of the project). The former indicates that the Mid Swan Project may have a significant adverse effect on the environment, and the latter undermines a central underlying purpose of the Project. Therefore, the Forest Service must candidly disclose, consider, and fully analyze the published scientific papers addressing climate change in these two contexts.

The Mid Swan Project purports to replicate past conditions created by fire by using logging; however, the effects of climate change were not adequately analyzed. These stands that are regeneration logged may not regrow due to increased temperatures drying out the understory. The Forest Service cannot use the past as a desired future condition because the future is uncertain due to climate change.

Challenges in predicting responses of individual tree species to climate are a result of "species competing under a never-before-seen climate regime – one forests may not have experienced before either." Achievable future conditions as a framework for guiding forest conservation and management, Forest Ecology and Management 360 (2016) 80–96, S.W. Golladay et al. (Attached)

NFMA requires restocking in five years. Forest managers must analyze and disclose the fact that the current conditions make old assumptions about natural regeneration obsolete. The Forest Service can no longer "insure that timber will be harvested from the National Forest system lands only where…there is assurance that such lands can be restocked within five years of harvest." (NFMA§6(g)(3)(E)(ii)).

Assuming large clearcuts will regenerate is no longer automatically consistent with NFMA's "adequate restocking" requirement. Scientific research can no longer be ignored.

At dry sites across our study region, seasonal to annual climate conditions over the past 20 years have crossed these thresholds, such that conditions have become increasingly unsuitable for

regeneration. High fire severity and low seed availability further reduced the probability of postfire regeneration. Together, our results demonstrate that climate change combined with high severity fire is leading to increasingly fewer opportunities for seedlings to establish after wildfires and may lead to ecosystem transitions in low-elevation ponderosa pine and Douglas-fir forests across the western United States. Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration*, PNAS* (2018)*,* Kimberley T. Davis, et al. (Attached)

The Forest Plan has no strategy for carbon reduction and this project will increase carbon emissions from log trucks driving to access the project area and from removing stored carbon from the forest.

The Mid Swan DEIS did not analyze the impacts to fish from rising stream temperatures, less water and increased peak flows due to climate change.

Other Issues

• In order to implement Alternative B the Flathead must deviate from its newly revised Forest Plan to allow excessive logging in lynx critical habitat and helicopters in recommended wilderness. It is unacceptable that you cannot even comply with the weakened Forest Plan.

• The DEIS fails to consider a reasonable range of alternatives.

• Whitebark pine "restoration" in the Mission Mountain Wilderness is an untested experiment that conflicts with the Wilderness Act and could have unknown consequences to the ecosystem that is not to be manipulated by humans.

This project should not proceed because it violates the National Environmental Policy Act, National Forest Management Act and Endangered Species Act.

/s/Arlene Montgomery
Program Director

FINAL

2 December 2014
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Comments on the

Revised Draft Recovery Plan for the Coterminous United States Population of Bull Trout (Salvelinus confluentus)

U. S. Fish and Wildlife Service Portland, Oregon

Available online at
http://www.fws.gov/pacific/bulltrout/pdf/Revised%20Draft%20Bull%20Trout%20Recovery%20Plan.pdf

SUMMARY

The 2014 Draft Recovery Plan for bull trout implies (and in a backhanded way specifies) that the USFWS assumes there is flexibility to make management choices deliberately allowing some core area populations of bull trout to go into decline or extinction, on the expectation others will appear from scratch, or disperse from severely depressed relict populations elsewhere in the Recovery Unit to arise in new locations. However this Draft Plan, the previous listing and recovery planning record, and the published literature present virtually no evidence to substantiate that new populations of bull trout have established in contemporary times, either at the Core Area scale or the next smaller scale of breeding populations. In this regard bull trout are the biological polar opposite of vagile species like wolves, which are demonstrated to be amenable to reintroduction and are proficient colonizers of new territory at the regional scale. On the other hand, we do have evidence that even small, so-called "relict" bull trout populations can rapidly re-establish migratory life histories, or expand extant spawning areas when changing habitat conditions allow it. But we do not know that they can establish new populations in previously unoccupied streams or watersheds under contemporary prevailing conditions. Hence from a scientific perspective, existing populations of bull trout, no matter how small and far-flung, must be viewed as the sole seed sources for future recovery.

The plan rejects biological and habitat data as the basis for recovery goals and delisting criteria, despite that the state of the art convention for making such determinations, the Viable Salmonids Framework, relies principally on such data and can be robustly informed with extant data sources for many, if not most populations of bull trout. Hence the treatment of threats as the prime basis for recovery in the Draft Plan becomes a construct unmoored not only from most relevant conservation science, but from the actual biology and ecology of bull trout in the field.

Several categories of threat are ignored or ill-described in the Draft Plan; any threat that is common in the real world but missed in this plan will likely not be recognized or will be given low priority in treatment, thereby seriously undermining actual biological recovery. Adaptive management is invoked in the plan, but not described or bounded in a way that would make it useful or meaningful for evaluating recovery actions and outcomes.

The central fallacy of the Draft Plan is its complete reliance on the wholly unexamined and undefended presumption that threats can be unambiguously identified, and with ease determined to be "managed" to the degree that biological recovery inexorably results. Threats can only be "managed" to the benefit of the species if assumptions about the relation of management actions to biophysical cause and effects, and thence the response of bull trout populations to habitat changes, are thoroughly evaluated with best available information *a priori* and verified *post hoc* with direct data on populations and habitat.

I. INTRODUCTION

I prepared the following technical comments pertaining to scientific elements and likely conservation effectiveness of the Revised Draft Recovery Plan for Bull Trout (USFWS 2014a) at the request from the Friends of the Wild Swan, Bigfork, Montana. This document expresses my professional opinion on matters related to the Draft Recovery Plan and the conservation biology of bull trout.

A CV is attached to outline my qualifications.

II. GOALS AND APPROACH: ECOLOGICAL REQUIREMENTS OF THE SPECIES

For the most part, the plan does an adequate job of characterizing in very general terms the habitat requirements of bull trout, and the general aspirational goals for recovery in terms of habitat conditions, connectivity, and robustness of population structure. In this vein, the Draft Plan draws on previous status reviews and draft recovery planning documents for the bull trout.

However the description of migratory life histories and habitat needs (p. iv-v and several subsequent text sections) oversimplifies what is known about bull trout movements, and the neglected information has a direct bearing on, and raises doubts about, the Service's ability to identify, protect, and "manage threats" to habitats critical to migratory bull trout. For example, Carnefix (2002) documented that some migratory adult bull trout in Rock Creek Basin, Montana, make pre-spawning forays well upstream into tributaries other than their natal streams. It appears likely these are foraging movements, and they may be essential to allowing fish to gain enough energy to complete successful spawning later in the year. After spending periods of days to weeks in non-natal tributaries, these fish move back to the mainstem of Rock Creek and thence to natal tributaries to spawn. These streams, though they may not support spawning and early juvenile rearing of bull trout, nevertheless play a potentially vital role in maintenance and recovery of migratory life histories that confer increased productivity, resilience and viability to their populations.

Loss of access to non-spawning tributaries, or decline or depletion of the productivity of forage fish or other food resources within these streams is likely among the many factors contributing to the simplification of life history diversity and migratory connectivity and resilience in many core areas across the range of the bull trout. If the Service does not recognize these complexities that contribute to bull life history diversity and population viability when assessing threats, then this draft recovery plan will likely fail to achieve its goals. This underscores the importance of developing and utilizing population-specific biological metrics to actually validate the hypothetical recovery gains that this draft plan posits will result from specific threat reductions. I.e., if we had not studied what these fish are actually doing in the field, we would have no inkling that "management of threats" to protect bull trout in Rock Creek should include protection of habitat and forage fish populations in specific tributaries that branch off the migratory pathway to headwater spawning areas.

III. DELINATION OF CONSERVATION UNITS FOR RECOVERY: CRITICAL UNSTATED PREMISES ABOUT DISPERSAL AND VAGILITY

The Service identifies three nested scales of biological units that are addressed to some extent in the plan. They are, in descending order of spatial extent, the Distinct Population Segment (DPS, the unitary aggregate of all bull trout in the lower 48 states as currently listed), the Recovery Unit delineated as an aggregate of watersheds with environmental and biological conditions for bull trout that are in some way (not specified in the Draft Plan) dissimilar from those in other recovery units, and finally Core Areas, which are watershed units within Recovery Units that contain one or more breeding populations of bull trout. Some Core Areas are connected to migratory corridors, but others are small and isolated from the surrounding landscape. The Draft Plan devolves the specific identification of threats and the assessment of whether they are "managed" and the degree of their likely impact to a future process of gathering expert opinion, rather than formal quantitative assessment methods, and these collections of opinion are focused at the Core Area scale. Delisting criteria, however, are left to determinations

made by USFWS staff, with few criteria identified for how they will be made, and no procedure identified for their validation.

The specific number and naming of core areas insofar as departure from previous draft recovery documents amounts to rearranging of deck chairs—it's not biologically consequential. Similarly, Recovery Areas, as constructs of management convenience that do not strictly accord with known genetic differences and inferred phylogeny of bull trout populations, also may amount to little more than deck chair arranging—except that the Draft Plan claims the option of delisting based on attainment of stated recovery criteria with Recovery Units. Hence, the crucial question is how the Service outlines that scientific and commercial data will be used to make biological determination at these scales.

I agree (as recommended in Ardren et al. [2011], and echoing earlier sources such as Dunham and Reiman [1999] that the Core Area scale is the most appropriate ecological and demographic unit for assessing biological viability and status of the species. The question as to how viability and status of core area can be rationally scaled up to assessing viability of the delineated Recovery Units remains extremely vague and unexamined in the Draft Plan, and is not self-evident biologically. The relatively high level of uncertainty about long-term persistence of bull trout in any demographically independent Core Area is correctly recognized by the Service at several places in this Draft Plan. The even greater uncertainty about whether past or future extirpated bull trout populations can be reliably replaced by translocation, stocking, or other artificial or natural means is not as clearly recognized in the Draft Plan, but it remains a major hurdle to recovery nevertheless (Clancy et al. 1996; reintroduction attempts in California have proven unsuccessful, and those in the Clackamas Basin in Oregon are too recently begun to be evaluated, and project managers expect it may take 20 years of intensive effort to ascertain success or failure [Koski et al. 2014]).

Therefore, to be reasoned and concordant with best available scientific information, the Draft Plan must explain why the extirpation of bull trout from any extant core area does not irretrievably foreclose future recovery opportunity, and therefore materially reduce the likelihood that the species will not be endangered in the future. It appears the 5-year status review (USFWS 2008) is the only quantitative assessment available to date of status of individual core areas across the range of bull trout. Of the then 121 core areas identified (Draft Plan p. 7), over half were of unknown population status—itself a most troubling statistic. Of the 55 core areas for which assessment was possible, bull trout populations in 42 percent (23) of them were reported to be declining; just 32 percent (18) were thought to be stable, and 25 percent (14) were reportedly increasing. Let's assume that the numbers for the known core areas represent the trend in the unknown other half—though that is likely grossly overoptimistic, given that many of the unknowns are among the smallest and most highly isolated and threatened populations in the DPS. With more than 40 percent of the total collection of core areas reported in decline, there is little apparent room for sanctioning further declines in the stable or increasing portion of the species range. Moreover, it appears that in preparing this Draft Plan, the Service both did not make an effort to revisit this Core Area status assessment to see how it might

have changed, and they took pains to ensure that delisting decisions would not rely on basic population-biology-informed assessments in the future.

High et al. (2008) reported that short-term trends toward seemingly stable bull trout populations in recent years appear to be a regional response to some regionally acting factor, such as more favorable climatic patterns in key seasons. Statistical analysis and spatial analysis did not support the hypothesis that positive and non-declining population trends were associated with watershed or stream-specific factors. Therefore it is erroneous to interpret this study as providing specific evidence for the effectiveness of recent land and water use practices, which are applied to individual watersheds in the course of timber cutting projects—not applied uniformly across the landscape. It should also be should be pointed out that scientifically there is no reason to expect short term changes in population trend in response to recent post-listing changes in management practice; biological response lags as a result of many complex lagged physical and population biological process-response feedbacks that are necessary for true recovery. Moreover, for simple reasons of statistical power, it is unwise to ascribe credence to trends in bull trout abundance that span periods of fewer than several generations, or a time series of ca. 15 years, and preferable any such inferences should rely on multiple metrics of population trend to allow checking survey methods for bias (Rieman and Myers 1997, Dunham et al. 2001).

Can the Service demonstrate and support with any actual data or scientific argument its hypothesis that the un-reversed loss of bull trout populations, the result of extirpation from a Core Area, can be compensated by increased populations in other Core Areas elsewhere in a Recovery Unit? The Service has not demonstrated this, and I sincerely doubt that they can. Moreover, this instrumental but never explicated assumption of the Service is most likely dead wrong. In effect, it appears the Recovery Units have been created in the Draft Plan as a bin for bean counting—not a unit of real and meaningful biological geography. In this Draft Plan, the Service essentially creates and uses Recovery Units to allow binning of Core Areas, to create the false perception that there is some kind of biological redundancy operating among Core Areas within a Recovery Unit. This use of the Recovery Unit to establish delisting criteria is essentially an administrative shell game—it is neither biologically defined nor scientifically justified. Most important, this sort of management shell game has irretrievable consequences in that it creates the putative discretion for the Service to allow the sacrifice of some Core Areas in favor of others. Yet this bean-counting and deeming flies in the face of the Service's own premise that environmental and demographic uncertainty—namely disturbance factors outside of management control, including for example, climate change effects, catastrophic drought, erosion or wildfire events—ultimately rules the fate of core areas. And in fact a large majority of those Core Areas expected to, in the Service's theory, replace the functions of Core Areas extirpated in the future are themselves showing ongoing decline or persistent lack of recovery.

Genetic research has shown a relatively high level of variation in genetic markers among stream-specific populations, even within basins, and a higher degree of differentiation among basins (Rieman and Allendorf 1999, Dunham et al. 1999. Whitely et al. 2006,

Ardren et al. 2011). Some of this observed variation is likely associated with strong selection for local adaptions that allow for persistence of indigenous populations in specific streams, rivers, and lakes; more important, most selected variation in physiological or behavioral traits is not reflected at all in genetic marker studies. Therefore, there is no assurance that translocated individuals will have behavior and tolerances favorable to survival in new habitats to which they are introduced. Ardren et al. (2011) conclude that focusing protection at grosser scales such as Recovery Units, at the expense of firm protection and restoration for Core Areas, risks irretrievable loss of genetic diversity within the species.

If there is functional biological redundancy present that is relevant for conservation strategy and recovery, it resides at a much smaller scale—that is, among populations within Core Areas, where metapopulation processes can operate to provide gene flow between breeding groups and to demographically reinforce or potentially naturally recolonize habitats where bull trout have been extirpated (Dunham and Reiman 1999, Dunham et al. 1999, Ardren et al. 2011).

Neither the Draft Plan nor any documents cited record a single example of a bull trout population having been established and persisting over time after an artificial introduction or a documented natural recolonization event, certainly not at the scale of a Core Area. (In the evolutionary history of bull trout, successful colonization of habitats obviously did occur, but those likely occurred under different environmental conditions more favorable to bull trout dispersal and survival.) Unless multiple such examples of success have been observed, any framework, such as that in this Draft Recovery Plan, that assumes extirpation from core areas is a retrievable loss to the species is not defensible. While the Service argues that recovery does not require the re-establishment of the species throughout its former range, a recovery policy for bull trout is not scientifically tenable if it expressly or passively allocates increased risk of extirpation of some Core Areas with the vague expectation that other Core Areas will remain to form the basis of some future hypothesized recovery.

To the contrary, it is consistent with all available scientific information to assume that bull trout range expansion occurred almost exclusively during more favorable climatic and hydrologic conditions that have not prevailed in recent centuries or decades, and that under current and foreseeable conditions, little or no recolonization of extirpated populations (or reintroduction success) is likely. Under this most-likely scenario, an effective recovery policy would need to be grounded in two principles: 1) preserving all extant populations and their present habitats to the extent this is possible (i.e., “keeping the seeds”), and 2) promoting range extension and the recovery of life history diversity by growing these extant populations through protection and restoration of habitats accessible to them. The basis for allocating resources in recovery under this more reasoned scenario is then a balanced portfolio: split between investing as needed to sustain extant populations no matter where they occur, and investing restoration dollars in those core areas with the greatest potential for near-term restorative actions. Protecting populations in extant core areas will often be more about implementing systemic policies for resource protection and maintenance and passive recovery of natural habitat-forming

processes than they are about investing in discrete habitat modifying projects (Kauffman et al. 1996, Ebersole et al. 1997).

Instead the stance of the FWS in the present Draft Recovery Plan appears to be presumptive that state and federal authorities, and all those with HCP permits, have for the most part already made all necessary improvements in natural resource policy already and that most or all habitat problems originate from "legacy" practices and events. Again, this is a crucial policy assumption on the part of the Service for which no shred of critical examination or specific evidence is provided. While legacy impacts may be large, the key effect of such legacy impacts can be that they render the affected streams and watersheds highly sensitive to even small, incremental disturbances caused by contemporary land and water use practices (e.g., Baxter et al. 1999, Foster et al. 2003).

IV. IMPORTANCE OF POPULATION AND DEMOGRAPHIC METRICS OF RECOVERY AND DELISTING

The Draft Recovery Plan discards reliance on quantitative targets for bull trout population size, number, life history diversity as factors for determining whether recovery is occurring; i.e., delisting criteria. Instead the Draft Plan relies almost exclusively on the Service's perception of whether identified threats have been "managed" to both structure and assess recovery. In so doing, the Service discards a large body of scientific literature and well-established scientific conventions for describing population and species status and recovery under the Endangered Species Act and in other fishery restoration programs. The Viable Salmon Populations concept (VSP), was developed by scientists at the National Marine Fisheries Service 15 years ago (McElhaney et al. 2000), and has been used extensively—in fact near universally—as the core biological framework for population and species assessment in status reviews, listing decisions, and recovery planning for salmonid fishes in the Pacific Northwest since then. The virtue of the VSP framework is that it relies on and integrates four major categories of biological information: abundance, population growth rate, population spatial structure (distribution), and diversity (expressed as life history traits and patterns within populations). In the scientific literature on fisheries and fish ecology, each of these four categories of information have been related to population and species extinction risk; each reflects factors that operate in all populations, and each is measurable and can be compared across populations or across time within a population, distinct population segment, or species (McElhany et al. 2000). A wide range of data sources can inform assessments and determinations with this framework (Ruckleshaus et al. 2002, Good et al. 2007), which provides a necessary biological platform for relating many kinds of information to the core question of species and population persistence and recovery. The VSP construct represents a significant improvement over earlier bull trout studies that relied on a narrower range of population characteristics to assess status (e.g., Rieman and McIntyre 1993).

Although the VSP framework is widely considered the state-of-art convention for status determination and recovery planning of salmonid fishes in the Pacific Northwest, the

Draft Plan offers no explanation of why the VSP concept was not adopted as an analytic tool in the recovery plan. Nor does the Recovery Plan explain why key data sources like bull trout redd count time series, observation of presence of migratory life histories within populations, field surveys of bull trout distribution and change in distribution over time, genetic estimates of effective population size, are no longer to be considered in establishing delisting criteria for bull trout. Each of these data sources has well-established protocols and methods, and known accuracy established through many years of research on bull trout (e.g., Rieman and McIntyre 1996, Rieman and Myers 1999, Dunham et al. 2001, High et al. 2008). Published scientific applications have shown how evaluation of time series in bull trout spawner counts has been instrumental in understanding the habitat and environmental factors underlying population trends and the robustness and recovery of populations within core areas (e.g., Rieman and McIntyre 1996, Baxter et al. 1999).

Relying on threats as the means to characterize recovery rests on a complex set of assumptions about the ways that threats act on the habitat and population ecology of bull trout to determine population viability. Erroneous assumptions about the outcomes of threat management are highly likely—in fact, virtually certain—if the underlying assumptions of cause and effect are not empirically checked against measured biological outcomes. The key biological outcomes are framed and captured in the VSP framework (Good et al. 2007).

More recent work has shown the potentially enormous value of allelic, genomic and population genetic information for understanding and predicting population viability, including size, exchange of individuals with other populations, and the development, fixation, or evolution of local genetic traits (Whitely et al. 2006, Ardren et al. 2011).

V. IMPORTANCE OF HABITAT METRICS IN RECOVERY AND DELISTING

The Draft Plan places no reliance on habitat metrics for assessing population status and delisting criteria. While the precision and inter-agency comparability of much stream habitat data are limited (Barnas and Katz 2010, Kershner and Roper 2010), specific kind of habitat data nevertheless play a crucial role in validation assumptions about threats and the direct and indirect effects of actions intended to alleviate threats—particularly where data strings of many years are available for important bull trout habitats. Carefully designed and executed habitat monitoring programs can have high precision and good power to detecting change in key habitat parameters (Larsen et al. 2003) known to determine bull trout survival and productivity. Habitat data in longer times series can provide important insight into the biophysical lags that determine bull trout population response to changes in threats (e.g., Baxter et al. 1999).

Some threats are in theory measurable, but they are expressed via numerous biophysical mechanisms. For example, roads influence many processes that affect aquatic ecosystems and fish: human behavior (poaching, debris removal, efficiency of access for logging, mining, or grazing, illegal species introductions), sediment delivery, and flow alterations

(Trombulak and Frissell 2000). Managers can influence multiple threats by reducing road impacts and density in key catchments. One could potentially reduce poaching by closing roads to public access seasonally; however if roads were physically obliterated, closure restrictions become unnecessary, and a much broader sweep of physical and biological impacts is reduced. Understanding these associations and the sweep of cause and effect relations associated with any single phenomena that we, for convenience, label a "threat" is only possible with good-quality information on processes and physical habitat conditions on the ground. Physical habitat information is crucial to assure that threat reduction is actual and effective, and not a paper exercise, and much useful information specific to bull rout conservation has already been published. Heedless, the Draft Plan lacks any critical examination or disclosure about how the effects of threats or their treatment on habitat and population viability will be informed prior to action, or verified after action. In my opinion this critical oversight throws the veracity of the entire Draft Plan into question.

VI. INADEQUATE AND ERRONEOUS CHARACTERIZATION OF THREATS AND THEIR MANAGEMENT

If the Service intends to govern recovery and delisting on the basis of threats, the Draft Plan evidences no great capacity for accuracy and fidelity in the Service's assessment of threats. For example, in 1999 Oregon's standing panel of independent scientists the Independent Multidisciplinary Scientific Team (IMST 1999) offered a detailed and expansive review highly critical of the adequacy Oregon's Forest Practices Rules to protect and restore the stream habitat of salmonid fishes in forested areas. Since that report, there have been only limited revisions of Oregon's rules, for a few narrowly defined stream categories only, and no known subsequent systematic assessment has determined the rules to be adequate to support bull trout recovery. Research in the past decade has shown that existing forest practice rules commonly result in stream harmful warming beyond the water quality standard for salmonid streams in western Oregon (Groom et al. 2011a, 2011b). Rulemaking to address the inadequacy of buffer rules to maintain shade is lagging many years behind the research, and there is no comparable study or rulemaking in process for Oregon forests east of the Cascades, where more bull trout populations are extant. Recent research by Pollock et al. (2014) demonstrates that wood debris recruitment is measurably impaired by selection logging or thinning in Douglas-fir-dominated forests within 100 feet of streamsides. Because most states and tribes within the range of the bull trout allow thinning within this zone, wood debris depletion is an ongoing threat to bull trout because of inadequate forest practices and stream protection rules on private, state, and federal forest lands.

Washington DNR's Forests and Fish HCP documented a great deal of unresolved uncertainty about the adequacy of the rules to protect bull trout habitat on private forestlands. As a result, forest practices rules under the HCP were implemented on a provisional basis pending the outcome of a series of monitoring studies (under the so-called CMER Program) taking a close look at in-stream habitat outcomes after logging in Washington (WDNR 2014). Hence, assurances that the HCP's requirements for forest

practices are sufficient to protect bull trout have not been made, and the plan is provisional based on the outcome of the ongoing studies. So while the FWS may be correct that improvements have been made in Washington stream protection rules, the adequacy of those rules to fully protect and restore bull trout habitat sufficient for recovery of the species in Washington remains unresolved—contrary to the impression left in the Draft Recovery Plan. To my knowledge Washington, Montana, and Idaho have completed no significant revisions of their private forest practices rules, and there have been no systematic scientific analyses of their adequacy to protect and restore habitat for recovery of bull trout conducted in recent decades.

Rashin's (2006) field study of forest practices and post-logging erosion in Washington showed numerous examples of harmful sediment entry to Washington headwater streams under existing Forests and Fish rules. They reported that current Washington rules (similar or weaker rules apply to headwaters streams in other PNW states) do not adequately protect streamside zones in headwater streams from equipment damage and skid trail, and yarding corridor erosion.

Rashin et al. (2006) reported that riparian vegetative buffers (approximately 30 ft or more wide on each side of channel), including leave tree areas and with required directional felling and restrictions on log yarding or ground-based equipment use, were wholly or partially effective in preventing near-term sediment delivery from logging disturbance. Current rules allow ground disturbance near enough to headwater stream channels (upstream from fish-bearing zone) that sediment delivery is likely (much of this was associated with skid trails and soil scars from felling and log yarding). Area of exposed soil that delivered sediment to stream channels was an order of magnitude higher in sites logged without stream buffers compared to those logged with substantial riparian buffers.

Effectiveness of stream buffers was breached at yarding corridors through buffers (associated with ground scars from cable yarding), and where selective logging occurred on steep inner gorge slopes within (or possibly adjacent to) the buffer areas (Rashin et al. 2006). Stream buffers were generally effective at reducing near-term sediment delivery to streams regardless of whether surrounding logging was by clearcut or partial cut. However, yarding corridors cutting through Riparian Management Areas (also expressly allowed by Oregon forest practice rules, and common practice in other states) measurably compromised the effectivenes of stream buffers in preventing sediment delivery. Stream buffers were most effective where timber felling and yarding activies were kept at least about 30 feet from streams and kept entirely away from steep, near-stream inner gorge slopes. Simple exclusion of logging-caused ground disturbance from near-stream areas and hillslopes with a high likelihood of delivery of sediment to channels accounted for most of the effectiveness of stream buffers. The primary observed causes of soil disturbance and erosion were skid trails from tractor yarding and yarding scars from cable yarding.

Rashin et al. (2006) reported that where riparian vegetative buffers were not left, erosion and channel response were clearly linked to surrounding logging practices. Specifically, clearcuts tended to generate more sustained active erosion and sediment delivery than

partial cuts. In winter-cold, interior forests, logging over snow and frozen ground appeared to substantially reduce observed erosion and sediment delivery.

Despite that most streams in the study were small headwater channels with steep channel slopes (exceeding six percent), surveys revealed numerous instances of fine sediment accumulation on streambeds, both in pools and across the entire bed, associated with localized sediment sources (and in some cases, accumulations of logging slash). In addition, the extent of actively eroding streambanks increased in streams logged without buffers. These observations contradict the oft-repeated but seldom-tested presumption that sediment entering steep headwater streams is "rapidly flushed out" and therefore is assumed to have little effect on instream biota and water quality. Gomi et al. (2005) have previously commented that sediment routing and fate in headwaters streams in the Pacific Northwest has received insufficient study relative to the inherent risks fine sediment poses to aquatic resources.

Note the Rashin et al. (2006) study is conservative in the sense that "controls" were previously-logged, second-growth sites, and hence "background" or baseline incidence of erosion was likely elevated over natural rates in watersheds not previously disturbed by logging and roads (see below, Keppeler 2012). This means that "background" sediment conditions in the Rashin (2006) study were likely already well elevated over conditions that would have prevailed in undisturbed natural forest. Nevertheless the analysis provides a useful basis for evaluating future management of the existing, largely second- or third-growth, road-private forest landscape.

The most important caveat is that Rashin et al. (2006) only evaluated localized erosion and sedimentation within the immediate two years after logging. Many potential sources of impact to streams, including those that result from hydrologic change caused by vegetation removal, and those that propagate over time and space, were not accounted for in the study design. Hence it is also important to recognize that some additional effects could occur that might not be fully mitigated by riparian buffers. Such processes include headward channel incision or gullying, landslides that increase with root strength depression occurring several years after harvest, and streamside erosion increases resulting from debris flow scour (Gomi et al. 2002, May 2002, May and Gresswell 2003), or the passage of coarse sediment waves (Beschta et al. 1995, Frissell 1992). To my knowledge, neither the US Fish and Wildife Service nor any other entity has conducted comprehensive research and assessments of the kind that would be necessary to assure that current forest practices of any state are fully protective of bull trout and their habitat.

VII. FAILURE TO CONTROL OBVIOUS THREATS TO CRITICAL CORE AREAS IN RECENT DECISIONS

While putatively this Draft Plan recognizes the importance of Core Areas and of maintaining those with more extensive habitat and larger and more robust populations, at the same time this plan was being prepared, the Fish and Wildlife Service prepared Biological Opinions permitting the development of two large mines that would together

deplete flows and depress or possibly extinguish bull trout in the only two remaining streams with reasonably robust bull trout populations in the Middle Clark Fork Core Area: East Fork Bull River (Montanore Mine [USFWS 2014b]) and East Fork Rock Creek (Rock Creek Mine, [USFWS 2006], and additional effects from Montanore). The notion lurking behind these decisions--that bull trout in two Core areas might be sustained by recovery of populations in heavily damaged streams where they have been nearly eliminated for decades (e.g., Prospect Creek)-- is assumed but not rationally defended by the Service and is on the face of it far-fetched and scientifically ludicrous. Again, the Service recognizes it has no established track record of major successful bull trout population recovery in response to habitat restoration actions, and no source of funds to support the massive restoration effort that would be required even to test the proposition.

VIII. THREATS NOT ADEQUATELY RECOGNIZED AND EVALUATED

Streamflow Depletion. It has long been known that reductions of low flow in streams can harm salmonids through reduced habitat capacity, survival and growth (e.g., Harvey et al. 2006). Recently published field research from New Zealand (McHugh et al. 2014) described the very important consequences of streamflow depletion (seasonal and across streams of different flow recession trends) on food webs in streams. The work provides dramatic evidence of food web truncation in stream pools as baseflow declines, which has potentially very large effects on productivity of streams for upper-level predators, such as bull trout. The models recently used for predicting the bull trout population consequences of flow depletion effects in relation to projected depletion of streamflows from construction and closure of the Montanore and Rock Creek Mine projects did not include any accounting for food web effects, which likely for bull trout outweigh the direct effects of flow depletion on simple habitat volume. Streamflow depletion is a consequence of other land and water uses besides mining, and these uses are widespread in the range of the bull trout (Luce and Holden 2009). For example, stream summer baseflows have been shown to decline in the decades after logging, with streamflow depletion effects lasting far longer than the relatively transient increases in flows seen in the initial 5 years or so after logging (Hicks et al. 1991, Moore and Wondzell 2005). These depletions appear to be associated with vigorous growth of hardwoods and other phreatophytic vegetation, or to general reduction in transpiration efficiency in second-growth forests, and a reduction in fog-drip water recovery. Few time series in experimental watershed studies have continued long enough to document the extent of this flow depletion effect in streams outside of western Oregon.

Chemical Use in Forests. Only very recently has science begun to directly tackle the difficult questions of fate, effects, and toxicity of pesticides and other chemicals associated with forestland uses on stream biota. Toxic contaminants come from various sources, including storm water runoff from roads (particularly those that discharge directly to surface waters pipes and ditches) (McCarthy et al. 2008). Herbicides are applied to tree plantations and roadsides to control unwanted vegetation. Until recently these activities were limited by court order on BLM and USFS lands, but now they are

increasing in extent and frequency, as well as continuing on adjacent private forest lands. The NMFS is reviewing the science concerning potential harm to listed species of Pacific salmon from application of commonly used pesticides. For example, use following label restrictions of the herbicide 2,4-D was determined to jeopardize Pacific salmon (NMFS 2011). Forest fire retardants that are aerially dropped in large quantities during wildfire suppression operations often reach surface waters, where they may be toxic to salmonids (Buhl and Hamilton 1998, Gaikowski et al. 1996).

While the science on toxic chemicals has advanced, management practices to protect bull trout and their habitat from chemical contaminants on state and private lands in the Pacific Northwest appear to have greatly lagged.

Roads. Roads are ecologically problematic in any environment because they affect biota, water quality, and a suite of biophysical processes through many physical, chemical, and biological pathways (Trombulak and Frissell 2000, Jones et al. 2000). The inherent contribution of forest roads to nonpoint source pollution (in particular sediment but also nutrients) to streams, coupled with the extensive occurrence of forest roads directly adjacent to streams through large portions of the range of bull trout in the coterminous US, adversely affects water quality in streams to a degree that is directly harmful to bull trout and their prey. This impairment occurs on a widespread and sustained basis; runoff from roads may be episodic and associated with annual high rainfall or snowmelt events, but once delivered to streams, sediment and associated pollutant deposited on the streambed causes sustained impairment of habitat for salmon and other sensitive aquatic and amphibian species. Current road design, management of road use and conditions, the locations of roads relative to slopes and water bodies, and the overall density of roads throughout most of the Pacific Northwest all contribute materially to this impairment. This effect is apart from, but contributes additively in effect to the point source pollution associated with road runoff that is entrained by culverts or ditches before being discharged to natural waters.

The magnitude of existing road impacts on watersheds and streams in the Draft Plan might equal or exceed the effect of all other activities combined. In the Columbia Basin, bull trout presence is strongly associated with road density (Quigley et al. 1996, Baxter et al. 1999, Carnefix and Frissell 2009). Other salmonid species are also known to be negatively associated with road density at a regional scale (e.g., Firman et al. ([2012] on coho salmon in coastal Oregon streams). Kaufmann and Hughes (2006) found that road density in Coast Range streams was associated negatively with 25-50% of the variability in condition of aquatic vertebrate assemblages. More recently, Meredith et al. (2014) showed that the abundance of habitat-forming wood in Columbia Basin streams declined with proximity to roads, and the effect was roughly the same magnitude as that of natural climate and vegetation differences or of long-term livestock grazing.

Roads are necessary to support logging, mining, grazing, and motorized recreation, but the existing federal forest road system far outstrips the extent of those demands. The number and poor condition of USFS and BLM roads, the agencies' inability to prevent current roads from deteriorating and harming streams, and the pervasive effects of roads

on the physical and biological environments were recognized in FEMAT (1993). In addition, forest roads have been the subject of high-profile national dialogue and policy reviews since the development of the Draft Plan (Gucinski et al. 2001, Pacific Rivers Council 2008). The Aquatic Conservation Strategy's (ACS) primary means of protecting streams from roads and encouraging effective restoration are twofold: first, ACS objectives discourage locating roads within Riparian Reserves, and second, roadless areas were to be maintained and overall road density reduced in Key Watersheds. For a small number of priority watersheds where road network reduction has been pursued, agency monitoring efforts have reported improvements of certain instream habitat conditions, a response not detected elsewhere (Reeves et al. 2006). Recent proposals to reduce the size of riparian reserves on federal lands in western Oregon and Washington could provide more free rein for the construction of roads and landings in closer proximity to streams, markedly increasing the likelihood of sediment delivery and alteration of near-stream hydrology.

How to substantially reduce road density in critical watersheds and improve road drainage, stream crossings, and other factors that affect streams and aquatic biota, while maintaining sufficient roads for other land uses, remain central challenges to forest planning and management. The Northwest Forest Plan and other operative plans and policies lack sufficient means and impetus to accomplish systematic reduction of net road impact on watersheds and freshwater ecosystems (Frissell et al. 2014).

Numerous studies have established watershed-scale statistical relationships between road density, stream habitat conditions, and fish populations (Trombulak and Frissell 2000, Carnefix and Frissell 2009) and road density often rises to the top of the list of landscape physical variables that correlate with fish population status in landscape-scale studies (Lee et al. 1997, Baxter et al. 1999, Bradford and Irvine 2000, Sharma and Hilborn 2001, Carnefix and Frissell 2009). Harms are likely expressed at much lower road densities in some watersheds, but in this range fish response may be muted by variation such as road condition and locations relative to sensitive habitats, time since road construction, occurrence of triggering storms, and other factors. Road segments at high road densities have redundant effects, and the first increments of road construction may impose the largest effects on aquatic ecosystems (Carnefix and Frissell 2009); that is, most ecological harm is likely expressed by the time moderate road densities are reached, and with only marginal further increments of harm at the highest road densities.

Moreover, while sediment delivery is of undoubted importance, it is not the only mechanism by which roads harm fish and other biota in streams; roads also affect peak and potentially low flows and may increase water temperature fluctuation by intercepting and diverting shallow groundwater, and roads are both sources and vectors for the delivery of nutrients and traffic-borne chemical contaminants to streams and rivers (Trombulak and Frissell 2000, McCarthy et al. 2008). No watershed or regional standard or metric of environmental acceptability of road network condition, e.g., for bull trout recovery, can be successful at explaining biological conditions and responses if it does not incorporate some measure of road density that accurately reflects the impact of all extant roads on the ground—including abandoned roads.

