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# Evaluation of potential effects of federal land management alternatives on trends of salmonids and their habitats in the interior Columbia River basin

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## Abstract

Aquatic species throughout the interior Columbia River basin are at risk. Evaluation of the potential effects of federal land management on aquatic ecosystems across this region is an important but challenging task. Issues include the size and complexity of the systems, uncertainty in important processes and existing states, flexibility and consistency in the analytical framework, and an ability to quantify results. We focused on salmonid fishes and their habitats as indicators of conditions in aquatic ecosystems and used Bayesian belief networks as a formal, quantitative framework to address the issues in our evaluation of land management alternatives proposed for the interior Columbia River basin. Because empirical information is limited at the scales relevant to our analysis, an ability to combine both empirical and more subjective information was key to the analysis. The representation of linkages through conditional probabilities made uncertainty explicit. We constructed two general networks. One represented the influence of landscape characteristics and existing and predicted management activities on aquatic habitats. A second represented the influence of habitat, existing biotic conditions, and for two anadromous species, ocean and migratory conditions, on the status of six widely distributed salmonid fishes. In the long term (100 years) all three land management alternatives were expected to produce positive changes in the status and distribution of the salmonids and their habitats. Trends were stronger for habitat than for the status of salmonids because of greater uncertainty in linking the fish and habitat networks and constraints outside spawning and rearing habitat on federal lands in the study area. Trends were stronger for resident salmonids than anadromous forms because of additional effects of the migratory corridor assumed for the latter. Alternative S2, which approached ecosystem restoration more conservatively, generally produced the strongest positive changes, and alternative S3, designed to promote more aggressive restoration, the weakest. Averaged across the basin, differences among the alternatives were small. Differences were greater at finer temporal and spatial scales. In the short term (10 years) alternative S3 was expected to lead to further degradation in some areas. By formalizing our understanding and assumptions in these networks, we provided a framework for exploring differences in the management alternatives that is more quantifiable, spatially explicit, and flexible than previous approaches. Published by Elsevier Science B.V.

*Keywords:* Bayesian belief network; Fish; Salmon; Trout; Columbia River basin; Aquatic habitat

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## 1. Introduction

The interior Columbia River basin landscape (hereafter referred to as Basin) has been substantially altered by humans over the past century, causing dramatic changes and declines in many native fish populations. Of 88 native fishes found in the Basin, 45 are now listed as threatened, endangered, sensitive, or otherwise of special concern by the agencies responsible for their management (Lee et al., 1997). These conditions led federal land managers to address conservation and restoration of aquatic habitat as part of comprehensive long-term land management. Evaluation of these long-term strategies is important to effective conservation planning.

Evaluation of land management plans in a region the size of the Basin is a daunting task. Such analyses are complicated by uncertainty in ecological processes, in implementation of multiple and often conflicting management objectives, and in the sheer size and diversity of the region. Currently, several efforts to develop tools to assist decision making involving aquatic resources in the Basin are underway (e.g., Mobrand and Kareiva, unpublished report, http://www.nwfsc.noaa.gov/cri/pdf_files/reconciling.pdf). The analytical processes are often complicated, computationally intensive, and limited by available data.

Past efforts to evaluate federal land management proposals consisted of arraying available information (e.g., projected activity levels, fish or habitat distribution and status) before one or more experts and then having them formulate opinions about the likely trend of populations across the affected range (e.g., Sedell et al., 1997). This approach proved to be unsatisfying for several reasons: (1) the huge body of information made it difficult to conceptualize and account for multiple interacting effects; (2) the influence of the experts' assumptions on results of analyses could not be evaluated; (3) the analyses could not be revisited or updated easily and consistently when management direction or key assumptions were modified; and (4) the results were not quantified or spatially explicit.

To address these difficulties, we used Bayesian belief networks (hereafter, BBNs or networks) for analysis of management plans outlined in a supplemental draft environmental impact statement (SDEIS) for Forest Service (FS) and Bureau of Land Management (BLM) lands in the Basin. Marcot et al. (2001) provide a more detailed description of the BBN approach. Briefly BBNs are a series of nodes representing states of nature and the causal dependencies among them (Marcot et al., 2001). The probabilities representing those linkages can be developed empirically or through expert judgement (Reckhow, 1999; Marcot et al., 2001). A distinct advantage of this approach is that BBNs do not have to incorporate the complete mechanistic detail of more process-based models (Reckhow, 1999). Uncertainty in ecological process or limited information is reflected in the vector of conditional probabilities for linkages that are defined.

We developed BBNs that represented current understanding and available information for what we believe are the key processes linking aquatic ecosystems and land management activities. The resulting networks included three basic elements: land management effects, landscape context, and biotic interactions. Our intent was to develop a consistent and transparent interpretation of likely system responses. We did not intend to provide new insights about ecological structure, interactions, or competing theories of management.

Analyses were conducted to examine the influence of three alternative land management strategies described in the SDEIS (S1–S3). Briefly these were the continuation of current management practices and direction (e.g., existing forest plans, biological opinions related to Endangered Species Act) under alternative S1; an increased focus on restoration of aquatic and terrestrial systems combined with a conservative conservation direction (alternative S2); and an accelerated and more aggressive approach to terrestrial restoration, particularly to benefit isolated communities economically dependent on federal lands (alternative S3). The aquatic management elements of alternatives S2 and S3 consisted of riparian conservation areas, subbasin and watershed analyses to identify finer scale restoration needs and minimize short-term risks of management, and a set of subwatersheds (i.e., ''A1'' and ''A2'') to conserve existing strong populations, genetically pure anadromous populations, ''fringe'' populations, and selected anadromous populations in the Snake River basin. Fringe populations have restricted distributions, are found on the margins of the species range, and may be of particular evolutionary significance (Lee et al., 1997). All of these elements were reduced under alternative S3. Detailed descriptions of these alternatives can be found in US Department of Agriculture and US Department of the Interior (USDA and USDI, 2000).

Our results described expected trends in aquatic habitat capacity and fish population status for each alternative. Our networks were based on professional opinion, empirical observations, and the predictions of both empirical and process-based models (see Appendix A). The goal of our study, therefore, was to describe the relative influence of the SDEIS alternatives on aquatic habitats and fish populations, not to predict the absolute numbers or locations of habitats or fish populations within the classes described. The BBNs we developed represented our (the authors and experts who contributed information or estimated probabilities) collective beliefs about these systems and how they will respond to future management. We view them as a means of organizing and synthesizing a large amount of information efficiently and consistently given those beliefs.

## 2. Methods

### *2.1. Overview*

Our analyses focused on six salmonid fishes identified in Lee et al. (1997) and their habitats. Salmonid conservation does not represent the only issue in the management of aquatic ecosystems within the Basin. However, the salmonids are, in our view, the best indicators of the condition and function of these systems given available information and the scale of the analysis (Lee et al., 1997).

The BBNs were constructed to use information with a resolution of 6th code hydrologic units (or ‘‘subwatersheds’’), which are nested within larger 4th code hydrologic units (or ‘‘subbasins’’) (Fig. 1; and see Maxwell et al., 1995). The BBNs were implemented for each subwatershed or collection of subwatersheds of interest to provide a spatially explicit result in habitat and fish population trends.