The states are remiss and tardy in addressing forest, rangeland, and residential road problems with planning and policy, with the possible partial exception of forest roads in Washington, managed under the Forests and Fish HCP. Oregon, Idaho, Nevada, and Montana have adopted no watershed-scale measures of road system condition to establish a benchmark for acceptable conditions for salmonid persistence and survival, water quality, and other water resources. This curtails the states' ability to measure progress toward water quality compliance and maintaining beneficial uses, and contributing to bull trout and salmon recovery. It is important to note that in each of these states a very large number of water bodies is listed as impaired by sediment under Clean Water act 303(d) rules.

Nutrients. Although relatively little studied in the Pacific Northwest compared to other parts of the world, much is known about the phenomena of eutrophication, which is the ultimate result of increased nutrient delivery to fresh and marine waters. Increased nutrients, particularly when N and P are combined, can cause a host of undesirable effects where they accumulate in downstream waters, including habitat changes that favor warmwater and introduced fishes over coldwater and native species. For example, eutrophication and rampant growth of underwater vegetation in reservoirs could favor introduced species such as northern pike, walleye, or bass that may prey on or compete with migratory bull trout (DeHaan and Bernall 2013, McMahon and Bennett 1996).

When delivered to steep headwater streams, nutrients commonly move swiftly through them. While traveling downstream, some portion of nutrients leach from stream waters into riparian aquifers where they may be taken up by the roots of riparian plants. Some portion is taken up by in-stream algae and other aquatic plants, cycling the aquatic food web. In anaerobic microhabitats, some nitrate-N may be taken up by denitrifying bacteria and reduced to elemental nitrogen, N_2, and lost to the atmosphere. P is commonly attached to inorganic or organic particles and can be deposited in overbank areas and cycled back into terrestrial vegetation. But most these riparian and in-stream cycles of uptake tend to keep much of the N and P in or within close proximity of the stream system. Nutrients taken up by near-stream vegetation, unless they are later consumed by animals and moved away from the stream, tend to return to the stream with leaf drop or litterfall. Nutrients cycled in aquatic plants and animals generally return to solution in the water when the organisms die. Hence the bulk of nutrients that enter headwater streams probably work their way to downstream receiving waters.

Increased algal growth in streams associated with nutrient inputs can result in increased oxygen consumption at night when the expanded plant community is respiring but not producing oxygen through photosynthesis. Large day-to-night swings in oxygen concentration and even pH can result, producing stressful conditions for fishes and other aquatic organisms. When these nutrients eventually work their way downstream to large pools, backwaters, wetlands, and coastal lakes, they can produce acute eutrophic effects. These effects include explosive growth of nuisance algae and aquatic macrophytes, oxygen depletion, high concentrations of plant-derived solutes in the water that result in acidic conditions, discoloration, and unpalatable odor and flavor. Even slight

eutrophication in streams, rivers, reservoirs, and lakes receiving increased nutrients from forestry or agricultural land use could favor introduced species that compete with or prey on bull trout, but to my knowledge this prospect has never been thoroughly assessed. Nutrient loading derives from multiple sources, including forest management, residential, recreational, and industrial development; ongoing tilling, irrigation, fertilization and herbicide application on agricultural lands; pollution of groundwater by surface runoff; construction and maintenance of roads and highways; and naturally caused events like flooding and wildfire. Many of these nutrient sources will increase in area or frequency with ongoing human population growth, and are likely to be further aggravated by climate change. Many of them are either poorly controlled by existing regulatory mechanisms (e.g., agricultural and land development practices, flood and wildfire hazard, groundwater quality and quantity), or are beyond direct human control (wildfire and flood initiation).

Roots and rhizomes of forest vegetation can effectively uptake N from soil and groundwater, particularly in riparian areas where phreatic water is often nearer the surface and rooting zone that it sometimes is in upland areas. P, on the other hand, is typically occurs attached to particulate solids, and as a result it is physically entrained by vegetation and by surface roughness and ponding at the soil surface created by downed woody debris or rock fragments. Vegetation and debris is less effective at filtering overland flow as slope steepness increases, hence P delivery to streams is commonly higher with steeper side slopes both because soil disturbance and erosion rates are higher and surface retention of particles is reduced, so that more soil reaches streams.

Experimental studies of nutrient uptake as water passes through forested buffer zones have established that wider buffers are needed where slopes are moderate or steep than when they are gentle and subsurface percolation is slow. Research in the Upper Midwest and other regions (Nieber et al. 2011, Sweeney and Newbold 2014) has established some general information on uptake rates relative to forest buffer widths that were not available 10 or 15 years ago. A recent meta-analysis of multiple field studies showed that, as a general rule, in terrain with gentle side slopes, a 100 foot forest buffer retains about 80% of the N and P passing through in surface and subsurface flow. That is, at 100 feet, about 20 percent of the N mobilized from upslope disturbance filters through and reaches the receiving water. A 50-foot buffer allows about 35% of mobilized N and P to pass through. Although few studies are available for wider buffers, by extrapolation it appears that forest buffers of between 150 and 250 feet width are likely necessary to ensure that *ca.* 90 to 95 percent of nutrient load is scrubbed from the water before it reaches the adjacent stream, wetland, or lake. The effective distances might in fact be greater (i.e., retention efficiencies lower) for most federal forest lands because side slopes are generally higher and soil porosity is likely greater than those in most available field studies. Research also shows that continuous buffers are needed to effectively protect receiving waters from retained nutrients; once nutrients reach surface waters in segments by penetrating areas with narrow or no forest buffers, they generally remain in the aquatic system.

Expanding fully forested stream buffers to at least 120 feet width from the edges of stream channels, including intermittent and ephemeral headwater streams, would likely be needed to attain nutrient retention efficiency above 90 percent (Sweeney and Newbold 2014, Frissell et al. 2014). This practice would apply to all land use categories, including forestry, cropland, rangeland, urban and exurban lands. Existing practices fall far short of this standard on all land ownerships, except designated federal Wilderness Areas and many areas within national parks.

Assessing Cumulative Consequences of Multiple Threats and Trends in Threats. Martinuzzi et al. (2014) provide a systematic conceptual framework and method for assessing threats and the likely effectiveness of conservation measures for freshwater ecosystems on regional and national scales. Despite the putative reliance on threats as the basis for determining status and delisting in this Draft Recovery Plan, the Draft Plan offers no scientifically defensible framework for the quantitative characterization of threats, nor for the many potential interactive effects of multiple threats acting on bull trout habitat and populations. The Draft Plan also utterly lacks a defensible, data-informed, forward-looking assessment of the full range of threats based on known status and trends of threat management, including most importantly, human population growth, land use change, climate change, and their many ecosystem effects. In sum, not only is the critical cumulative analysis of threat factors for bull trout lacking here, no framework is provided to develop or inform such an analysis.

If the Draft Plan is to provide a credible basis for measuring recovery in terms of threats, a systematic framework and valid prospective analysis are necessary elements. These do not exist in this Draft Plan; instead we have vague and wishful language about threats being "managed" or not, lacking any analysis or even definition of what a "managed" threat is, and what is biologically accomplished when a threat is "managed" v. when it is "unmanaged." It is wholly inadequate to rely on existing general or generic published assessments of threats; few are specific to bull trout habitat and ecological requirements, and even fewer have been systematically evaluated to ascertain their effectiveness to protect and restore bull trout. For example, even among state and federally sanctioned "Best Management Practices" to protect water quality, very few have been the subject of effectiveness monitoring or research that established their sufficiency to protect habitat required by bull trout.

Appendix E in the Draft Plan provides a simple categorization of threats and offers some organization to how the categories are arrayed, but identifies no systematic means of informing or objectively determining the relative and potentially changing importance of different threats, their relationship to each other, and how they are likely to trend in the future. The foremost challenges with assessing threats both in terms of scientifically describing their effects and in terms of management to alleviate them are not fine scale and local, but broadly regional and global. State and federal laws and policies largely dictate what authority exists to manage threats to any given degree or purpose, but limits on authority relative to recovery needs of bull trout are not defined in the Draft Plan. The text on this subject points generally backwards, to what the Service believes has been

accomplished in the past, rather than pointing forward to what must be accomplished in the future if bull trout recovery is to be a likely outcome.

In fact a systematic and reasoned analysis, verified with ecological and biological information about real world habitat and bull trout population response, would reveal that the notion that threats can be classified with a simple bifurcated choice as "managed" or "not managed" is spurious. Threats mostly occupy a gray area both in their degree of management control, and the nature and magnitude of their environmental impact on bull trout. In some places at some times a particular threat may cause severe and immediate harm, at other times lesser ongoing harm. At other times they only cause harm when another threat factor is also acting; in other circumstances the biological harm caused may not be evident until some time in the future. In still other circumstances, a perceived threat may cause no harm, or in some limited contexts be seen as beneficial to bull trout recovery. The result is that threats predominantly occupy a gray area of relative impact or harm to bull trout; if present, there is a complex array of conditions or levels of "management" that affects whether or not harm is caused. There is no simple way to classify threats as "managed,"

In this Draft Plan the idea of a "managed" threat is tautological; it is implied that "managed" means harms from the threat have in theory been alleviated. No framework, method, or criteria are identified for how such a determination is to be made. This is a core fallacy in the present Draft Plan: *It relies entirely on the wholly unexamined and undefended presumption that threats can be unambiguously identified and with ease determined to be "managed" to the degree that biological recovery inexorably results.* This presumption may be a matter of convenience to managers and policymakers, but it is of absolutely critical biological importance to the survival and recovery of bull trout.

IX. A VAGUELY DEFINED ROLE OF ADAPTIVE MANAGEMENT

Adaptive management is prescribed in the Draft Plan with no sideboards or definition for what is intended. Like Hell, the road to adaptive management is paved with good intentions. Effective adaptive management for biological restoration and species recovery requires close attention to defining testable treatments, adequate scientific engagement in monitoring, analysis, and reporting of results, followed by a formal decision process that is triggered by a predetermined threshold response in the data, and is bounded by meaningful *a priori* biological criteria to gauge outcomes (McDonald et al. 2002, Murphy and Weiland 2014, Westgate et al. 2014). Lacking formal hypotheses about processes and outcomes, planning for specific triggers, adaptive management is generally a euphemism for trying to get away with minimum action and small, incremental changes in practice, in hopes that future data will be less incriminating to the practice than current data are (Murphy and Weiland 2014). Ecological lags between management, stream habitat response, and fish population response preclude adaptive management except over very long time frames and with multiple replicated treatments (usually impossible in watersheds or other natural ecosystems, McDonald et al. 2002). This necessitates a highly precautionary approach to ensure that a listed species is not

further jeopardized by the actions taken or not taken on the basis of experimental research, even when that research is labeled as adaptive management (Alverts et al. 2001, Murphy and Weiland 2014). Rather than identifying any of these criteria, the Draft Plan simply identifies adaptive management as an undefined means of resolving (entirely unspecified) uncertainties and potentially improving (in no specified way) future decisions and actions—with no identified means of providing the species a hedge against experimental actions or inactions that could cause widespread, irreversible harm.

Therefore, if bull trout recovery decisions are to rest on adaptive management the Draft Plan needs to spell out a formal process for design and implementation of it. That policy should also spell out under what conditions adaptive management is appropriate, and under what conditions it is not. Resource requirements and administrative responsibilities to adequately service the considerable scientific component of design, data gathering, reporting, and evaluation of results are needed.

X. CLIMATE CHANGE AND IMPORTANCE FOR RECOVERY OF GROUNDWATER AND HYPORHEIC PROCESSES AND THEIR RESTORATION

Deep groundwater is vulnerable to warming from climate change, roughly in proportion to the mean annual air or ground temperature of the groundwater region. Recently Menberg et al (2014), for example, reported annual incremental warming of soil groundwater temperature that closely tracks the regional climate change trend. Previous published work by Tague and Grant (2009) projects that deep groundwater dominated streams in the Cascade Range would respond similarly and become less cold in roughly linear response to change in average annual air temperature. Probably the largest effect of this groundwater warming is to increase stream temperature at the point of origin of surface flows, where stream flow arises from bedrock interface springs. By contrast, hyporheic flow in alluvial aquifers (Poole and Berman 2001, Baxter and Hauer 2001, Poole et al. 2008), especially in larger forested floodplain systems where water discharges in midsummer was entrained during spring snowmelt runoff, might in fact be better-buffered against climate forcing then deep groundwater is. However, where most snowmelt runoff is converted to rainfall upstream in the catchment, that would likely be associated with a nonlinear jump in warming of the hyporheic aquifer and the surface waters it influences, because of the increased temperature of spring peak flows that recharge the hyporheic aquifer.

The best-buffered systems for coldwater species in the face of climate change are likely to be those with a high proportion of very high elevation terrain and some well-developed alluvial floodplains, as well as a lot of natural sinuosity and channel complexity in the mid-reaches that promote entrainment of spring runoff into the local alluvial aquifers, then slow discharge delayed 2-4 months, offsetting summer warm flows with cold hyporheic discharges.

Methods have been widely applied to characterizing the influence of upwelling cold water on temperature refugia for coldwater species of salmonids (e.g., Ebersole et al. 2003, Torgerson et al. 1999, 2012). There is a great deal of scientific knowledge about thermal complexity and fish response in the Pacific Northwest that should be applied to bull trout to better predict likely specific population responses to anticipated climate warming. There are also studies specific to bull trout that elucidate these factors (Baxter et al. 1999, 2000). For example, Baxter et al. (1999) reported that bull trout spawning streams with more extensive alluvial landforms, associated with alluvial aquifers and hyporheic flow exchange that buffer stream temperature, supported more stable spawning populations than nearby streams with fewer alluvial landforms. Some research methods used in prior studies for other salmonids need to be applied to bull trout to determine how they exploit thermal refugia for survival (e.g., thermal recording radio tags on individual fish in the wild). This is particularly a major issue of concern for migratory bull trout, for which managers have relatively little information about habitat needs to guide restoration.

Streams that currently support bull trout, but might warm in the future in response to projected influences of climate change (Dunham et al. 2003, Rieman et al. 2007, Jones et al. 2014) should not be written off as certain or likely extinctions in the Draft Recovery Plan. The Service errs in this regard. While conditions in and surrounding such streams, rivers and lakes were likely to become more harsh under climate change scenarios, bull trout isolates presently persist at many of these locations, and have done so for centuries or millennia--because the habitats are sustained by hydrologic or other biophysical anomalies that sustain cooler water temperatures than general regional models would predict. Streams whose thermal regime is strongly influence by hyporheic discharge in summer of cold water that was entrained months previous during spring runoff are not likely to warm in direct response to atmospheric temperature increases—rather they will respond more strongly to runoff changes and possible changes in channel complexity that cause increased or decreased entrainment of flood flows into shallow alluvial aquifers. Research suggests that natural recovery of riparian vegetation (e.g., Arismendi at al. 2012) and active, large-scale restoration of channel planform complexity and appropriate land and water use that increases entrainment of high flows onto the floodplains and into the hyporheic system (e.g. Arrigoni et al. 2008) can potentially buffer stream temperatures with enough magnitude to offset and potentially nullify expected warming caused by climate change. Climate change increases, rather than decreases, the need for and potential recovery benefit of active restoration of floodplain and channel processes (Furniss et al. 2010, Seavey et al. 2009).

The Draft Plan appears to assume that because climate change may hasten the extinction of some populations and thwart restoration efforts in others, that is grounds for not adopting any quantitative criteria for population persistence or distribution. However, the continued persistence of bull trout in arid and semiarid regions within the range of the bull trout indicates that persistence and potentially expansion of extant populations possible under climatically stressed conditions, if the natural hydrologic mechanisms that maintain water quality and habitat for bull trout are protected and restored. It is critical that these hydrologic and ecological factors that confer resilience to extant bull trout populations be specifically identified and protected. Furthermore, recovery will rest on

the ability to restore these factors and functions in other locations within and between Core Areas and migratory corridors. Limiting and reversing threats to these particular functions are likely among the most crucial for long-term survival and recovery of the bull trout. The Draft Plan fails to incorporate any clear strategic assessment, prioritization, or specific slate of actions pertinent to sustaining these critical environmental actors and functions. Because even isolated and relict populations could form the nucleus of recovered Core Areas in the future, each population should be examined closely with regard to the habitat factors that currently contribute to its persistence and the potential for expansion or restoration of conditions that could favor recovery, before their potential significance to future species recovery is discounted.

However, regardless of these various potential sources of local resilience, stochastic factors are likely to take a toll in extinction (Reiman and McIntyre 1993, Reiman et al. 1997) and smaller isolates will be more vulnerable to extinction because they are less diverse in spatial occupancy and life history pattern, and vulnerable to other small population size effects. The stochasticity of extinction from natural stressors and events, taken together with the existing reality that across much of the range of bull trout, recovery must rest on growing populations from a few scattered isolates, means that scientifically speaking, the FWS cannot have the luxury of allocating extant populations to future decline or to conditions in which their local recovery is unlikely or impossible. Stochastic events can act without regard to the Service's plans, therefore beyond accepting that extant relatively robust populations that include at least some migratory life histories have higher intrinsic resistance to extinction, the Service cannot in fact anticipate which populations might be essential for supporting recovery in the medium and long term.

XI. EMERGING RESEARCH RELEVANT TO THE RECOVERY PLAN

The following areas of emerging research could imminently re-shape the science on which the Draft Recovery Plan rests, and more dramatically increase the feasibility and reduce the cost of gaining direct information on population-level demography—vital information for conservation that is black-boxed in the present Draft Plan.

Improved methods of effective population size estimation using genetic markers, which can replace cumbersome, time-consuming, and expensive traditional fish capture methods for census. There are also pending and recent advances in using DNA sequencing to more precisely characterize pedigrees in field sampled organisms, determine the local structure of breeding groups or populations, and measure gene flow among nearby populations (Gordon Luikart, Flathead Lake Biological Station, Polson, MT, Personal Communication; Kovach et al. MS in Review).

Improved methods of detecting presence of bull trout (and other species) upstream from DNA fragments in water samples, resulting in the possibility of more accurate and quick mapping of distribution and presence/absence of populations of unknown status.

(Emerging technology being developed in numerous labs, including Dr. Mike Young, USFS RMRS, Missoula).

Climate change modeling projection studies, especially including several at Rocky Mountain Research Station, Young, Isaak, and Luce PIs. Note that predictions from climate forcing models can be seriously inaccurate to predict instream conditions if they do not explicitly take in to account for the routing of flows through shallow groundwater and hyporheic flow compartments (see discussion above).

XII. LITERATURE CITED

The following sources are incorporated in my comments above by reference, are available either in hard copy or via the web. The author can provide a copy of any of the following studies not otherewise available to the US Fish and Wildlife Service upon request.

Alverts, R., Calhoun, J. M., and Lee, R. L., (technical editors). 2001. Organizational Learning: Adaptive Management for Salmon Conservation Conference Proceedings. University of Washington, Olympic Natural Resources Center, Forks, WA USA. Online at: http://www.adaptivemanagement.net/content/organizational-learning-adaptive-management-salmon-conservation-conference-proceedings

Ardren, W. R., DeHaan, P. W., Smith and 10 others. 2011. Genetic structure, evolutionary history, and conservation units of bull trout in the coterminous United States. *Transactions of the American Fisheries Society*, *140*(2), 506-525. Online at: http://www.tandfonline.com/doi/abs/10.1080/00028487.2011.567875?journalCode=utaf20#.VHzl_4em1-Y

Arismendi, I., S.L. Johnson, J.B. Dunham, R.Haggerty, and D. Hockman-Wert. 2012. The paradox of cooling streams in a warming world: Regional climate trends do not parallel variable local trends in stream temperature in the Pacific continental United States. Geophysical Research Letters 39, L10401, doi:10.1029/2012GL051448. [online] URL: http://ir.library.oregonstate.edu/xmlui/bitstream/handle/1957/36506/ArismendiIvanCEOASParadoxCoolingStreams.pdf?sequence=1

Arrigoni, A. S., G.C. Poole, L.A. Mertes, S.J. O'Daniel, W.W. Woessner, and S.A. Thomas. 2008. Buffered, lagged, or cooled? Disentangling hyporheic influences on temperature cycles in stream channels. *Water Resources Research*, *44*(9). DOI: 10.1029/2007WR006480

Barnas, K., & Katz, S. L. 2010. The challenges of tracking habitat restoration at various spatial scales. *Fisheries* 35(5):232-241. DOI: 10.1577/1548-8446-35.5.232 Online at: http://www.tandfonline.com/doi/abs/10.1577/1548-8446-35.5.232#.VH6jEYem1-Y

Baxter, C.V. and F.R. Hauer. 2000. Geomorphology, hyporheic exchange, and selection of spawning habitat by bull trout (Salvelinus confluentus). *Canadian Journal of Fisheries and Aquatic Sciences* 57: 1470-1481. Online at: http://deptweb.wwu.edu/huxley/huxweb/gis/EGEO452/04_projects/Subsurface%20Discharge/CJFA-hyporheicexchangeofspawninghabitatofbulltrout.pdf

Baxter, C.V., C.A. Frissell, and F.R. Hauer. 1999. Geomorphology, logging roads and the distribution of bull trout *(Salvelinus confluentus)* spawning in a forested river basin: implications for management and conservation. *Transactions of the American Fisheries Society,* 128:854-867. Online at: http://deptweb.wwu.edu/huxley/huxweb/gis/EGEO452/04_projects/Subsurface%20Discharge/CJFA-hyporheicexchangeofspawninghabitatofbulltrout.pdf

Bradford, M.J., and J. R Irvine. 2000. Land use, fishing, climate change, and the decline of Thompson River, British Columbia, coho salmon. C*anadian Journal of Fisheries and Aquatic Sciences*, 2000, 57(1): 13-16, 10.1139/f99-283.

Beschta, R.L., and 12 others. 1995. Cumulative effects of forest practices in Oregon. Report prepared for Oregon Department of Forestry, Salem, OR. 705 pp. http://ferm.forestry.oregonstate.edu/sites/default/files/pdf/CumulativeEffectsofForestPractices.pdf

Buhl, K.J. and S.J. Hamilton. 1998. Acute toxicity of fire-retardant and foam-suppressant chemicals to early life stages of Chinook salmon (*Oncorhynchus tshawytscha*). Environmental Toxicology and Chemistry 17(8):1589-1599. Online at: http://onlinelibrary.wiley.com/doi/10.1002/etc.5620170821/abstract?deniedAccessCustomisedMessage=&userIsAuthenticated=false

Carnefix, G. M.S. in Organismal Biology and Ecology, The University of Montana, 2002. Thesis title: *Movements and ecology of bull trout in Rock Creek, MT.*

Clancy, C., C. Frissell, and T. Weaver. 1996. Assessment of methods for removal or suppression of introduced fish to aid bull trout recovery. Report prepared by the Montana Bull Trout Scientific Group for the Montana Bull Trout Restoration Team. Montana Fish, Wildlife and Parks, Helena, MT.

DeHaan, P. W., & Bernall, S. R. 2013. Spawning Success of Bull Trout Transported above Main-Stem Clark Fork River Dams in Idaho and Montana. *North American Journal of Fisheries Management*, *33*(6):1269-1282.

Dunham, J. B., & Rieman, B. E. 1999. Metapopulation structure of bull trout: influences of physical, biotic, and geometrical landscape characteristics. *Ecological*

Applications, *9*(2):642-655. Online at: http://www.fs.fed.us/rm/boise/publications/fisheries/rmrs_1999_dunhamj001.pdf

Dunham, J., Rieman, B., & Davis, K. 2001. Sources and magnitude of sampling error in redd counts for bull trout. *North American Journal of Fisheries Management*, *21*(2):343-352. Online at: http://gis.fs.fed.us/rm/boise/publications/fisheries/rmrs_2001_dunhamj004.pdf

Dunham, J., Rieman, B., & Chandler, G. 2003. Influences of temperature and environmental variables on the distribution of bull trout within streams at the southern margin of its range. *North American Journal of Fisheries Management*,*23*(3):894-904. Online at: http://fresc.usgs.gov/products/papers/1406_Dunham.pdf

Ebersole, J.L., W.J. Liss, and C.A. Frissell. 1997. Restoration of stream habitats in the western United States: restoration as re-expression of habitat capacity. *Environmental Management* 21:1-14.

Ebersole, J.L., W.J. Liss, and C.A. Frissell. 2003. Cold water patches in warm streams: Physicochemical characteristics and the influence of shading. *Journal of the American Water Resources Association* 39:355-368. (Published online 8 June 2007).

FEMAT (Forest Ecosystem Management and Assessment Team). 1993. Forest Ecosystem Management: An Ecological, Economic and Social Assessment. USDA Forest Service, BLM, USFWS, NOAA, EPA and National Park Service, Portland, Oregon.

Firman, J. C., E.A. Steel, D.W. Jensen, K.M. Burnett, K. Christiansen, B.E. Feist, Blake E., D.P. Larsen, and K. Anlauf. 2011. Landscape Models of adult coho salmon density examined at four spatial extents. *Trans. Am. Fisheries So.c* 140:440-455. DOI: 10.1080/00028487.2011.567854.

Foster, D., F. Swanson, J. Aber, I. Burke, I., N. Brokaw, D. Tilman, and A. Knapp, 2003) The importance of land-use legacies to ecology and conservation. *BioScience* 53(1), 77-88. Online at: https://sustainability.asu.edu/docs/agtrans/FosterEtAl.pdf

Frissell, C.A. 1992. Cumulative effects of land use on salmon habitat in southwest Oregon coastal streams. Doctoral dissertation, Oregon State University, Corvallis, OR.

Frissell, C.A. 2014. Declaration Of Christopher A. Frissell, Ph. D. In Support Of The U.S. Environmental Protection Agency's And The National Oceanic And Atmospheric Administration's Proposal To Disapprove The State Of Oregon's Coastal Nonpoint Pollution Control Program For Failing To Adopt Additional Management Measures For Forestry. Prepared for Washington Forest Law Center, Seattle, WA, and Northwest Environmental Advocates, Portland, OR. Online at: http://northwestenvironmentaladvocates.org/blog/wp-content/uploads/2014/03/Declaration-of-Christopher-Frissell-3-14-14.pdf

Frissell, C.A., R.J. Baker, D.A. DellaSala, R.M. Hughes, J.R. Karr, D. A. McCullough, R.K. Nawa, J. Rhodes, M.C. Scurlock, and R.C. Wissmar. 2014. Conservation of Aquatic and Fishery Resources in the Pacific Northwest: Implications of New Science for the Aquatic Conservation Strategy of the Northwest Forest Plan. Report prepared for the Coast Range Association, Corvallis, OR. 35 pp. Available online at: http://coastrange.org

Furniss, M.J., B.P. Stabb, S. Hazelhurst, C.F. Clifton, K.B. Roby, B.L. Ilhadrt, E.B. Larry, A.H. Todd, L.M. Reid, S.J. Hines, K.A. Bennett, C.H. Luce, and P.J. Edwards. 2010. Water, climate change, and forests: watershed stewardship for a changing climate. USDA Forest Service General Technical Report PNW-GTR-812, Portland, Oregon, 75pp. http://www.fs.fed.us/pnw/pubs/pnw_gtr812.pdf

Gaikowski, M.P., S.J. Hamilton, K.J. Buhl, S.F. McDonald, and C.H Summers. 1996. Acute toxicity of three fire retardant and two fire suppressant foam formulations to the early life stages of rainbow trout (*Oncorhynchus mykiss*). *Environmental Toxicology and Chemistry* 15(8):1365-1374. Online at: http://onlinelibrary.wiley.com/doi/10.1002/etc.5620150816/abstract;jsessionid=09F3CC7C26D8265602FDF60BCB1D7998.f02t01

Gomi, T., R. C. Sidle, and J. S. Richardson. 2002. Understanding processes and downstream linkages of headwater streams. *BioScience* 52:905-916.

Gomi, T., R. D. Moore, and M. A. Hassan. 2005. Suspended sediment dynamics in small forest streams of the Pacific Northwest. *Journal of the American Water Resources Association* 41(4):877-898. Online at: http://www.wou.edu/las/physci/taylor/andrews_forest/refs/gomi_etal_2005.pdf

Good, T. P., Beechie, T. J., McElhany, P., McClure, M. M., & Ruckelshaus, M. H. (2007). Recovery planning for Endangered Species Act-listed Pacific salmon: using science to inform goals and strategies. *Fisheries* 32(9):426-440. Online at: http://www.webpages.uidaho.edu/fish510/PDF/Good%20et%20al%202007%20Fisheries.pdf

Groom, J.D., L. Dent, and L.J. Madsen. 2011a. Response of Western Oregon Stream Temperatures to Contemporary Forest Management. Forest Ecology and Management, 262.8 (2011): 1618-1629. Online at: http://ir.library.oregonstate.edu/xmlui/bitstream/handle/1957/23358/groomjeremy.ferm.responsewesternoregon.pdf?sequence=1

Groom, J.D., L. Dent, and L.J. Madsen. 2011b. Stream temperature change detection for state and private forests in the Oregon Coast Range. Water Resources Research 47.1 Accessed November 27, 2013. doi:10.1029/2009WR009061 Online at: http://onlinelibrary.wiley.com/doi/10.1029/2009WR009061/full

Gucinski, H., M. J. Furniss, R. R. Ziemer, and M. H. Brookes. 2001. Forest roads: a synthesis of scientific information. Gen. Tech. Rep. PNWGTR-509. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR. http://www.fs.fed.us/pnw/pubs/gtr509.pdf

Hicks, B. J., R. L. Beschta. and R. D. Harr. 1991b. Longterm changes in streamflow following logging in western Oregon and associated fisheries implications. *Water Resources Bulletin* 27:217-226.

High, B., Meyer, K. A., Schill, D. J., & Mamer, E. R. 2008. Distribution, abundance, and population trends of bull trout in Idaho. *North American Journal of Fisheries Management* 28(6): 1687-1701.

IMST. 1999. Recovery of Wild Salmonids in Western Oregon Forests: Oregon Forest Practices Act Rules and the Measures in the Oregon Plan for Salmon and Watersheds. Independent Multidisciplinary Science Team (IMST). Technical Report 1999-1 to the Oregon Plan for Salmon and Watersheds, Governor's Natural Resources Office, Salem, Oregon, 1999. Online at: http://www.fsl.orst.edu/imst/reports/1999-1.pdf

Jones, J.A., F. J. Swanson, B.C. Wemple, and K.U. Snyder. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. *Conservation Biology* 14:76-85.

Jones, L. A., Muhlfeld, C. C., Marshall, L. A., McGlynn, B. L., & Kershner, J. L. 2013. Estimating thermal regimes of bull trout and assessing the potential effects of climate warming on critical habitats. *River Research and Applications 30*(2):204-216.

Kauffman, J. B., Beschta, R. L., Otting, N., & Lytjen, D. 1997. An ecological perspective of riparian and stream restoration in the western United States. *Fisheries*, *22*(5), 12-24.

Kershner, J. L., & B. B. Roper. 2010. An evaluation of management objectives used to assess stream habitat conditions on Federal lands within the Interior Columbia Basin. *Fisheries* 35(6):269-278. DOI: 10.1577/1548-8446-35.6.269 Onlien at: http://www.tandfonline.com/doi/abs/10.1577/1548-8446-35.6.269#.VHzqxIem1-Y

Koski, M.L., J.M. Hudson, B.P. Silver, and T.A. Whitesel. 2012. Clackamas River Bull Trout Reintroduction. FY2010-11 Progress Report. U.S. Fish and Wildlife Service, Columbia River Fisheries Program Office, Vancouver, WA. 47 pp.

Kovach, R., C. Muhlfeld, A. Wade, B. Hand, D. Whited, P. DeHaan, R. Al Chokhacy, and G. Luikart. In Review. Genetic diversity is related to climatic and habitat variation in bull trout: implications for climatic vulnerability. [Because MS is being redrafted pursuant to review, please obtain current copy from authors.]

Lee, D. C., J. R. Sedell, B. E. Rieman, R. F. Thurow, and J. E. Williams. 1997. Broadscale assessment of aquatic species and habitats. Pages 1059-1496 in T. M.

Quigley, and S. Arbelbide, editors. An assessment of ecosystem components in the Interior Columbia Basin. Volume III. General technical report PNW-GTR-405. U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Luce, C.H. and Z.A. Holden. 2009. Declining annual streamflow distributions in the Pacific Northwest United States, 1948–2006. *Geophysical Research Letters* 36, L16401, doi:10.1029/2009GL039407, 2009.

McCarthy S.G., Incardona J.P., and Scholz N.L. 2008. Coastal storms, toxic runoff, and the sustainable conservation of fish and fisheries. American Fisheries Society Symposium 64: 1-21.

McDonald, L. L., Bilby, R., Bisson, P. A., Coutant, C. C., and 12 others. 2007. Research, monitoring, and evaluation of fish and wildlife restoration projects in the Columbia River Basin: lessons learned and suggestions for large-scale monitoring programs. *Fisheries 32*(12):582-590. Online at: http://ir.library.oregonstate.edu/xmlui/bitstream/handle/1957/25471/ResearchMonitoringAndEvaluationOfFish.pdf?sequence=1

McElhany P., M. H. Ruckelshaus, J. Ford, T. Wainwright, and E. Bjorkstedt. 2000. Viable salmonid populations and the recovery of evolutionarily significant units. U.S. Department of Commerce, NOAA Tech. Memo NMFS-NWFSC-42. Online at: http://www.westcoast.fisheries.noaa.gov/publications/protected_species/salmon_steelhead/vsp_2000.pdf

McHugh, P.A., R.M. Thompson, H. S. Greig, H.J. Warburton, and A.R McIntosh. Habitat size influences food web structure in drying streams. *Ecography* 11/2014: DOI:10.1111/ecog/01193. Online at: https://www.researchgate.net/publication/268528946_Habitat_size_influences_food_web_structure_in_drying_streams

McMahon, T. E., and D. H. Bennett. 1996. Walleye and Northern Pike: boost or bane to Northwest fisheries? Fisheries 21(8):6–13. Online at: http://www.montana.edu/mcmahon/SppIntro-walleye.pdf

Martinuzzi, S., S. R. Januchowski-Hartley, B. M. Pracheil, P. B. McIntyre, A. J. Plantinga, D. J., Lewis, & V. C. Radeloff, 2014. Threats and opportunities for freshwater conservation under future land use change scenarios in the United States. *Global Change Biology 20*(1):13-124. Online at: http://limnology.wisc.edu/personnel/mcintyre/publications/Martinuzzi%20etal%202013%20GCB%20policy%20scenarios%20of%20land%20use%20change%20affect%20aquatic%20BD.pdf

May, C.L. 2002. Debris flows through different forest age classes in the Central Oregon Coast Range. *Journal of the American Water Resources Association* 38(4): 1097-1113. Online at: http://wwwtest.fs.fed.us/psw/publications/may/May_JAWRA.pdf

May, C. L., and R. E. Gresswell. 2003. Large wood recruitment and redistribution in headwater streams in the southern Oregon Coast Range, U.S.A. *Can. J. For. Res.* 33: 1352–1362. Online at: http://www.southernregion.fs.fed.us/psw/publications/may/May_Gress_CJFR.pdf

Moore, R.D. and S.M. Wondzell. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: A review. *Journal of the American Water Resources Association* 41(4):763-784. DOI: 10.1111/j.1752-1688.2005.tb03770.x

Murphy, D. D., and P. S Weiland. 2014. Science and structured decision making: fulfilling the promise of adaptive management for imperiled species. *Journal of Environmental Studies and Sciences*:1-8. DOI: 10.10107/as13412-014-0165-0 Online at: http://www.calwatercenter.org/images/Science and structured decision making_-_fulfilling_the_promise_of_adaptive_management_for_imperiled_species.pdf

Nieber, J., 2011. Evaluation of Buffer Width on Hydrologic Function, Water Quality, and Ecological Integrity of Wetlands. Final Report 2011-06, Minnesota Department of Transportation, Office of Policy, Analysis, and Innovation. St. Paul, MN. 182 pp.

Pollock, M.M. and T.J. Beechie. 2014. Does riparian forest thinning enhance biodiversity? The ecological importance of large wood. *Journal of the American Water Resources Association* 50(3):543-559. DOI: 10.1111/jawr.12206

Poole, G.C. and C.H. Berman. 2001. An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. *Environmental Management* 27(6):787–802.

Poole, G.C., S.J. O'Daniel, K.L. Jones, W.W. Woessner, E.S. Bernhardt, A.M. Helton, J.A. Stanford, B.R. Boer, and T.J. Beechie. 2008. Hydrologic spiralling: the role of multiple interactive flow paths in stream ecosystems. *River Research and Applications* 24(7):1018-1031.

Rashin, E.B., C.J. Clishe, A.T. Loch, and J.M. Bell. 2006. Effectiveness of timber harvest practices for controlling sediment. *Journal of the American Water Resources Association* 42:1307-1347.

Reeves, G.H., J.E. Williams, K.Gallo, and K.M.Burnett. 2006. The aquatic conservation strategy of the Northwest Forest Plan. *Conservation Biology* 20: 319-329. Online at: http://andrewsforest.oregonstate.edu/pubs/pdf/pub4036.pdf

Rieman, B. E., & Allendorf, F. W. 2001. Effective population size and genetic conservation criteria for bull trout. *North American Journal of Fisheries*

Management, *21*(4):756-764. Online at:
http://www.flatheadtu.org/indexFiles/WebDocs/BT10.pdf

Rieman, B. E., & Myers, D. L. 1997. Use of Redd Counts to Detect Trends in Bull Trout (Salvelinus confluentus) Populations. *Conservation Biology*, *11*(4):1015-1018. Online at:
http://gis.fs.fed.us/rm/boise/publications/fisheries/rmrs_1997_riemanb002.pdf

Rieman, B. E. and J. D. McIntyre. 1993. Demographic and Habitat Requirements for Conservation of Bull Trout. Gen. Tech. Rep. INT-302. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Boise, ID. 38p

Rieman, B. E., & McIntyre, J. D. 1996. Spatial and temporal variability in bull trout redd counts. *North American Journal of Fisheries Management*, *16*(1):132-141. Online at:
http://gis.fs.fed.us/rm/boise/publications/fisheries/rmrs_2001_dunhamj004.pdf

Rieman, B. E., Lee, D. C., & Thurow, R. F. 1997. Distribution, status, and likely future trends of bull trout within the Columbia River and Klamath River basins. *North American Journal of Fisheries Management*, *17*(4):1111-1125.

Rieman, B. E., Lee, D., Chandler, G., & Myers, D. 1997. Does wildfire threaten extinction for salmonids: responses of redband trout and bull trout following recent large fires on the Boise National Forest. In *Proceedings of the symposium on fire effects on threatened and endangered species and their habitats. International Association of Wildland Fire, Fairfield, Virginia* (pp. 47-57). Online at:
http://gis.fs.fed.us/rm/boise/publications/fisheries/rmrs_1997_riemanb004.pdf

Rieman, B., Peterson, J. T., Clayton, J., Howell, P., Thurow, R., Thompson, W., & Lee, D. (2001). Evaluation of potential effects of federal land management alternatives on trends of salmonids and their habitats in the interior Columbia River basin. *Forest ecology and management*, *153*(1):3-62. Online et:
http://svinet2.fs.fed.us/psw/publications/lee/lee2.pdf

Rieman, B. E., Isaak, D., Adams, S., Horan, D., Nagel, D., Luce, C., & Myers, D. (2007). Anticipated climate warming effects on bull trout habitats and populations across the interior Columbia River basin. *Transactions of the American Fisheries Society*, *136*(6):552-1565. Online at:
http://www.fs.fed.us/rm/boise/publications/fisheries/rmrs_2007_riemanb001.pdf

Rhodes, J.J., D.A. McCullough, and F. A. Espinosa. 1994. A coarse screening process for evaluation of the effects of land management activities on salmon spawning and rearing habitat in ESA consultations. Columbia River Inter-Tribal Fish Commission, Technical Report 94-4. Portland, OR. 245pp.

Rich Jr, C. F., McMahon, T. E., Rieman, B. E., & Thompson, W. L. 2003. Local-habitat, watershed, and biotic features associated with bull trout occurrence in Montana streams. *Transactions of the American Fisheries Society*, *132*(6):1053-1064.

Ruckelshaus, M. H., Levin, P., Johnson, J. B., & Kareiva, P. M. 2002. The Pacific salmon wars: what science brings to the challenge of recovering species. *Annual Review of Ecology and Systematics* 665-706. Online at: http://www.olemiss.edu/courses/bisc677/Ruckelshaus2002.pdf

Seavy, N. E., T. Gardali, G. H. Golet, F. T. Griggs, C. A. Howell, R. Kelsey, S. L. Small, J. H. Viers, J. F. Weigand. 2009. Why climate change makes riparian restoration more important than ever: recommendations for practice and research. *Ecological Restoration* 27(3): 330-338. URL: http://er.uwpress.org/content/27/3/330.full.pdf+html

Sharma, R. and R. Hilborn. 2001. Empirical Relationships between Watershed Characteristics and Coho Salmon (*Oncorhynchus kisutch*) Smolt Abundance in 14 Western Washington Streams. *Can J. Fish Aquat Sci.* 58:1453-1463.

Spruell, P., Rieman, B. E., Knudsen, K. L., Utter, F. M., & Allendorf, F. W. 1999. Genetic population structure within streams: microsatellite analysis of bull trout populations. *Ecology of Freshwater Fish*, *8*(3):114-121.

Sweeney, B. W., & Newbold, J. D. (2014). Streamside Forest Buffer Width Needed to Protect Stream Water Quality, Habitat, and Organisms: A Literature Review. *JAWRA Journal of the American Water Resources Association 50*(3):560-584. Online at: http://www.stroudcenter.org/about/pdfs/Sweeney2014-JAWRA-StreamsideBufferWidth_LiteratureReview.pdf

Tague, C., & Grant, G. E. 2009. Groundwater dynamics mediate low-flow response to global warming in snow dominated alpine regions. *Water Resources Research 45*(7). DOI: 10.1029/2008WR007179

Torgersen, C.E., Price, D.M., Li, H.W., and McIntosh, B.A. 1999. Multiscale thermal refugia and stream habitat associations of chinook salmon in northeastern Oregon. *Ecological Applications*, 9(1):301-319.

Torgersen, C.E., J.L. Ebersole, and D.M. Keenan. 2012. Primer for Identifying Cold-Water Refuges to Protect and Restore Thermal Diversity in Riverine Landscapes. EPA 910-C-12-001, U.S. Environmental Protection Agency, Seattle, Washington. p. 91. Online at: http://www.epa.gov/region10/pdf/water/torgersen_etal_2012_cold_water_refuges.pdf

Trombulak, S.C., and C.A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology* 14:18-30.

USFWS. 2014a. Final Biological Opinion on the Effects to Bull Trout and Bull Trout

Critical Habitat From the Implementation of Proposed Actions Associated with the Plan of Operations for the Montanore Minerals Corporation Copper/Silver Mine. 238 p. with appendices and attachments.

USFWS. 2014b. Revised Draft Recovery Plan for the Coterminous United States Population of Bull Trout (*Salvelinus confluentus*). 151 p. Available online at: http://www.fws.gov/pacific/bulltrout/pdf/Revised%20Draft%20Bull%20Trout%20Recovery%20Plan.pdf

USFWS. 2006. Biological Opinion on the Effects to Grizzly Bears, Bull Trout, and Bull Trout Critical Habitat from the Implementation of Proposed Actions Associated with Plan of Operation for the Revett RC Resources Incorporated Rock Creek Copper/Silver Mine as proposed by the U.S. Forest Service, Kootenai National Forest. U.S. Fish and Wildlife Service Montana Ecological Services Field Office, Helena. MT. Online at: http://www.revettminerals.com/sites/default/files/rock_creek_bo_final_document_10-11.pdf

WDNR (Washington Department of Natural Resources). 2014. Active cooperative monitoring, evaluation and research committee projects. Web site, accessed 27 Nov. 2014, online at: http://www.dnr.wa.gov/BusinessPermits/Topics/FPAdaptiveManagementProgram/Pages/fp_cmer_active_projects.aspx

Whiteley, A. R., Spruell, P., Rieman, B. E., & Allendorf, F. W. 2006. Fine-scale genetic structure of bull trout at the southern limit of their distribution. *Transactions of the American Fisheries Society*, *135*(5), 1238-1253. Online at: http://www.fs.fed.us/rm/pubs_other/rmrs_2006_whiteley_a001.pdf

Curriculum Vitae *19 August 2014* *18pp.*

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Education: Ph.D. in Fisheries Science, Oregon State University, 1992
M.S. in Fisheries Science, Oregon State University, 1986
B.A. with High Honors in Zoology, University of Montana, 1982

Professional Appointments:

Affiliate Research Professor, Flathead Lake Biological Station, and Systems Ecology Program (*pending*), The University of Montana, 2014-present
Self Employed, Consulting Ecologist 2012-present
Director of Science and Conservation and Senior Staff Scientist, The Pacific Rivers Council, 2000-2012
Research Associate Professor, The University of Montana, Flathead Lake Biological Station, 1998-2000
Research Assistant Professor, The University of Montana, Flathead Lake Biological Station, 1993-1998
Research Assistant Professor, Department of Fisheries and Wildlife, Oregon State University, 1994-1997
Postdoctoral Research Associate (Faculty), Department of Fisheries and Wildlife, Oregon State University, 1992-1994
Research Assistant (Faculty), Oak Creek Laboratory of Biology, Department of Fisheries and Wildlife, Oregon State University, 1985-1992

Fields of Interest:

- Land-aquatic ecosystem linkages
- Ecology, biogeography, and conservation biology of fishes and freshwater biota in relation to landscape and hydrologic change.
- Cumulative impacts of natural processes and human activities on stream habitat and stream biota.
- Aquatic ecosystem conservation and restoration strategies.
- Geomorphic, hydrophysical, and landscape ecology considerations in design of integrated conservation reserves.
- Restoration and recovery planning and design for freshwater ecosystems and species.

Theses and Dissertations:

Frissell, C.A. 1992. Cumulative effects of land use on salmon habitat in southwest Oregon coastal streams. Doctoral dissertation, Oregon State University, Corvallis.

Frissell, C. A. 1986. A hierarchical stream habitat classification system: development and demonstration. M.S. thesis, Oregon State University, Corvallis.

Frissell. C. A. 1982. Colonization and development of community structure in coexisting Ephemerellid mayflies (Ephemeroptera, Ephemerellidae). Senior Thesis, Watkins Scholarship Program, The University of Montana, Missoula.

Professional Societies:

Society for Conservation Biology, 1991-present
American Fisheries Society, 1985-present
Ecological Society of America, intermittent
North American Benthological Society, intermittent

Graduate Students Mentored

Cavallo, B.J. M.S. in Organismal Biology and Ecology, The University of Montana, 1997. Thesis title: *Floodplain habitat heterogeneity and the distribution, abundance, and behavior of fishes and amphibians in the Middle Fork Flathead River Basin, Montana.*

Adams, S. B. Ph.D.in Organismal Biology and Ecology, The University of Montana, 1999. Dissertation title: *Mechanisms Limiting a Vertebrate Invasion: Brook Trout in Mountain Streams of the Northwestern USA*.

Hitt, N.P. M.S. in Organismal Biology and Ecology, The University of Montana, 2002, *Distribution and potential invasion of introduced rainbow trout in the upper Flathead River drainage.*

Carnefix, G. M.S. in Organismal Biology and Ecology, The University of Montana, 2002. Thesis title: *Movements and ecology of bull trout in Rock Creek, MT.*

Hastings, K. Ph.D.in Organismal Biology and Ecology, The University of Montana, 2005. Dissertation title: *Long-term persistence of isolated fish populations in the Alexander Archipelago.*

Reviewer for Journals and Agency Publications:

Canadian Journal of Fisheries and Aquatic Sciences, Conservation Biology, Ecological Applications, Environmental Management, Fisheries (AFS), Freshwater Biology, North American Journal of Fisheries Management, Oikos, Transactions of the American Fisheries Society, Fundamental and Applied Limnology, USDA Forest Service General Technical Reports

Member of Board of Editors for Journals:

Conservation Biology, 1996-2000

Appointments to Review Panels and Scientific Advisory Committees:

USEPA Bristol Bay Watershed Assessment Team, Subcontractor on road and pipeline impacts, through University of Alaska Anchorage and NatureServe, 2011-2012.

Independent Expert Review Panel for King County Water and Land Resources Division's Project Scoping and Implementation Practices. 2011-12. Subcontractor to MWH (Montgomery Watson Harza) for King County Dept. of Natural Resources and Parks, Seattle, WA. http://www.kingcounty.gov/environment/dnrp/publications/wlrd-expert-review-report.aspx

Umpqua Watersheds Science Advisory Council, Sponsored by Umpqua Watersheds, Inc., 16-17 November 2010, Roseburg, OR.

Wychus Creek Restoration Monitoring Plan Review Panel, sponsored by Upper Deschutes Watershed Council and Bonneville Environmental Foundation. 2 October 2009, Bend, OR.

Landscape Pattern Task Group, *State of the Nation's Ecosystems* report. 2003-2007. H. John Heinz III Center For Science, Economics and the Environment. Washington, DC. http://www.heinzctr.org/Programs/Reporting/Working%20Groups/Fragmentation/index.shtml

Science Review Team, King County Normative Flow Studies Project. 2002-2005, Seattle, WA. http://dnr.metrokc.gov/wlr/BASINS/flows/science-review-team.htm

Science Advisory Panel, Westside. Governor's Salmon Restoration Funding Board, Washington State, February 2000.

Ecological Work Group, Multi-species Framework Process and Subbasin Assessment Process, Northwest Power Planning Council 1998-2000.

Peer review panelist for U.S. Environmental Protection Agency/National Science Foundation Water and Watersheds Grants Program for 1997. 7-9 May 1997.

Scientific Group for the Governor's Bull Trout Restoration Team, State of Montana, 1994-2000

Oregon Department of Environmental Quality, 1992-95: Temperature Standards Review Subcommittee of the Technical Advisory Committee, Triennial Water Quality Standards Review

Scientific Assessment Panel for amphibian species, Eastside Oregon-Washington and Upper Columbia Basin EIS, US BLM and US Forest Service, 1994

Oregon Department of Forestry, 1990-93: Technical Advisory Group for the Forest Practices Monitoring Program; Wetlands Technical Group; Stream Protection Advisory Panel.

Peer-Reviewed Articles Published in Scientific Journals:

DellaSala, D. A., R.G. Anthony, M.L. Bond, Monica, E.S. Fernandez, C.A. Frissell, Chris, C.T. Hanson, and R. Spivak. 2014. Alternative Views of a Restoration Framework for Federal Forests in the Pacific Northwest. *Journal of Forestry* 111(6):420-429.

Williams, J. E., R. N. Williams, R. F. Thurow, L. Elwell, D. P. Philipp, F. A. Harris, J. L. Kershner, P. J. Martinez, D. Miller, G. H. Reeves, C. A. Frissell, and J. R. Sedell. 2011. Native Fish Conservation Areas: a vision for large-scale conservation of native fish communities. *Fisheries* 36:267-277. http://www.tu.org/sites/www.tu.org/files/documents/Williams%20et%20al.%202011%20Fisheries%20NFCA.pdf

Whiteley, A.R., K. Hastings, J. K. Wenburg, C. A. Frissell, J. C. Martin and F. W. Allendorf. 2010. Genetic variation and effective population size in isolated populations of coastal cutthroat trout. *Conservation Genetics* 11(5):1929-1943. DOI: 10.1007/s10592-010-0083-y

Olson, D.H., P.D. Anderson, C.A. Frissell, H.H. Welsh, Jr., and D.F. Bradford. 2007. Biodiversity management approaches for stream-riparian areas: perspectives for Pacific Northwest headwater forests, microclimates, and amphibians. *Forest Ecology and Management* 246(1):81-107. **[Forest Ecology and Management "Highly Cited Author" award for 2007-2010]*

Poole, G.C., J.A. Stanford, S.W. Running, and C.A. Frissell. 2006. Multiscale geomorphic drivers of groundwater flow paths: subsurface hydrologic dynamics and hyporheic habitat diversity. *Journal of the North American Benthological Society* 25(2): 288-303.

Poole, G. C., J. A. Stanford, S. W. Running, C. A. Frissell, W. W. Woessner, and B. K. Ellis. 2004. A patch hierarchy approach to modeling surface and sub-surface hydrology in complex flood-plain environments. *Earth Surface Processes and Landforms* 29**:** 1259–1284.

Karr, J. R., J. J. Rhodes, G. W. Minshall, F. R. Hauer, R. L. Beschta, C. A. Frissell, and D. A. Perry. 2004. The effects of postfire salvage logging on aquatic ecosystems in the American West. *BioScience* 54:1029-1033. http://www.sierraforestlegacy.org/Resources/Conservation/FireForestEcology/SalvageLoggingScience/Salvage-Karr04.pdf

Hitt, N.P., and C.A. Frissell. 2004. A case study of surrogate species in aquatic conservation planning. *Aquatic Conservation: Marine and Freshwater Ecosystems*. 14:625–633.

Beschta, ,R.L., J. J. Rhodes, J.B. Kauffman, R.E. Gresswell, G.W. Minshall, J. R. Karr, D.A. Perry, F.R. Hauer, C. A. Frissell. 2004. Postfire Management on Forested Public Lands of the Western United States. *Conservation Biology* 18: 957–967. http://www.researchgate.net/publication/227654964_Postfire_Management_on_Forested_Public_Lands_of_the_Western_United_States?ev=prf_pub

Articles Published in Scientific Journals, continued:

Hitt, N.C., C. Muhlfeld, C.A. Frissell, and F. Allendorf. 2003. Hybridization between native westslope cutthroat trout and non-native rainbow trout. *Canadian Journal of Fisheries and Aquatic Sciences* 60:1440-1451.

Ebersole, J. L., W.J. Liss, and C.A. Frissell. 2003. Thermal heterogeneity, stream channel morphology, and salmonid abundance in northeastern Oregon streams. *Canadian Journal of Fisheries and Aquatic Sciences* 60(10):1266-1280.

Ebersole, J. L., W.J. Liss, and C.A. Frissell. 2003. Cold water patches in warm streams: Physicochemical characteristics and the influence of shading. *Journal of the American Water Resources Association* 39:355-368.

Poole, G.C., J. A. Stanford, C.A. Frissell and S.W. Running. 2002. *Three-dimensional mapping of geomorphic controls on flood-plain hydrology and connectivity from aerial photos*. ***Geomorphology*** 48(4):329-347.

Adams, S.B., and C.A. Frissell. 2002. Changes in distribution of nonnative brook trout in an Idaho drainage over two decades. *Transactions of the American Fisheries Society*, 131:561-568.

Adams, S.B., and C.A. Frissell. 2001. Thermal habitat use and evidence of seasonal migration by tailed frogs, *Ascaphus truei*, in Montana. *Canadian Field-Naturalist* 115: 251-256.

Adams, S.B., C.A. Frissell, and B.E. Rieman. 2001. Geography of invasion in mountain streams: consequences of headwater lake fish introductions. *Ecosystems* 296-307.

Ebersole, J.L., W.J. Liss, and C. A. Frissell. 2001. Relationship between stream temperature, thermal refugia, and rainbow trout *Oncorhynchus mykiss* abundance in arid-land streams in the northwestern United States. *Ecology of Freshwater Fish* 10:1-10.

Adams, S.A., C.A. Frissell, and B.E. Rieman. 2000. Movements of non-native brook trout in relation to stream channel slope. *Transactions of the American Fisheries Society* 129:623-638

Trombulak, S.C., and C.A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology* 14:18-30.

Baxter, C.V., C.A. Frissell, and F.R. Hauer. 1999. Geomorphology, logging roads and the distribution of bull trout *(Salvelinus confluentus)* spawning in a forested river basin: implications for management and conservation. *Transactions of the American Fisheries Society,* 128:854-867.

Williams, R.N., P.A. Bisson, D.L. Bottom, L.D. Calvin, C.C. Coutante, M.W. Erho Jr., C.A. Frissell, J.A. Lichatowich, W.J. Liss, W.E. McConnaha, P.R. Mundy, J.A. Stanford & R.R. Whitney. 1999. Return to the River: Scientific Issues in the Restoration of Salmonid Fishes in the Columbia River. *Fisheries* (Bethesda) 24(3):10-19.

Articles Published in Scientific Journals, continued:

Currens, K.P., F.W. Allendorf, D. Bayles, D.L. Bottom,. C.A. Frissell, D. Hankin, J.A. Lichatowich, P.C. Trotter, and T.A. Williams. 1998. Conservation of Pacific salmon: response to Wainwright and Waples. *Conservation Biology* 12:1148-1149.

Poole, G.C., C.A. Frissell, and S.C. Ralph. 1997. In-stream habitat unit classification: inadequacies for monitoring and some consequences for management. *Journal of the American Water Resources Association* 33:879-896.

Ebersole, J.L., W.J. Liss, and C.A. Frissell. 1997. Restoration of stream habitats in the western United States: restoration as re-expression of habitat capacity. *Environmental Management* 21:1-14.

Allendorf, F.W., D. Bayles, D.L. Bottom, K.P. Currens, C.A. Frissell, D. Hankin, J.A. Lichatowich, W. Nehlsen, P.C. Trotter, and T.H. Williams. 1997. Prioritizing Pacific salmon stocks for conservation. *Conservation Biology* 11:140-152.

Frissell, C.A., and D. Bayles. 1996. Ecosystem management and the conservation of aquatic biodiversity and ecological integrity. *Water Resources Bulletin* 32:229-240.

Stanford, J.A., J.V. Ward, W.J. Liss, C.A. Frissell, R.N. Williams, J.A. Lichatowich, and C.C. Coutant. 1996. A general protocol for restoration of regulated rivers. *Regulated Rivers: Research and Management* 12:391-413. http://tinyurl.com/c4wbcwo

Nawa, R., and C.A. Frissell. 1994. Measuring scour and fill of gravel streambeds with scour chains and sliding bead monitors. *North American Journal of Fisheries Management* 13:634-639.

Frissell, C.A. 1993. Topology of extinction and endangerment of native fishes in the Pacific Northwest and California, USA. *Conservation Biology* 7:342-354.

Frissell, C.A., R.K. Nawa, and R. Noss. 1992. Is there any conservation biology in "New Perspectives?" A response to Salwasser. *Conservation Biology* 6:461-464.

Frissell, C.A., and R.K. Nawa. 1992. Incidence and causes of failure of artificial habitat structures in streams of western Oregon and Washington. *North American Journal of Fisheries Management* 12:182-197.

Frissell, C.A., W.J. Liss, C.E. Warren, and M.D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. *Environmental Management* 10:199-214. *

**[Recognized as among the ten most cited papers in benthic ecology in Resh, V.H. 2003. J. of the North American Benthological Society 22 (3): 341-35.]*

Symposium Articles Published:

Frissell, C., and G. Carnefix. 2007. The geography of freshwater habitat conservation: roadless areas and critical watersheds for native trout. Pp. 210-217 in R. F. Carline, and C. LoSapio, (eds*.) Sustaining Wild Trout in a Changing World: Proceedings of Wild Trout IX Symposium*, *October 9-12, 2007, West Yellowstone, Montana.* 308pp. http://www.wildtroutsymposium.com/proceedings-9.pdf

Poole, G.C., J.A. Stanford, S.W. Running, and C.A. Frissell. 2000. A Linked GIS/modeling approach to assessing the influence of flood-plain structure on surface- and ground-water routing in rivers. *Proceedings of the 4th International Conference on Integrating Geographic Information Systems (GIS) and Environmental Modeling*. Held 2-8 September 2000, Banff, Alberta. B. Parks, editor.

Stahl, R.G., J. Mille, R. Frederick, D. Courtemanch, C. Frissell, M. Kaplan, M., K. Sappington, and M. Zeeman, 1999. Managing Ecological Risks Posed by Multiple Stressors. Pages 51-66 in Foran, J.A., and S. A. Forenc (eds.) *Multiple Stressors in Ecological Risk and Impact Assessment: Proceedings from the Pellston Workshop on Multiple Stressors in Ecological Risk and Impact Assessment.* 13-18 September 1997, Pellston, Michigan. *SETAC Special Publications Series,* SETAC Press, The University of Michigan. 100pp.

Clancy, C., C. Frissell, and T. Weaver. 1998. Removal or suppression of introduced fish to aid bull trout recovery. *Proceedings of the Wild Trout XI Conference*, held August, 1997 in Bozeman, MT. http://www.wildtroutsymposium.com/proceedings-6.pdf

Li, H.W., K. Currens, D. Bottom, S. Clarke, J. Dambacher, C. Frissell, P. Harris, R.M. Hughes, D. McCullough, A. McGie, K. Moore, R. Nawa, and S. Thiele. 1995. Safe havens: refuges and evolutionarily significant units. *American Fisheries Society Symposium* 17:371-380.

Frissell, C.A., W.J. Liss, and D. Bayles. 1993. An integrated, biophysical strategy for ecological restoration of large watersheds. In D.F. Potts ed., *Changing Roles in Water Resources Management and Policy*. Proceedings of a symposium of the American Water Resources Association, held 27-30 June, 1993, Bellevue, WA.

Frissell, C.A., and R.K. Nawa. 1989. Cumulative impacts of timber harvest on fisheries: "All the King's horses and all the King's men..." In C. Toole, (ed.), *Proceedings of the Seventh California Salmon, Steelhead and Trout Restoration Conference*. February 24-26, Arcata, CA. California Sea Grant Publication UCSGEP-89-02.

Frissell, C.A., and T. Hirai. 1988. Life history patterns, habitat change, and productivity of fall chinook stocks of southwest Oregon. In B. Sheperd (ed.) *Proceedings of the Northeast Pacific Chinook and Coho Workshop*, Bellingham, Washington, 3-4 October 1988. North Pacific International Chapter, American Fisheries Society.

Books and Book Chapters Published:

Frissell, C.A., and C.W. Bean. 2009. Responding to environmental threats. In: Assessing The Conservation Value Of Fresh Waters (Boon, P.J. & Pringle, C. eds.) pp. 91-116. Cambridge University Press Books, Cambridge, UK. 293pp.

Langford T.E.L., & Frissell C.A. 2009. Evaluating restoration potential. Pp. 117-141 in P.J. Boon & C.M. Pringle (eds.) *Assessing the Conservation Value of Freshwaters. An International Perspective*. Cambridge University Press,Cambridge, UK. 293pp.

Stanford, J. A., C. A. Frissell and C. C. Coutant. 2006. Chapter 5: The Status of Freshwater Habitats. Pp. 173-248 in Williams, R. N. (ed.), Return to the River: Restoring Salmon to the Columbia River. Elsevier Academic Press, Amsterdam. 720 pp. http://www.sciencedirect.com/science/book/9780120884148

Frissell, C.A., N.L. Poff, and M.E. Jensen. 2001. Assessment of biotic patterns in freshwater ecosystems. Chapter 27 in Bourgeron, P., M. Jensen, and G. Lessard (eds.) *A Guidebook for Integrated Ecological Assessments*. Springer-Verlag, NY.

Jensen, M.E., I. Goodman, and C.A. Frissell. 2001. Design and use of aquatic biophysical classifications and maps. Chapter 26 in Bourgeron, P., M. Jensen, and G. Lessard (eds). *A Guidebook for Integrated Ecological Assessments.* Springer-Verlag, NY.

Welsh, H.H., T.D. Roelofs, and C.A. Frissell. 2000. Aquatic ecosystems of the redwood region. Pages 165-199 in R. Noss (ed.) *The Redwood Forest: History, Ecology, and Conservation of the Coast Redwoods.* Island Press, Washington, DC.

Frissell, C.A., and S.C. Ralph. 1998. Stream and watershed restoration. Pages 599-624 in R.J. Naiman and R.E. Bilby (eds.) *Ecology and Management of Streams and Rivers in the Pacific Northwest Coastal Ecoregion*. Springer-Verlag, NY.

Frissell, C.A. 1997. Ecological principles. Pages 96-115 in J.E. Williams, M.P. Dombeck, and C.A. Wood (eds.) *Watershed Restoration: Principles and Practices.* The American Fisheries Society, Bethesda, MD.

Frissell, C.A., W.J. Liss, R.K. Nawa, R.E. Gresswell, and J.L. Ebersole. 1997. Measuring the failure of salmon management. Pages 411-444 in D.J. Stouder, P.A. Bisson,and R.J. Naiman (eds.) *Pacific Salmon and their Ecosystems: Status and Future Options*. Chapman and Hall, New York, NY.

Frissell, C.A. 1996. A new strategy for watershed protection, restoration and recovery of wild native fish in the Pacific Northwest. Pages 1-24 in B. Doppelt (ed.) *Healing the Watershed: A Guide to the Restoration of Watersheds and Native Fish in the West*. The Pacific Rivers Council, Eugene, OR.

Frissell, C.A., and D.G. Lonzarich. 1996. Habitat use and competition among stream fishes. Pages 493-510 in F.R. Hauer and G.A. Lamberti (eds.) *Methods in Stream Ecology*. Academic Press, San Diego, CA.

Books and Book Chapters Published, continued:

Doppelt, B., M. Scurlock, C. Frissell, and J. Karr. 1993. *Entering the Watershed: A New Aproach to Save America's River Ecosystems* . Island Press, Washington, DC.

Final Research Reports and Miscellaneous Publications since 1993:

Frissell, C.A., R.J. Baker, D.A. DellaSala, R.M. Hughes, J.R. Karr, D. A. McCullough, R.K. Nawa, J. Rhodes, M.C. Scurlock, and R.C. Wissmar. 2014. Conservation of Aquatic and Fishery Resources in the Pacific Northwest: Implications of New Science for the Aquatic Conservation Strategy of the Northwest Forest Plan. Report prepared for the Coast Range Association, Corvallis, OR. 35 pp. Available online at: http://coastrange.org

Frissell, C.A., with R. Shaftel. 2013. Foreseeable Environmental Impact of Potential Road and Pipeline Development on Water Quality and Freshwater Fishery Resources of Bristol Bay, Alaska. Appendix G (52pp) in *An Assessment of Potential Mining Impacts on Salmon Ecosystems of Bristol Bay, Alaska, Second External Review Draft*. USEPA, Washington, DC 910-R-004a-c. 30 April 2013. Final Report for University of Alaska Anchorage Environment and Natural Resources Institute And Alaska Natural Heritage Program (NatureServe), under contract to USEP. Available online at: http://ofmpub.epa.gov/eims/eimscomm.getfile?p_download_id=513558

Pacific Rivers Council (Scurlock, M., and C.A.Frissell). 2012. Conservation of Freshwater Ecosystems on Sierra Nevada National Forests: Policy Analysis and Recommendations for the Future. Pacific Rivers Council, Portland Oregon, report prepared for Sierra Forest Legacy. 156pp. http://www.sierraforestlegacy.org/Resources/Conservation/Biodiversity/Conservation%20of%20Freshwater%20Ecosystems%20on%20Sierra%20Nevada%20Forests%202012%20PRC.pdf

Frissell, C.A., M. Scurlock, and R. Kattelmann. 2012. SNEP Plus 15 Years: Ecological & Conservation Science for Freshwater Resource Protection & Federal Land Management in the Sierra Nevada. Pacific Rivers Council Science Publication 12-001. Portland, Oregon, USA. 39 pp. http://www.sierraforestlegacy.org/Resources/Conservation/FireForestEcology/ThreatenedHabitats/Aquatic/RETROSNEP_PRC_Report_2012.pdf

MWH. (Montgomery Watson Harza). 2012. Independent Expert Panel Review of Water and Land Resources Division's Project Scoping and Implementation Practices. Prepared for King County Dept. of Natural Resources and Parks, Seattle, WA. 24 January 2012. 67 pp. + appendices. http://kingcounty.gov/environment/dnrp/publications/wlrd-expert-review-report.aspx

Final Research Reports and Misc. Publications since 1993, cont:

Frissell, C.A. 2011. Comment on the environmental effects on Bull Trout (*Salvelinus confluentus*) as considered in the *Supplemental Draft Environmental Impact Statement for the Montanore Project*. Report prepared for Save Our Cabinets, Heron, MT. http://www.earthworksaction.org/files/pubs-others/montanore-comments_christopher-frissell_FINAL_20111220.pdf

Pacific Rivers Council (Wright, B., and C. Frissell). 2010. Roads and Rivers II: An Assessment of National Forest Roads Analyses. Report for the Pacific Rivers Council, Portland, OR. http://pacificrivers.org/science-research/resources-publications/roads-and-rivers-ii/download

Carnefix, G. and C.A. Frissell. 2010. Science for Watershed Protection in the Forest Service Planning Rule: Supporting Scientific Literature and Rationale. Report for the Pacific Rivers Council, 6 October 2010. 22pp. http://pacificrivers.org/files/nfma/supporting-scientific-rationale-for-nfma-language

Carroll, C., D.C. Odion, C.A. Frissell, D.A. Dellasala, B.R. Noon, and R. Noss. 2009. Conservation implications of coarse-scale versus fine-scale management of forest ecosystems: are reserves still relevant? Report for Klamath Center for Conservation Research. http://www.klamathconservation.org/docs/ForestPolicyReport.pdf

Carnefix, G., and C. A. Frissell. 2009. Aquatic and Other Environmental Impacts of Roads: The Case for Road Density as Indicator of Human Disturbance and Road-Density Reduction as Restoration Target, A Concise Review. Pacific Rivers Council Science Publication 09-001. Pacific Rivers Council, Portland, OR and Polson, MT. http://pacificrivers.org/science-research/resources-publications/road-density-as-indicator/download

Duane,T.P., G. Carnefix, S.Chattopadhyay, C. Davidson, D.A. DellaSala, J.Duffield, C. Frissell, M.P. Hayes**,** M. Jennings, J. Kerkvliet, G. LeBuhn, P. Morton, E. Niemi, D. Spooner, and M. Weber. 2008. Economics of Critical Habitat Designation and Species Recovery: Consensus Statement of a Workshop. Report prepared for Pacific Rivers Council after a two-day workshop, October 4-5, 2007, San Francisco, CA. http://pacificrivers.org/science-research/resources-publications/economics-of-critical-habitat-designation-and-species-recovery-consensus-statement-of-a-workshop-sponsored-by-the-pacific-rivers-council-environmental-studies-program-at-san-francisco-state-university-ecotrust-and-the-national-center-for-conservati/download

Williams, J.E., D.A. DellaSala, J. F. Franklin, C.D. Williams, and C. Frissell. 2004. A new vision for wildfire preparation in the western USA. Media report presented at the Society for Conservation Biology Annual Meeting, Aug. 2, 2004, Columbia University, New York, NY.

Final Research Reports and Misc. Publications since 1993, cont:

Frissell, C. A. and G. Carnefix. 2002. Environmental correlates of spatial variation in spawning abundance of bull trout (Salvelinus confluentus) in Rock Creek Basin, Montana, USA. FLBS Report 168-02. Prepared for Rocky Mountain Research Station, USDA Forest Service, Boise, Idaho by Flathead Lake Biological Station, The University of Montana, Polson, Montana. 76 pp. + 2 appendices.

Merrill, T., D.J. Mattson, and C. Frissell. 2001. Life history, reserve design and umbrella effects: grizzly bears and aquatic systems in western Montana. Unpublished manscript, available online at http://y2y.net/files/673-merrill-reserve-design-and-umbrella-effects.pdf

Franklin, J. F., D.A. Perry, R.F. Noss, D. Montgomery, and C. Frissell. 2000. Simplified Forest Management to achieve watershed and forest health. Report for the National Wildlife Federation, Seattle, Washington. 46pp.

Frissell, C.A., P. H. Morrison, S.B. Adams, L. H. Swope, and N.P. Hitt. 2000. Conservation Priorities: an Assessment of Freshwater Habitat for Puget Sound Salmon. Trust for Public Land, Northwest Regional Office, 1011 Western Suite 605, Seattle, WA. http://www.tpl.org/tier3_cd.cfm?content_item_id=9280&folder_id=262.

Frissell, C.A. 1999. An ecosystem approach for habitat conservation for bull trout: groundwater and surface water protection. Flathead Lake Biological Station, Open File Report 156-99, The University of Montana, Polson, MT.

Hitt, N.P. and C.A. Frissell. 1999. Wilderness in a landscape context: a quantitative approach to ranking aquatic diversity areas in western Montana. Paper presented at Wilderness Science Conference, 23-27 May, Missoula, MT.

Montana Bull Trout Scientific Group. 1998. The relationship between land management activities and habitat requirements of bull trout. Report prepared for the Montana Bull Trout Restoration Team, Office of the Governor, Helena, MT.

Frissell, C.A. 1998. Landscape refugia for conservation of Pacific salmon in selected river basins of the Olympic Peninsula and Hood Canal, Washington. Flathead Lake Biological Station, Open File Report 147-98, The University of Montana, Polson, MT.

Frissell, C.A. 1997. Ecological benefits of wildland reserves: The proposed Copper Salmon Wilderness in southwest Oregon. Flathead Lake Biological Station, Open File Report 150-97, The University of Montana, Polson, MT.

Huntington, C.W., and C.A. Frissell. 1997. Aquatic conservation and salmon recovery in the North Coast Basin of Oregon: A crucial role for the Tillamook and Clatsop State Forests. Report prepared for Oregon Trout, Portland, OR.

Williams, R.N., L.D. Calvin, C.C. Coutant, M.W. Erho, Jr., J.A. Lichatowich, W.J. Liss, W. E. McConnaha, P.R. Mundy, J.A. Stanford, R.R. Whitney, D.L. Bottom, and C.A. Frissell. In press. *Return to the River: Restoration of Salmonid Fishes in the Columbia River Ecosystem.* Independent Scientific Group, Northwest Power Planning Council, Portland, OR.

Final Research Reports and Misc. Publications since 1993, cont:

Frissell, C.A., J.L. Ebersole, W.J. Liss, B.J. Cavallo, and G.C. Poole. 1996. Potential effects of climate change on thermal complexity and biotic integrity of streams: seasonal intrusion of non-native fishes. Final Report for USEPA Environmental Research Laboratory, Duluth, MN. Oak Creek Laboratory of Biology, Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR.

Bottom, D.L., J.A. Lichatowich, and C.A. Frissell. 1996. Variability of marine ecosystems and relation to salmon production. Report prepared for Theme 2 of the Pacific Northwest Coastal Ecosystem Region Study Workshop, Troutdale, OR, 12-14 August.