We considered effects of the management alternatives on habitat on all lands, but for salmonids we focused only on the potential spawning and rearing areas. We excluded consideration of areas classified only as corridors or seasonal habitats for two reasons: (1) spawning and rearing habitats are the critical areas found predominantly on federal land and are the habitats most sensitive to federal land-use management; and (2) spawning and rearing areas are more likely to be in headwater systems; we have poor understanding and ability to predict the influence of multiple effects over very large and complex catchments contributing to downstream habitats. We do not imply that federal land management does not influence more downstream areas, but that its effects will be more evident and predictable in spawning and rearing areas.

Predictions of the landscape conditions used as inputs for our networks were made for current, 10-year, and 100-year points of evaluation. Stream habitats and fish populations are dynamic. They are continually responding to changes in landscapes as a result of management and natural processes. Some responses may occur quickly, whereas others may lag years or decades behind the changes on the landscape (Swanston, 1991). For the purposes of our evaluation, we assumed that landscape conditions and management effects at the evaluation points initiated the watershed responses that would influence habitat and populations; we also assumed that initiating or controlling conditions did not change after that point. This is equivalent to a population viability analysis that extrapolates to some future state assuming that initial conditions are valid throughout the period (Lee and Rieman, 1997). Obviously, conditions can change, but we cannot be certain how such change will occur.

### *2.2. Structure of BBNs*

We identified the physical and biological processes we believed most strongly influence distribution and dynamics of the salmonids and their habitats. Our BBNs were characterized by a collection of components (nodes) that represented environmental states or processes and two variables of primary interest, aquatic habitat capacity and the future status of each salmonid (Fig. 2).

Within the geomorphic constraints of any watershed, upslope disturbances (e.g., logging, roads, fire) that cause accelerated production of sediment, alter hydrologic regimes, or alter the characteristics of riparian areas are generally accepted as primary drivers influencing the potential capacity of habitats for

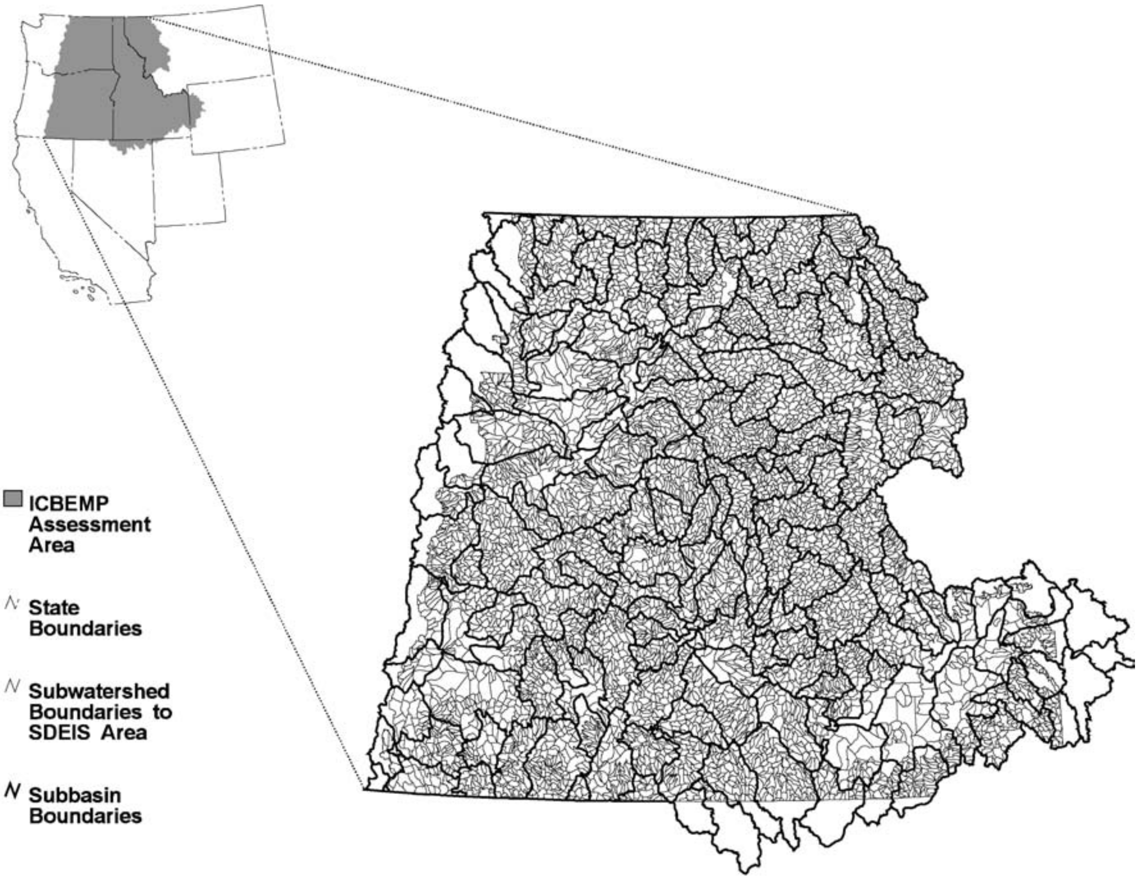

Fig. 1. The interior Columbia River basin analysis area. Subwatersheds (sixth code hydrologic units), which were the basic unit of analysis are shown within their encompassing subbasins.

fishes in wildland systems (Beechie and Bolton, 1999; Kauffman et al., 1997; and see papers in Meehan, 1991; Naiman, 1992). In our network aquatic habitat capacity depended on: (1) generation and delivery of sediment; (2) the occurrence of large channel reorganizing floods; and (3) the condition or integrity of the riparian corridor (Fig. 2a). Sediment was defined as the relative amount of sediment entering streams above natural rates. It was influenced by *road density*, *ground disturbance* (i.e., logging, thinning and prescribed fire), topographic conditions (*slope steepness*), and management activities designed to mitigate erosion or sediment delivery (*standards and guides*). Hydrologic effects included the probability of flood/debris-flow events that could reorganize large portions of the steam network. These events were influenced by slope angle (*slope 2*) and the combined probability of a large fire followed by a flood generating storm (*fire-rain*). Riparian condition was viewed as those characteristics influencing shading, climate moderation, bank stabilization, water storage, and delivery of coarse and fine organic material. It was influenced by *prior riparian condition*, *future grazing*, and management activities intended to conserve or restore riparian function (*standards and guides*).