Clancy, C., C. Frissell, and T. Weaver. 1996. Assessment of methods for removal or suppression of introduced fish to aid bull trout recovery. Report prepared by the Montana Bull Trout Scientific Group for the Montana Bull Trout Restoration Team. Montana Fish, Wildlife and Parks, Helena, MT.

Frissell, C.A., J. Doskocil, J. Gangemi, and J. Stanford. 1995. Identifying priority areas for protection and restoration of riverine biodiversity: a case study in the Swan River basin, Montana, USA. Flathead Lake Biological Station, Open File Report 136-95, The University of Montana, Polson, MT.

Beschta, R.L., C.A. Frissell, R. Gresswell, R. Hauer, J.R. Karr, G.W. Minshall, D.A. Perry, and J.J. Rhodes. 1995. Wildfire and salvage logging: recommendations for ecologically sound post-fire salvage logging and other post-fire treatments on federal lands in the West. The Pacific Rivers Council, Eugene, OR.

Frissell, C.A. 1993. The shrinking range of the Pacific Salmon. Report and status and range maps prepared for the Pacific Northwest Salmon Study, The Wilderness Society, Washington, DC.

Frissell, C.A., and W.J. Liss. 1993. Valley segment classification for the streams of Great Basin National Park, Nevada. Report prepared for the National Park Service Cooperative Park Studies Unit, College of Forestry, Oregon State University, Corvallis, OR.

Frissell, C.A. 1993. Panacea or placebo? An ecologist's view of captive breeding. *Wild Fish* July/August 1993:7-12. The Wilderness Society, Portland, OR.

Frissell, C.A. 1993. A new strategy for watershed restoration and recovery of Pacific salmon in the Pacific Northwest. Report prepared for The Pacific Rivers Council, Eugene, Oregon. Oak Creek Laboratory of Biology, Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR.

Selected Papers and Seminars Presented Since 1993 (__=presenter):

Frissell, C.A., M. Scurlock, and K Crispen. 2011. Forest thinning in Pacific Northwest riparian areas: rationale, risks, and policy calibration. (Abstract) Annual Meeting of the American Fisheries Society, Symposium on Forest Management: Can Fish and Fiber Coexist? 4-8 September, Seattle, WA. http://pacificrivers.org/science-research/resources-publications/dr.-chris-frissells-american-fisheries-society-presentation-on-riparain-thinning/download

Frissell, C.A. 2008. Water, watersheds and forest stewardship: the shared landscape (Abstract). Paper presented at the Western Stewardship Summit: Restoring Community and the Land, Bend, OR, September 24-26 2008.

Frissell, C.A., and N.P. Hitt. 2008. Four biological quanta: a conceptual framework for conservation of stream ecosystems. (Abstract) Society for Conservation Biology Annual Meeting Symposium: Advances in Freshwater Conservation Planning. Chattanooga, TN, July 13-19, 2008.

Frissell, C.A. 2008. Ecological impacts of roads in an era of climate change (Abstract). Watershed Restoration and Forest Roads Symposium, Pacific Rivers Council, 4 April 4, Tacoma, WA. http://pacificrivers.org/conservation-priorities/land-management/roads/watershed-restoration-and-forest-roads-symposium

Frissell, C.A., and G, Carnefix. 2007. (Abstract) Spawning abundance of bull trout (*Salvelinus confluentus*) in relation to geomorphology, temperature and roads in tributaries of Rock Creek Basin (Missoula and Granite Counties), Montana, US. Annual Meeting of the Montana Chapter of the American Fisheries Society, 13-16 February, Missoula, MT. http://www.fisheries.org/units/AFSmontana/2007%20MCAFS%20Annual%20Meeting%20Program.pdf

Frissell, C.A. 2007. Setting regional priorities for watershed restoration. 25th Salmonid Restoration Conference, Salmonid Restoration Federation, 9-10, Santa Rosa, CA.

Frissell, C.A. 2006. Post-fire management effects on streams. NCSSF Disturbance, Management, and Biodiversity Symposium, **National Commission for Science and Sustainable Forestry,** 26-27 April, **Denver, CO.**

Frissell, C.A., and G. Carnefix. 2005. (Abstract) Indicators of landscape pattern for freshwater ecosystems. 20th Annual Symposium of the US-International Association for Landscape Ecology, 12-16 March, Syracuse, NY.

Frissell, C.A. 2004. Managing risk and uncertainty: National Forest management and freshwater conservation. Regional Centennial Forum: The Forest Service In the Pacific Southwest Region. US Forest Service, 5-6 November, Sacramento, CA.

Williams, J.E., D.A. DellaSala, J. F. Franklin, C,D.Williams, and C. Frissell. 2004. Scientific findings require a new vision for successful wildlfire preparation. News briefing at the Society for Conservation Biology Annual Meeting, Aug. 2, 2004., Columbia University, New York, NY. http://www.conbio.org/Media/Fire/

Selected Papers and Seminars Presented Since 1993, continued:

Frissell, C.A. 2001. (Abstract) What to do first with limited time, money, and staff. Watershed Restoration Workshop: Integrating Practical Approaches. Oregon Chapter of the American Fisheries Society, 13-15 November, Eugene, OR.

Ebersole, J.L., Colden V. Baxter, Hiram W. Li, and William J. Liss, and Frissell, C.A. 2001. (Extended abstract) Detecting temporal dynamics and ecological effects of smallmouth bass invasion in northeast Oregon streams. *In*: Proceedings, American Fisheries Society Special Symposium: Practical Approaches for Conserving Native Inland Fishes of the West. Montana Chapter and Western Division of the American Fisheries Society, 6-8 June, The University of Montana, Missoula, MT.

Carnefix, G., C. Frissell, and E. Reiland. 2001. (Extended abstract) Complexity and stability of bull trout (*Salvelinus confluentus*) movement patterns in the Rock Creek drainage, Missoula and Granite counties, Montana. *In*: Proceedings, American Fisheries Society Special Symposium: Practical Approaches for Conserving Native Inland Fishes of the West. Montana Chapter and Western Division of the American Fisheries Society, 6-8 June, The University of Montana, Missoula, MT.

Frissell, C.A. 1999. (Abstract) Groundwater processes and stream classification in the montane West. Invited paper, Symposium #7: Aquatic Classification Schemes for Ecosystem Management: Making the Transition from Methods Development to Application and Validation. Annual Meeting of the Ecological Society of America 7-12 August, Spokane, WA.

Frissell, C.A. 1999. Fisheries and watershed processes: strategies for protection and restoration. Invited paper, Annual Meeting of the Cal-Neva Chapter of the American Fisheries Society, 24-27 March, Redding, CA.

Frissell, C.A. 1999. Surface-subsurface flow linkages in rivers and their importance for river flow conservation. Invited paper, Symposium on Water Quality and Hydropower Re-licensing, Annual Meeting of the Cal-Neva Chapter of the American Fisheries Society, 24-27 March, Redding, CA.

Frissell, C.A. 1999. Dams, uncertainty, and the salmon ecosystem. *Keynote Address*, Annual Meeting of the Idaho Chapter of the American Fisheries Society and The Wildlife Society, 4-6 March, Boise, ID.

Frissell, C.A. 1998. Climate forcing of thermal habitat in Pacific Northwest rivers: Buffering effects of floodplain forests and hyporheic processes. (Abstract) Symposium on Climate Change Impacts to Freshwater Fish Habitats, Annual Meeting of the American Fisheries Society, 23-27 August, Hartford, CT.

Frissell, C.A. 1998. Ecosystem concepts in large-scale restoration. (Abstract). Montana Chapter of the American Fisheries Society, 3-5 February, Helena, MT.

Selected Papers and Seminars Presented Since 1993, continued:

Frissell, C.A. and B.J. Cavallo 1997. Aquatic habitats used by larval western toads (*Bufo boreas*) on an intermontane river floodplain and some landscape conservation implications (Abstract). Annual Meeting of the Ecological Society of America, 10-14 August, Albuquerque, NM.

Stanford, J.A. (presented by C.A. Frissell). 1997. Conservation and enhancement of alluvial rivers: the importance of hyporheic linkages. (Abstract). Symposium on Ecological Effects of Roads, Society for Conservation Biology, 7-10 June, Victoria, British Columbia, Canada.

Frissell, C.A., and G.C. Poole . 1997 Management of Riparian Zones in Western Montana: Present Issues and Emerging Challenges. (Abstract). Annual Meeting of the American Fisheries Society, 23-28 August, Monterey, CA.

Frissell, C.A., and J.T. Gangemi. 1997. Roads and the conservation of aquatic biodiversity and ecological integrity. (Abstract). Society for Conservation Biology, Victoria, British Columbia, Canada, 7-10 June.

Frissell, C.A. 1997. Spatial assessment of biological status and biodiversity loss. Invited seminar, National Research Center for Statistics and the Environment, University of Washington, Seattle, WA, 14 January.

Frissell, C.A., and B.J. Cavallo 1996. Thermal and hydrologic diversity of aquatic habitats mediated by floodplain complexity and hyporheic flow exchange in an alluvial segment of the Middle Fork Flathead River, Montana, USA. (Abstract). Annual Meeting of the N. Am. Benthological Society, Kalispell, MT, 3-8 June.

Frissell, C.A. 1995. Ecological principles for watershed restoration. (Abstract). Invited paper for Workshop on Watershed Restoration: Principles and Practices, Annual Meeting of the American Fisheries Society, Tampa, FL, 27-31 August.

Frissell, C.A. 1995. Managing native fish and their ecosystems: let's get (spatially) explicit! (Abstract). Invited panel presentation at Montana Chapter of the American Fisheries Society, Chico Hot Springs, MT, 6-10 February.

Frissell, C.A. 1995. Birth in the fast lane: sediment transport, human disturbance, and reproductive strategies of salmonid fishes in Pacific Northwest streams. (Abstract). Invited paper for Symposium on Influence of Geomorphic Processes on Terrestrial and Aquatic Ecosystem patterns and Processes, Annual meeting of the Ecological Society of America, Snowbird, UT, 31 July-3 August.

Frissell, C.A. 1995. Resource management impacts on bull trout populations. Invited panel presentation for Searching for Solutions: Solving the Bull Trout Puzzle Science and Policy Conference, Andrus Center for Public Policy, Boise State University, Boise, ID, 1-2 June.

Selected Papers and Seminars Presented Since 1993, continued:

Frissell, C.A. 1995. Watershed dynamics: natural pattern and process and some consequences for ecosystem management. Invited presentations at Managing Terrestrial Ecosystems Relative to Past and Present Disturbances: A Workshop Integrating Fire, Range, Fish and Wildlife Habitat and the Practice of Silviculture in the Northern Region. U.S. Forest Service, Missoula, MT, 14-16 March.

Ebersole, J.L., C.A. Frissell, and W.J. Liss (co-presenters). 1995. Invasion of non-native fishes in northeast Oregon and western Montana streams: potential impacts of climate change. (Abstract). Oregon Chapter of the American Fisheries Society, Ashland, OR, 15-17 February.

Frissell, C.A. 1994. Watershed restoration strategies. (Invited presenter and session convenor) Watersheds '94 Expo, US Environmental Protection Agency and Center for Streamside Studies, University of Washington. Bellevue, WA, 27-30 September.

Frissell, C.A. 1994. A hierarchical approach to restoration of riverine ecosystems. Invited paper at Symposium on Aquatic Habitat Restoration in Northern Ecosystems, Alaska Chapter of the American Fisheries Society, Girdwood, AK, 20-22 September.

Frissell, C.A. 1994. An integrated, biophysical strategy for ecological restoration of large watersheds (Abstract). Annual Conference of The Universities Council on Water Resources, Big Sky, MT, 3-5 August.

Frissell, C.A., and J.A. Stanford. 1994. Designing a watershed reserve network to protect and restore aquatic biodiversity in the northern Rocky Mountains (Abstract). Annual meeting of the Montana Chapter of the American Fisheries Society, Billings, Montana, Billings, MT, 9 February.

Frissell, C.A. 1994. The Endangered Species Act: principles for the protection and recovery of fishes. Invited panel presentation, annual meeting of the Idaho Chapter of the American Fisheries Society, McCall, ID, 24-26 February.

Frissell, C.A., W.J. Liss, B. Doppelt, and D. Bayles. 1993. A new, ecologically based restoration strategy for Pacific salmon in the Pacific Northwest (Abstract). Annual meeting of the American Fisheries Society, Portland, OR, 29 August-2 September.

Technical Workshops Organized (selected):

Lead organizer and facilitator, New Science Implications for the Aquatic Conservation Strategy of the Northwest Forest Plan. Sponsored by the Coast Range Association, 2-3 December 2013, Portland, OR.

Co-organizer, with M. Scurlock and R. Kattelmann: SNEP Plus 15 Years: Ecological & Conservation Science for Freshwater Resource Protection & Federal Land Management in the Sierra Nevada. Sponsored by Pacific Rivers Council, Sierra Forest Legacy, UC Berkeley School Environmental Design, UC Davis Center for Watershed Science, and CaliforniaTrout; 12-13 December 2011, Davis, CA.

Technical Workshops Organized (selected), continued:

Organizer and facilitator, Workshop on Science for River and Watershed Conservation. Sponsored by Campaign for Montana's Headwaters, 7 October 2010, Flathead Lake Biological Station, Polson, MT.

Co-convener, with M. Scurlock and Kristen Boyles: Technical Workshop on Science for Forest Planning. Sponsored by Pacific Rivers Council and Earthjustice, 29 June 2010, Seattle, WA.

Organizer and panelist, Umpqua Independent Science Council. Sponsored by Pacific Rivers Council, 2010-2011.

Co-organizer and panelist, with Deanne Spooner and David Bayes: Workshop on Economics of ESA Critical Habitat Policy, sponsored by Pacific Rivers Council and San Francisco State University, October 4-5, 2007, San Francisco, CA.

Organizer and coordinator of Science Panel on Roads and Watersheds, sponsored by Pacific Rivers Council, 10-11 November 2006, Forest Grove, OR.

Organizer and coordinator of the Recovery Science Panel for the Western Native Trout Campaign. Sponsored by Pacific Rivers Council, meeting 2-3 March 2002, Portland, OR.

Organizer and coordinator of Biodiversity Workshop, Consortium for the Study of North Temperate Montane Ecosystems. A cooperative research venture of The University of Montana and Montana State University, supported by the NSF EPSCoR program. 4 February, 1997 Missoula, MT.

Scientific Workshop on Large Basin Restoration: Grande Ronde River (co-organizer). 21-22 March 1993, La Grande, OR. Sponsored by The Pacific Rivers Council.

Scientific Workshop on Large Basin Restoration: South Umpqua River. 16-18 September 1992, Roseburg, Oregon. Sponsored by The Pacific Rivers Council.

Scientific Workshop on Large Basin Restoration: Lower Rogue River. 21-23 October 1992, Gold Beach, OR. Sponsored by The Pacific Rivers Council.

Other Panels and Workshops Attended by Invitation since 1994:

Invited Review Panelist, Workshop on Linking Habitat Characteristics to Salmon Data. 29-30 September 1999, National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, WA.

Invited participant, Yellowstone to Yukon Aquatic Conservation Science Workshop. 20-22 August 1999, Flathead Lake Biological Station, The University of Montana, Polson, MT.

Invited Panelist, Workshop on Options for Restoring Salmon Habitat in the Mainstem Snake and Columbia Rivers. Pacific Northwest National Laboratory-Battelle, 19 August 1999, Kennewick, WA

Panelist at State of Oregon/National Marine Fisheries Service Memorandum of Agreement Committee Workshop: Cumulative Effects of State and Private Forest Practices on Salmon Habitat. 21April 1998, Salem, OR.

Other Workshops Attended since 1994, continued:

Invited participant in a scientific workshop, Multiple Stressors in Ecological Risk Management. Sponsored by the Society for Environmental Chemistry and Toxicology and the USEPA, 13-18 September 1997, Pellston, MI.

Society for Conservation Biology Workshop: Communicating with the Media (panel member). 9 June 1997, Victoria, British Columbia, Canada.

Invited speaker for a workshop, Continuing Education in Ecosystem Management. Sponsored by the University of Idaho. Catchment scale processes and linkages between landscape and stream conditions. 31 January 1997, Moscow, ID.

The Nature Conservancy, Aquatic Classification Workshop (invited presenter). 9-11 April 1996, Cedar Creek Farm, MO.

Kenai River Community Forum (keynote speaker and panelist). The Nature Conservancy of Alaska, USEPA and USFWS, 19-21 April, Soldotna, AK.

Conservation Biology and Management of Interior Salmonids (invited presenter and session co-moderator). USDA Forest Service Intermountain Research Station and Utah State University, 4-5 October 1995, Logan, UT.

Eastside Ecosystem Planning Workshop. Sierra Club Legal Defense Fund, 16 December 1994, Portland, OR.

Co-instructor at workshop series on Watershed Restoration and the "Rapid Biotic Response Strategy" for Riverine Ecosystem Restoration, sponsored by The Pacific Rivers Council, 1993-95, California, Oregon, and Washington.

Fire/Salvage and Aquatic Ecosystems Policy Workshop. The Pacific Rivers Council, 15 December 1994, Portland, OR.

Panel on Forest Health Issues, Native Forest Network annual conference, 13 November 1994, Missoula, MT.

Workshop on Watershed/Fisheries Cumulative Effects Analysis, sponsored by Headwaters, The Pacific Rivers Council, USDA Forest Service, and Bureau of Land Management. 29 September-2 October, 1994, Ruch, OR.

Boise Funders' Scoping Meeting, sponsored by Bullit, Harder, and Lazar Foundations, 30-31 August 1994, Boise, Idaho.

Workshop for a statewide process to prioritize restoration of watersheds and salmon populations, by invitation of Oregon Senate President Bill Bradbury, 18 May 1994, Salem, OR.

Other Presentations (Selected):

Scientists Briefing for U.S. Senate staff on post-fire logging and forest management and freshwater resources. Washington, D.C., 18-19 September 2006.

Invited testimony on federal land management and the future of salmon and aquatic biodiversity in the Pacific Northwest, to the U.S. House of Representatives, Subcommittee on National Parks and Public Lands, Washington, D.C., 11 March 1993.

Briefing for Congressional representatives and staff on federal lands management and conservation and recovery of salmonid fishes and riverine ecosystems, Washington, D.C., 22 January 1993.

Invited testimony to the 1991 Oregon State Legislature, on panel representing the Oregon Chapter of the American Fisheries Society, on the status of native fishes, impacts of forest practices on fish habitat, and the need or changes in environmental regulation.

Invited testimony to the Oregon Board of Forestry Forest Issues Forum, December 1990, on cumulative impacts of forest practices on native aquatic species and the need for changes in forest management.

Worked with Oregon Public Broadcasting to describe our research project and its significance in a 15-minute segment of the television program, Oregon Field Guide, first aired in June 1990.

Presented seminars, informal presentations, lectures, and discussions at research review meetings, as guest speaker in classrooms and public interest groups, at state board meetings, at workshops, and on field trips with foresters, geotechnical personnel, fishery and watershed managers, and conservationists.

Table __. Median percentage of streambed material smaller than 6.35 mm in McNeil core samples collected from bull trout spawning areas in the Swan River drainage from 1987-2018.

Stream	1987	1988	1989	1990	1991	1992	1993	1994	1995
Elk	34.1	35.9	37.6	39.8	35.0	35.7	39.0	34.9	27.1
Goat	26.7	29.8	35.0	27.9	34.1	37.3	27.2	26.7	26.2
Squeezer (Old)	33.6	38.9	42.2	42.4	--	40.1	39.8	40.9	40.8
Squeezer (New)	--	--	--	--	--	--	--	24.9	25.3
Lion	36.7	38.4	39.0	44.5	41.2	42.6	43.4	39.8	37.0
Jim	--	41.5	50.3	46.4	39.0	40.6	41.2	42.6	40.5
Soup	--	--	--	--	--	--	34.2	34.9	34.2
South Lost	--	--	--	--	--	--	--	30.4	28.8
Woodward	--	--	--	--	--	--	--	--	--
South Woodward	--	--	--	--	--	--	--	--	--

Stream	1996	1997	1998	1999	2000	2001	2002	2003	2004
Elk	30.8	31.6	32.1	35.1	35.4	32.1	31.6	30.1	31.8
Goat	28.9	28.2	26.7	27.4	28.0	26.9	24.8	26.6	29.4
Squeezer (Old)	41.3	--	--	--	--	--	--	--	--
Squeezer (New)	24.8	25.2	24.1	25.3	26.1	24.2	23.6	25.0	26.3
Lion	37.6	38.1	39.0	38.4	39.2	37.1	36.6	35.3	36.2
Jim	38.3	39.6	40.1	38.1	38.9	37.8	36.1	35.1	34.6
Soup	34.1	33.9	35.3	37.0	38.2	36.9	37.2	38.0	39.7
South Lost	23.4	26.8	25.1	30.1	31.6	30.4	29.6	30.1	32.2
Woodward	35.6	37.2	35.8	38.1	36.9	37.0	35.8	36.7	35.9
South Woodward	30.0	34.1	33.1	26.0	28.7	27.8	25.1	24.9	27.6

Stream	2005	2006	2007	2008	2009	2010	2011	2012	2013
Elk	32.3	30.9	32.4	31.1	30.1	30.8	26.6	29.2	28.9
Goat	31.3	31.6	33.7	34.0	33.2	28.9	28.0	31.4	29.0
Squeezer (Old)	--	--	--	--	--	36.1	33.8	33.0	35.7
Squeezer (New)	30.1	31.0	33.6	35.8	32.8	30.6	33.4	29.7	30.4
Lion	36.8	36.9	37.4	37.1	36.6	42.9	39.0	32.4	42.9
Jim	35.7	37.1	36.9	34.0	34.2	33.7	35.0	36.7	40.6
Soup	37.1	35.9	36.6	36.1	35.2	39.4	38.4	37.6	35.4
South Lost	33.0	31.6	32.3	30.2	29.6	32.1	27.5	24.2	26.6
Woodward	--	34.2	35.0	34.1	33.3	36.4	38.8	32.8	41.3
South Woodward	--	28.0	31.2	32.8	29.8	28.2	35.4	32.2	29.0

Stream	2014	2015	2016	2017	2018	2019			
Elk	29.4	29.4	31.2	28.8	29.3				
Goat	31.2	30.0	31.4	23.8	26.9				
Squeezer (Old)	36.6	34.1	36.8	42.0	40.0				
Squeezer (New)	31.2	30.8	32.1	31.6	33.3				
Lion	42.6	39.1	41.3	38.7	36.1				
Jim	41.4	37.6	39.8	37.4	39.1				
Soup	35.6	29.3	33.6	33.4	36.2				
South Lost	31.0	30.3	32.5	33.2	33.4				
Woodward	41.7	35.3	37.4	37.4	40.0				
South Woodward	30.5	29.8	32.9	32.2	31.0				

Stream									
Elk									
Goat									
Squeezer (Old)									
Squeezer (New)									
Lion									
Jim									
Soup									
South Lost									
Woodward									
South Woodward									

Stream									
Elk									
Goat									
Squeezer (Old)									
Squeezer (New)									
Lion									
Jim									
Soup									
South Lost									
Woodward									
South Woodward									

Stream									
Elk									
Goat									
Squeezer (Old)									
Squeezer (New)									
Lion									
Jim									
Soup									
South Lost									
Woodward									
South Woodward									

SENSITIVE SPECIES LIST Forest Service, Region 1 Februrary 2011	States Where Sensitive (a)				State Ranking				Forests Where Species is Known (K) or Suspected (S) to Occur													Comments
	MT	ID	ND	SD	MT	ID	ND	SD	B/D	BRT	CLW	CUS	DPG	FLAT	GAL	HEL	IPNF	KOOT	L&C	LOLO	NEZ	
BIRDS																						
American peregrine falcon (*Falco peregrinus anatum)*	X	X	X	X	S3	S2B	S1	SX	K	K		K		K	K	K	K	K	K	K	K	Federally delisted on August 25, 1999. USFWS monitoring of status for 5-year intervals after delisting. Species of Concern in MT, State Endangered in SD. ND CWCS Level 3 spp.
Baird's sparrow (*Ammodramus bairdii)*			X	X	S3B		SU	S2B SZN				K	K									MT CFWCS as a Priority 2 spp. Listed in SD CWCS, and as a Level 1 species in ND CWCS.
Bald eagle (*Haliaeetus leucocephalus*)	X	X	X	X	S3	S3B S4N	S1	S1B S2N	K	K	K	K	K	K	K	K	K	K	K	K	K	Federally delisted on June 28, 2007. USFWS monitoring for 5-year intervals after delisting. State Threatened in SD. ND CWCS Level 1 spp.
Black-backed woodpecker (*Picoides arcticus*)	X	X			S3	S3		S3	K	K	K	K		K	K	K	K	K	K	K	K	Species of Concern in MT, and in MT CFWCS as a Priority 2 spp. Listed in SD CWCS, and as a Level 1 species in ND CWCS.
Black swift *(Cypseloides niger)*		X			S1B	S1B											K				S	Colonial nester with few known nesting sites. IPNF has known nesting sites.
Blue-gray gnatcatcher (*Polioptila caerulea)*	X				S2B			S1B SZN				K										Species of Concern in MT, and in MT CFWCS as a Priority 2 spp. South end of Priors Mtns. in MT.
Burrowing owl (*Athene cunicularia)*	X		X	X	S3B	S2B	SU	S3 S4B SZN				K	K						S			Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. Listed in SD CWCS, and as a Level 2 species in ND CWCS.
Common loon *(Gavia immer)*	X	X			S3B	S1B S2N	S4	S1B S3						K			K	K		K	S	Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. ID CWCS spp.
Flammulated owl *(Otus flammeolus)*	X	X			S3B	S3B		S1B SZN	K	K	K			K	S	K	K	K	S	K	K	Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. ID CWCS spp.
Greater prairie chicken *(Tympanuchus cupido)*			X		SX		S2	S4					K									Listed in SD CWCS, and as a ND CWCS Level 2 spp.

SENSITIVE SPECIES LIST Forest Service, Region 1 Februrary 2011	States Where Sensitive (a)				State Ranking				Forests Where Species is Known (K) or Suspected (S) to Occur													Comments
	MT	ID	ND	SD	MT	ID	ND	SD	B/D	BRT	CLW	CUS	DPG	FLAT	GAL	HEL	IPNF	KOOT	L&C	LOLO	NEZ	
BIRDS continued																						
Greater sage-grouse *(Centrocercus urophasianus)*	X		X	X	S2	S2	SU	S2	K			S	K						S			Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. Listed in SD CWCS, and as a Level 2 species in ND CWCS. No breeding sites on BDNF.
Harlequin duck *(Histrionicus histrionicus)*	X	X			S2B	S1B			K		K	K		K	K	S	K	K	K	K	K	Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. ID CWCS spp.
Loggerhead shrike *(Lanius ludovicianus)*			X	X	S3B		SU	S3				K	K									Species of Concern in MT, and in MT CFWCS as a Priority 2 spp. ND CWCS Level 2 spp.
Long-billed curlew *(Numenius americanus)*			X	X	S3B	S2B	S2	S3B SZN				K	K								S	Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. SD CWCS
Mountain quail *(Oreortyx pictus)*		X				S1															K	ID CWCS spp.
Pygmy nuthatch *(Sitta pygmaea)*		X			S4	S2		S2 S3			K						K				K	MT CFWCS Priority 2 spp. ID CWCS spp.
Sprague's pipit *(Anthus spragueii)*			X	X	S3B		S3	S2B SZN					K									Species of Concern in MT, and in MT CFWCS as a Priority 2 spp. Listed in SD CWCS, and as a Level 1 species in ND CWCS.
Trumpeter swan *(Cygnus buccinator)*	X				S3	S1B S2N	SX	S3	K						K							Species of Concern in MT, and in MT CFWCS as a Priority 2 spp. Listed in
White-headed woodpecker *(Picoides albolarvatus)*		X			SNA	S2															K	ID CWCS spp.
MAMMALS																						
Black-tailed prairie dog *(Cynomys ludovicianus)*	X		X	X	S3		SU	S4				K	K									Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. ND CWCS
Bighorn sheep *(Ovis canadensis)*	X	X	X	X	S4	S1	S2		K	K		K	K	K	K	K		K	K	K	K	MT CFWCS as a Priority 3 spp.

SENSITIVE SPECIES LIST Forest Service, Region 1 Februrary 2011	States Where Sensitive (a)				State Ranking				Forests Where Species is Known (K) or Suspected (S) to Occur													Comments
	MT	ID	ND	SD	MT	ID	ND	SD	B/D	BRT	CLW	CUS	DPG	FLAT	GAL	HEL	IPNF	KOOT	L&C	LOLO	NEZ	
MAMMALS continued																						
Fisher *(Martes pennanti)*	X	X			S3	S1	S2		K	K	K			K		K	K	K	K	K	K	Species of Concern in MT, and in MT CFWCS as a Priority 2 spp. ID CWCS spp.
Fringed myotis *(Myotis thysanodes)*		X			S3	S2		S2	K		K	K	K			K	K	K	K		K	Sub Species of Concern does not occur on FS in SD.
Gray wolf *(Canis lupus)*	X	X			S4	S2			K	K	K	K	K	K	K	K	K	K	K	K	K	Delisted in Idaho and Montana in 2011. However, wolves remain federally listed in North and South Dakota.
Great Basin pocket mouse *(Perognathus parvus)*	X				S2 S3				S													Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. At periphery of range on BDNF.
Long-eared myotis *(Myotis evotis)*				X	S4		SU	S1	K	K	K	K	K	K	K	K	K	K	K	K	K	MT CFWCS Priority 3 spp. ND CWCS Level 3 spp. Limited distribution, but does occur on NFS lands based on survey results.
Long-legged myotis *(Myotis volans)*				X	S4		SU	S5	K	K	K	K	K	K	K	K	K	K	K	K	K	MT CFWCS Priority 3 spp. ND CWCS Level 3 spp. Limited distribution, but does occur on NFS lands based on
North American wolverine *(Gulo gulo luscus)*	X	X			S3	S2			K	K	K	K		K	K	K	K	K	K	K	K	Species of Concern in MT, and in MT CFWCS as a Priority 2 spp. ID CWCS spp.
Northern bog lemming *(Synaptomys borealis)*	X	X			S2	S1			K	K				K		S	K	K	K	K		Species of Concern in MT, and in MT CFWCS as a Priority 1 spp.
Pallid bat *(Antrozous pallidus)*	X				S2	S1						K										Species of Concern in MT, and in MT CFWCS as a Priority 1 spp.
Pygmy rabbit *(Brachylagus idahoensis)*	X				S3	S2			K													Species of Concern in MT, and in MT CFWCS as a Priority 1 spp.
Spotted bat *(Euderma maculatum)*	X				S2	S3			K			K										Species of Concern in MT, and in MT CFWCS as a Priority 1 spp.
Towsend's big-eared bat *(Corynorhinus townsendii)*	X	X		X	S2	S3		S2 S3	K	K	K	K	K	K	K	K	K	K	K	K	K	Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. SD CWCS spp. ID CWCS spp. Occurs on Nez

SENSITIVE SPECIES LIST Forest Service, Region 1 Februrary 2011	States Where Sensitive (a)				State Ranking				Forests Where Species is Known (K) or Suspected (S) to Occur													Comments
	MT	ID	ND	SD	MT	ID	ND	SD	B/D	BRT	CLW	CUS	DPG	FLAT	GAL	HEL	IPNF	KOOT	L&C	LOLO	NEZ	
White-tailed prairie dog *(Cynomys leucurus)*	X				S1							K										Species of Concern in MT, and in MT CFWCS as a Priority 1 spp.
AMPHIBIANS																						
Coeur d'Alene salamander (*Plethodon idahoensis)*	X	X			S2	S2				K	K						K	K		K	K	Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. ID CWCS spp.
Great Plains toad *(Bufo cognatus)*	X				S2		SU	S5				K										MT CFWCS Priority 2 spp. Probable reduction in occurance/range.
Northern leopard frog *(Rana pipiens)*	X				S1-w S4-e	S2	SU	S5		S		K		S	K	K		K	K	S		Species of Concern in MT, and in MT CFWCS as a Priority 1 spp.
Plains spadefoot (*Spea bombifrons*)	X				S3		SU	S5				S				K						MT CFWCS Priority 2 spp. ND CWCS Level 1 spp. Probable reduction in occurance/range.
Western toad *(Bufo boreas)*	X	X			S2	S4			K	K	K	K		K	K	K	K	K	K	K	K	Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. Loss of breeding sites is ongoing.
REPTILES																						
Greater short-horned lizard (*Phrynosoma hernandesi*)	X				S3		SU	S2				K							S			Species of Concern in MT, and in MT CFWCS as a Priority 2 spp. ND CWCS Level 2 spp.
Milk snake *(Lampropeltis triangulum)*	X				S2			S4				K										Species of Concern in MT, and in MT CFWCS as a Priority 1 spp.
Ringneck snake *(Diadophis Punctatus)*		X				S2		S2			K										S	ID CWCS spp. Unconfirmed occurance on NFS lands on Nez Perce NF.
Western hognose snake *(Heterodon nasicus)*	X				S2		SU	S5				K										Species of Concern in MT, and in MT CFWCS as a Priority 1 spp. ND CWCS
INSECTS																						
Arogos skipper *(Atrytone arogos iowa)*			X		SNR		SU	S2					K									

SENSITIVE SPECIES LIST Forest Service, Region 1 Februrary 2011	States Where Sensitive (a)				State Ranking				Forests Where Species is Known (K) or Suspected (S) to Occur													Comments
	MT	ID	ND	SD	MT	ID	ND	SD	B/D	BRT	CLW	CUS	DPG	FLAT	GAL	HEL	IPNF	KOOT	L&C	LOLO	NEZ	
INSECTS continued																						
Broad-winged skipper *(Poanes viator)*			X				S2	S2					K									
Dakota skipper (*Hesperia dacotae)*			X	X			S2	S2					K									
Dion skipper *(Euphyes dion)*			X				S1						K									
Mulberry wing *(poanes massasoit)*			X				S2	S1					K									
Ottoe skipper *(Hesperia ottoe)*			X	X	S2-w S3-e		SU	S2					K									Species of Concern in MT. SD CWCS spp.
Powesheik skipper *(Oarisma powesheik)*			X	X			SU	S2					K									SD CWCS spp.
Regal fritillary *(Speyeria idalia)*			X				S2	S3					K									SD CWCS spp.
Tawny crescent *(Phyciodes batessi)*			X		S2-w S3-e		S3	S2					K									

(a) Species are listed as Sensitive by State. The State where a species is listed as Sensitive is indicated by an "X" in the State/species column. A species iedntified as Sensitive within a State, will be considered as Sensitive on all Units within the state where it occurs, unless described otherwise.

(b) National Forest (Grasslands) where a species is known or suspected to occur, within States where a species is listed as Sensitive, are identified by shading and either a known "K" or suspected "S" in the Forest/species column.

CWCS = Comprehensive Wildlife Conservation Strategy

CFWCS = Comprehensive Fish and Wildlife Conservation Strategy

SD bird species may have two state ranks, one for breeding (S#B) and one for nonbreeding seasons (S#N)

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Review and synthesis

Achievable future conditions as a framework for guiding forest conservation and management

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ABSTRACT

We contend that traditional approaches to forest conservation and management will be inadequate given the predicted scale of social-economic and biophysical changes in the 21st century. New approaches, focused on anticipating and guiding ecological responses to change, are urgently needed to ensure the full value of forest ecosystem services for future generations. These approaches acknowledge that change is inevitable and sometimes irreversible, and that maintenance of ecosystem services depends in part on novel ecosystems, i.e., species combinations with no analog in the past. We propose that ecological responses be evaluated at landscape or regional scales using risk-based approaches to incorporate uncertainty into forest management efforts with subsequent goals for management based on Achievable Future Conditions (AFC). AFCs defined at a landscape or regional scale incorporate advancements in ecosystem management, including adaptive approaches, resilience, and desired future conditions into the context of the Anthropocene. Inherently forward looking, ACFs encompass mitigation and adaptation options to respond to scenarios of projected future biophysical, social-economic, and policy conditions which distribute risk and provide diversity of response to uncertainty. The engagement of science-management-public partnerships is critical to our risk-based approach for defining AFCs. Robust monitoring programs of forest management actions are also crucial to address uncertainty regarding species distributions and ecosystem processes. Development of regional indicators of response will also be essential to evaluate outcomes of management strategies. Our conceptual framework provides a starting point to move toward AFCs for forest management, illustrated with examples from fire and water management in the Southeastern United States. Our model is adaptive, incorporating evaluation and modification as new information becomes available and as social–ecological dynamics change. It expands on established principles of ecosystem management and best management practices (BMPs) and incorporates scenarios of future conditions. It also highlights the potential limits of existing institutional structures for defining AFCs and achieving them. In an uncertain future of rapid change and abrupt, unforeseen transitions, adjustments in management approaches will be necessary and some actions will fail. However, it is increasingly evident that the greatest risk is posed by continuing to implement strategies inconsistent with current understanding of our novel future.

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1. Introduction

The future is increasingly uncertain due to the rapid and compounded environmental, economic, and social changes that characterize the so-called Anthropocene, the geological epoch dominated by human modification of the Earth System (Steffen et al., 2007). High rates of landscape modification and species extinctions are unprecedented, and few, if any, ecosystems remain beyond the influence of human activity (e.g., Likens, 2001; Seastedt et al., 2008; Hobbs et al., 2009). Modern landscapes are social–ecological matrices of patches ranging from "natural/wild" to "intensive commodities-oriented" to "urban" (Hobbs et al., 2014). Novel ecosystems – the product of direct or indirect human activity – are increasingly prevalent and are often characterized by species assemblages and biophysical conditions with no analog in the past (Hobbs et al., 2006). The combined effects of changing climate and land-use, habitat fragmentation, species loss and introductions, and altered nutrient and hydrologic cycles at times exceed the ability of contemporary ecosystems to maintain their structure and function. Such disruptions can result in rapid unanticipated transitions and irreversible thresholds, which have significant social and ecological consequences (see Research Alliance Thresholds Database for examples, http://www.resalliance.org/index.php/thresholds_database). At the same time, there are societal expectations that ecosystems can and will be restored or rehabilitated to functional states, even while climate change, population growth, water diversion, the proliferation of chemicals and numerous other environmental changes impose additional burdens in ways that are not adequately understood (Naiman, 2013). Indeed, a primary goal of ecosystem management is to sustain ecosystem structure and function (Christensen et al., 1996). However, we contend that ongoing changes will in some cases exceed our ability to sustain existing ecosystems, and in such cases, a shift in focus to mitigation and adaptation for ecosystem services will be necessary and therefore produce "novel" ecosystems (e.g., Millar et al., 2007; Hobbs et al., 2014).

The rate and magnitude of environmental and socio-economic change expected over the next several decades will require innovative conservation and management perspectives, as these anthropogenic changes will alter (e.g., increase or decrease) the ability of ecosystems to provide ecosystem services (Hobbs et al., 2014, AIBS, http://actionbioscience.org/environment/esa.html). Ecosystem services are values associated with human well-being and are comprised of needs (i.e., *life sustaining*) and desires (i.e., *quality of life sustaining*), with both tightly tied to ecosystem structure and function. The capacity to maintain or enhance these services is a significant concern, as reductions hold negative and in some cases, potentially dire consequences for human well-being (e.g., www.millenniumassessment.org).

Although many of the concepts presented in this paper can be applied to a wide range of ecosystems, our focus is primarily on forests. Forests are an especially critical component of the modern landscape, providing diverse services such as wood and fiber, climate regulation, carbon storage, biodiversity support, and regulation of water yields and quality (FAO and JRC, 2012; Agrawal et al., 2013; Haddad et al., 2015). Current approaches to forest management in areas dominated by private land ownership are generally fragmented and uncoordinated. While management *goals* may be intended to ensure productivity, environmental quality, and conservation of biodiversity, management *approaches* are often limited in their ability to protect key ecosystem services given the rate and scale of biophysical and social-economic changes. We attribute this deficiency, at least in part, to an outdated view of ecosystems and the Earth System as static or inherently stable rather than dynamic (Pickett et al., 1992; Milly et al., 2008). New approaches focused on anticipating and guiding ecological responses to change are urgently needed to ensure ecosystem services for future generations. This need will likely require challenging some widely accepted principles of forest management and restoration, revising and expanding long-held guidelines and best management practices, and reappraisal of current regulations and laws. For example, focusing conservation efforts on public lands, local preserves, protection of rare species assemblages, and restoration of historic forest ecosystems may prove insufficient. Change is inevitable and might often be irreversible, so the provision of ecosystem services will depend, in part, on the development of novel ecosystems and the emergence of regionally coordinated forest conservation strategies and management

approaches that consider both public and private land (Dale et al., 2000; Seastedt et al., 2008; Hobbs et al., 2013; Rieman et al., 2015). Managing in the Anthropocene builds on many of the concepts of ecosystem management (Christensen et al., 1996), but incorporates long-term risk and uncertainty, and recognizes that future biophysical and social-economic conditions constrain desired management outcomes. As a starting point, we propose the following tenets:

1.1. Science-management-public partnerships as a foundation for conservation strategies

It has long been acknowledged that ecological dynamics operate at scales that are independent of ownership and political boundaries, so the effectiveness of conservation strategies will depend on the spatial arrangement of decisions at multiple scales (Christensen et al., 1996). However, this challenge presented by ecosystem management has yet to be resolved. Landscape assessments and management planning that cross these boundaries and incorporate stakeholder concerns and desires may provide the only means of anticipating the cumulative effects of multiple drivers operating at various scales. The emergence of novel ecosystems also creates challenges that span traditional management and technical boundaries (terrestrial, aquatic, game and non-game wildlife, endangered species, etc.), and a need for integrated management across these boundaries.

1.2. A risk-based approach is required to assess current conditions and develop conservation strategies in the face of future uncertainty

It is frequently difficult or impossible to provide rigorously quantified estimates of future uncertainty for decision-making (Carpenter et al., 2009). However, there is increasing certainty that future biophysical conditions will continue to diverge from the past. Further, risks can be quantified by using technical information to develop possible future scenarios (risk assessment) using best available information and accounting for uncertainty. Thus, effective conservation planning and policies must include the envelopes of potential future social-economic and biophysical conditions. We call these envelopes 'Achievable Future Conditions' AFC (see below), and their development moves beyond projections of future biophysical conditions. The AFCs should be designed within the boundaries of social acceptance and capacity for decision-making. AFCs are intended to facilitate social and ecological adaptive capacity to respond to uncertain, even unforeseen, conditions (e.g., Gunderson, 2000; Chapin et al., 2009). Applying a risk-based framework to alternative future scenarios can help identify potential problems and limitations, improving the likelihood that conservation objectives are achievable.

1.3. Achievable future conditions provide the foundation for prioritizing conservation and management actions

The primary objective of the AFC approach is to identify an envelope of *achievable* ecosystem service futures that incorporate current understanding of projected biophysical constraints, social-economic demands, and political realities of land ownership and development. From this foundation, risk management approaches can be used to consider and prioritize management actions that can mitigate undesirable conditions or provide adaptive responses to reduce adverse consequences of anticipated change (Yohe and Leichenko, 2010). Mitigation and adaptation distribute risk and provide diversity of response in the face of uncertainty. The process is inherently iterative, as new information becomes available, strategies for mitigation and adaptation can be revised or reprioritized as needed (Yohe and Leichenko, 2010).

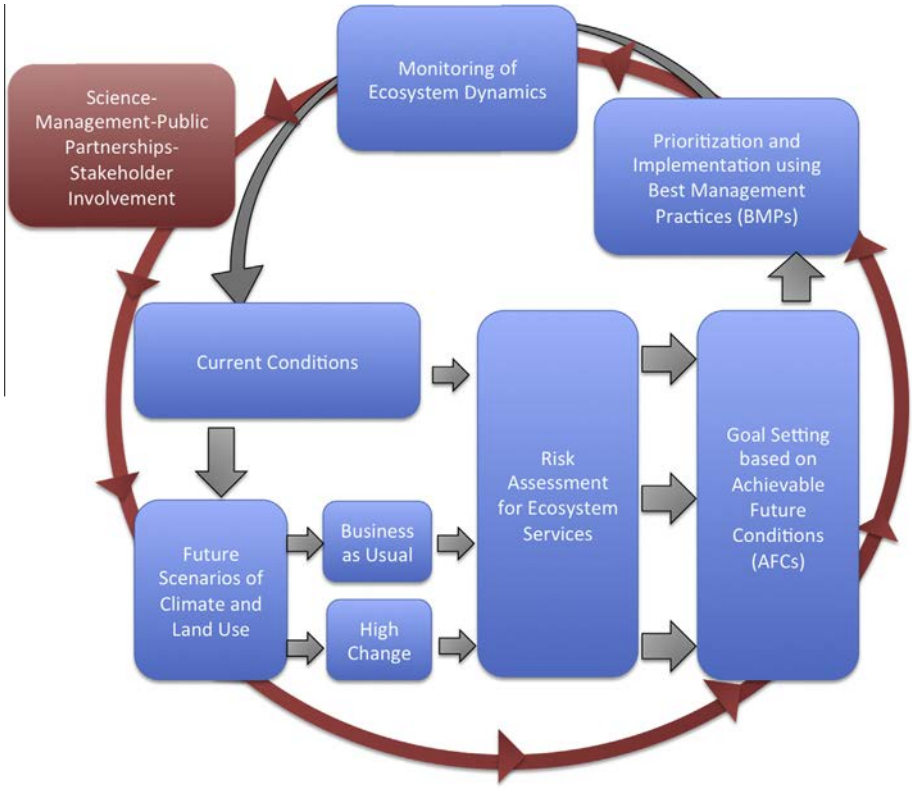

Fig. 1. Framework for conservation management for achievable future conditions, based on scenario and risk assessments. The red line and arrows show that stakeholder engagement is a continuous process, with an expectation of frequent engagement at all stages of conservation management. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

We apply a conceptual approach to conservation and management that examines projected environmental changes and ecological responses using the southeastern United States as a template. At the core of our approach is the delineation of AFCs that constrain outcomes within the boundaries set by future biophysical, social-economic and political scenarios. The dominance of diverse private landowners, abundance of forest area and forestry operations, and the dynamics of land uses make the southeastern U.S. challenging and illustrative example.

2. Developing conservation strategies

Our conceptual approach emphasizes that human population growth and development will be rapid, and as a result, biophysical and social-economic systems will continue to change in ways that are uncertain and may result in unprecedented or novel conditions. To ensure that forest ecosystems will continue to provide essential ecosystem services, planning will need to apply risk-based scenarios of future conditions. By necessity these strategies will be forward looking, interdisciplinary, large scale, and include mitigative (actions that reduce vulnerability) and adaptive (actions that cope with consequences) elements. In the following section, we outline critical elements that guide forest responses to rapid environmental and social-economic changes, and provide a conceptual model for the process we propose (Fig. 1). We begin by describing the critical elements required to convert our tenets into effective strategies, provide scenario-based projections for future biophysical and social conditions in the southeastern U.S., discuss the challenges that remain to implement our strategies, and use natural resource management case studies to demonstrate how strategies could be applied to anticipate and manage ecological responses.

2.1. Element 1: Science-management-public partnerships are the foundation for successful conservation strategies

Partnerships between researchers, managers and the public are foundational to identifying priorities for ecosystem services, where meaningful changes and/or scarcities of ecosystem services might arise. Without communication among scientists, managers, and stakeholders at appropriate spatial and temporal scales, the ability to understand ecosystem service tradeoffs will be limited. The ecosystem services that societies depend upon are often not well aligned with political boundaries, so coordination across jurisdictional borders is imperative. Because they are not traded and are by-products of management for other services (i.e., classic production externalities) most ecosystem services are unlikely to be produced at socially-optimal rates.

Stakeholder engagement remains an enormous challenge (e.g., Reed et al., 2009) especially in mixed ownership settings of the Southeast, where the autonomy of landowner choices creates difficulties for coordinating management strategies to achieve regional benefits. Large-scale conservation strategies will be most successful when stakeholders are engaged in developing priorities and understand how risks could be reduced and outcomes for local conditions enhanced through coordinated regional management approaches. Research into the potential benefits and costs of institutional mechanisms, including regulation, or landowner incentives, easements, and trading between landowners and ecosystem service consumers is needed to encourage the coordinated decision-making that can lead to better ecosystem service outcomes. Existing institutions may need to be modified to support coordinated activities within and among constituents to focus on landscape or regional outcomes.

2.2. Element 2: A risk-based approach is required to develop and manage conservation strategies in the face of future uncertainty

There is a growing consensus that future forest dynamics will be altered in ways we will not anticipate, resulting in outcomes that will challenge our understanding (Lindenmayer and Likens, 2010). Beyond ecological dynamics, our understanding of how to guide the responses of coupled social–ecological systems to sustain ecosystem services is a continually evolving area of research (Carpenter et al., 2009). Management planning and evaluation of ecosystem services are predominantly based on current and/or historic conditions, and goals are set to sustain the existing ecosystem structures and functions that support them (Christensen et al., 1996) However, given the pace of climate and land-use changes, historical conditions and ecological communities might become increasingly improbable (e.g., Maes et al., 2012). Current conditions are best viewed as baselines for projecting future scenarios and developing risk management strategies (Fig. 1), but this approach is seldom applied to projections for the provision of ecosystem services at relevant scales (Iverson et al., 2012; Wear and Greis, 2013). The uncertainty of any specific future projection may be high, but a scenario-based approach generates an envelope around the range of possible forest futures (Wear and Greis, 2013) from which risk can be assessed.

Assessing risk can help define outcomes and guide prioritization of potential problems and conservation strategies at large spatial scales. Risk can be described as the product of the frequency of a "hazardous" event (e.g., drought, storms, wildfire) and the consequences of the event (IPCC, 2014; Yohe and Leichenko, 2010), where "hazardous" is defined as an event whose frequency or severity is sufficient to cause undesirable outcomes. Characterizing risk requires consideration of the physical, environmental, and socioeconomic factors that determine the ability to resist or recover from a hazardous event, referred to as vulnerability. For example, high severity events (e.g., catastrophic flooding and landslides following extreme rainfall, forest wildfires during droughts) are always high risk, even when the likelihood of occurrence is low. Conversely, low severity events (e.g., tree growth reduction due to short-term drought) generally are low risk, even when the likelihood of occurrence is high. Risk management involves a variety of mitigative and adaptative options that reduce risk or vulnerability to acceptable levels (IPCC, 2007; Yohe and Leichenko, 2010). Examples using this bivariate approach in the ecological literature are rare (Iverson et al., 2012); however, implementing a risk-based framework for management simply requires that assessments are based on projected frequency and estimated consequences (Ojima et al., 2012). The utility of risk-based approaches has been recognized in forest management (e.g., Ojima et al., 2014).

Quantifying risks at fine spatial and temporal scales is likely to be highly uncertain, and in cases where there is insufficient knowledge, it may be necessary to apply qualitative rather than quantitative analyses (e.g., IPCC, 2007; Richter et al., 2011; Melillo et al., 2014). Both quantitative and qualitative information can be used to determine vulnerabilities as well as acceptable levels of risk in response to the changing biophysical and socioeconomic drivers.

2.3. Element 3. Achievable future conditions provide the foundation for prioritizing conservation and management actions

A cornerstone for conservation planning has been the identification of desired future conditions (DFC). Guided by historical endpoints and legal requirements like the Endangered Species Act in the US, DFCs typically include maintenance or restoration to a historical condition of existing species, communities or ecosystems. Although the DFC approach has proven useful, it fails to address

how rapidly changing biophysical and social ecological conditions could lead to novel conditions at a landscape or regional scale with unattainable historical endpoints. For example, a DFC for a restored freshwater wetland in coastal areas subject to sea level rise and saltwater intrusion is not feasible (Ardón et al., 2013). Instead, we propose more forward looking DFCs that grant equal or greater weight to projected future scenarios of biophysical and social-economic change (Prato, 2008); more appropriately termed "Achievable Future Conditions" (AFC). Defining AFCs requires a science-management-public partnership to develop social-economic needs and ecological possibilities within the risk-based and scenario building framework. The AFCs along with the identification of vulnerabilities and thresholds for acceptable risk can be used to determine the goals for management.

To be successful at large spatial scales, management actions to achieve AFCs will require widespread acceptance and application. The best analogy is the current implementation of best management practices (BMP) to minimize or mitigate impacts of timber harvesting and site preparation on water quality and site productivity (Prud'homme and Greis, 2002; Aust and Blinn, 2004). We build upon the concept of BMPs, but our discussion extends beyond the current scope and authority of most water-based BMP programs in the Southeast. Our suggestion is that the scope of BMPs should be broadened to include management practices that guide ecological change to attain AFCs. Broadening also implies an ongoing need for monitoring of BMPs (discussed below) for efficacy and the establishment of adaptive mechanisms for adjusting practices to reflect new and anticipated ecosystem conditions (e.g., Westgate et al., 2013). These new BMPs will need to include management options that address suites of ecosystem services and the potential tradeoffs among them, such as regional water yields or climate buffering (Bagley et al., 2014). For BMPs to be effective, they must be consistently implemented at regional scales. Further, it should be noted that while BMPs can be effective for guiding management toward AFCs, some problems may be resolved only through elimination of certain practices altogether, implying a need to structure compensation for landowners, perhaps through a payment/trading system within the relevant landscape.

Determination of BMP effectiveness requires monitoring to assess if desired outcomes are achieved. Hence, continuous monitoring of physical, biological, and social conditions will be critical to ensure that ecosystem service based management goals are attained. Monitoring will also allow approaches to be modified as priorities and evaluations for AFCs shift in response to changing values and/or updated future forecasts. This evaluation process will require substantial investment in "on-the-ground" environmental monitoring (Lindenmayer and Likens, 2010; Fekete et al., 2015). Remote sensing can serve some monitoring needs (Hargrove et al., 2009; Famiglietti et al., 2015), but it has limitations in the types of parameters that can be monitored (Kerr and Ostrovsky, 2003; Pettorelli et al., 2005; Arvor et al., 2013). Existing and emerging networks (e.g., Long Term Ecological Research (LTER), Long Term Agroecosystem Research (LTAR), National Ecological Observatory Network (NEON), Global Lake Ecological Observatory Network (GLEON)), and new remote sensing platforms (e.g., GRACE; Thomas et al., 2014; Reager et al., 2014) may also be useful in meeting some landscape and continental scale monitoring needs (Kao et al., 2012; McDowell, 2015; Schimel and Keller, 2015), but these networks remain largely untested with regards to detecting large scale changes in response to management actions. In addition, emerging networks do not directly address change in forests, particularly on private land. Although not focused on detecting responses to specific management actions, the United Forest Service Forest Inventory and Analysis Program has provided a critical database for assessing changes in forest condition (Woodall et al., 2011). The success of programs such as data on bird populations generated by the Audubon Society's Christmas Bird Count indicate the value of citizen science to meet some of these data needs (Dickinson et al., 2012). Citizen science programs also involve and raise awareness among stakeholders on conservation issues (Aceves-Bueno et al., 2015).

3. The Southeastern template

In this section, we characterize the social and ecological complexity of the Southeastern US and examine future projections for the region as a guide for developing scenario-based management actions. We define the Southeast as the 13 state region included in the Southern Forest Futures Assessment (Wear and Greis, 2013), extending from Virginia and Kentucky, south to Florida and west to Texas and Oklahoma.

3.1. A region of high social and ecological complexity

The Southeastern region is dominated by private land ownership and highly diverse forests, streams, rivers, and wetlands. Forests are the dominant land cover type, exceeding 65% in some states. One third of the private forest lands are under corporate ownership, and these lands are experiencing a rapid shift from ownership and management by integrated forest management companies to real-estate investment trusts (REITs) and timber investment management organizations (TIMOs). This shift toward REITs and TIMOs, which can divest rapidly, makes land ownership and management more difficult to forecast. The remaining two-thirds of the private forest land is owned by individuals or families, who typically lack long-term management plans (Butler and Wear, 2013). Developing and implementing management plans is further challenged by the size of landholdings, with an average family holding of less than 12 ha and trending smaller as land is passed down to multiple heirs (Butler and Wear, 2013).

The abundance of small, private landholdings creates a mosaic of developed areas adjacent to wildlands, with up to 22% of the Southeastern landscape included in wildland–urban interface (WUI) (Zhang et al., 2008). The greatest extent of WUI extends from Virginia to South Carolina, due to development outside mid-sized cities such as Columbia, SC (Zhang et al., 2008, see below). Expansion of WUI causes habitat fragmentation, biodiversity loss, and increases wildfire risk (Radeloff et al., 2005; Haddad et al., 2015).

With an abundance of terrestrial and aquatic habitats, the Southeast is noted for biological richness and high productivity (Burr and Mayden, 1992; Neves et al., 1997; Wear and Greis, 2013), providing diverse ecosystem services (www.teebweb.org). The Southeast leads the nation in timber production, a well-quantified ecosystem service (Wear and Greis, 2013). Payments for other forest-based services, including biodiversity (conservation easements and banks, wildlife viewing, and hunting), carbon offsets, and bundled services (e.g., Wetland Reserve Program that provides multiple services) totaled $1.7–$1.9 billion across the US from 2005 to 2007 (Mercer et al., 2011). The greatest payment rates occurred in the Southeast, particularly in Georgia, Florida and Louisiana (Mercer et al., 2011). Incentive programs for water resources are frequently provided to agricultural landowners, but payments for conservation measures or water quality trading programs are difficult to estimate on regional scales (Mercer et al., 2011).

The major river systems in the southeastern U.S. are influenced by multiple landowners and land uses, across diverse political jurisdictions, and have varying stakeholder interests. Municipal, industrial, and power generation are the predominant drivers of water management programs; however, most drinking water in

Table 1
Organizations currently working at landscape scale management coordination in the Southeast. These organizations can be key players in any new efforts in conservation management; however, a key distinction between existing efforts and the proposed framework is the focus on scenario-based planning, risk assessment, and landscape monitoring.

Organization	• Description • Goal • Guiding documents	• Public–private partnerships • Ownership focus	• Scenario-based future planning, risk assessment, or landscape monitoring?
Landscape Conservation Cooperatives (LCCs)	• Collaboration among FWS, USGS, state agencies. • Support integrated landscape conservation. • Spatially explicit Conservation Blueprint	• Yes • Primarily focused on public land	• Capacity to update conservation blueprint
Coalition of Prescribed Fire Councils	• Promotes prescribed fire management, supports landowners.	• Yes- • Primarily focused on private land	• No
America's Longleaf Initiative	• Federal, state and private industry collaboration across nine Southeastern states • Goal to more than double the area of longleaf pine in 15 years	• Yes • Focused across ownerships	• Local implementation teams
Greater Okefenokee Association of Landowners (GOAL	• Federal, state and private industry landowners in the Okefenokee region of GA, FL • Goal to respond to wildfire and reduce risk.	• Yes	• No
Flint Riverkeeper	• Stakeholder NGO • Goal to "restore and preserve the habitat, water quality and flow of the Flint River for the benefit of current and future generations and dependent wildlife"	• Yes	• No
ACF Stakeholders	• Stakeholder NGO • Goal to "achieve equitable water-sharing solutions among stakeholders that balance economic, ecological and social values, while ensuring sustainability for current and future generations"	• Yes	• No

the Southeast U.S. is derived from surface water sources (Caldwell et al., 2014). Water quality guidelines for large rivers are set through federal regulation and administered by states (i.e., Clean Water Act and NPDS permitting). Adjacent land use has a large effect on patterns of runoff and water quality. Non-point source runoff is often unregulated on private lands and Best Management Practices (BMP's) are voluntary and not always implemented, particularly for smaller streams (Shortle and Horan, 2013). Thus, individual landowner management decisions can have a large effect on runoff into stream reaches crossing the landowner's property. Many larger rivers in the Southeast are regulated by impoundments, whose operating authority and procedures are under control of the US Army Corps of Engineers or other agencies (Benke, 1990).

There is very little practical coordination of management actions at regional scales in the Southeast, despite the objective of "all lands" management approaches by some agencies and conservation groups (Table 1). Notable exceptions include a relatively new effort at range-wide restoration of longleaf pine. Current conservation and knowledge transfer programs are developed and implemented by organizations operating at different scales, sometimes with contrasting objectives and guiding principles. For example, public land management plans emphasize a spectrum of goals, from a broad complement of multiple uses (e.g., National Forests, US Forest Service (USFS)) to long-term protection (e.g., US National Park Service, or Wilderness Areas). Management plans for large private landowners (e.g., The Nature Conservancy, forest industry, land trusts, hunting preserves) typically focus on a narrower set of goals specific to institutional objectives. Goals and objectives may be narrower still for small non-industrial private forest landowners. Communication and knowledge transfer in the Southeast derives from a culture of independence and a strong southeastern legacy of private property rights, therefore, is often site-specific and based on local experience. This situation suggests that private landowner participation in coordinated, large-scale conservation efforts is a key challenge requiring innovative strategies to incentivize engagement (Wear and Greis, 2013).

3.2. Scenarios for the future

Projections for the Southeast indicate a future climate that is hotter with more variable precipitation (IPCC, 2014). Although precipitation projections are uncertain, greater evaporative loss from increased temperatures is expected to increase water stress (Liu et al., 2013a; Lockaby et al., 2013). The growing human population will rely on an increasingly urbanized landscape for ecosystem services, suggesting the risk of ecosystem service losses or disruptions is increasing (Wear and Greis, 2013). Climate models project steady warming across the region, though at varying rates and with subregional differences (IPCC, 2007; McNulty et al., 2013). Temperatures are projected to increase between 0.5 and >3.5 °C over the next 50 years, with greater increases in the western part of the region. Precipitation projections vary across models, suggesting potentially high uncertainty. As the climate changes, estimates indicate rapid land use-land cover (LULC) changes due to increasing population, with 12–17 million ha of new development by 2060 (Wear and Greis, 2013). Projected loss of rural land ranges from 4 to 9 million ha (6.5–13.1%) for forest, and 2–7 million ha (6–19%) for agriculture. Development and population growth are expected to be greatest at the periphery of urban centers, particularly in the Piedmont of the Southern Appalachian Mountains from Raleigh NC to Atlanta GA, and along the Atlantic and Gulf of Mexico coastal zones (Fig. 2, Wear and Greis, 2013; Terando et al., 2014). Scenarios with high economic growth result in more urbanization spread across a larger area (Fig. 2). Increasing urban populations are correlated with increasing real incomes; however, the economic conditions in depopulating rural areas are likely to decline, increasing the vulnerability of rural communities to extreme events and reduction of ecological services (Gaither et al., 2011). Differences in economic position and general perspectives on land use (rural-utilitarian vs. urban-esthetic as an example) suggest both economic and political tensions between stakeholders in contrasting segments of the Southeastern landscape.

Projections of forested LULC in the Southeast depend on population- and income-growth drivers as well as agricultural

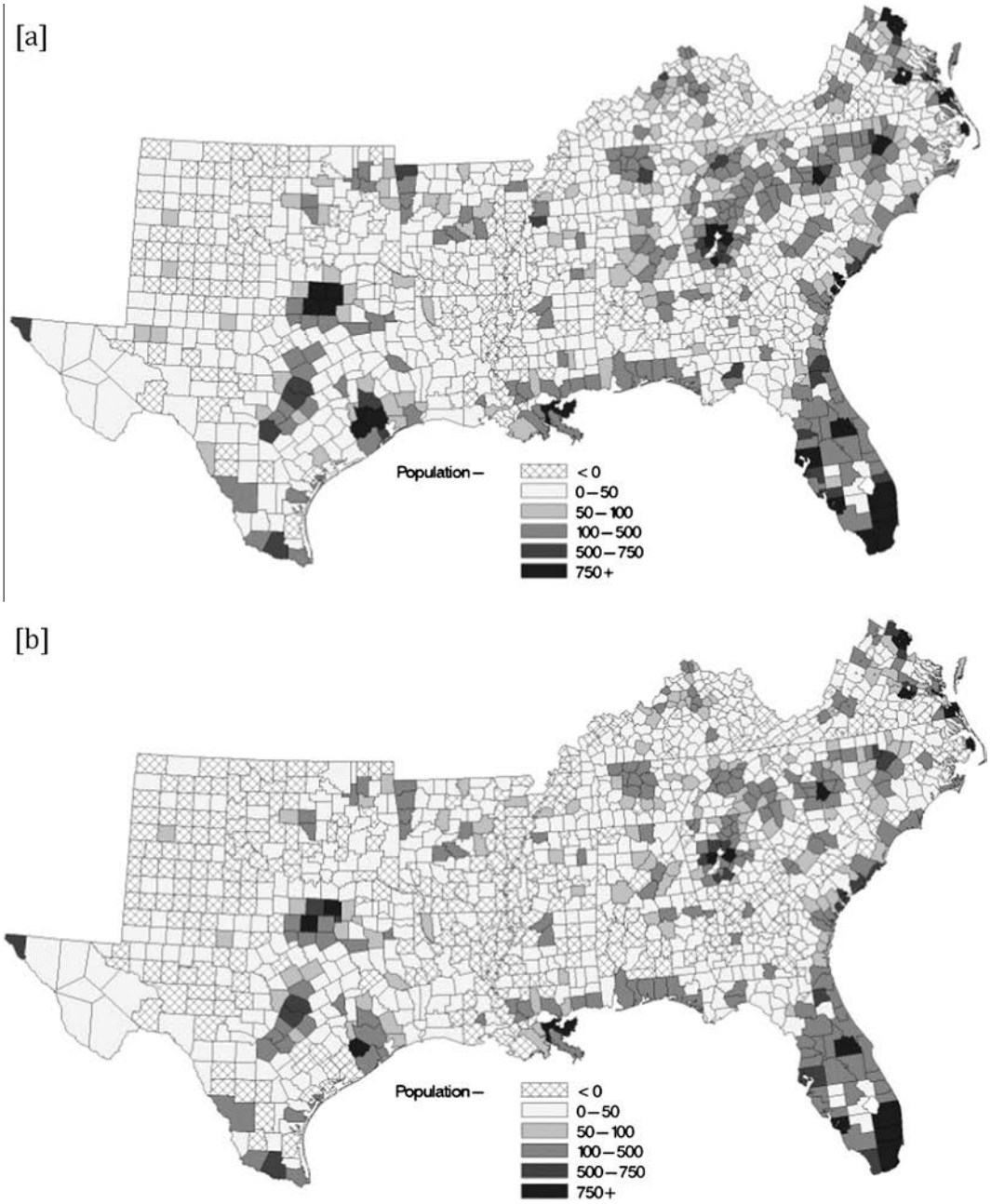

Fig. 2. Projection of population change in the Southeastern United States, 2010–2060, assuming the (a) A1B storyline of low population growth, high economic growth, high-energy use; and (b) B2 storyline of moderate growth and energy use from the 2010 Resources Planning Act (RPA) assessment. Note that counties in crosshatch have forecasted population losses. Adapted from Wear and Greis (2013).

and timber prices. Until recently, forest losses associated with development were offset by afforestation of marginal agricultural land. Recent projections indicate agricultural LULC will stabilize or even increase due to strong future agricultural markets (Wear and Greis, 2013). Forest losses are projected to be greatest from northern Georgia through North Carolina and into parts of Virginia (Fig. 3). Other areas of projected concentrated forest loss are the Atlantic Coast, along the Gulf of Mexico, and outside of Houston, TX. Forest type is particularly responsive to market conditions, with projections indicating an increasing trend in planted pines that began in the 1950s (Wear and Greis, 2013). Planted pines currently occupy almost 16 million ha, or 19% of the 83 million ha of forest in the Southeast (Wear and Greis, 2013; Klepzig et al., 2014). Projections of urbanization and timber prices suggest that by 2060, planted pines could increase by 11.3–27.3 million ha (34% of forest area), replacing much of the remaining natural (non-plantation) pine forest (Wear and Greis, 2013).

Scenarios of future climate and land use suggest forest conservation and management challenges will increase in the Southeast (Wear and Greis, 2013). The region faces a future of increasing fragmentation, loss of deciduous forest area in the Piedmont and Southern Appalachians, and potential conversion of natural forest to pine plantations in the Coastal Plain. These changes create uncertainty for important ecosystem services including biodiversity conservation, maintenance of hydrologic function, carbon sequestration, and climate buffering (Wear and Greis, 2013). Responding to change will require new thinking and strategies. For example, future forests may be subject to large-scale severe

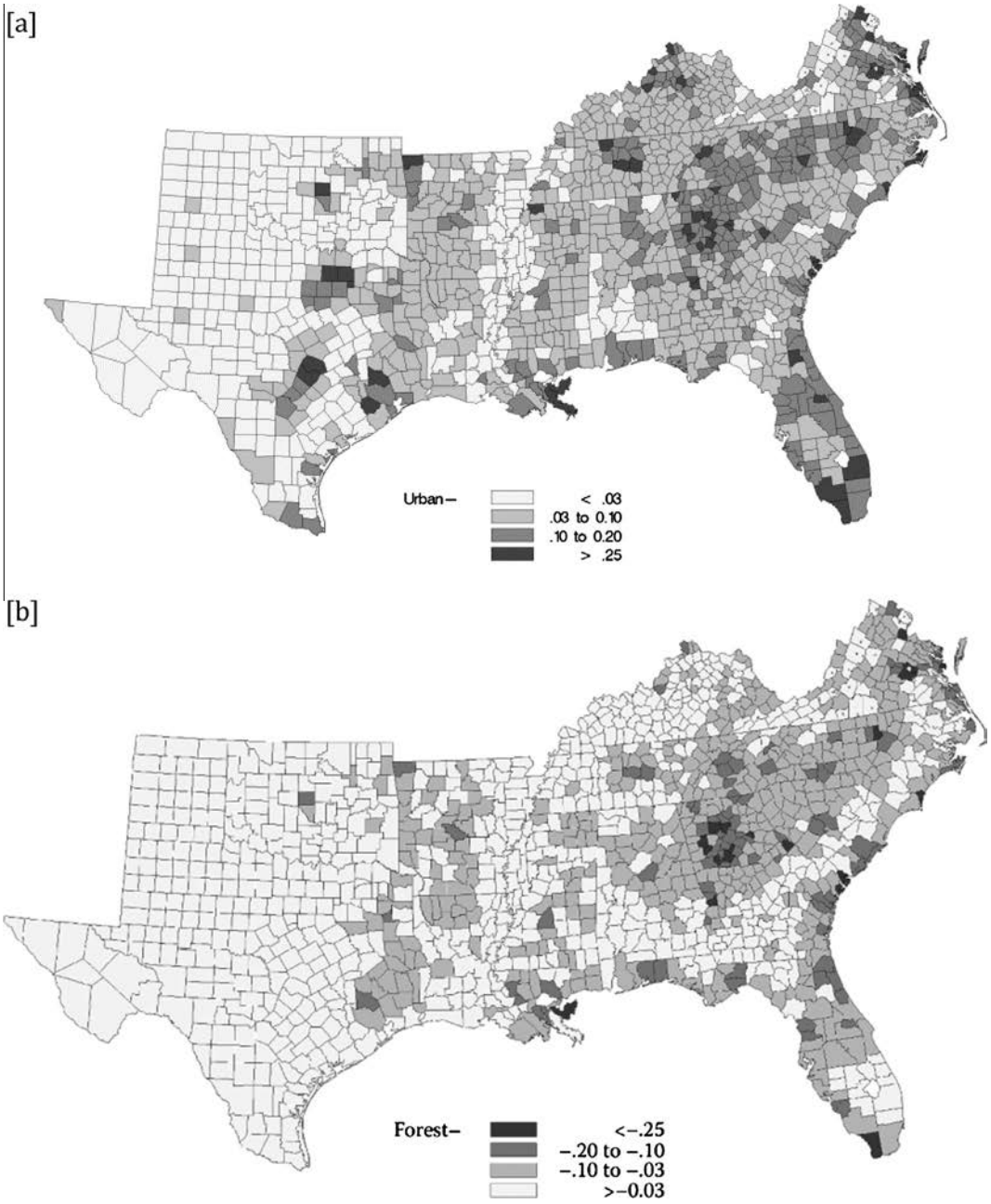

Fig. 3. Proportional change in land 1997–2060. (a) Percent change (increase) in urban land use. (b) Percent change (decrease) in forest land use. Adapted from Wear and Greis (2013).

wildfires, which have not been a primary concern in the past due to management including prescribed fire (Melvin, 2012). Predicted hotter conditions and thus, increased Keetch-Byram Drought Index (KBDI) seem likely to increase wildfire risk (Liu et al., 2013a, Fig. 4), causing concerns for the safety and health of an expanding population of the WUI and in depopulating rural areas. The confluence of population growth, urbanization, climate change, and loss/fragmentation of forests will also increase the water supply stress (water demand/water supply) between 10% and 100% throughout much of the Southeast (Fig. 5).

4. Applying the framework to conservation challenges in the Southeast US

In the following section, we use case studies focused on wildfire risk and water scarcity to explore how our framework could be used to develop and attain AFCs in a real-world setting. Characterizing current conditions is a logical starting point for the process. In the examples that follow, regional drought conditions caused undesirable events (wildfires and exceptionally low stream flows). These observed events, along with climate, land use, and population projections, have created concerns about future risk and exposed vulnerabilities to both stakeholder and management communities. We discuss these case studies in the context of our conceptual model and framework, not to criticize past efforts, but to show how regional conservation strategies and actions might be enhanced to address these concerns.