Salmonids can be strongly influenced by the physical capacity and quality of habitat. The status of many populations, however, is not strictly a function of local habitat conditions. Introductions of exotic species (e.g., Li and Moyle, 1981; Moyle et al., 1986) or the loss of some ‘‘keystone’’ forms such as the anadromous salmonids (Willson and Halupka,

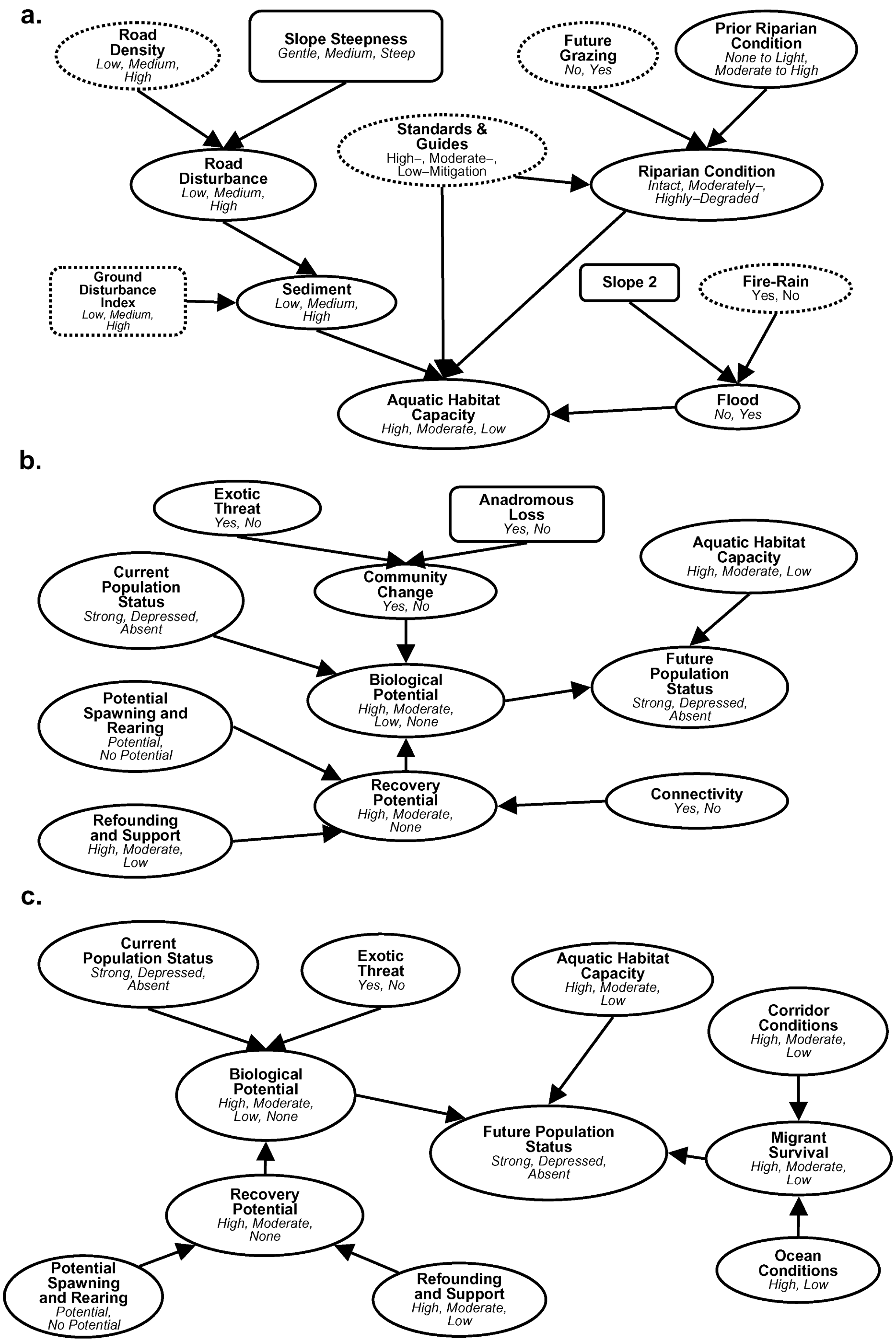

Fig. 2. BBNs for estimation of (a) aquatic habitat capacity and future population status for (b) resident and (c) anadromous salmonids for subwatersheds under all alternatives. Definitions of each node and a brief summary of the source information are in Appendix A. Dashed nodes represent inputs projected from land management activities as described by Hemstrom et al. (2001).

1995) may have profound effects. Larger scale habitat fragmentation and isolation from surrounding populations can also be important (e.g., Dunham and Rieman, 1999; Dunning et al., 1992). For these reasons the future status of the salmonids in each subwatershed was conditioned by one or two factors that were independent of habitat capacity (Fig. 2b and c). The future status of resident salmonids (bull trout *Salvelinus confluentus*, Yellowstone and westslope cutthroat trout *Oncorhynchus clarki bouvieri* and *O.c. lewisi*, redband trout *Oncorhynchus mykiss gibbsi*) depended on the *biological potential* of the existing population. In addition, status of anadromous salmonids (stream-type chinook salmon *Oncorhynchus tshawytscha*, steelhead *Oncorhynchus mykiss mykiss*) depended on conditions of the river corridors and ocean environments they migrate through (*migrant survival*) to complete their life cycle (e.g., NRC, 1996; Williams et al., 1996). Biological potential represented biological constraints on population resilience, productivity, and size that are associated with the subwatershed. These depended on current condition of the population (*current population status*), the biotic effects associated with introduction or loss of members of the associated fish community (*exotic threat*, *anadromous loss*) and potential for demographic support from surrounding populations (*refounding and support*, *connectivity*). *Migrant survival* depended both on conditions in the migratory corridor (i.e., number of dams that must be passed by migrating fish) and in the ocean.

Conditional dependencies in the aquatic habitat and population status networks were estimated using a combination of expert judgement and empirical relationships. Two sets of conditional probabilities were developed in the habitat network to represent the 10- and 100-year points of evaluation. The conditional probabilities linked with salmonid status were estimated independently for each species. For many of the key nodes we relied on multiple experts and averaged their estimated probabilities to reflect the relative uncertainty in collective beliefs. To provide the best possible synthesis of current understanding of the relevant processes, we engaged a number of scientists and biologists noted for their work in this region. Any disagreement among these experts produced a more uniform distribution of probabilities across states reflecting greater uncertainty in the conditional dependencies. Definitions for each node and a brief summary of information used to estimate the associated conditional probabilities are given in Appendix A.

### 2.3. Available information

Existing syntheses of landscape characteristics (Jensen et al., 1997), fish assemblages (Lee et al., 1997), and an interpretation of planned management activities based on the alternatives in the SDEIS (e.g., Hemstrom et al., 2001) represented the primary information available for our analyses. The biophysical coverages summarized to subwatersheds were obtained from the scientific assessment prepared for the project (Quigley and Arbelbide, 1997). Predictions of the land management activities (Hemstrom et al., 2001; Hemstrom et al., 2000) included estimates of road density, mechanical ground disturbance, livestock grazing, and probability of large wildfire, for the current condition and at 10 and 100 years from current under each alternative. In addition, we developed our own series of rules to assign a level of aquatic conservation and restoration (i.e., high, moderate, low; see Appendix A) in each subwatershed based on the management direction in the SDEIS.

All inputs for the networks were summarized from subwatershed (e.g., species status and distribution) or finer resolution (landscape data derived at 1 km pixel) information. Variables represented by the nodes in our network then were viewed as conditions representative of entire subwatersheds. Our summaries were based on mean probabilities for and counts of subwatersheds expected to be in a particular state.