5. Fire management and the Georgia Bay Complex wildfires

The Southeast experiences more wildfire than any other region of the country, averaging 45,000 fires a year over the six year

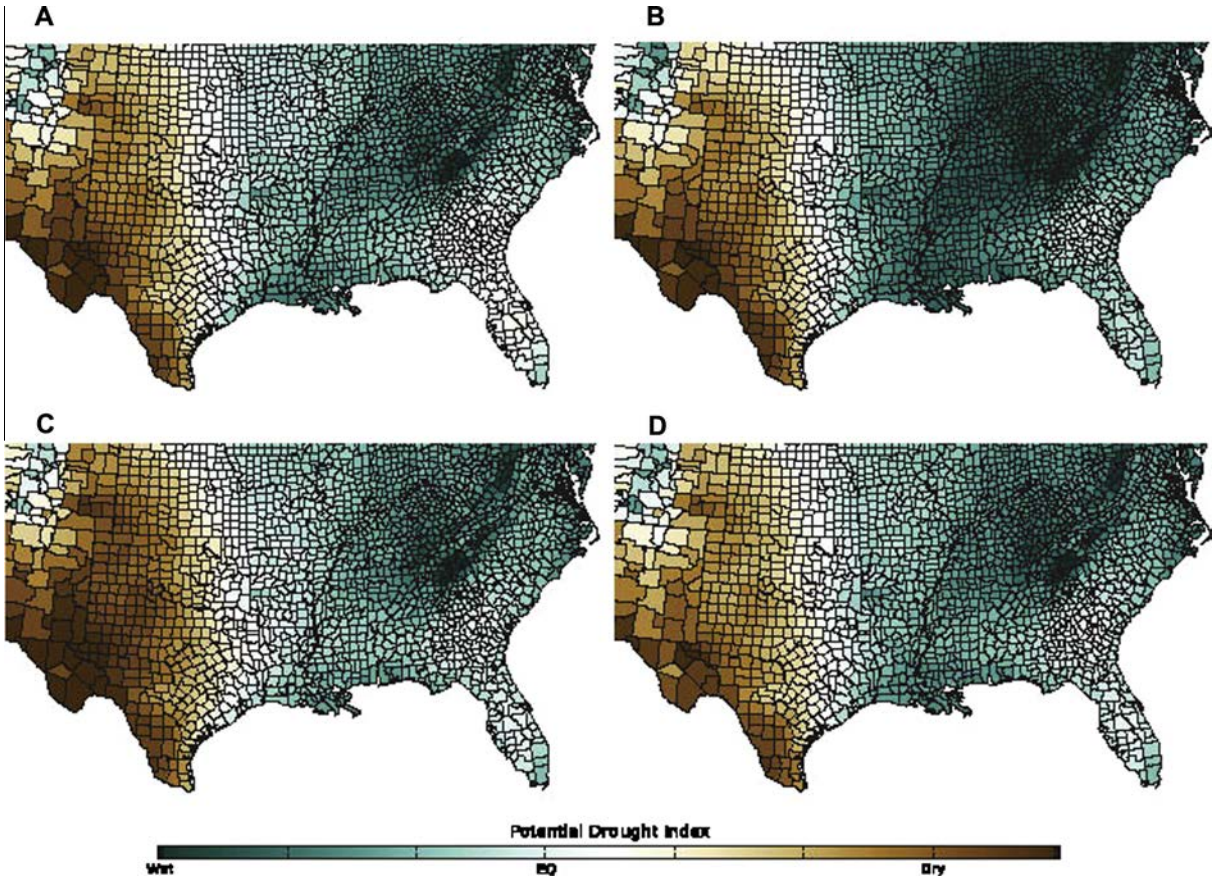

Fig. 4. Comparison of annual fire potential for future conditions (2060). Adapted from Stanturf and Goodrick (2013).

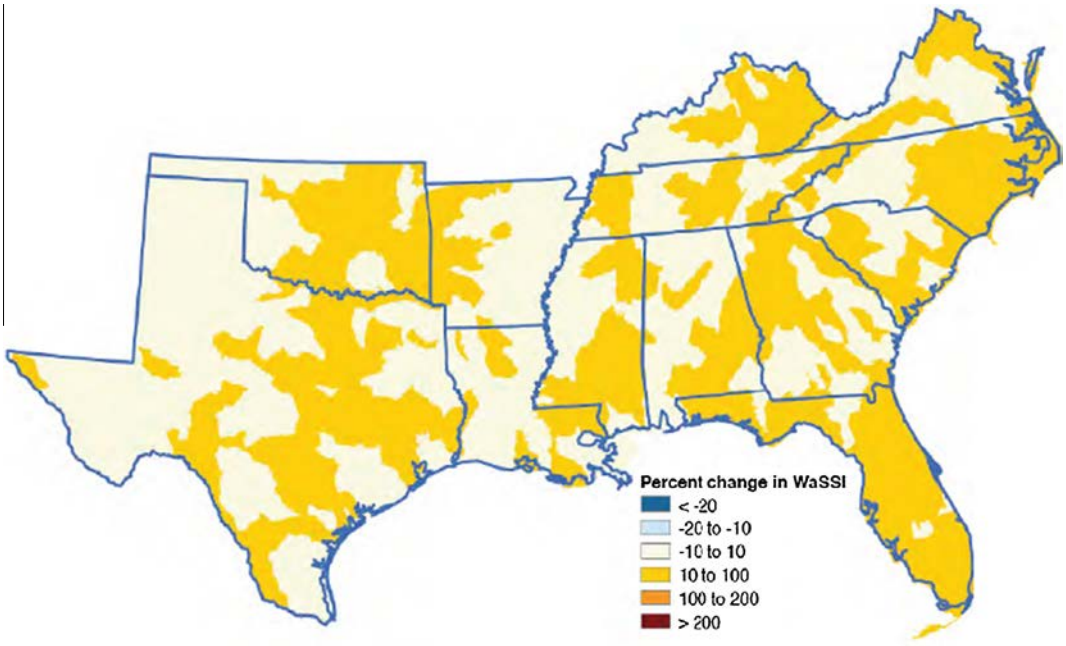

Fig. 5. Water supply stress index as identified by the Water Supply Stress Index (WaSSI) and calculated as the ratio of total demand and total water supply. Values represent change in 2050 from current conditions, based on population projections. Adapted from Wear and Greis (2013).

period from 1997 to 2003 (Mitchell et al., 2014). Typically, wildfires are not severe or large, as many Southeastern ecosystems are adapted to short fire return intervals that reduce fuel loads (Liu et al., 2013a). The Southeast also has a strong tradition of prescribed fire management, implementing more burns than other regions combined (Melvin, 2012). Prescribed burns have

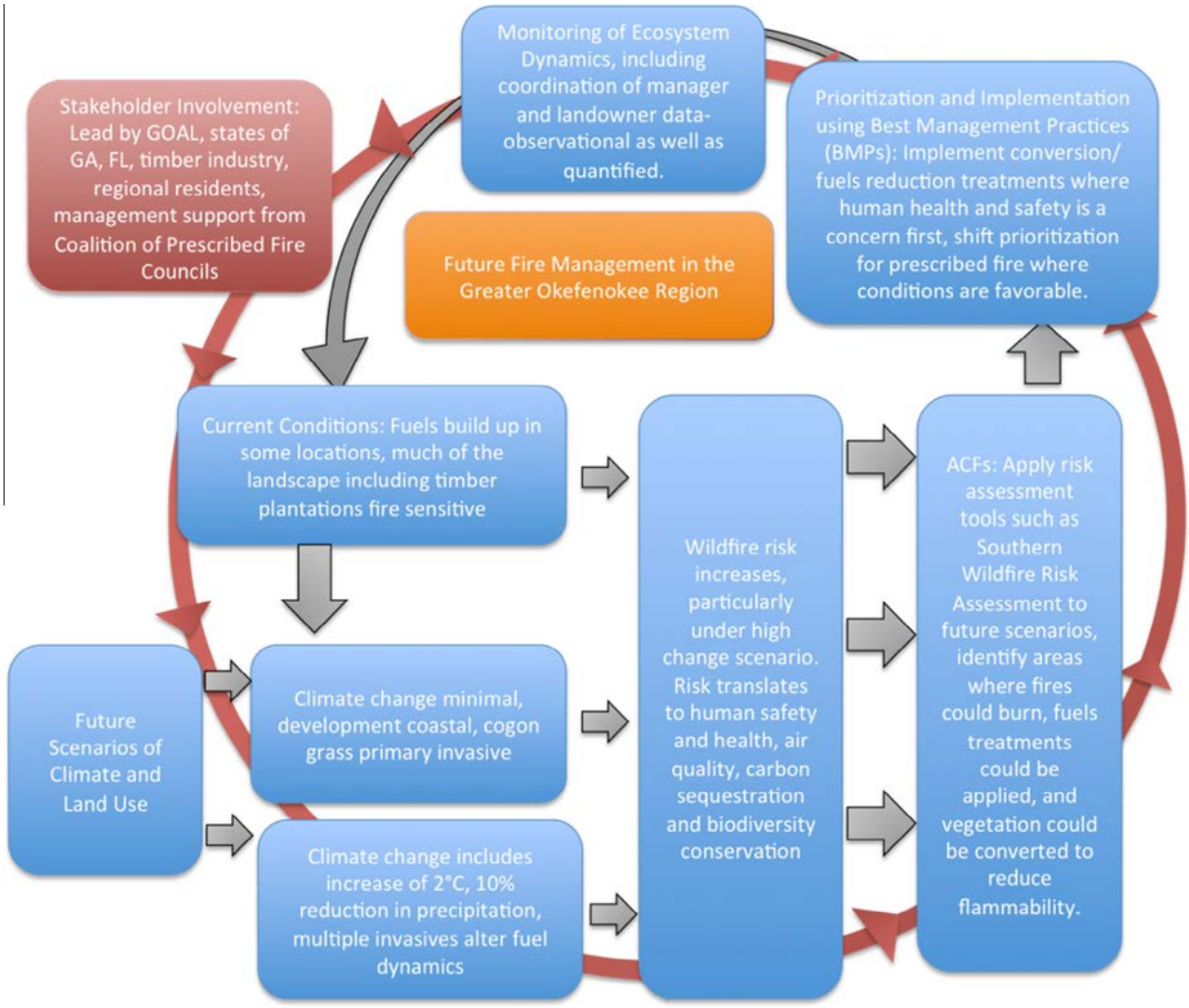

Fig. 6. Application of conservation framework to fire management in the Okefenokee region of Georgia and Florida. Increasingly quantitative data could be incorporated as it becomes available.

historically been implemented for management of fire-dependent biodiversity, including game species. Despite regular fuels management, Southeastern wildfires destroy more structures than in other regions and can lead to substantial timber losses for affected landowners (Stanturf and Goodrick, 2013). Projections for climate and land use in the Southeast indicate wildfires will be more frequent, intense, and occur over a fire season that will be extended by 1–3 months (Liu et al., 2013a, 2013b). These conditions suggest that wildfires will burn under more extreme conditions and fire frequency will increase in ecosystems with longer fire return intervals, including wetlands that historically burn only during significant droughts (Stanturf and Goodrick, 2013). The risk imposed by future wildfire will be compounded because of projected expansion of WUI and dispersed rural housing (Gaither et al., 2011; Liu et al., 2013a; Zhang et al., 2013). As wildfire risk increases, fuel management will be constrained because fewer days will meet criteria for prescribed fire implementation (Liu et al., 2013a; Mitchell et al., 2014). Recent wildfires in Georgia and Florida provide useful guidelines for potential future wildfire in the Southeastern U.S., and we use the Greater Okefenokee region as an application of our risk-based management framework (Fig. 6).

In 2007, the Georgia Bay Complex Fires (including the Sweat Farm Road Fire, Big Turnaround Fire Complex and the Bugaboo Scrub Fire) burned 242,800 ha across federal, state and private lands under severe drought conditions. The fires exhibited novel, erratic fire behavior, including flame lengths as high as 30 m (Edwards et al., 2013). Timber losses on private lands were in excess of $58 million (Mitchell et al., 2014). Smoke from the fires degraded air quality from Mississippi to North Carolina (Odman et al., 2007; Mitchell et al., 2014) and caused the temporary closure of two interstate highways in southern Georgia and northern Florida (I-10 and I-75) (Edwards et al., 2013). Adjacent communities were at particularly high risk, as many of them had been identified as economically vulnerable (Gaither et al., 2011). Only four years later, the 2011 Honey Prairie Complex Fire became the second largest in the Okefenokee region's history (125,129 ha) (http://www.fws.gov/okefenokee/HoneyPrairieArchive.html). The estimated historic return interval for such large fires was approximately 150 years (Yin, 1993).

5.1. Science-management-public partnerships as a foundation for conservation strategies

Collaboration and coordination are particularly critical for fire management in the Southeast because of fragmented land ownership, high WUI, economically vulnerable rural residents, and valuable, fire-sensitive timber resources. The Greater Okefenokee Association of Landowners (GOAL) was formed between US Fish

and Wildlife Service, state agencies from Georgia and Florida, and private landowners to coordinate fire responses, motivated by an active wildfire season in 1994 (Table 1). While it includes representatives from adjacent rural communities, GOAL would need to expand to facilitate wildfire mitigation for socio-economically vulnerable rural residents, including renters and their properties, to mitigate wildfire risk (Gaither et al., 2011). Further, GOAL does not currently include significant participation from the scientific research community, which could facilitate estimates of future fire risk, and move the organization toward more anticipatory strategies.

5.2. A risk-based approach to assess current conditions and develop conservation strategies in the face of future uncertainty

Risk assessment and prioritization are key goals of the Stewardship and Fireshed Assessment (SFA) process, which is increasingly implemented on public lands in the West (Bahro et al., 2007). The SFA uses a collaborative approach incorporating expert opinion, stakeholder involvement, and simulation modeling of management outcomes to prioritize landscape-scale wildfire risk reduction (Bahro et al., 2007). Landscapes are delineated into firesheds, usually several times the spatial extent of large, severe wildfires (Bahro et al., 2007; North et al., 2012). Delineations are based on historic fire data, fuels, LULC including WUI, expert opinion and stakeholder involvement. Coordinated strategies for fire and fuels management are developed and implemented using firesheds as management units (Bahro et al., 2007; North et al., 2012).

Application of fireshed management concepts in our conservation and management framework (Fig. 6) begins with a fuels assessment to establish baseline risk conditions. The Southern Wildfire Risk Assessment Tool is a potentially important resource to establish baseline conditions (www.southernwildfirerisk.com) that the scientific community could add to as a resource for GOAL. The current map indicates that the Okefenokee fireshed includes areas of moderate and high severity fire risk, primarily in the uplands. Established baselines can then be incorporated into future scenarios to determine the impacts of projected changes in climate and LULC. For example, the fire season in the Okefenokee region is expected to extend two months longer into the autumn by 2060 (Liu et al., 2013a), potentially increasing the intensity and thus severity of fire. This kind of future scenario projection is a key contribution science partners could provide to GOAL. Even if detailed assessments of future fire risk were to prove difficult to project, scenario based planning could use an experimental approach to test the robustness of management planning. For example, if fuel moisture were reduced by 10% across the fireshed, would it be important to consider revisions or shifts in areas at high risk?

Fire management in forests also will be challenged by increasingly fragmented landscapes with shifting and novel species compositions. Given that Jacksonville, FL is a major port and the center of urban development in the Okefenokee region, commercial transport and fragmentation from development will likely facilitate the spread of invasive species (Wear, 2013). The expansion of invasive species will alter fuel dynamics and in turn, fire behavior. Climatic envelope mapping can inform potential invasive species spread across regional scales for inclusion in potential fuels mapping (Sheppard et al., 2014). Using this approach, Bradley et al. (2010) indicate ongoing range expansion of cogongrass (*Imperata cylindrica*), which is highly flammable. Mapping potential scenarios of fuels will be a necessary part of fireshed risk mapping. This mapping could include both continuous updating of invasive occurrence and abundance from monitoring programs that might include a citizen science component, as well as more technical species occurrence projections provided by scientific partners.

5.3. Achievable future conditions provide the foundation for prioritizing conservation and management actions

In the context of fire management, current strategies that could be thought of as AFC-based BMPs include "Fire-wise" community guidelines (www.firewise.org), which provide recommendations for landowners to reduce impacts and spread of wildfires, such as using less flammable landscaping and building materials (Stephens et al., 2013). Such programs historically emphasized fire on a short term and case-by-case basis according to the landowner and community resources. Expanding BMPs to firesheds and incorporating preventative approaches that account for future climate and land use change would broaden protection of human health, safety, and economic interests across the Southeast, while maintaining biodiversity in fire-dependent communities (e.g. Gaither et al., 2011; Stanturf and Goodrick, 2013; Mitchell et al., 2014).

In firesheds where the risk and potential costs of wildfire are increasing, BMPs could guide the determination of acceptable levels of risk and effective management responses. Within such risked-based approaches, high severity crown fire might be acceptable where human health and safety are not jeopardized, such as across interior wilderness areas of Okefenokee NWR. This approach is consistent with fireshed management, where severe wildfires are allowed to burn in certain parts of the landscape to protect other areas (e.g., North et al., 2012; Stephens et al., 2013). Prioritizing areas of risk, including potential fire spread across the landscape will require mapping not only future biophysical conditions (including drought stress and fuel loads) but also social-economic conditions from future WUI and economically vulnerable areas (e.g. Gaither et al., 2011). Establishing priorities will require technical guidance from scientists in collaboration with land managers and owners. In high-risk areas of firesheds such as WUI areas, recommended BMPs might include physical fire breaks (Agee et al., 2000) or establishment of novel vegetation communities with lower fuel loads and flammability. Recommendations might also mean that traditional Southeastern landscaping with flammable materials, such as pine straw, will need to be reevaluated and/or moved away from structures. Landowners who wish to reduce the risk to timber investments might consider planting fire-resistant and resilient species such as longleaf pine (*Pinus palustris*) in place of loblolly (*Pinus taeda*) or slash pine (*Pinus elliotti*) and designing pine plantations to minimize risks to developed areas (Mitchell et al., 2014). These types of management approaches are unlikely to be perfectly aligned with the interests of landowners – for example, fire breaks may only provide benefits to adjacent areas – so they probably would not be adopted without some type of compensatory exchange between landowners and beneficiaries, or policy intervention, most likely at the state level.

Prescribed fire, along with wildland fire acceptance, or allowing wildfires in wilderness areas to burn, will continue to be important BMPs for wildfire and biodiversity management in the Southeast (Kirkman et al., 2001; Melvin, 2012; Stanturf and Goodrick, 2013; Mitchell et al., 2014). Prescribed fire is often viewed as the most cost effective and ecologically beneficial fuels management strategy (Melvin, 2012). However, managers should expect that opportunities to use fire would occur less frequently under future conditions, because of altered climate and concerns about smoke management and air quality (Bhoi et al., 2009; Mitchell et al., 2014). Public acceptance of prescribed fire will also be necessary, and outreach from scientists and managers can provide the public with information as a key part of the decision making process as to where and when prescribed fire will be acceptable. Further, it will not be sufficient to reduce fire risk on public land alone. Scientists and managers can provide private landowners with information and technical support to implement prescribed burning. Already, states including Georgia and Florida provide assistance to

landowners who seek it; however, states might need to move to a more proactive recruitment process as wildfire risk increases.

Where prescribed fire is not feasible, implementation of firebreaks and/or conversion to less flammable vegetation, particularly in the WUI, will be important BMPs to reduce future wildfire risk across property boundaries. Even with aggressive management of fuels, many areas are likely to experience high severity fires that may substantially alter ecosystem structure and function. During the Georgia Bay Complex Fires, some recently prescribed burned (<5 years) stands in the Osceola National Forest experienced high severity burns due to extremely dry weather and high-risk fire conditions (Fire Behavior Assessment Team 2007). Further, GOAL sought to create a wildfire buffer around the Okefenokee (goalpartners.org) following the 2007 fire season; so, the spread of the 2011 fires suggests that management will need adaptive and increasingly aggressive approaches to contain fire under projected warmer and drier climatic conditions (Stanturf and Goodrick, 2013).

For the Okefenokee fireshed, determining acceptable levels of risk for management prioritization will require coordination among state and federal agencies, GOAL, Prescribed Fire Councils, and private landowners. Additional collaboration with climate and fire scientists will be required to provide the technical information necessary to support informed risk assessment and decision making. Risk assessment will most likely include multiple priorities and therefore tradeoffs across the fireshed. Public agencies will likely prioritize risks to human health and safety, whereas private landowners might prioritize protecting high-value timber resources. Such conflicting priorities could increase the overall fireshed risk. What is needed is an institutional structure to incorporate scenario-based forecasts across ownership boundaries within firesheds, to establish consensus-based goals for managing fire risk at a landscape scale, and regulatory or compensatory mechanisms for facilitating private management activities that support broader social benefits.

6. Water resources in the Flint River Basin

Human population growth, urbanization, and climate change are increasing regional water demands and depleting water sources in many areas of the Southeast (Sun et al., 2008; Carlisle et al., 2011; Rugel et al., 2012; Famiglietti and Rodell, 2013). The ability of water resources to meet higher demand is limited, as the Southeast is characterized by small river basins, modest groundwater resources, and relatively small, shallow storage reservoirs that are designed primarily for flood control (Sun et al., 2013). In areas of high population growth such as the Piedmont, which includes Atlanta GA, Charlotte NC, and Raleigh-Durham NC, regional heat-island effects (e.g., www.epa.gov/heatisland/) formed by the combination of rapid urban/suburban expansion and climate change may further stress water resources (Terando et al., 2014). Increased municipal water demands will affect aquatic biodiversity, as the development of water resources is associated with a decrease in faunal richness and regional homogenization of fauna (e.g., Moyle and Mount, 2007).

We use the Flint River in Georgia to illustrate how our risk-based conservation framework could address water resource and aquatic biodiversity conservation concerns in the region. Draining over two million ha, the Flint River watershed is approximately 50% forested, 40% agriculture, and 5% urban. Hence, the region is a matrix of mixed land uses that interact in complex ways. Originating in the metro Atlanta area, the Flint flows southward across the Piedmont and Coastal Plain to its confluence with the Chattahoochee River, forming the Apalachicola River, which flows from southwestern Georgia to the Florida panhandle. Water is withdrawn primarily for power generation and municipal supply in the upper basin near Atlanta, and for agricultural irrigation (e.g., row crops, pecan orchards) in the lower portions of the basin. Water use has been expanding rapidly throughout the basin since the 1970s and growing season stream flows are declining, particularly during droughts (Rugel et al., 2012; Emanuel and Rogers, 2012). During this period, there was no significant trend in annual or seasonal precipitation (Rugel et al., 2012; Emanuel and Rogers, 2012). Between 2010 and 2050, the population of the upper basin is expected to increase 63%. In the lower Flint, water demand increased rapidly with the adoption and expansion of crop irrigation during the 1970s and 80s (Couch et al., 1996). Total water use in the Flint River Basin is projected to increase from 4.3 million m^3 per day (Mm^3 D) in 2010 to 4.9 Mm^3 D in 2050 (Lower Flint Ochlockonee Watershed Council, 2011; Upper Flint Watershed Council, 2011). With a future of increasing population, land use changes, temperatures, and uncertain precipitation, water resource issues will become more critical. Allocation of water from the Apalachicola–Chattahoochee–Flint River Basin has been a long-term source of contention between Georgia, Florida and Alabama (see acfstakeholders.org). However, the development of water management guidelines for the Flint River, and many other southeastern rivers, is hindered by the lack of systematic assessment of hydrologic change across mixed land uses. Particularly lacking is information about changes in water yield or water balance in watersheds undergoing development or land conversion (e.g., Sun et al., 2008). Information concerning biological responses, essential for developing water management guidelines, is also not well developed for southeastern rivers. In the following sections, we explore the use of our conservation framework to address water resource issues in the Flint basin (Fig. 7).

6.1. Science-management-public partnerships as a foundation for conservation strategies

Stakeholder concerns about the availability of clean, fresh water in the Flint River basin led to the formation of the Flint Riverkeeper (FRK) in 2008 (flintriverkeeper.org). Major stakeholders within the basin include municipalities, agriculture, canoe liveries, and anglers. Concern arose from an apparent increase in frequency and duration of droughts and an increase in the frequency and duration of extreme low flows. Collectively, those observations suggested that the ability to meet water demands, recreational needs, and support instream biota were increasingly uncertain (Emanuel and Rogers, 2012). There was also recognition that addressing water resource problems in the Flint River required a broad integrative approach and coordinated efforts among diverse stakeholder interests.

6.2. A risk-based approach to assess current conditions and develop conservation strategies in the face of future uncertainty

Future water resources and aquatic biodiversity depend on sustained streamflow, but this requirement encompasses the magnitude, duration, frequency, timing, and rate of change in both common and uncommon events (i.e., low flows, base flows, and flood pulses) (Olden and Poff, 2003; Poff et al., 2010). Characterizing flow alteration is difficult because of the complex nature of the hydrologic regimes and because information concerning biotic responses to altered stream flows is site specific and/or often lacking. The Sustainable Boundary Approach (SBA) is a method of stream flow assessment that simplifies data needs and reduces complexity. Using a combination of stakeholder consensus, expert-opinion and evidence, an acceptable range of daily stream flow is developed based on historical records (Richter, 2009). This approach incorporates societal values, technical expertise, along with flow augmentation and reduction, in water management planning. The SBA approach has been further refined with the

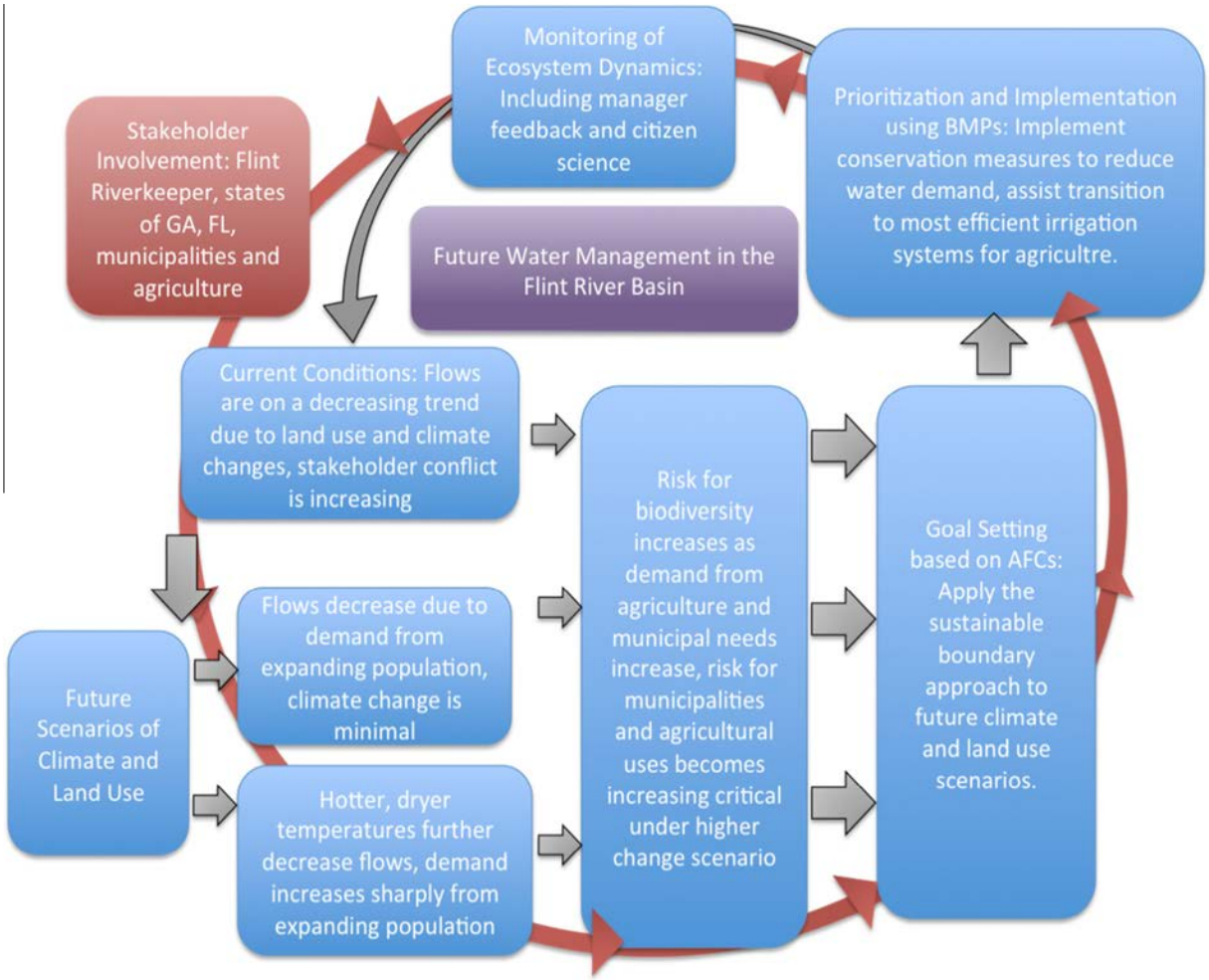

Fig. 7. Application of conservation framework to the Flint River watershed in Georgia.

introduction of a 'presumptive standard' for environmental flow protection (Richter et al., 2011). Presumptive standards are risk-based guidelines using case studies and real world experience with the SBA template as a conceptual model.

We used the SBA and Presumptive Standard approach to assess hydrologic alteration and develop AFCs for water resources in the Flint River (Richter et al., 2011). The main stem of the Flint River has a number of US Geological Survey (USGS) gauging stations that provided data (i.e., daily flow) for the SBA analysis. Based on demographic and land use data, we used 1975 as a breakpoint for pre- and post-hydrologic alteration (e.g., Rugel et al., 2012; Emanuel and Rogers, 2012); i.e., data from WY 1940–1974 were used to estimate 'pre-alteration' conditions, and data from 1975 to 2012 represented 'altered' flows. We calculated median average daily flow for each day of the year, and upper and lower boundaries for SBA were calculated as the median daily flow ±20% (i.e., presumptive standard; Richter et al., 2011).

Median daily flow for the altered flow period shows substantial departure from the pre-alteration period (Fig. 8). From April through mid-October for WY 1974–2012, median daily flows are often at or below the lower SBA boundary. Even during winter, when the WY 1974–2012 flow generally resided within the SBA band, median daily flow seldom equaled or exceeded the pre-alteration median value. This analysis suggests that substantial hydrologic alteration has already occurred in the Flint River and is reflected in lower flows, particularly during late spring and summer.

Under climate change scenarios, warmer temperatures, along with variable rainfall, will result in a continuing trend of hydrologic alteration. Human population growth would create additional stress on water resources, exacerbating climate effects. Reduced summer stream flow and increased stream temperature have negative implications for ecological communities in the river, such as freshwater mussels (Golladay et al., 2004; Emanuel and Rogers, 2012), native crayfish (Sargent et al., 2011), and fish (van den Avyle and Evans, 1990; Freeman et al., 2012; Emanuel and Rogers, 2012). In addition to ecological effects, low flows would reduce the seasonal volume of water available to receive permitted discharges. Increased discharge concentration, along with ecological changes may alter river assimilative capacity and increase water treatment costs for downstream users.

The range of potential outcomes emerging from this analysis could be a starting point for developing AFCs for the Flint River. This scenario building and risk assessment process would provide an evaluation of whether costs (economic, social, and ecological) associated with departures from the SBA are acceptable, followed by the development of achievable strategies for addressing flow deficits. Sufficient technical information exists to guide management responses and a monitoring network is in place to provide feedback. The challenge lies with developing an institutional framework for engaging diverse social and economic interests in a process leading to AFCs defined by environmental stream flows that could address tradeoffs between ecological structure and function of the Flint River and the provision of various human

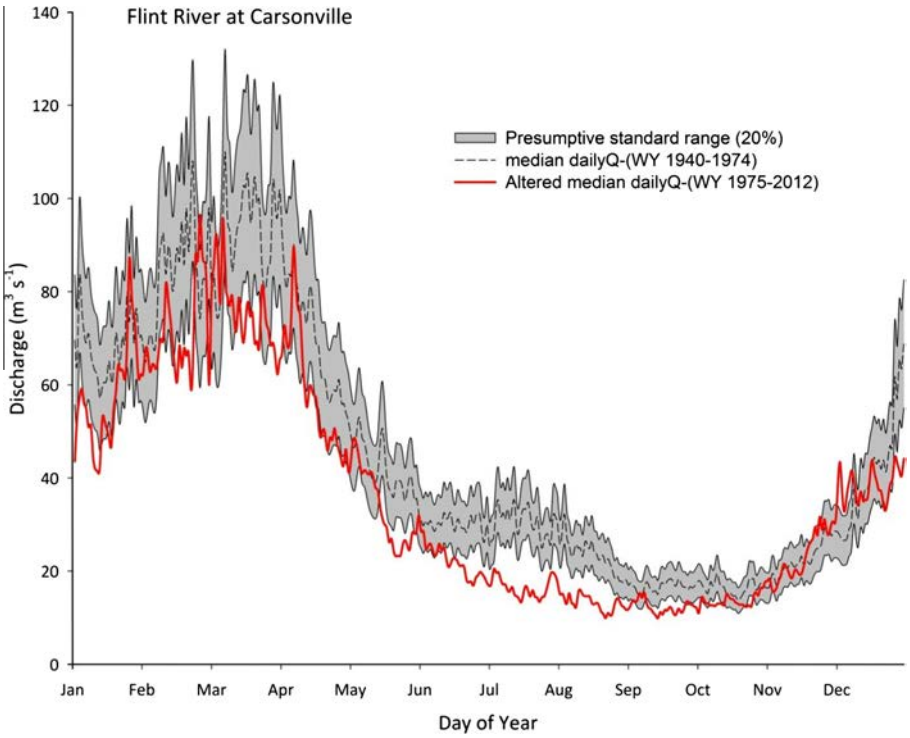

Fig. 8. Calculation of Presumptive Standard using the SBA from Richter et al. (2011). Data obtained from the Carsonville, GA gauge (USGS 02347500). Grey range represents median daily flow ±20%. Solid red line represents 'altered' flow conditions based on median daily flows calculated from WY 1975 to 2012.

water uses. Key constituents include water utilities, industrial users, recreational users, and the agricultural sector. The SBA approach is inherently adaptive, once goals for AFCs are set, data collection and assessment can proceed and management activities adjusted based on river- specific knowledge and economic/ ecological interests.

6.3. Achievable future conditions provide the foundation for prioritizing conservation and management actions

Concerns over future water resources motivate immediate consideration of strategies to address vulnerabilities associated with extended periods of low flows in the Flint River. Increasing the availability of storage reservoirs is expensive and may be geologically challenging (Sun et al., 2013); therefore, efforts at reducing consumption might be emphasized over the short term. A number of approaches have been suggested for the Flint River (e.g., Emanuel and Rogers, 2012; Emanuel, 2014), such as better early recognition of drought conditions and faster responses in reducing *per capita* water use in response to anticipated shortages. The Flint River is already part of a National Oceanographic and Atmospheric Administration (NOAA) test program for regional drought early warning efforts (www.drought.gov/drought/content/regional-programs). Inclusion within this program indicates that monitoring capability is already in place, and water utilities could use NOAA seasonal outlooks to more aggressively impose water conservation measures before water storage reaches critical thresholds. Changing landscaping practices and improving the efficiency of lawn irrigation systems can also reduce water demand during seasonal dry periods. In municipal systems user demand approximately doubles during the growing season, largely due to landscape watering (Emanuel and Rogers, 2012). Another approach is improving water distribution and use efficiency through repair of leaks in distribution systems and incentives for improving end-user efficiencies. Several municipalities in the upper Flint have initiated programs to improve efficiency of distribution and household water use (e.g., Emanuel and Rogers, 2012). With the exception of infrastructure repair and improvement, many of the changes described above require water conservation encouraged either through public information campaigns or with progressive water pricing mechanisms.

Longer term responses to potential water shortage could involve systematic examination of water yield/balance at basin scales and adapting BMPs to enhance yields and reduce water loss. In urban areas, BMPs might include converting storm drain networks, which accelerate runoff, to 'green' infrastructure, which encourages water storage and infiltration (e.g. Jaffe, 2011). In rural areas of the lower Flint River basin, conversion of native forests to agriculture has accelerated water stress by replacing water conservative vegetation with water demanding crops (Ford et al., 2008; Brantley unpublished data). Intensive agriculture also requires irrigation during periods of low rainfall, putting additional seasonal stresses on water resources (Perry and Yager, 2011). In the lower Flint, improvements in agricultural irrigation efficiency are being adopted to reduce seasonal demand (Perry and Yager, 2011). However, there are limits to water savings that can be acquired through improved efficiency. Under some futures, retirement of some agricultural lands might provide the only feasible path to AFCs, suggesting some institutional structure to compensate landowners for foregone revenue (similar perhaps to the Conservation Reserve Program, but targeted to important components of this specific watershed).

At landscape scales, economically viable alternatives to water intensive irrigated agriculture need to be developed and evaluated. Adoption could be encouraged through conservation easements and incentives programs for reforestation. Programs have been initiated within the southeastern US and include USDA Regional Conservation Partnership Programs (RCPP, http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/programs/farmbill/rcpp/) and the America's Longleaf Restoration Initiative (ALRI, http://www.

americaslongleaf.org/). While these programs promote land conversion, their potential to affect regional water yield/balance and contribute to restoration of river flows has not been assessed. This is a critical information need in developing achievable future conditions for water resources in the southeastern US.

7. Summary and conclusions

In the context of using AFCs to guide conservation and management strategies, establishing current conditions, developing scenarios for the future, and assessing risk is a largely *technical* process; while determining AFCs is a collaborative *social* process between stakeholders and technical experts. Technical information can provide insight into future biophysical envelopes; however, stakeholder involvement is essential for determining social and policy goals. This process emphasizes the need for strategies or forest management actions that are realistic given ecological and social-economic constraints, have specific timelines for implementation, and have measurable outcomes to evaluate success (e.g., Maxwell et al., 2015). Critical to this process is developing a new suite of mitigative and adaptive BMP's that are applied at a regional scale. Although uncertainty is inherent in projections of future climate and land use, there is abundant evidence that rapid changes are altering forest ecosystem function and species distributions, and some of these changes are irreversible, resulting in novel ecosystems. Management approaches based solely on historic or current conditions are limited in their ability to sustain ecosystem services critical to an expanding human population. Understanding of the ecological past and current dynamics is necessary to establish baseline conditions, but not sufficient to guide decision-making. Instead, these baselines must be applied in scenario-based forecasting to generate a range of possible futures that can then be analyzed in a risk-based approach. These futures should include a diversity of land uses. Strategies for management must be broadened, as public lands are not sufficient to sustain ecosystem services for society in regions dominated by private land, such as the southeastern U.S. The notion of public versus private management responsibility is particularly untenable for rivers, which cross large sections of landscape under a variety of uses. Scientists, managers, policymakers, and stakeholder are therefore challenged to collaborate across political boundaries and spatial and temporal scales. New approaches are needed that anticipate and guide ecosystems in directions that mitigate undesirable outcomes. This process will necessarily be adaptive as baseline conditions change, future scenario projections are updated, and societal needs shift with a growing and diversifying population. The scientific community is tasked with advancing understanding of ecological dynamics and their contribution to ecosystem services, improving simulation models, and communicating their findings with managers and stakeholders. Together with scientists, managers will have to design and implement monitoring programs that are crucial to appropriate management responses. The critical need to provide an early warning of unexpected changes suggests that it will be necessary to expand monitoring networks, including approaches such as citizen science initiatives (Conrad and Hilchey, 2011; Aceves-Bueno et al., 2015).

Our framework provides a starting point to move towards AFCs, illustrated with examples from forest wildfire and water management in the southeastern U.S. It builds on work including the forest management resistance- resilience- transition framework of Millar et al. (2007), which could be incorporated as part of the process to determine ACFs. It is also consistent with environmental flows methodology, often proposed as an approach for developing water allocation strategies (Poff and Zimmerman, 2010). We anticipate that points along the cycle will be updated, and our framework will be adapted as information becomes available and dynamics change. In an uncertain future of rapid change and abrupt, unforeseen transitions, adjustments in management approaches will be necessary and some actions will fail. However, it is increasingly evident that the greatest risk is posed by continuing to implement strategies inconsistent with and not informed by current understanding of our novel future.

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References

Aceves-Bueno, E., Adeleye, A.S., Bradley, D., Brandt, W.T., Callery, P., Feraud, M., Garner, K.L., Gentry, R., Huang, Y., McCullough, I., Pearlman, I., Sutherland, S.A., Wilkinson, W., Yang, Y., Zink, T., Anderson, S.E., Tague, C., 2015. Citizen science as an approach for overcoming insufficient monitoring and inadequate stakeholder buy-in in adaptive management: criteria and evidence. Ecosystems 18, 493–506.

Agee, J.L., Bahro, B., Finney, M.A., Omi, P.N., Sapsis, D.B., Skinner, C.N., van Wagtendonk, J.W., Weatherspoon, C.P., 2000. The use of shaded fuelbreaks in landscape fire management. Forest Ecol. Manag. 127, 55–66.

Agrawal, A., Cashore, B., Hardin, R., Shepherd, G., Benson, C., Miller, D., 2013. Economic contributions of forests. In: Background Paper 1, United Nations Forum on Forests 10th Session. Instanbul.

Ardón, M., Morse, J., Colman, B., Bernhardt, E.S., 2013. Drought-induced saltwater incursion leads to increased wetland nitrogen export. Glob. Change Biol. 19, 2976–2985. http://dx.doi.org/10.1111/gcb.12287.

Arvor, D., Durieux, L., Andres, S., LaPorte, M.-A., 2013. Advances in geographic object-based image analysis with ontologies: a review of main contributions and limitations from a remote sensing perspective. ISPRS J. Photogramm. 82, 125–137.

Aust, W.M., Blinn, C.R., 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: and overview of water quality and productivity research during the past 20 years (1982–2002). Water Air Soil Poll.: Focus 4, 5–36.

Bagley, J.E., Desai, A.R., Harding, K.J., Snyder, P.K., Foley, J.A., 2014. Drought and deforestation: has land cover change influenced recent precipitation extremes in the Amazon? J. Climate 27. http://dx.doi.org/10.1175/JCLI-D-12-00369.1.

Bahro, B., Barber, K.H., Sherlock, J.W., Yasuda, D.A., 2007. Stewardship and fireshed assessment: a process for designing a landscape fuel treatment strategy. In: USDA Forest Service GTR PSW-203.

Benke, A.C., 1990. A perspective on America's vanishing streams. J. North Am. Benthol. Soc. 9, 77–88.

Bhoi, S., Qu, J.J., Dasgupta, S., 2009. Multi-sensor study of aerosols from 2007 Okefenokee forest fire. J. Appl. Remote Sens. 3 (031501). http://dx.doi.org/10.1117/1.3078070.

Bradley, B.A., Wilcove, D.S., Oppenheimer, M., 2010. Climate change increases risk of plant invasion in the Eastern United States. Biol. Invasions 12, 1855–1872.

Burr, B.M., Mayden, R.L., 1992. Phylogenetics and North American freshwater fishes. In: Mayden, R.W. (Ed.), Systematics, Historical Ecology, and North American Freshwater Fishes. Stanford University Press, Stanford, pp. 18–75.

Butler, B.J., Wear, D.N., 2013. Forest Ownership dynamics of Southern forests, in: Wear, D.N., Greis, J.G., (Eds.), The Southern Forest Futures Project, USDA Forest Service, GTR SRS-178. Ashville, pp. 103–122.

Caldwell, P., Muldoon, C., Ford-Miniat, C., Cohen, E., Krieger, S., Sun, G., McNulty, S., Bolstad, P.V., 2014. Quantifying the role of National Forest System lands in providing surface drinking water supply for the Southern United States. In: USDA Forest Service, GTR SRS-197. Asheville.

Carlisle, D.M., Wolock, D.M., Meador, M.R., 2011. Alteration of streamflow magnitudes and potential ecological consequences: a multiregional assessment. Front. Ecol. Environ. 9, 264–270.

Carpenter, S.R., Folke, C., Scheffer, M., Westley, F., 2009. Resilience: accounting for the non-computable. Ecol. Soc. 14, 13, <http://www.ecologyandsociety.org/vol14/iss1/art13/>.
Chapin III, F.S., Folke, C., Kofinas, G.P., 2009. A framework for understanding change. In: Chapin, F.S. (Ed.), Principles of Ecosystem Stewardship. Springer-Verlag, Berlin, pp. 3–28. http://dx.doi.org/10.1007/978-0-387-73033-2.
Christensen, N.L., Bartuska, A.M., Brown, J.H., Carpenter, S., D'Antonio, C., Francis, R., Franklin, J.F., MacMahon, J.A., Noss, R.F., Parsons, D.J., Peterson, C.H., Turner, M. G., Woodmansee, R.G., 1996. The report of the ecological society of America committee on the scientific basis for ecosystem management. Ecol. Appl. 6, 665–691. http://dx.doi.org/10.2307/2269460.
Conrad, C.C., Hilchey, K.G., 2011. A review of citizen science and community-based environmental monitoring: issues and opportunities. Environ. Monit. Assess. 176, 273–291.
Couch, C.A., Hopkins, E.H., Hardy, P.S., 1996. Influences of environmental settings on aquatic ecosystems in the Apalachicola–Chattahoochee–Flint River basin. In: USGS Water-Resources Investigations Report 95-4278. Reston.
Dale, V.H., Brown, S., Haeuber, R., Hobbs, N.T., Huntley, N., Naiman, R.J., Riebsame, W.E., Turner, M.G., Valone, T., 2000. Ecological principles and guidelines for managing the use of land. Ecol. Appl. 10, 639–670.
Dickinson, J.L., Shirk, J.L., Bonter, D.N., Bonney, R.E., Crain, R.L., Martin, J., Phillips, T., Purcell, K.A., 2012. The current state of citizen science as a tool for ecological research and public engagement. Front. Ecol. Environ. 10, 291–297.
Edwards, L., Ambrose, J., Kirkman, L.K., 2013. The Natural Communities of Georgia. University of Georgia Press, Athens.
Emanuel, B., 2014. Upper Flint River Resiliency Action Plan. American Rivers. Atlanta <http://www.AmericanRivers.org/flintworkinggroup>.
Emanuel, B., Rogers, G., 2012. Running Dry: Challenges and Opportunities in Restoring Healthy Flows in Georgia's Upper Flint River Basin. American Rivers. Atlanta <http://www.AmericanRivers.org/RunningDry>.
Famiglietti, J.S., Rodell, M., 2013. Water in the balance. Science 340, 1300–1301.
Famiglietti, J.S., Cazenave, A., Eiker, A., Reager, J.T., Rodell, M., Velicogna, I., 2015. Satellites provide the big picture. Science 349, 684–685.
FAO and JRC, 2012. Global forest land-use change 1990–2005. In: Lindquist, E.J., D'Annunzio, R., Gerrand, A., MacDicken, K., Achard, F., Beuchle, R., Brink, A., Eva, H.D., Mayaux, P., San-Miguel-Ayanz J., Stibig, H-J., (Eds.), FAO Forestry Paper No. 169. Food and Agriculture Organization of the United Nations and European Commission Joint Research Centre, Rome.
Fekete, B.M., Robarts, R.D., Kumagai, M., Nachtnebel, H.-P., Odada, E., Zhulidov, A.V., 2015. Time for in situ renaissance. Science 349, 685–686.
Ford, C.R., Mitchell, R.J., Teskey, R.O., 2008. Water table depth affects productivity, water use, and the response to nitrogen addition in a savanna system. Can. J. Forest Res. 38, 2127–2218.
Freeman, M.C., Buell, G.R., Hay, L.E., Hughes, W.B., Jacobson, R.B., Jones, J.W., Jones, S. A., LaFontaine, J.H., Odum, K.R., Peterson, J.T., Riley, J.W., Schlinder, J.S., Shea, C., Weaver, J.D., 2012. Linking river management to species conservation using dynamic landscape-scale models. River Res. Appl. http://dx.doi.org/10.1002/rra.2575.
Gaither, C.J., Poudyal, N.C., Goodrick, S., Bowker, J.M., Malone, S., Gan, J., 2011. Wildland fire risk and social vulnerability in the Southeastern United States: an exploratory spatial data analysis approach. Forest Policy Econ. 13, 24–36.
Golladay, S.W., Gagnon, P., Kearns, M., Battle, J.M., Hicks, D.W., 2004. Response of freshwater mussel assemblages (Bivalvia: Unionidae) to a record drought in the Gulf Coastal Plain of southwestern Georgia. J. North Am. Benthol. Soc. 23, 494–506.
Gunderson, L.H., 2000. Ecological resilience – in theory and application. Ann. Rev. Ecol. Syst. 31, 425–439.
Haddad, N.M., Brudvig, L.A., Clobert, J., Davies, K.F., Gonzalez, A., Holt, R.D., Lovejoy, T.E., Sexton, J.O., Austin, M.P., Collins, C.D., Cook, W.M., Damschen, E.I., Ewers, R. M., Foster, B.L., Jenkins, C.N., King, A.J., Laurance, W.F., Levey, D.J., Margules, C.R., Melbourne, B.A., Nicholls, A.O., Orrock, J.L., Song, D.-A., Townshend, J.R., 2015. Habitat fragmentation and its lasting impact on Earth's ecosystems. Sci. Adv. 2015 (1), e1500052.
Hargrove, W.H., Spruce, J.P., Gasser, G.E., Hoffman, F.M., 2009. Toward a national early warning system for forest disturbances using remotely sensed canopy phenology. Photogramm. Eng. Rem. Sens. 75, 1150–1156.
Hobbs, R.J., Arico, S., Aronson, J., Baron, J.S., Bridgewater, P., Cramer, V.A., Epstein, P. R., Ewel, J.J., Klink, C.A., Lugo, A.E., Norton, D., Ojima, D., Richardson, D.M., Sanderson, E.W., Valladares, F., Vila, M., Zamora, R., Zobel, M., 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. Global Ecol. Biogeogr. 15, 1–7.
Hobbs, R.J., Higgs, E., Harris, J.A., 2009. Novel ecosystems: implications for conservation and restoration. Trends Ecol. Evol. 24, 599–605.
Hobbs, R.J., Higgs, E.S., Hall, C.M., 2013. Novel Ecosystems: Intervening in the New Ecological World Order. John Wiley & Sons Ltd, Chichester.
Hobbs, R.J., Higgs, E., Hall, C.M., Bridgewater, P.M., Chapin III, F.S., Ellis, E.C., Ewel, J.J., Hallett, L.M., Harris, J., Hulvey, K.B., Jackson, S.T., Kennedy, P.L., Kueffer, C., Lach, L., Lantz, T.C., Lugo, A.E., Mascaro, J., Murphy, S.D., Nelson, C.R., Perring, M.P., Richardson, D.M., Seastedt, T.R., Standish, R.J., Starzomski, B.M., Suding, K.N., Tognetti, P.M., Yakob, L., Yung, L., 2014. Managing the whole landscape: historical, hybrid, and novel ecosystems. Front. Ecol. Environ. 12, 557–564.
IPCC, 2007. Climate change 2007: synthesis report. In: Core Writing Team, Pachauri, R.K., Reisinger, A. (Eds.), Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Geneva.
IPCC, 2014. Climate change 2014: synthesis report. In: Core Writing Team, Pachauri, R.K., Meyer, L.A. (Eds.), Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Geneva.
Iverson, L.R., Matthews, S.N., Prasad, A.M., Peters, M.P., Yohe, G., 2012. Development of risk matrices for evaluating climatic change responses of forested habitats. Climatic Change 114, 231–243.
Jaffe, M., 2011. Reflections on green infrastructure economics. Environ. Pract. 12, 357–365.
Kao, R.H., Gibson, C.M., Gallery, R.E., Meier, C.L., Barnett, D.T., Docherty, K.M., Blevins, K.K., Travers, P.D., Azuaje, E., Springer, Y.P., Thibault, K.M., McKenzie, V. J., Keller, M., Alves, L.F., Hinckley, E.S., Parnell, J., Schimel, D., 2012. NEON terrestrial field observations: designing continental-scale, standardized sampling. Ecosphere, 3. http://dx.doi.org/10.1890/ES12-00196.1.
Kerr, J.T., Ostrovsky, M., 2003. From space to species: ecological applications for remote sensing. Trends Ecol. Evol. 18, 299–305.
Kirkman, L.K., Mitchell, R.J., Helton, R.C., Drew, M.B., 2001. Productivity and species richness across an environmental gradient in a fire-dependent ecosystem. Am. J. Bot. 88, 2119–2128.
Klepzig, K., Shelfer R., Choice, Z., 2014. Outlook for coastal plain forests: a subregional report from the Southern Forest Futures Project. In: USDA Forest Service, GTR SRS-GTR-196. Ashville.
Likens, G., 2001. Biogeochemistry, the watershed approach: some uses and limitations. Mar. Freshwat. Res. 52, 5–12.
Lindenmayer, D., Likens, G., 2010. The science and application of ecological monitoring. Biol. Conserv. 143, 1317–1328.
Liu, Y.Q., Goodrick, S.L., Stanturf, J.A., 2013a. Future U.S. wildfire potential trends projected using a dynamically downscaled climate change scenario. Forest Ecol. Manag. 294, 120–135.
Liu, Y.Q., Prestemon, J., Goodrick, S.L., Holmes, T.P., Stanturf, J.A., Vose, J.M., Sun, G., 2013b. Future wildfire trends, impacts and mitigation options in the Southern United States. In: Vose, J.M., Klepzig, K. (Eds.), Climate Change Adaptation and Mitigation Management Options. CRC Press, Boca Raton, pp. 85–126.
Lockaby, G., Nagy, C., Vose, J. M., Ford, C. R., Sun, G., McNulty, S., Caldwell, P., Cohen, E., Moore Myers, J., 2013. Forests and water. In: Wear, D.N., Greis, J.G., (Eds.), The Southern Forest Futures Project: Technical Report. USDA Forest Service GTR SRS-178. Asheville, pp. 309–339.
Lower Flint-Ochlockonee Regional Water Plan, 2011. <http://www.flintochlockonee.org/documents/LFO_Adopted_RWP.pdf> (accessed September 2015).
Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schager, J.P., Grizzetti, B., Drakou, E.V., La Notte, A., Zulian, G., Bouraoui, F., Paracchini, M.L., Bratt, L., Bidoglio, G., 2012. Mapping ecosystem services for policy support and decision making in the European Union. Ecosyst. Serv. 1, 31–39.
Maxwell, S.L., Milner-Gulland, E.J., Jones, J.P.G., Knight, A.T., Bunnefeld, N., Nuno, A., Bal, P., Earle, S., Waterson, J.E.M., Rhodes, J.R., 2015. Being smart about SMART environmental targets. Science 347, 1075–1076.
McDowell, W.H., 2015. NEON and STREON: opportunities and challenges for the aquatic sciences. Freshwat. Sci. 34, 386–391.
McNulty, S., Myers, J.M., Caldwell P., Sun, G., 2013. Climate change summary. In: Wear, D.N., Greis, J.G., (Eds.), The Southern Forest Futures Project. USDA Forest Service GTR SRS-178. Ashville, pp. 27–44.
Melillo, J.M., Richmond, T., Yohe, G.W., 2014. Climate change impacts in the United States: the third national climate assessment. Global Change Res. Prog. http://dx.doi.org/10.7930/J0Z31WJ2.
Melvin, M., 2012. National Prescribed Fire Use Survey Report. Coalition of Prescribed Fire Councils Technical Report 01-12. Newton.
Mercer, D.E., Cooley, D., Hamilton, K., 2011. Taking Stock: Payments for Forest Ecosystem Services in the United States. Forest Trends Association, Washington, DC.
Millar, C.I., Stephenson, N.L., Stephens, S.L., 2007. Climate change and forests of the future: managing in the face of uncertainty. Ecol. Appl. 17, 2145–2151.
Milly, P.C.D., Betancourt, J., Falkenmark, M., 2008. Climate change: stationarity is dead: whither water management? Science 319, 573–574.
Mitchell, R.J., Youngqiang, L., O'Brien, J.J., Elliott, K.J., Starr, G., Miniat, C.F., Hiers, J.K., 2014. Future climate and fire interactions in the southeastern region of the United States. Forest Ecol. Manag. 327, 316–326.
Moyle, P.B., Mount, J.F., 2007. Homogenous rivers, homogenous faunas. Proc. Nat. Acad. Sci. USA 104, 5711–5712.
Naiman, R.J., 2013. Socio-ecological complexity and the restoration of river ecosystems. Inland Waters 3, 391–410.
Neves, R.J., Bogan, A.E., Williams, J.D., Ahlstedt, S.A., Hartfield, P.W., 1997. Status of aquatic mollusks in the southeastern United States: a downward spiral of diversity. In: Benz, G.W., Collins, D.E., (Eds.), Aquatic Fauna in Peril: The Southeastern Perspective. Southeastern Aquatic Research Institute, Lenz Design and Communications, Decatur, pp. 43–85.
North, M., Collins, B.M., Stephens, S., 2012. Using fire to increase the scale, benefits, and future maintenance of fuels treatments. J. Forest. 110, 392–401.
Odman, T., Hu, Y., Russel, A., 2007. Forecasting the impacts of wildfires. In: 2007 CMAQ User Workshop, Chapel Hill <http://www.cmascenter.org>.
Ojima, D.S., Iverson, L.R., Sohngen, B.L., 2012. Appendix 2: risk-based framework and risk case studies: Risk-based framework for evaluating changes in response thresholds and vulnerabilities. In: Vose, J.M., Peterson, D.L., Patel-Weynand, T., (Eds.), Effects of Climatic Variability and Change on Forest Ecosystems: A Comprehensive Science Synthesis for the U.S. Forest Sector. USDA Forest Service, GTR PNW-GTR-870. Portland, pp. 243–245.

Ojima, D.S., Iverson, L.R., Sohngen, B.L., Vose, J.M., Woodall, C.W., Domke, G.M., Peterson, D.L., Littell, J.S., Matthews, S.N., Prasad, A.M., Peters, M.P., Yohe, G.W., Friggens, M.M., 2014. Risk assessment. In: Petersen, D.L., Vose, J.L., Patel-Weyland, T., (Eds.), Climate Change and United States Forests. Advances in Global Change Research 57, pp. 223–244, http://dx.doi.org/10.1007/978-94-007-7515-2_9.

Olden, J.D., Poff, N.L., 2003. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. Riv. Res. Appl. 19, 101–121.

Perry, C., Yager, R., 2011. Irrigation water conservation efforts at the UGA C.M. Stripling Irrigation Research Park. In: Proceedings of the 2011 Georgia Water Resources Conference. Athens.

Pettorelli, N., Vik, J.O., Mysterud, A., Gaillard, J-.M., Tucker, C.J., Stenseth, N.C., 2005. Using satellite-derived NDVI to assess ecological responses to environmental change. Trends Ecol. Evol. 20, 503–510.

Pickett, S.T.A., Parker, V.T., Fiedler, P.L., 1992. The new paradigm in ecology: implications for conservation biology above the species level. In: Fiedler, P.L., Jain, S.K. (Eds.), Conservation Biology: The Theory and Practice of Nature Conservation, Preservation, and Management. Chapman and Hall, New York, pp. 65–88.

Poff, N.L., Zimmerman, J.K.H., 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwat. Biol. 55, 194–205.

Poff, N.L., Richter, B.D., Arthington, A.H., Bunn, S.E., Naiman, R.J., Kendy, E., Acreman, M., Apse, C., Bledsoe, B.P., Freeman, M.C., Henriksen, J., Jacobson, R.B., Kennen, J.G., Merritt, D.M., O'Keefe, J.H., Olden, J.D., Rogers, K., Tharme, R.E., Warner, A., 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. Freshwat. Biol. 55, 147–170.

Prato, T., 2008. Accounting for risk and uncertainty in determining preferred strategies for adapting to future climate change. Mitig. Adapt. Strateg. Glob. Change 13, 47–60.

Prud'homme, B.A., Greis, J.G., 2002. Best management practices in the south. In: Wear, D.N., Greis, J.G., (Eds.), The Southern Forest Resource Assessment: Technical Report. USDA Forest Service, GTR SRS-53. Ashville, pp. 519–535 (chapter 22).

Radeloff, V.C., Hammer, R.B., Stewart, S.I., Fried, J.S., Holcomb, S.S., McKeefry, J.F., 2005. The wildland–urban interface in the United States. Ecol. Appl. 15, 799–805.

Reager, J.T., Thomas, B.F., Famiglietti, J.S., 2014. River basin flooding potential inferred using GRACE gravity observations at several months lead time. Nat. Geosc. 7, 588–592.

Reed, M.S., Graves, A., Dandy, N., Posthumus, H., Hubacek, K., Morris, J., Prell, C., Quinn, C.H., Stringer, L.C., 2009. Who's in and why? A topology of stakeholder analysis methods for natural resource management. J. Environ. Manag. 90, 1933–1949.

Richter, B.D., 2009. Re-thinking environmental flows: from allocations and reserves to sustainability boundaries. Riv. Res. Appl. http://dx.doi.org/10.1002/rra.1320.

Richter, B.D., Davis, M.M., Apse, C., Konrad, C., 2011. A presumptive standard for environmental flow protection. Riv. Res. Appl. http://dx.doi.org/10.1002/rra.1511.

Rieman, B.E., Smith, C.L., Naiman, R.J., Ruggerone, G., Wood, C.C., Huntly, N., Merrill, E.N., Alldredge, J.R., Bisson, P.A., Congleton, J., Fausch, K., Levings, C., Pearcy, W., Scarnecchia, D., Smouse, P., 2015. A comprehensive approach for habitat restoration in the Columbia Basin. Fisheries 40, 124–135.

Rugel, K., Jackson, C.R., Romeis, J.J., Golladay, S.W., Hicks, D.W., Dowd, J.F., 2012. Effects of irrigation withdrawals on streamflows in a karst environment: lower Flint River Basin, Georgia, USA. Hydrol. Process. 26, 523–534.

Sargent, L.W., Golladay, S.W., Covich, A.P., Opsahl, S.P., 2011. Physicochemical habitat association of a native and non-native crayfish in the lower Flint river, Georgia: implications for invasion success. Biol. Invasions 13, 499–511.

Schimel, D., Keller, M., 2015. Big questions, big science: meeting the challenges of global ecology. Oecologia. http://dx.doi.org/10.1007/s00442-015-3236-3.

Seastedt, T.R., Hobbs, R.J., Suding, K.N., 2008. Management of novel ecosystems: are novel approaches required? Front. Ecol. Environ. 6, 547–553.

Sheppard, C.S., Burns, B., Stanley, M.C., 2014. Predicting plant invasions under climate change: are species distribution models validated by field trials? Glob. Change Biol. 20, 2800–2814.

Shortle, J.S., Horan, R.D., 2013. Policy instruments for water quality protection. Ann. Rev. Resour. Econ. 5, 111–138.

Stanturf, J.A., Goodrick, S.L., 2013. Fire. In: Wear D.N., Greis, J.G., (Eds.), The Southern Forest Futures Project. USDA Forest Service GTR SRS-178. Ashville, pp. 509–529.

Steffen, W., Crutzen, P.J., McNeill, J.R., 2007. The Anthropocene: are humans now overwhelming the great forces of nature. Ambio 36, 614–621.

Stephens, S.L., Agee, J.K., Fule, P.Z., North, M.P., Romme, W.H., Swetnam, T.W., Turner, M.G., 2013. Managing forests and fire in changing climates. Science 342, 41–42.

Sun, G., McNulty, S.G., Moore, J.A., Cohen, E.C., 2008. Impacts of multiple stresses on water demand and supply across the southeastern United States. J. Am. Water Resour. Assoc. 44, 1441–1457.

Sun, G., Caldwell, P.V., McNulty, S.G., Georgakakos, A.P., Arumugam, S., Cruise, J., McNider, R.T., Terando, A., Conrads, P.A., Feldt, J., Misra, V., Romolo, L., Rasmussen, T.C., Marion, D.A., 2013. Impacts of Climate Change and Variability on Water Resources in the Southeast USA. USDA Forest Service, NCA Southeast Technical Report 204-234. Ashville.

Terando, A.J., Costanza, J., Belyea, C., Dunn, R.R., McKerrow, A., Collazo, J.A., 2014. The southern megalopolis: using the past to predict the future of urban sprawl in the southeast U.S. PLOSone 9, e102261, http://dx.doi.org/10.1371/journal.pone.0102261.

Thomas, A.C., Reager, J.T., Famiglietti, J.S., Rodell, M., 2014. A grace-based water storage deficit approach for hydrological drought characterization. Geophys. Res. Lett. 41, 1537–1545.

Upper Flint Regional Water Plan, 2011. <http://www.upperflint.org/documents/UFL_Adopted_RWP.pdf> (accessed September 2015).

van den Avyle, M.J., Evans, J.W., 1990. Temperature selection by Striped Bass in a Gulf of Mexico coastal river system. N. Am. J. Fish. Manag. 10, 58–66.

Wear, D.N., 2013. Forecasts of land uses. In: Wear, D.N., Greis, J.G., (Eds.), The Southern Forest Futures Project. USDA Forest Service, GTR SRS-178. Ashville, pp. 45–72.

Wear, D.N., Greis, J.G., 2013. The Southern Forest Futures Project: Technical Report, USDA Forest Service, GTR SRS-178. Ashville.

Westgate, M.J., Likens, G.E., Lindenmayer, D.B., 2013. Adaptive management of biological systems: a review. Biol. Conserv. 158, 128–139.

Woodall, C.W., Amacher, M.C., Bechtold, W.A., Coulston, J.W., Jovan, S., Perry, C.H., Randolph, K.C., Schulz, B.K., Smith, G.C., Tkacz, B., Will-Wolf, S., 2011. Status and future of the forest health indicators program of the USA. Environ. Monit. Assess. 177, 419–436.

Yin, Z., 1993. Fire regime of the Okefenokee Swamp and its relation to hydrological and climatic conditions. Int. J. Wildland Fire 3, 229–240.

Yohe, G., Leichenko, R., 2010. Adopting a risk-based approach. Ann. NY. Acad. Sci., Spec. Iss. New York City Panel Clim. Change 196, 29–40.

Zhang, Y., He, H.S., Yang, J., 2008. The wildland–urban interface dynamics in the southeast U.S. from 1990 to 2000. Landsc. Urban Plan. 85, 155–162.

Zhang, R., Delworth, T.L., Sutton, R., Hodson, D.L.R., Dixon, K.W., Held, I.M., Kushnir, Y., Marshall, J., Ming, Y., Msadek, R., Robson, J., Rosati, A.J., Ting, M.-F., Vecchi, G. A., 2013. Have aerosols caused the observed Atlantic multidecadal variability? J. Atmos. Sci. 70, 1135–1144.

Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration

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Climate change is increasing fire activity in the western United States, which has the potential to accelerate climate-induced shifts in vegetation communities. Wildfire can catalyze vegetation change by killing adult trees that could otherwise persist in climate conditions no longer suitable for seedling establishment and survival. Recently documented declines in postfire conifer recruitment in the western United States may be an example of this phenomenon. However, the role of annual climate variation and its interaction with long-term climate trends in driving these changes is poorly resolved. Here we examine the relationship between annual climate and postfire tree regeneration of two dominant, low-elevation conifers (ponderosa pine and Douglas-fir) using annually resolved establishment dates from 2,935 destructively sampled trees from 33 wildfires across four regions in the western United States. We show that regeneration had a nonlinear response to annual climate conditions, with distinct thresholds for recruitment based on vapor pressure deficit, soil moisture, and maximum surface temperature. At dry sites across our study region, seasonal to annual climate conditions over the past 20 years have crossed these thresholds, such that conditions have become increasingly unsuitable for regeneration. High fire severity and low seed availability further reduced the probability of postfire regeneration. Together, our results demonstrate that climate change combined with high severity fire is leading to increasingly fewer opportunities for seedlings to establish after wildfires and may lead to ecosystem transitions in low-elevation ponderosa pine and Douglas-fir forests across the western United States.

ecosystem transition | climate change | wildfire | ponderosa pine | Douglas-fir

As climate and disturbance regimes change, abrupt transitions are increasingly recognized and predicted in ecological systems (1, 2). Abrupt transitions between different ecosystem states can occur when gradually changing environmental conditions cross critical thresholds beyond which small changes produce large ecosystem responses (1, 2). Before reaching a critical threshold or "tipping point," gradual changes in environmental conditions may cause increased variance but do not necessarily lead to distinct changes in ecosystem states (1, 3, 4). In stochastic environments, systems may oscillate between two states before a transition (3, 5). For example, tree species that have climatic thresholds for regeneration may experience episodic recruitment as climate temporally varies between conditions that are suitable and unsuitable for regeneration (refs. 6 and 7, Fig. 1). Despite widespread interest in understanding ecological thresholds in an era of rapid global change (8–10), quantifying abrupt climate-induced vegetation shifts remains a substantial and important scientific challenge.

Stand-replacing fires can catalyze vegetation shifts during periods of directional climate change, accelerating changes that would otherwise take decades to centuries to play out (11–13). This is particularly true in forest ecosystems, where adult trees can live for centuries and tolerate a broader range of climate conditions than juveniles of the same species (6, 14, 15). Disturbance-catalyzed change at lower treeline, where trees grow at the warm, dry margin of their climatic tolerances, may be one of the first visible signs of forest ecosystems adjusting to new climate conditions. Recent evidence suggests that wildfires may already be catalyzing vegetation shifts in forests across the western United States (16), with limited tree regeneration following fires in recent decades (e.g., refs. 17–19). This is particularly acute in low-elevation forests (17, 20–23), implicating climate change as an important driver of regeneration failures. However, the annual climate conditions which limit tree regeneration are poorly resolved, and potential thresholds to regeneration have not been identified. Understanding if recent reductions in postfire tree regeneration signal an ecosystem transition (e.g., to a nonforested state) requires a quantitative understanding of how seasonal to interannual variations in climate impact tree seedling germination and establishment.

Here we demonstrate that dry low-elevation *Pinus ponderosa* (ponderosa pine) and *Pseudotsuga menziesii* (Douglas-fir) forests of the western United States have crossed a critical climate threshold for postfire tree regeneration. We focused on ponderosa pine and Douglas-fir because they are widespread ecologically and

Significance

Changes in climate and disturbance regimes may cause abrupt shifts in vegetation communities. Identifying climatic conditions that can limit tree regeneration is important for understanding when and where wildfires may catalyze such changes. This study quantified relationships between annual climate conditions and regeneration of *Pinus ponderosa* (ponderosa pine) and *Pseudotsuga menziesii* (Douglas-fir), two ecologically and economically important conifer species in low-elevation forests of western North America. We found that regeneration exhibited a threshold response to annual climate conditions and the forests we sampled crossed these climate thresholds in the past 20 years, resulting in fewer recruitment opportunities through time. In areas that have crossed climatic thresholds for regeneration, stand-replacing fires may result in abrupt ecosystem transitions to nonforest states.

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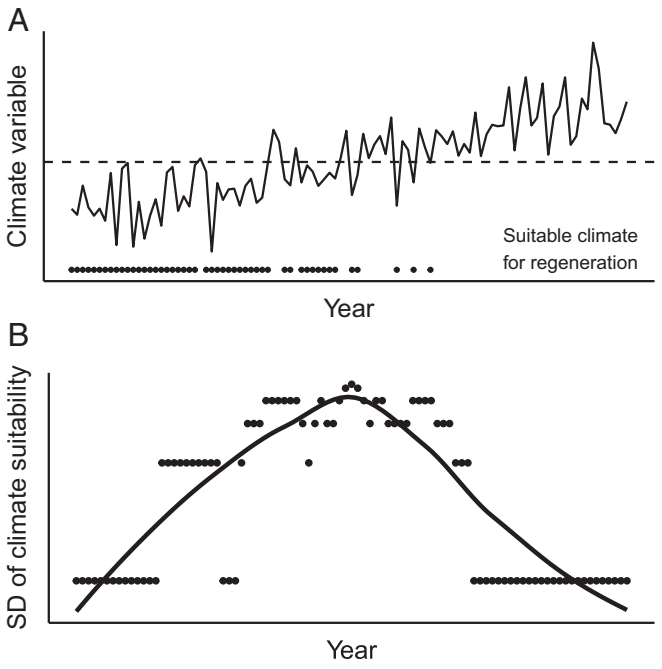

Fig. 1. Conceptual diagram depicting a simulated hypothetical annual climate variable (e.g., mean summer temperature or vapor pressure deficit; solid line) and the corresponding expected changes to the frequency of years suitable for regeneration (dots), as mean climate conditions increase and cross a threshold for regeneration (dashed line) (*A*). When climate conditions are suitable for regeneration every year, other factors such as cone production may still cause episodic establishment. As climatic thresholds are approached, the SD of climate suitability for regeneration increases, shown here using a moving 10-y window and summarized with a locally weighted polynomial regression smoothing (LOESS) (*B*). Once a critical threshold is crossed, variability declines again, as fewer years are suitable for regeneration.

economically important conifers in low-elevation forests of western North America. We sampled across climate gradients in space and time (1988–2015) using annually resolved establishment dates from 2,935 trees that established after 33 wildfires in four regions across the western United States (Fig. 2) to (*i*) quantify the relationship between seasonal to annual climate and regeneration, (*ii*) identify critical climate thresholds for regeneration, and (*iii*) assess how climate suitability for postfire regeneration has changed over recent decades. To isolate the effect of annual climate, we accounted for the effect of other drivers of postfire regeneration, including fire severity and distance to seed source. Our results demonstrate threshold responses of annual tree recruitment to vapor pressure deficit (VPD), surface temperature, and soil moisture. Climate conditions in the low-elevation forests we sampled have repeatedly crossed these thresholds over the past 20 y, revealing a decline in the climate suitability for postfire tree regeneration across broad regions of the western United States. These findings imply that increased frequency of stand-replacing fires could initiate abrupt ecosystem transitions in low-elevation ponderosa pine and Douglas-fir forests.

Results and Discussion

Annual rates of tree regeneration exhibited strongly nonlinear relationships with annual climate conditions, with distinct threshold responses to summer VPD, soil moisture, and maximum surface temperatures (Figs. 3 and 4). Across the study region, seasonal to annual climate conditions from the early 1990s through 2015 have crossed these climate thresholds at the majority of sites (*SI Appendix*, Fig. S8), indicating conditions that are increasingly unsuitable for tree regeneration, particularly for ponderosa pine. We assessed changes in the climate suitability for postfire regeneration by projecting our boosted regression tree models [mean area under the curve (AUC) 0.81; *SI Appendix*, Table S4] with climate time series for each site from 1980 to 2015, while holding nonclimatic factors constant (Figs. 3*F* and 4*F*). We found abrupt declines in modeled annual recruitment probability in the 1990s for both species and across all regions, with the exception of Douglas-fir in California and Colorado. The cumulative probability of postfire recruitment calculated on a 5- and 10-y basis also declined in the Southwest and Northern Rockies for both species, and in Colorado and California for ponderosa pine (*SI Appendix*, Figs. S9 and S10). Change-point detection analysis identified significant changes in modeled recruitment probability in the early to mid-1990s (*SI Appendix*, Table S5). These declines in annual and cumulative recruitment probability correspond directly to years when mean climate conditions approached threshold values for recruitment (Figs. 3 and 4), making it increasingly unlikely that future climate conditions will be favorable (i.e., a cool or wet year) for subsequent regeneration pulses.