### 2.4. Sensitivity analysis

Prior to our analysis of the management alternatives, we examined characteristics of the BBNs via sensitivity analysis. Sensitivity analysis was used to identify network components that have the greatest influence on the outcomes of interest (i.e., aquatic habitat capacity, future status). Sensitivity analysis was conducted by systematically varying the values of one of the network components to determine effects on aquatic habitat capacity or future population status. Although these two components consisted of three states, we restricted our summary to *high* aquatic habitat capacity and *strong* future population status because previous work indicated that divergent states, such as *high* vs. *low* habitat capacity, tended to be equally sensitive to the same network components.

The ultimate importance of input variables to the analysis depended both on the relative sensitivity to those inputs and the actual estimated variation in inputs among subwatersheds and alternatives. To identify the most influential variables, we used correlation analysis to associate changes in probabilities in all management driven inputs with the estimated probabilities for habitat capacity outputs.

### *2.5. Evaluation of the alternatives*

Three sets of analyses were used to examine the relative influence of SDEIS alternatives on expected trends for aquatic habitat and salmonid populations. In the first set, we examined trends in aquatic habitat capacity and salmonid population status in relation to current for 10-year and 100-year conditions predicted for the three alternatives. For habitat, we limited our summary to counts of subwatersheds expected to have high habitat capacity and the mean probabilities of high habitat capacity. For salmonid status, we included summaries both for strong and ''present''(the sum of the probabilities for strong and depressed). The latter summary was included to provide better resolution of trends for anadromous salmonids because there are very few existing strong populations.

In the second and third sets, we revisited key issues and assumptions related to environmental conditions that can be directly influenced by federal land management (i.e., habitat) and those that cannot (e.g., exotic species and biotic interactions, dams). The nature of our BBNs made it difficult to interpret the relative magnitude of the trends we observed. For example, did a given increase in the number of subwatersheds expected to support strong populations of bull trout represent major or minor progress? In the second scenario, we estimated the ''maximum'' possible change in salmonid status given the constraints of our BBNs. The maximum was estimated by assuming that all disruptive effects of current and future land management were removed from federal lands. We used the results to compare the change in results expected with any alternative relative to what was hypothetically possible given the constraints in biological potential and migrant survival.

In the third scenario, we considered the confounding effects of mainstem hydroelectric dams in the Snake River on populations of steelhead and stream-type chinook in the Snake River basin. In our network for the anadromous salmonids, we assumed that migrant survival strongly influenced future status. We also assumed that migrant survival was strongly dependent on the number of mainstem dams in the migratory corridor. Because status is so strongly linked to the corridor, it was possible that the effects of dams may have masked the potential benefits associated with each alternative. For this scenario, we assumed that dams in lower Snake River corridor did not exist and kept all other inputs the same as the original alternatives.

## 3. Results

### *3.1. Sensitivity analysis*

Although the patterns varied among the different parameterizations of the BBNs (e.g., 10 vs. 100 years, redband trout vs. bull trout), the differences were not dramatic. We present four examples to illustrate the range of results (Fig. 3). Aquatic habitat capacity was most sensitive to riparian condition, sediment, and the occurrence of large floods. Of the nodes directly influenced by management activities (standards and guides, ground disturbance, fire-rain), standards and guides had a substantially larger influence on our expectations for habitat capacity than the others (Fig. 3). In general, the aquatic habitat network reflected a collective belief that conditions of riparian corridors and their management (e.g., riparian protection defined by ''standards and guides'') would strongly influence the condition of habitats.

Biological potential and aquatic habitat capacity had similar influences on resident salmonid status (Fig. 3). Current population status was clearly the most influential component within biological potential. For anadromous salmonids migrant survival was expected to be more important than either biological potential or habitat capacity, although the differences among those three were not large (Fig. 3). The salmonid networks reflected a collective belief that no single element would dominate (or assure) the future status of these species.

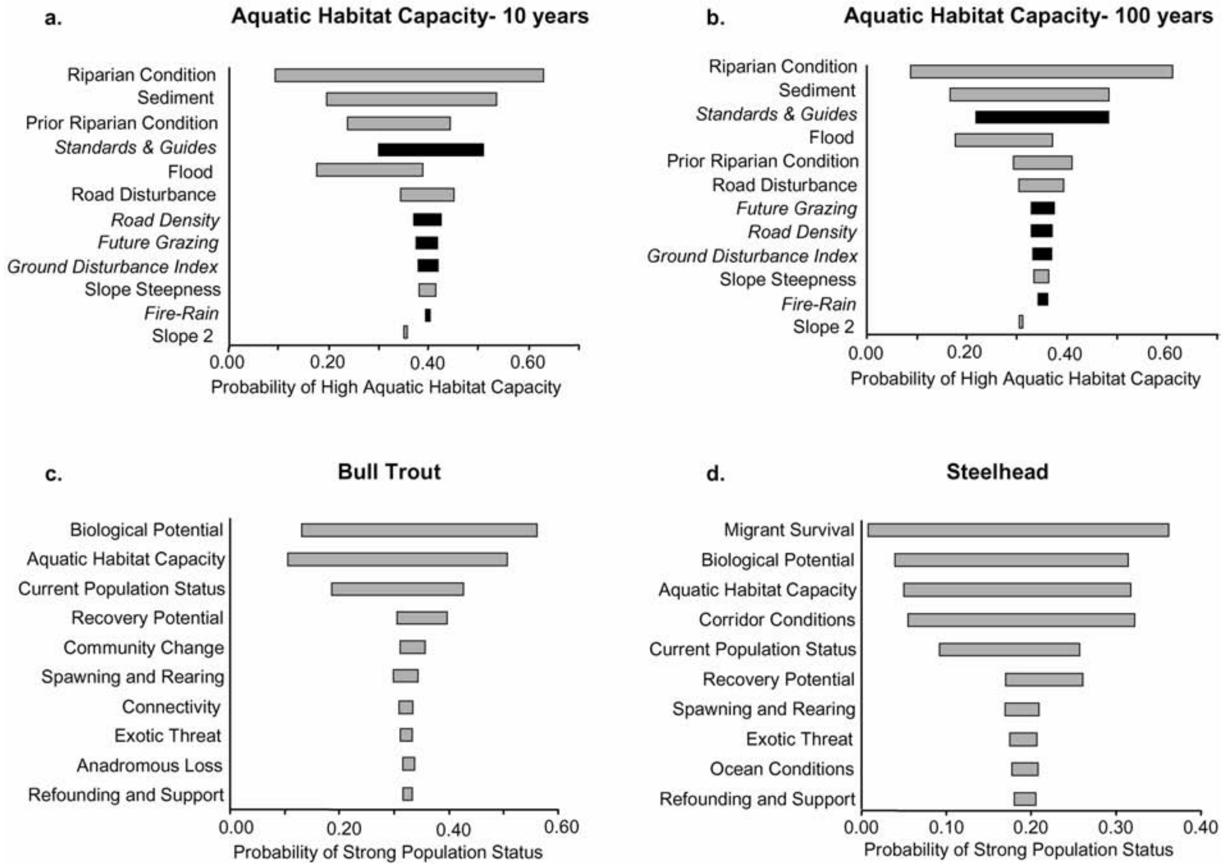

Fig. 3. Sensitivity of the mean probability for high aquatic habitat capacity at (a) 10 years and (b) 100 years and for *strong* population status for (c) bull trout and (d) steelhead to changes in nodes of the BBNs. Nodes are listed from most (top) to least influential. The bars represent the range of variation observed in the habitat when values for the states in each node on the *y*-axis were varied over their possible ranges and all other input nodes were held constant at the values associated with current conditions. Black bars represent inputs influenced by land management activities addressed in the interior Columbia basin SDEIS.