Theory predicts that variance in state variables will increase as critical transitions are approached (3, 4), and then decrease once thresholds are crossed (ref. 2, Fig. 1). Consistent with this, in most regions we found that the variability in the modeled annual recruitment probability increased initially and then declined over time, once climate thresholds were crossed (Figs. 3*E* and 4*E*). There were two exceptions to this pattern, which highlight important complexities in how climate change may impact postfire tree regeneration. First, the annual recruitment probability of ponderosa pine in Colorado varied little over time (Fig. 3*E*), because as mean VPD increased, so too did interannual variability in VPD, such that some years were still suitable for recruitment (Fig. 3*B*). Increased variability in climate metrics was not seen in other regions. Second, the probability of Douglas-fir regeneration in California sites likewise varied little over time, reflecting little change in the mean and variability of the climate predictors of Douglas-fir recruitment (Fig. 4 *B* and *D*). Across our study regions, we also observed higher variability in observed recruitment at sites closer to the dry margins of tree distributions. By analyzing the annual patterns of observed recruitment across sites, we

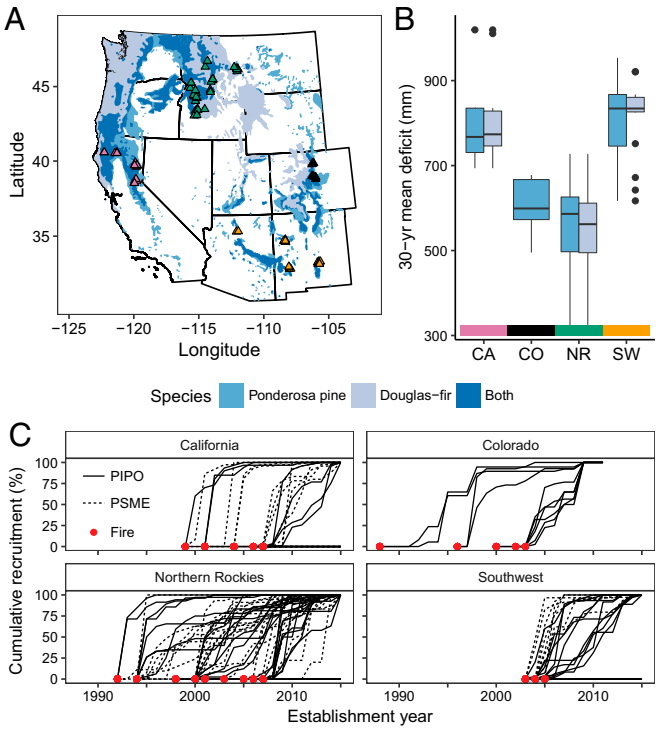

Fig. 2. Map of study site locations in the western United States (*A*), 30-y (1981–2010) mean climatic water deficit for sample sites within each region (*B*), and cumulative recruitment following fires for each site (*C*). Overall, 90 sites were sampled (see *SI Appendix*, Tables S1 and S2 for details). Ponderosa pine (PIPO) and Douglas-fir (PSME) ranges are shown in blue in *A* (72).

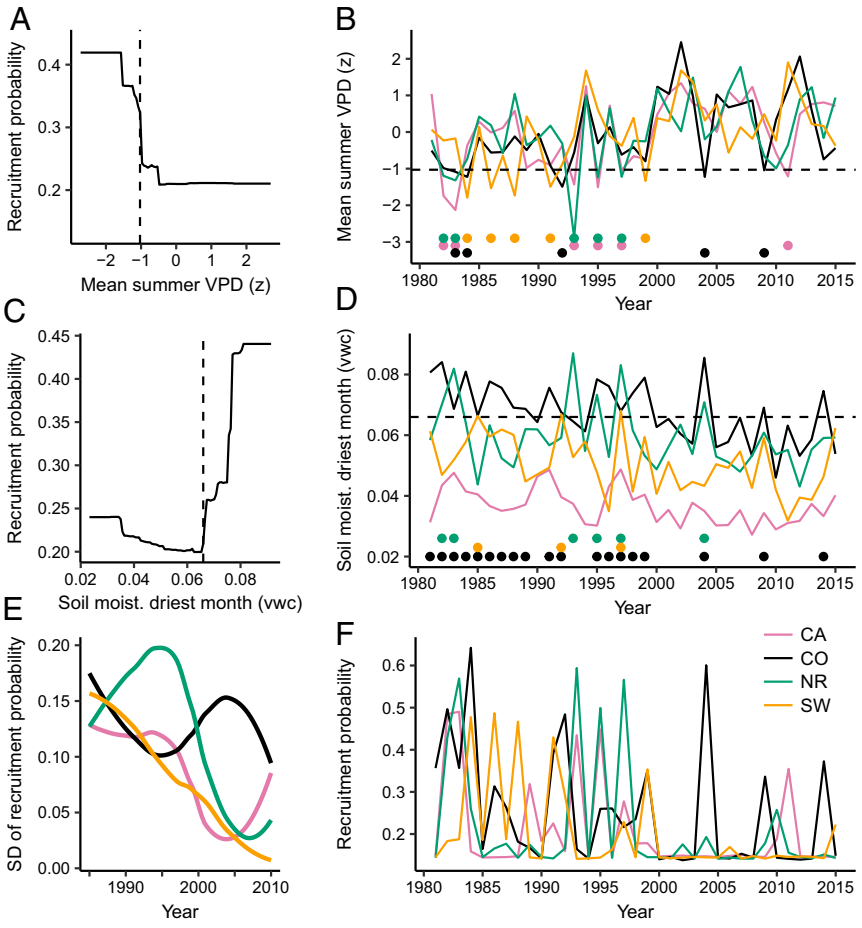

Fig. 3. Threshold response of recruitment to annual climate and modeled annual recruitment probability for ponderosa pine. Partial dependency plots from a boosted regression tree model show the marginal effect of the two most important climate variables on annual recruitment probability, after accounting for the average effects of all other variables in the model (*A* and *C*). Annual time series of climate variables at each site averaged by region (*B* and *D*). Climate thresholds are identified with vertical (*A* and *C*) and horizontal (*B* and *D*) dashed lines. Dots below the lines in *B* and *D* represent years when that specific climate variable was suitable for regeneration. The influence of both climate variables on regeneration are summarized by the modeled annual recruitment probability (*F*), while holding constant time since fire (1 y), distance to seed source (50 m), and fire severity (dNBR 412). Variability in annual recruitment probability (*E*) is shown as the SD of recruitment probability values from *F*, calculated in 10-y moving windows and plotted with a LOESS. The metric vwc indicates the ratio of water volume to soil volume.

showed that sites with drier 30-y mean climate conditions exhibited more episodic recruitment than wetter sites (*SI Appendix*, Fig. S13; $t = -2.9$, df = 69, $P = 0.005$), highlighting the prevalence of episodic recruitment at sites near climatic thresholds (24, 25).

Differences between the sensitivity of ponderosa pine and Douglas-fir to annual climate have important implications for species-specific responses to climate change (26, 27). Both species were sensitive to soil moisture; first-year conifer seedlings have shallow root systems and frequently succumb to desiccation (28–31). However, Douglas-fir regeneration was more strongly related to spring soil moisture, which has not declined in recent decades, than to soil moisture of the driest month, which has recently crossed threshold values. This pattern explains the larger declines in annual recruitment probability for ponderosa pine relative to Douglas-fir (Figs. 3 and 4). Ponderosa pine was additionally sensitive to high summer VPD, which leads to increased transpiration rates and increased plant water stress. VPD, which has increased over the past several decades (ref. 32, Fig. 3), is recognized as an important factor determining tree mortality, growth rates, and seedling survival (31, 33–36). VPD is predicted to increase in the future (32). Consequently, the probability of ponderosa pine regeneration at low-elevation sites will likely decline. Douglas-fir recruitment was sensitive to high maximum surface temperatures, which kill seedlings by damaging vascular tissue (37–39). The threshold response we observed is consistent with previous experiments that show threshold mortality responses to soil surface temperatures at or near 55 °C (37, 40). This mechanism of mortality is especially important in disturbed areas where there is no remaining canopy cover to ameliorate high maximum temperatures near the soil surface (41, 42). Intraspecific differences in climatic tolerances are also likely to exist (43–45). For example, across the study area, ponderosa pine recruitment was more strongly related to site-specific VPD anomalies than raw VPD values, suggesting that ecotypic variation or local adaptation among populations in part influences how annual climate impacts germination and survival (36). However, in our models, geographic region had a negligible effect on annual recruitment probability and no strong interactions with annual climate variables. Thus, our results suggest that consistent threshold relationships with annual climate were identifiable across the entire range of both species.

While annual climate was an important driver of postfire regeneration, our findings also highlight that the nature of a fire event strongly influences postfire regeneration. For example, the combined relative influence of annual climate variables on tree recruitment in our boosted regression tree (BRT) models was 24% for ponderosa pine and 34% for Douglas-fir (*SI Appendix*, Fig. S3), while the relative influence of distance to seed source, which is largely determined by fire severity, was 32% for ponderosa pine and 21% for Douglas-fir (*SI Appendix*, Fig. S3). The importance of seed tree availability in determining postfire regeneration has been demonstrated across forest types in the western United States (e.g., refs. 17, 20, and 46), suggesting that increases in high severity burned patch sizes (e.g., ref. 47) will significantly reduce tree regeneration. Beyond affecting seed availability, fire severity can also affect regeneration by altering microclimate (41, 42, 48) or soil properties and biota (49). Furthermore, high severity fire is correlated with high postfire shrub dominance in some regions, which can limit ponderosa pine regeneration (19). Accordingly, we found that ponderosa pine regeneration was lower at sites that experienced higher fire severity (*SI Appendix*, Fig. S3). High shrub cover also corresponded with more episodic establishment of ponderosa pine (*SI Appendix*, Fig. S13; $t = -3.2$, df = 69, $P = 0.002$), with pronounced peaks of regeneration immediately following fire at sites with high shrub cover. Fire severity was much less influential for Douglas-fir regeneration (*SI Appendix*, Fig. S3). Overall, our results indicate that

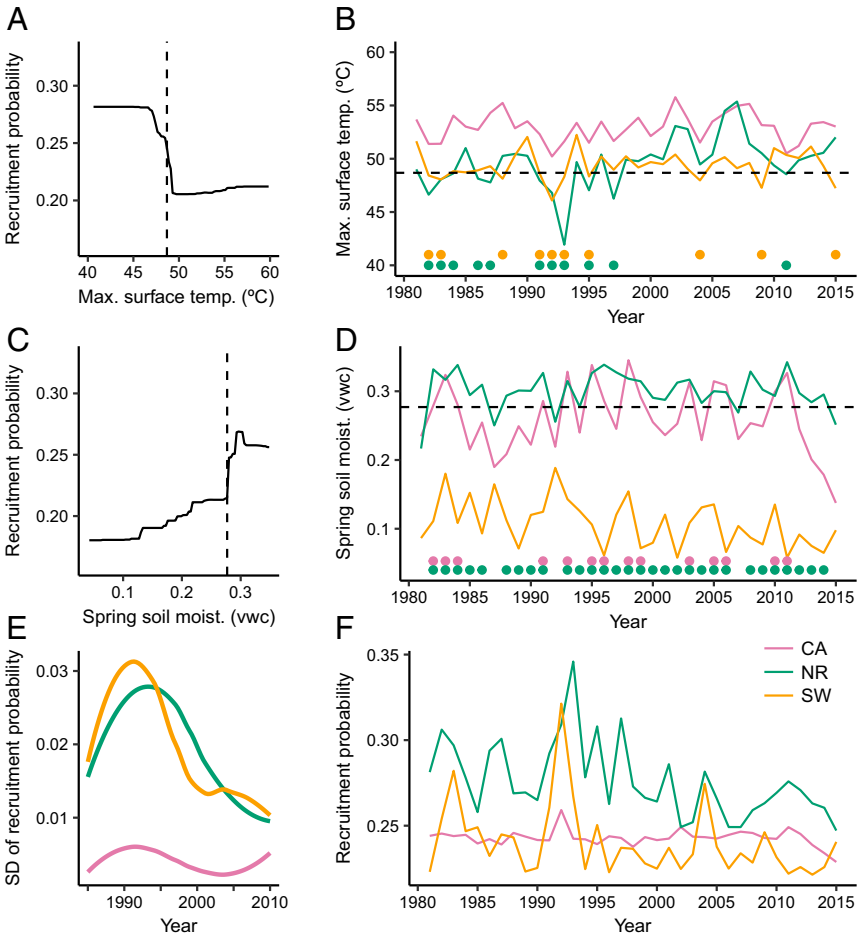

Fig. 4. Threshold response of recruitment to annual climate and modeled annual recruitment probability for Douglas-fir. Partial dependency plots from a boosted regression tree model show the marginal effect of the two most important climate variables on annual recruitment probability, after accounting for the average effects of all other variables in the model (*A* and *C*). Annual time series of climate variables at each site averaged by region (*B* and *D*). Climate thresholds are identified with vertical (*A* and *C*) and horizontal (*B* and *D*) dashed lines. Dots below the lines in *B* and *D* represent years when that specific climate variable was suitable for regeneration. The influence of both climate variables on regeneration are summarized by the modeled annual recruitment probability (*F*), while holding constant time since fire (1 y), distance to seed source (50 m), and fire severity (dNBR 412). Variability in annual recruitment probability (*E*) is shown as the SD of recruitment probability values from *F*, calculated in 10-y moving windows and plotted with a LOESS. The metric vwc indicates the ratio of water volume to soil volume.

the impacts of annual climate conditions on tree regeneration are strongly mediated by multiple biotic and abiotic factors.

While our results reveal clear relationships between annual climate and the probability of tree regeneration, they are constrained by two limitations. First, our statistical models focus on climate conditions during the year of seedling germination. Theory and observations highlight that mortality during the year of germination is a key bottleneck affecting conifer demography (28, 30, 31, 39, 50). Our results are consistent with this perspective: climate during the year of germination was a significant predictor of tree demography at sites sampled years to decades after germination. However, annual climate conditions in the years following germination through the year of sampling, may also affect seedling survival. Second, our statistical models did not account for cone or seed production, an important determinant of tree regeneration that also varies with annual climate conditions (51, 52). As years with suitable conditions for regeneration become increasingly rare, the episodic nature of cone production will likely further limit regeneration, particularly if years with high cone production do not align with years with suitable climate for germination and survival.

Our results reveal an important pathway linking climate change and interannual climate variability to recently observed reductions in postfire tree regeneration. Our findings suggest that many low-elevation mixed conifer forests in the western United States have already crossed climatic thresholds beyond which the climate is unsuitable for regeneration. Once climate exceeds climatic thresholds for regeneration and fire results in adult mortality, sites that currently experience episodic recruitment may lose local tree populations (6). We sampled dry ponderosa pine and Douglas-fir sites within each region, but as climate continues to get warmer and drier (32, 53), more areas will cross climatic thresholds that limit conifer recruitment. Increasing frequency of extreme fire weather could lead to more high severity fire (54), while decreases in summer precipitation (55) coupled with changes in fuel aridity and timing of spring snowmelt, have led to more area burned across the western United States in the past several decades (56, 57). The combination of more area burned, potentially at high severity, with decreasing climate suitability for postfire regeneration could lead to rapid transitions from ponderosa pine and Douglas-fir forests to nonforest vegetation. Our results highlight the potential for ecological processes to exhibit rapid transitions, and we expect that fire will increasingly catalyze vegetation shifts at lower treeline in the future. The nonlinear relationships between annual climate and regeneration observed in this study are likely not unique to these two species (e.g., ref. 30). Thus, the combination of fire and climate change may lead to abrupt ecosystem changes where other tree species exhibit similar threshold responses to climate.

Materials and Methods

Our study was designed to understand the relationship between annual climate conditions and postfire tree regeneration in dry conifer forests dominated by *P. ponderosa* (ponderosa pine) and/or *P. menziesii* (Douglas-fir). We used dendrochronology to determine the germination year of trees established after 33 fires in 90 sites across four regions in the western United States: the Northern Rockies (NR), Colorado Front Range (CO), Southwest (SW), and Northern California (CA) (Fig. 2). Within each region, we sampled sites near the warm/dry limits of regional forest extent, to bracket the climatic conditions suitable for recruitment; further, these areas are where we would expect tree regeneration to be most sensitive to climate change. We constructed a statistical model predicting the annual recruitment probability at each site as a function of biophysical variables. The model results provide insights into the nature of climatic and nonclimatic controls of postfire tree regeneration, and through hindcasting, the model allowed us to assess how climate suitability for tree regeneration has varied over the past several decades across our study regions. The data and code used in this study are publicly available via the Dryad Data Repository: https://doi.org/10.5061/dryad.pc3f9d8 (58).

Field Sampling. In each region, we selected sites that burned at moderate to high severity between 1988 and 2007 [based on Monitoring Trends in Burn Severity (MTBS) data (59) and later field verified], had no postfire planting and had 30-y (1980–2009) mean climatic water deficits within the top (i.e., driest) 50th percentile for each species within each region (*SI Appendix, Supplemental Methods*). From a random set of sites that fit these criteria, we sampled a total of 19 sites in CA (from six fires), 10 in CO (from five fires), 40 in NR (from 18 fires), and 21 in SW (from four fires) (*SI Appendix*, Tables S1 and S2). Historically, forests in these regions experienced mixed severity fire regimes, with SW and some CA sites characterized by low-severity surface fire regimes (60–62).

In NR, SW, and CA, sampling occurred in 60-m long belt transects with variable width (2–40 m), with the goal of destructively sampling ∼30 trees per transect. All trees that established following fire in each transect were sampled; as all sampled trees were less than 25 y old, we use the term "juvenile" hereafter to refer to sampled individuals. We destructively sampled juveniles to obtain precise germination years, because field-based methods for estimating tree ages are not accurate with annual precision (63). If no juveniles were present at initial randomly identified sample points, site data were recorded and the zero density was retained in the dataset; this occurred at 30% of sites. Where none of the preselected random sample points within a fire yielded at least 30 juveniles for sampling, new plots were located in areas with more regeneration. Thus, field sampling was designed to accurately reconstruct tree age structures (which inherently requires trees to be present), not the probability of juvenile presence/absence at each site.

To destructively sample juveniles, soil was excavated and a segment of the stem from at least 10 cm below and 10 cm above the root–shoot boundary was removed. At three points along each transect, shrub cover was estimated in 2 × 3 m plots. Distance to the nearest seed source (i.e., live reproductive tree) of each species was recorded from the center of the transect with a laser range finder. Data collection in CO followed similar protocols, although not enough Douglas-fir juveniles were sampled to be included in this analysis. Additionally, juveniles in CO were defined as trees <150 cm in height, which excluded three trees at two sites from sampling (25).

Dendrochronology. To identify the germination year of juvenile trees with annual accuracy, sample stems were cut into 2.5-cm-long segments; the bottom of each segment was progressively sanded with finer-grit sandpaper (up to 600–1,500 grit; ref. 64) to reveal ring boundaries. Tree rings on each segment were counted at 10–40× with a Nikon SMZ stereomicroscope, and the segment with the most rings, which also corresponded with the first appearance of pith, was used to age the sample (24, 25, 65). We recorded visual marker years, but the young age of the trees precluded more formal cross-dating methods. To test the precision of our aging methods, 555 random samples were recounted by three technicians. The mean (SD) difference in ring-count-based ages among the technicians was 0.298 (0.461) y. If ring boundaries were indistinct or pith dates were otherwise ambiguous, then the sample was not included in the final dataset. In total, we used 2,935 aged juveniles in our analyses (*SI Appendix*, Table S1).

Climate and Biophysical Data. We chose a suite of biophysical variables as potential predictors of tree recruitment, based on their direct effects on seed availability and plant–water relations impacting germination and survival. As potential predictors of seed availability and site conditions suitable for germination, we used the MTBS-calculated fire severity value for the 30 × 30 m pixel including each site [differenced normalized burn ratio (dNBR)] and the field-measured distance to seed source.

Bioclimatic variables included mean summer (June–August) VPD (in kilopascals), maximum surface temperature (in Celsius), mean spring (March–May) soil moisture [volumetric water content (vwc), ratio of water volume to soil volume], mean soil moisture of the driest month (vwc), and climatic water deficit (in millimeters). Bioclimatic variables were calculated from gridded climate data from 1979 to 2015 (55) with a resolution of 250 m at daily or subdaily timescales and then summarized to seasonal or annual values (*SI Appendix, Supplemental Methods*). Soil moisture and maximum land surface temperature (LST) were modeled using the ECH_20 ecohydrology model (66) following methods described by Simeone et al. (31). Soil moisture was characterized at 0–10 cm, the depths reached by young conifer seedling roots.

Data Analysis. We constructed BRT models for each species separately to predict annual recruitment as a function of annual climate during the year of germination and other biophysical variables (67). Annual recruitment was modeled as a binomial process, with "success" defined by annual recruitment rates (no. juveniles $ha^{-1}{\cdot}yr^{-1}$) exceeding a region-specific threshold: the 25th percentile of annual recruitment rates from among all years with recruitment for a given species in a given region. This threshold accounts for varying forest density among regions (*SI Appendix*, Table S1), and we use this value to represent the annual recruitment rate needed for successful "regeneration" at each site. We also conducted our analysis using a 50th percentile threshold and juvenile presence/absence alone, which produced similar results to those reported in the main text (*SI Appendix*, Figs. S4–S7).

Predictors used for the BRT models included both static and dynamic, time-varying variables. Static variables, specific to each site, included distance to seed source and fire severity. Time-varying variables, specific to each year and site combination, included time since fire, mean summer VPD (calculated as a site-specific z score), maximum surface temperature, annual climatic water deficit (calculated as a site-specific z score), mean soil moisture of the driest month, and mean spring soil moisture. We developed an initial BRT model using all of the above predictor variables, and then used the relative influence of each bioclimatic variable to select the most influential moisture-related (i.e., deficit or soil moisture) and energy-related (i.e., VPD or maximum surface temperature) variables; this resulted in a final BRT model with five predictor variables: distance to seed source, fire severity, time since fire, a moisture-related annual bioclimatic variable, and an energy-related annual bioclimatic variable. We used R version 3.3.3 (68), the package "dismo" (69), and the function "gbm.step" to fit BRT models. To account for lack of spatial independence in our observations, each fold in the k-fold cross-validation used by gbm.step included data from one fire (70), resulting in 25 folds for the ponderosa pine model and 23 folds for the Douglas-fir model. We tested predictive performance by leaving out each site, fitting a BRT model with the same settings as above, predicting the holdout site, and then calculating accuracy (defined as the proportion of years with correct prediction) and the AUC statistic.

To examine how the climate suitability for recruitment has changed with shifts in annual climate over the past 35 y, we used the species-specific BRT models to hindcast annual recruitment probability at each site based on the site-specific annual climate time series (1981–2015; *SI Appendix*, Fig. S8) and a constant distance to seed source (50 m), time since fire (1 y), and dNBR value (412, the median across all sites). This median dNBR value loosely corresponds to the median observed fire-related tree mortality of 90%; such high mortality likely resulted from torching of individual trees, active crown fire, or smoldering at the base of trees, given the thick bark of both study species. A range of distance-to-seed-source and dNBR values were initially tested when hindcasting the models; while the mean annual recruitment probability differed based on these initial values, the temporal patterns were consistent among different initial values (*SI Appendix*, Figs. S11 and S12). Thus, because distance to seed source, time since fire, and fire severity were held constant, the modeled annual recruitment probability at each site is a measure of climate suitability for recruitment in the first year after a hypothetical fire. Modeled annual recruitment probabilities for sites within each region were averaged to create a single time series for each region and species combination (Figs. 3*F* and 4*F*). We characterized the temporal variability in the regional annual recruitment probability values by calculating the SD of recruitment probability in consecutive 10-y windows (Figs. 3*E* and 4*E*). We identified significant shifts in the regional time series of annual (year 1 postfire) and cumulative (years 1–5 or 1–10 postfire) (*SI Appendix, Supplemental Methods*) recruitment probabilities using a change-point detection algorithm [Sup(F)] in the "strucchange" package in R (9, 71).

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1. Scheffer M, Carpenter S, Foley JA, Folke C, Walker B (2001) Catastrophic shifts in ecosystems. *Nature* 413:591–596.
2. Bestelmeyer BT, et al. (2011) Analysis of abrupt transitions in ecological systems. *Ecosphere* 2:1–26.
3. Scheffer M, et al. (2009) Early-warning signals for critical transitions. *Nature* 461: 53–59.
4. Carpenter SR, Brock WA (2006) Rising variance: A leading indicator of ecological transition. *Ecol Lett* 9:311–318.

5. Dakos V, van Nes EH, Scheffer M (2013) Flickering as an early warning signal. *Theor Ecol* 6:309–317.
6. Jackson ST, Betancourt JL, Booth RK, Gray ST (2009) Ecology and the ratchet of events: Climate variability, niche dimensions, and species distributions. *Proc Natl Acad Sci USA* 106:19685–19692.
7. Andrus RA, Harvey BJ, Rodman KC, Hart SJ, Veblen TT (2018) Moisture availability limits subalpine tree establishment. *Ecology* 99:567–575.
8. Groffman P, et al. (2006) Ecological thresholds: The key to successful environmental management or an important concept with no practical application? *Ecosystems (NY)* 9:1–13.
9. Andersen T, Carstensen J, Hernández-García E, Duarte CM (2009) Ecological thresholds and regime shifts: Approaches to identification. *Trends Ecol Evol* 24:49–57.
10. Cavanaugh KC, et al. (2014) Poleward expansion of mangroves is a threshold response to decreased frequency of extreme cold events. *Proc Natl Acad Sci USA* 111:723–727.
11. Svenning JC, Sandel B (2013) Disequilibrium vegetation dynamics under future climate change. *Am J Bot* 100:1266–1286.
12. Crausbay SD, Higuera PE, Sprugel DG, Brubaker LB (2017) Fire catalyzed rapid ecological change in lowland coniferous forests of the Pacific Northwest over the past 14,000 years. *Ecology* 98:2356–2369.
13. Gavin D, Brubaker L, Greenwald D (2013) Postglacial climate and fire-mediated vegetation change on the western Olympic Peninsula, Washington (USA). *Ecol Monogr* 83:471–489.
14. Bell DM, Bradford JB, Lauenroth WK (2014) Early indicators of change: Divergent climate envelopes between tree life stages imply range shifts in the western United States. *Glob Ecol Biogeogr* 23:168–180.
15. Dobrowski SZ, et al. (2015) Forest structure and species traits mediate projected recruitment declines in western US tree species. *Glob Ecol Biogeogr* 24:917–927.
16. Davis KT, Higuera PE, Sala A (2018) Anticipating fire-mediated impacts of climate change using a demographic framework. *Funct Ecol* 32:1729–1745.
17. Stevens-Rumann CS, et al. (2018) Evidence for declining forest resilience to wildfires under climate change. *Ecol Lett* 21:243–252.
18. Roccaforte JP, Fulé PZ, Chancellor WW, Laughlin DC (2012) Woody debris and tree regeneration dynamics following severe wildfires in Arizona ponderosa pine forests. *Can J For Res* 42:593–604.
19. Welch KR, Safford HD, Young TP (2016) Predicting conifer establishment post wildfire in mixed conifer forests of the North American Mediterranean-climate zone. *Ecosphere* 7:e01609.
20. Tepley AJ, Thompson JR, Epstein HE, Anderson-Teixeira KJ (2017) Vulnerability to forest loss through altered postfire recovery dynamics in a warming climate in the Klamath Mountains. *Glob Change Biol* 23:4117–4132.
21. Donato DC, Harvey BJ, Turner MG (2016) Regeneration of montane forests 24 years after the 1988 yellowstone fires: A fire-catalyzed shift in lower treelines? *Ecosphere* 7: e01410.
22. Rother MT, Veblen TT (2016) Limited conifer regeneration following wildfires in dry ponderosa pine forests of the Colorado Front Range. *Ecosphere* 7:e01594.
23. Kemp KB, Higuera PE, Morgan P, Abatzoglou JT (2019) Climate will increasingly determine post-fire tree regeneration success in low-elevation forests, Northern Rockies, USA. *Ecosphere* 10:e02568.
24. League K, Veblen T (2006) Climatic variability and episodic *Pinus ponderosa* establishment along the forest-grassland ecotones of Colorado. *For Ecol Manage* 228: 98–107.
25. Rother MT, Veblen TT (2017) Climate drives episodic conifer establishment after fire in dry ponderosa pine forests of the Colorado Front Range, USA. *Forests* 8:159.
26. Williams JW, Shuman BN, Webb T, Bartlein PJ, Leduc PL (2004) Late-quaternary vegetation dynamics in North America: Scaling from taxa to biomes. *Ecol Monogr* 74:309–334.
27. Fisichelli N, et al. (2014) First-year seedlings and climate change: Species-specific responses of 15 North American tree species. *Oikos* 123:1331–1340.
28. Johnson DM, McCulloh KA, Reinhardt K (2011) The earliest stages of tree growth: Development, physiology and impacts of microclimate. *Size- and Age-Related Changes in Tree Structure and Function, Tree Physiology*, eds Meinzer FC, Lachenbruch B, Dawson TE (Springer, Dordrecht, The Netherlands), Vol 4, pp 65–87.
29. Rother MT, Veblen TT, Furman LG (2015) A field experiment informs expected patterns of conifer regeneration after disturbance under changing climate conditions. *Can J For Res* 45:1607–1616.
30. Reinhardt K, Germino MJ, Kueppers LM, Domec JC, Mitton J (2015) Linking carbon and water relations to drought-induced mortality in *Pinus flexilis* seedlings. *Tree Physiol* 35:771–782.
31. Simeone C, et al. (September 27, 2018) Coupled ecohydrology and plant hydraulics modeling predicts ponderosa pine seedling mortality and lower treeline in the US Northern Rocky Mountains. *New Phytol*, 10.1111/nph.15499.
32. Ficklin DL, Novick KA (2017) Historic and projected changes in vapor pressure deficit suggest a continental-scale drying of the United States atmosphere. *J Geophys Res D Atmospheres* 122:2061–2079.
33. Restaino CM, Peterson DL, Littell J (2016) Increased water deficit decreases Douglas fir growth throughout western US forests. *Proc Natl Acad Sci USA* 113:9557–9562.
34. Eamus D, Boulain N, Cleverly J, Breshears DD (2013) Global change-type drought-induced tree mortality: Vapor pressure deficit is more important than temperature per se in causing decline in tree health. *Ecol Evol* 3:2711–2729.
35. Will RE, Wilson SM, Zou CB, Hennessey TC (2013) Increased vapor pressure deficit due to higher temperature leads to greater transpiration and faster mortality during drought for tree seedlings common to the forest-grassland ecotone. *New Phytol* 200: 366–374.
36. McCullough IM, Davis FW, Williams AP (2017) A range of possibilities: Assessing geographic variation in climate sensitivity of ponderosa pine using tree rings. *For Ecol Manage* 402:223–233.
37. Daubenmire RF (1943) Soil temperature versus drought as a factor determining lower altitudinal limits of trees in the Rocky Mountains. *Bot Gaz* 105:1–13.
38. Kolb PF, Robberecht R (1996) High temperature and drought stress effects on survival of *Pinus ponderosa* seedlings. *Tree Physiol* 16:665–672.
39. Hermann RK, Chilcote WW (1965) *Effect of Seedbeds on Germination and Survival of Douglas-Fir* (Forest Research Laboratory, Oregon State University, Corvallis, OR).
40. Seidel KW (1986) Tolerance of seedlings of ponderosa pine, Douglas-fir, grand fir, and Englemann spruce for high temperatures. *Northwest Sci* 60:1–7.
41. Davis KT, Dobrowski SZ, Holden ZA, Higuera PE, Abatzoglou JT (2018) Microclimatic buffering in forests of the future: The role of local water balance. *Ecography* 41:1–11.
42. Feddema JJ, Mast JN, Savage M (2013) Modeling high-severity fire, drought and climate change impacts on ponderosa pine regeneration. *Ecol Modell* 253:56–69.
43. St Clair JB, Mandel NL, Vance-Borland KW (2005) Genecology of Douglas fir in western Oregon and Washington. *Ann Bot* 96:1199–1214.
44. Bansal S, Harrington CA, Gould PJ, St Clair JB (2015) Climate-related genetic variation in drought-resistance of Douglas-fir (*Pseudotsuga menziesii*). *Glob Change Biol* 21: 947–958.
45. Shinneman DJ, Means RE, Potter KM, Hipkins VD (2016) Exploring climate niches of ponderosa pine (*Pinus ponderosa* Douglas ex Lawson) haplotypes in the Western United States: Implications for evolutionary history and conservation. *PLoS One* 11: e0151811.
46. Kemp KB, Higuera PE, Morgan P (2016) Fire legacies impact conifer regeneration across environmental gradients in the U.S. Northern Rockies. *Landsc Ecol* 31:619–636.
47. Stevens JT, Collins BM, Miller JD, North MP, Stephens SL (2017) Changing spatial patterns of stand-replacing fire in California conifer forests. *For Ecol Manage* 406: 28–36.
48. Montes-Helu MC, et al. (2009) Persistent effects of fire-induced vegetation change on energy partitioning and evapotranspiration in ponderosa pine forests. *Agric For Meteorol* 149:491–500.
49. Certini G (2005) Effects of fire on properties of forest soils: A review. *Oecologia* 143: 1–10.
50. Moyes AB, Castanha C, Germino MJ, Kueppers LM (2013) Warming and the dependence of limber pine (*Pinus flexilis*) establishment on summer soil moisture within and above its current elevation range. *Oecologia* 171:271–282.
51. Moreira X, Abdala-Roberts L, Linhart YB, Mooney KA (2015) Effects of climate on reproductive investment in a masting species: Assessment of climatic predictors and underlying mechanisms. *J Ecol* 103:1317–1324.
52. Eis S (1973) Cone production of Douglas fir and Grand fir and its climatic requirements. *Can J For Res* 3:61–70.
53. IPCC (2014) Climate change 2014: AR5 synthesis report. Contribution of working groups I, II and III to the fifth assessment report of the intergovernmental panel on climate change, eds Pachauri RK, Meyer LA (IPCC, Geneva), p 151.
54. Parks SA, et al. (2018) High-severity fire: Evaluating its key drivers and mapping its probability across western US forests. *Environ Res Lett* 13:044037.
55. Holden ZA, et al. (2018) Decreasing fire season precipitation increased recent western US forest wildfire activity. *Proc Natl Acad Sci USA* 115:E8349–E8357.
56. Westerling AL (2016) Increasing western US forest wildfire activity: Sensitivity to changes in the timing of spring. *Philos Trans R Soc Lond B Biol Sci* 371:20150178, and erratum (2016) 371:20160373.
57. Abatzoglou JT, Williams AP (2016) Impact of anthropogenic climate change on wildfire across western US forests. *Proc Natl Acad Sci USA* 113:11770–11775.
58. Davis KT, et al. (2019) Data from "Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration." Dryad. Available at https://doi.org/10.5061/dryad.pc3f9d8. Deposited February 5, 2019.
59. Eidenshink J, et al. (2007) A project for monitoring trends in burn severity. *Fire Ecol* 3: 3–21.
60. Baker WL (2009) *Fire Ecology in Rocky Mountain Landscapes* (Island Press, Washington, DC), p 544.
61. Sugihara NG, Van Wagtendonk JW, Fites-Kaufman J, Shaffer KE, Thode AE (2006) *Fire in California's Ecosystems* (Univ California Press, Berkeley, CA), p 596.
62. Agee JK (1993) *Fire Ecology of Pacific Northwest Forests* (Island Press, Washington, DC), p 493.
63. Hankin LE, Higuera PE, Davis KT, Dobrowski SZ (2018) Accuracy of node and bud-scar counts for aging two dominant conifers in Western North America. *For Ecol Manage* 427:365–371.
64. Speer JH (2010) *Fundamentals of Tree-Ring Research* (Univ. Arizona Press, Tucson, AZ), p 333.
65. Telewski FW (1993) Determining the germination date of woody plants: A proposed method for locating the root/shoot interface. *Tree Ring Bull* 53:13–16.
66. Maneta MP, Silverman NL (2013) A spatially distributed model to simulate water, energy, and vegetation dynamics using information from regional climate models. *Earth Interact* 17:1–44.
67. Elith J, Leathwick JR, Hastie T (2008) A working guide to boosted regression trees. *J Anim Ecol* 77:802–813.
68. R Core Team (2017) R: A Language and Environment for Statistical Computing Version 3.3.3 (R Foundation for Statistical Computing, Vienna).
69. Hijmans RJ, Phillips S, Leathwick J, Elith J (2017) dismo: Species Distribution Modeling. R Package Version 1.1-4. Available at https://CRAN.R-project.org/package=dismo. Accessed February 26, 2019.
70. De'ath G (2007) Boosted trees for ecological modeling and prediction. *Ecology* 88: 243–251.
71. Zeileis A, Leisch F, Hornik K, Kleiber C (2002) strucchange: An R package for testing for structural change in linear regression models. *J Stat Softw* 7:1–38.
72. Little EL (1971) *Atlas of United States Trees* (US Department of Agriculture, Forest Service, Washington, DC).