The correlation analysis (Table 1) showed that changes in standards and guides were consistently and most strongly associated with the projected responses. Changes in grazing, ground disturbance and road density were occasionally, but less strongly, associated with the response depending on the alternative and point of evaluation (Table 1).

## *3.2. Analysis of alternatives*

We used a variety of strata reflecting land ownership/management (e.g., BLM, FS) to consider patterns in aquatic habitat and salmonid status resulting from predicted differences in current and future management.

### *3.2.1. Habitat*

The most likely state (MLS) in aquatic habitat capacity under current conditions was high in 33% of federal land in the Basin but varied considerably by ownership and management category (Table 2). Nearly half (49%) of the current FS land was classified as high, in contrast to about 4% of BLM land. Virtually all designated wilderness (99.9%) and about 38% of the non-wilderness FS lands were classified as high. Mean probabilities of high aquatic habitat capacity ranged from 0.53 in wilderness to 0.27 in BLM lands. Federal land as a whole was 0.33.

Predicted changes in aquatic habitat capacity at 10 years were relatively small (Table 2). Changes were

Table 1
Pearson correlations between the changes in the probability of high aquatic habitat capacity and probabilities for the inputs for the 10- (top) and 100-year (bottom) estimates for three alternatives addressed in the SDEIS[a]

| Input node | State | Alternative | | |
|---|---|---|---|---|
| | | S1 | S2 | S3 |
| *10-year estimates* | | | | |
| Road density | Low | −0.0062 | 0.0044 | 0.0153 |
| | High | 0.0054 | 0.0039 | −0.0090 |
| Future grazing | Yes | −0.2561 | *−0.3083* | −0.2497 |
| Standards and guides | High | *0.6458* | *0.8905* | *0.9332* |
| | Low | *−0.6512* | *−0.9371* | *−0.9451* |
| Ground disturbance | Low | *0.5546* | 0.0941 | 0.0985 |
| | High | −0.1740 | −0.0426 | −0.0355 |
| Fire-rain | Yes | −0.2604 | −0.0646 | −0.0533 |
| *100-year estimates* | | | | |
| Road density | Low | 0.0074 | 0.0007 | 0.0003 |
| | High | −0.2010 | *−0.3210* | −0.1699 |
| Future grazing | Yes | 0.1138 | −0.2127 | −0.1183 |
| Standards and guides | High | *0.3430* | *0.5458* | *0.6203* |
| | Low | *−0.3503* | *−0.7056* | *−0.7487* |
| Ground disturbance | Low | 0.0752 | 0.0492 | 0.0358 |
| | High | 0.0121 | 0.0075 | 0.0035 |
| Fire-rain | Yes | −0.0220 | −0.0529 | −0.0206 |

[a] The estimates are summarized for all federal land to show the relative strength of the associations between input and output variables. Correlations of $r \geq 0.30$ are italicized.

generally positive under alternatives S1 (status quo) and S2 (active but cautious), but negative under S3 (active and aggressive). For non-wilderness FS land the decline from high under S3 included about 13% (89 of 703 subwatersheds) of the subwatersheds.

Because the changes at 10 years were minor, we focused our interpretation on the results at 100 years. All alternatives were projected to increase aquatic habitat capacity across federal lands, based on both number of subwatersheds expected as high aquatic habitat capacity and mean probability of high capacity (Table 2). The magnitude of the increase, however, varied among alternatives and strata of ownership/management. Alternative S2 resulted in the largest number of subwatersheds classified as high and highest probabilities of high for all ownership/management strata. Alternative S3 had the lowest expectation for high in all strata except BLM land. The smallest increases in aquatic habitat capacity were projected for wilderness. This was due in part to the currently high probabilities of high aquatic habitat capacity for wilderness and to the limited range of management options that might actually affect aquatic habitat in wilderness.

The largest increases in aquatic habitat capacity were expected on BLM and non-wilderness FS lands where substantial opportunities for restorative management exist. Increases in the probabilities were greater than 20% over the current projected conditions for both BLM and non-wilderness FS lands at 100 years (Table 2). The number of subwatersheds classified as high doubled at 100 years for non-wilderness FS and increased by more than 10-fold on BLM land under alternative S2.

### 3.2.2. Salmonids

We summarized our results for future salmonid status by species for all lands and for federal lands alone (Table 3). Although the patterns varied among the six salmonids, the results were generally similar at these broad scales.

For five of the six species, federal lands contained most of the subwatersheds representing potential habitat (Table 3). Federal lands contained about 21% of expected occupied habitat for Yellowstone cutthroat trout. Management of federal lands can be expected to have important implications for status of all of these fishes, but it will be particularly important for bull trout (66%), stream-type chinook salmon (70%) and westslope cutthroat trout (69%).

Changes between species status at current and 10 years for each alternative were always small and differed little among alternatives; therefore, we focused our results on the 100-year predictions. Predictions for each species across all alternatives showed increases in both the counts and probabilities for strong and present status. Patterns varied among species, but when summarized across the entire Basin, alternative S2 generally produced the strongest change and alternative S3 the weakest change (Table 3). The mean increases associated with one alternative never exceeded that of another by more than 20–30% for the resident forms and was negligible for anadromous forms (Table 3).

Increases in expected status between current and 100 years were smaller than observed in the habitat predictions. The magnitude of change ranged from about 4 to 10% in the mean probabilities and 7–16% in MLS counts for fish status (relative to about 20% in

Table 2
Counts of subwatersheds with a MLS of high habitat capacity and mean probabilities for high habitat capacity for current 10-, and 100-year estimates for three alternatives in the SDEIS[a]

| Land management strata | Variable | *N* | Current | 10-year |  |  | 100-year |  |  |
|---|---|---|---|---|---|---|---|---|---|
|  |  |  |  | S1 | S2 | S3 | S1 | S2 | S3 |
| Federal land[b] | MLS count of "high" | 3558 | 1189 | 1233 | 1207 | 1105 | 2254 | 2429 | 1948 |
|  | Mean probability of high class |  | 0.332 | 0.333 | 0.339 | 0.333 | 0.385 | 0.396 | 0.381 |
| FS | MLS count of "high" | 2301 | 1131 | 1175 | 1137 | 1044 | 1871 | 1912 | 1549 |
|  | Mean probability of high class |  | 0.365 | 0.366 | 0.370 | 0.362 | 0.421 | 0.427 | 0.411 |
| Wilderness (FS) | MLS count of "high" | 430 | 428 | 428 | 430 | 430 | 430 | 430 | 430 |
|  | Mean probability of high class |  | 0.534 | 0.534 | 0.536 | 0.536 | 0.573 | 0.575 | 0.575 |
| Non-wilderness (FS) | MLS count of "high" | 1871 | 703 | 747 | 707 | 614 | 1441 | 1482 | 1119 |
|  | Mean probability of high class |  | 0.326 | 0.327 | 0.332 | 0.323 | 0.386 | 0.393 | 0.373 |
| BLM | MLS count of "high" | 1135 | 47 | 46 | 58 | 52 | 345 | 475 | 37 |
|  | Mean probability of high class |  | 0.273 | 0.273 | 0.284 | 0.279 | 0.319 | 0.339 | 0.327 |

[a] The estimates were summarized by five strata representing ownership or management history. *N* is the total number of subwatersheds in the strata.

[b] Includes FS, BLM, and other federal lands (e.g., national parks).

Table 3
Counts of subwatersheds with a MLS of strong and present future status and mean probabilities for strong and present future status for six salmonids on federal lands[a]

| Species | Variable | *N* | Strong | | | | | Present | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Current | 100-year | | | | Current | 100-year | | | |
| | | | | Maximum | S1 | S2 | S3 | | Maximum | S1 | S2 | S3 |
| Bull trout | MLS count | 1860 | 310 | 423 | 352 | 352 | 347 | 1069 | 1139 | 1099 | 1100 | 1089 |
| | Mean probability | | 0.183 | 0.219 | 0.198 | 0.198 | 0.195 | 0.451 | 0.494 | 0.468 | 0.467 | 0.463 |
| Westslope cutthroat trout | MLS count | 1527 | 459 | 547 | 503 | 500 | 490 | 1289 | 1300 | 1293 | 1292 | 1291 |
| | Mean probability | | 0.289 | 0.340 | 0.308 | 0.309 | 0.304 | 0.627 | 0.680 | 0.645 | 0.645 | 0.639 |
| Yellowstone cutthroat trout | MLS count | 77 | 11 | 22 | 14 | 15 | 15 | 50 | 50 | 50 | 50 | 50 |
| | Mean probability | | 0.215 | 0.276 | 0.224 | 0.235 | 0.231 | 0.561 | 0.596 | 0.566 | 0.570 | 0.568 |
| Redband trout | MLS count | 1896 | 497 | 855 | 649 | 674 | 627 | 1335 | 1376 | 1346 | 1347 | 1340 |
| | Mean probability | | 0.266 | 0.319 | 0.283 | 0.287 | 0.282 | 0.580 | 0.608 | 0.588 | 0.590 | 0.587 |
| Stream-type chinook salmon | MLS count | 751 | 2 | 7 | 4 | 5 | 5 | 50 | 50 | 50 | 50 | 50 |
| | Mean probability | | 0.053 | 0.063 | 0.057 | 0.058 | 0.057 | 0.202 | 0.215 | 0.207 | 0.208 | 0.207 |
| Steelhead | MLS count | 804 | 6 | 15 | 14 | 14 | 14 | 101 | 101 | 101 | 101 | 101 |
| | Mean probability | | 0.104 | 0.122 | 0.111 | 0.112 | 0.111 | 0.325 | 0.342 | 0.331 | 0.332 | 0.331 |

[a] Results are shown for current-, and 100-year estimates for three alternatives in the SDEIS. Results are also shown for the hypothetical maximum possible assuming that land management effects could be eliminated. *N* is the total number of subwatersheds within the potential range for each salmonid.

habitat). The difference in the strength of response between habitat and status summaries was due to attenuation in the networks with the additional link and confounding effects of other components that influence status.

Although differences among alternatives were not large across the entire Basin, differences were clearly stronger for some species than others. This was, in part, the result of the spatial stratification that occurred in the summaries for each species. In other words, results of each alternative depended on the resolution in the summaries. To illustrate this point, we summarized trends in habitat capacity among alternatives for different land management classes, species distributions or current species conditions (Fig. 4). Results showed that the alternatives may play out quite differently even within a species depending on where in the landscape we consider the effects. The relative benefits for alternative S2, for example, were obviously accentuated in the subwatersheds that were included in the aquatic conservation network and received additional emphasis for habitat protection and restoration (e.g., A2 subwatersheds). Alternative S1, however, was expected to produce greater benefits for bull trout populations that are currently depressed.

Differences among alternatives also became apparent in the spatially explicit representation of basin wide maps. Expected improvements in bull trout status were spread relatively evenly across the species distribution in alternative S1 but showed stronger or

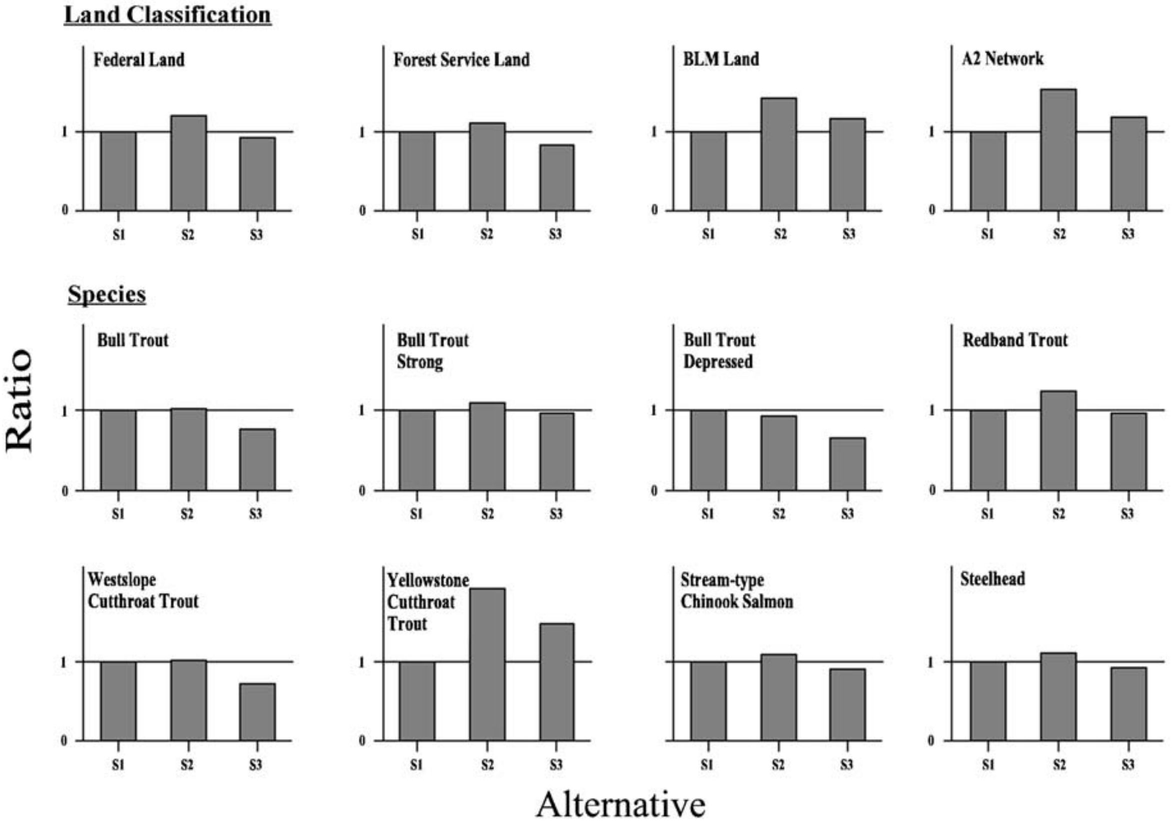

Fig. 4. Relative increase in the mean probability of high habitat capacity for subwatersheds associated with several different land classes or species distributions for the three management alternatives at 100 years. The relative increase was estimated as the ratio of the net increase from current or base relative to alternative S1. "A2" are subwatersheds selected for an aquatic emphasis in alternatives S2 and S3. Subwatersheds associated with different parts of the existing bull trout distribution are displayed to show differences that result because of the combined variation in species status, management direction and landscape characteristics. Summaries for other species cover the complete distribution without reference to current status.

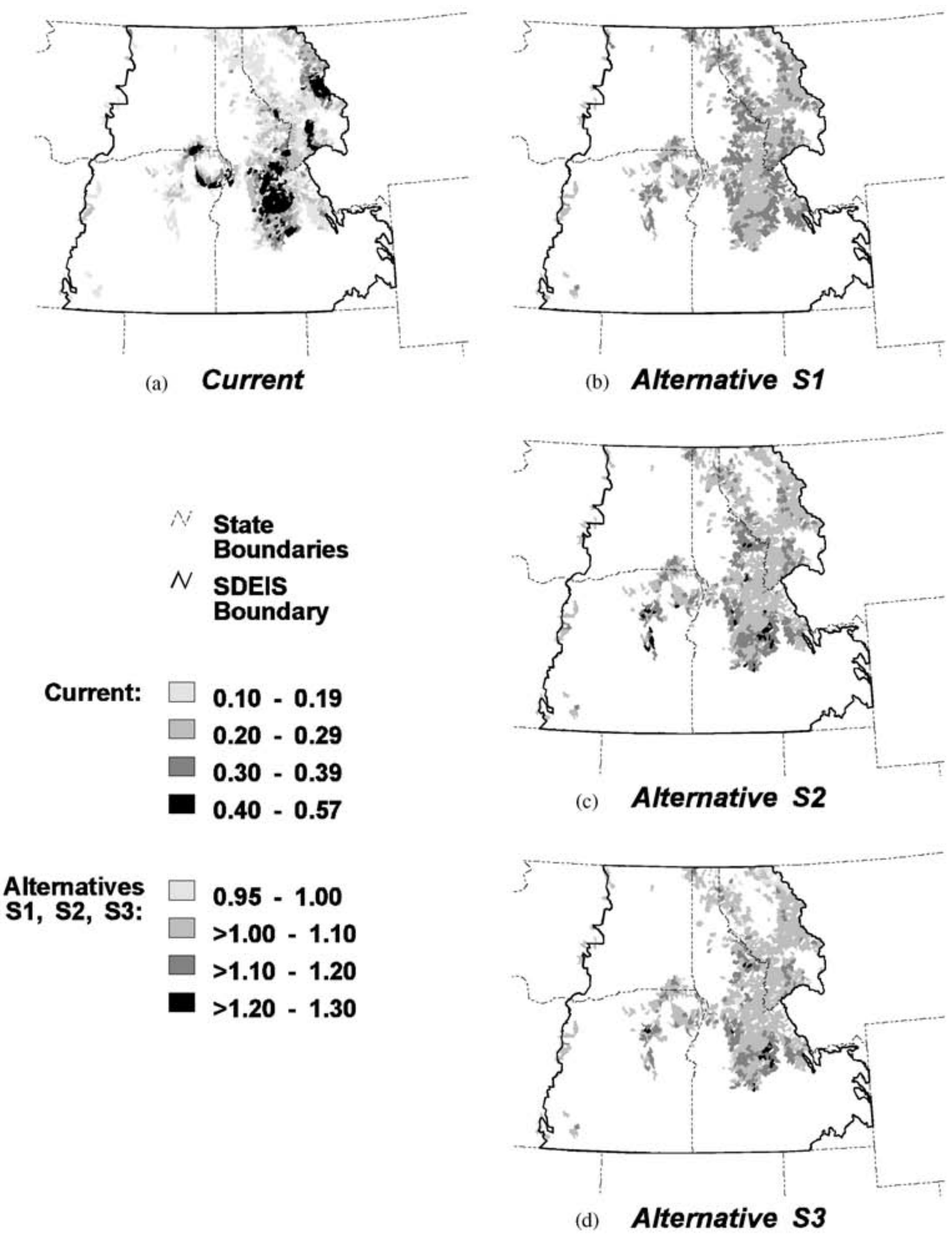

Fig. 5. Mean probability of strong bull trout across all subwatersheds supporting potential bull trout habitat in the current or base condition (a), and the proportional change in probabilities between current and the 100-year estimate for alternative S1 (b), alternative S2 (c), and alternative S3 (d). The proportional change was calculated as the 100 year probability divided by the current probability.

more focused increases in some areas in alternative S2 (Fig. 5). Alternative S3 showed some of the same focusing as in S2, but also showed little or no improvement through much of the area relative to alternative S1.

The results of the scenario for the hypothetical maximum change provided an additional context for considering the effects. For example, the primary summaries showed an increase of 8% in the number

Table 4
Comparison of the predicted status at 100 years (mean probabilities of strong and present) of stream-type chinook salmon and steelhead for federal lands in the Snake River basin under the original assumptions of effects of Snake River dams on migrant survival and a hypothetical scenario where we assumed that dams in the Snake River migratory corridor did not exist

| Species | Scenario | Strong | | | | Present | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Current | S1 | S2 | S3 | Current | S1 | S2 | S3 |
| Stream-type chinook salmon | Dams | 0.042 | 0.045 | 0.045 | 0.044 | 0.172 | 0.177 | 0.178 | 0.177 |
| | No dams | 0.153 | 0.164 | 0.165 | 0.163 | 0.487 | 0.495 | 0.495 | 0.494 |
| Steelhead | Dams | 0.079 | 0.084 | 0.084 | 0.083 | 0.274 | 0.280 | 0.280 | 0.279 |
| | No dams | 0.265 | 0.280 | 0.281 | 0.279 | 0.635 | 0.646 | 0.647 | 0.645 |

of subwatersheds expected to support strong bull trout for alternative S2 (Table 3). Comparison with the ‘‘maximum’’ indicated that these changes are notable. The increase for bull trout, for example, represented about 40% of what was hypothetically possible.

The scenario hypothesizing absence of several Snake River dams also produced a large change. In relation to the original analysis, the new scenario showed no increase in number of expected strong subwatersheds for chinook or steelhead, but a nearly threefold increase in the probability of strong (Table 4). There was a seven to 10-fold increase in the number of subwatersheds where steelhead and chinook were expected to be present and more than a twofold increase in the probability of present. The differences among the alternatives were small and essentially identical between scenarios: alternative S2 produced slightly stronger positive trends than either S1 or S3. Despite the dramatic increase in total numbers, our assumptions about the influence of dams on the status of anadromous salmonids did not change our interpretation of the relative trends among alternatives.

## 4. Discussion

The development of BBNs to evaluate the management alternatives for the interior Columbia River basin SDEIS represents an important advance. Previous expert panels struggled to conceptualize multiple effects and the complexity of a huge and diverse system. Formalization through the use of a network allows the representation of key known or anticipated relationships without the full complexity of process-based models. Because uncertainty in a particular linkage can be acknowledged in the probabilistic statement of relationships, the models are not necessarily limited by the mechanistic detail of existing information or understanding (Reckhow, 1999).

Our assumptions and the logic for the analysis are made explicit in the formal structure of a network. Those assumptions can be challenged and revised, and they can be directly evaluated to determine whether results are robust. Our example with the Snake River dams demonstrates this point. Although the political and scientific debate about the relative influence of dams and habitat on the long term persistence of anadromous salmonids is far from resolved (e.g., Lee et al., 1997; NRC, 1996; Schaller et al., 1999; Williams et al., 1996), our evaluation does not hinge on the assumptions we made in that regard.

The formal and relatively simple structure of our BBNs also allows them to be implemented with spatial detail. It was a relatively simple process to estimate outcomes for more than 6000 subwatersheds in the Basin. Because subwatersheds or their aggregates represent a useful resolution from the perspective of population dynamics and biological interactions (Lee et al., 1997), our results provide a logical synthesis of population level trends. Summarizing the results across any spatial strata allows a full interpretation of the differences that may emerge with varying landscapes. The results suggest important differences among the alternatives with relatively fine resolution but are less clear in broader summaries. For example, the relative difference projected for the mean probability of high habitat capacity for alternative S2 was only about 1.01 times than that in alternative S1 for all FS lands but about 1.50 times when summarized for the subwatersheds included in a conservation network (A2) in alternative S2 (Figs. 4 and 5).

Although broad scale results indicate that differences among alternatives would not be large, several important trends emerged. At 10 years, changes from base were relatively small presumably because only a small portion of the basin can actually be affected by management during that period. Differences in the direction of change, however, were observed among the alternatives. Alternatives S1 and S2 produced improvements in conditions, whereas alternative S3 produced some declines. The differences reflect a general belief that aggressive restoration, such as management to improve forest health (e.g., fuel reduction, removal of diseased and dying timber) that is accompanied by less strategic planning and habitat protection measures carries greater risks, especially where the approaches used are unproven (see Rieman et al., 2000).

At 100 years all of the alternatives produced positive results in both habitat and salmonid status. The trends appeared to be due primarily to the conservative nature of standards and guides in all of the alternatives compared with past management and secondarily due to predicted reductions in road densities and grazing effects. Differences associated with ground disturbance or changing fire regimes had negligible effects in our results. In general, alternative S2 produced stronger positive changes because we assumed that the mitigative and restorative management direction was stronger, particularly for subwatersheds that were included in a conservation network. The results were clearly variable spatially, however, which implied an important interaction between proposed management and the existing biological and physical conditions. Both alternatives S2 and S3 produced more focused changes in some subwatersheds (e.g., some subwatersheds improved noticeably more than others), whereas alternative S1 produced more even changes across the Basin (e.g., Fig. 5), a difference that is not apparent in the basin-wide summary tables. Clearly, tradeoffs are inherent in the implementation of any alternative, and no alternative can provide an optimum result across the entire system.

Even though the overall differences in trends among alternatives are not large, we believe the differences are important for three reasons. First, we believe the conservation strategy in alternative S2 generally is a more efficient and effective approach than that proposed in alternative S1. In essence conservation priorities are identified where management resources are directed rather than dispersed and diluted throughout the Basin. The differences in patterns resulted, in part, from the network of subwatersheds with a focus on conservation and restoration of aquatic habitats. A general theme in aquatic conservation has been to focus limited resources first on conserving the best remaining examples of aquatic biological integrity and diversity (Moyle and Sato, 1991; Reeves and Sedell, 1992). Essentially, that approach attempts to focus limited resources where the best chance of success exists for the least cost. Alternatively, some urge prioritization based on both evolutionary distinction or rarity, and risk of extinction (e.g., Allendorf et al., 1997). Both of these ideas are encompassed in the conservation strategy associated with alternatives S2 and S3.

Second, although all of the alternatives produced positive and often similar results in the 100-year evaluation, some declines were expected at 10 years under alternative S3. The changes at 10 years were small, but they do suggest greater risks in the short term with the more aggressive strategy. Because the BBNs we used are not temporally dynamic, the 100-year predictions are independent of those for 10 years. The BBNs assumed a linear response between current conditions and those at the point of evaluation. If, in reality, conditions declined first and then began to improve, some habitats and populations might decline to a point where recovery would be more difficult than we projected at 100 years. Because some currently depressed populations may be on the verge of extinction, for example, short term habitat loss could make the difference in long term persistence. Thus, our evaluation of alternative S3 could be more optimistic than justified by a more dynamic view of the system.

Third, the simplicity and inherent uncertainty of our BBNs may mask larger differences that would emerge in the implementation of the alternatives. In our BBNs, we necessarily made broad generalizations about the overall success of activities under the different alternatives. The linkages in our BBNs are based on estimates of conditional probabilities that "average" across a wide range of conditions or personal experience. This generalization leads to greater uncertainty among states in the network and tends to obscure rather than emphasize differences among alternatives. Models and analyses applied at finer levels, with better local information could provide better understanding and conceivably a more effective resolution of differences than can be anticipated with the BBNs and analysis at the scale we have attempted.

Assumptions played a critical role in the structure of the models, the probabilities assigned, and the resulting outcomes we projected. Because we used inputs from a landscape modeling team (Hemstrom et al., 2000), we indirectly assumed that vegetative restoration for forest health will occur with minimal ground disturbance. It was also assumed that aquatic restoration needs are apparent and that the methods used will be effective, when any of the alternatives are implemented. In some cases this may be optimistic (e.g., Kauffman et al., 1997). It was assumed that the alternatives will be fully implemented to achieve the objectives and meet the standards as described. This includes the assumption that the FS and BLM will be adequately staffed and have the necessary resources to implement the alternatives. This could be particularly challenging given the recent downward trend in the budgets and staffing of those agencies, especially at the field unit where much of the implementation occurs. Violations of these assumptions could influence the magnitude of effects but would only change the relative effects among the alternatives if the validity of the assumptions also varied among alternatives.

## 5. Summary and conclusions

Our analyses based on the BBNs indicate that all of the alternatives can be expected to produce positive and noteworthy improvements in aquatic habitats and salmonid population status. Trends were stronger in habitat capacity than in salmonid status because of uncertainty and attenuation in the networks and because salmonid populations are constrained by more than habitat. Trends were stronger in the resident salmonids than in the anadromous forms principally because of the additional influence of the migratory corridor assumed for the latter. This does not imply that habitat improvements are not important for persistence of anadromous salmonids (see Lee et al., 1997), but simply that in much of the range we believe that strong rebuilding of populations is unlikely without also improving conditions in the migratory corridor (e.g., Schaller et al., 1999).

On average, differences among alternatives were small, but larger differences were observed with finer resolution in the summaries. Important differences may be obscured by uncertainty, and larger differences might emerge if we could anticipate the nature and effectiveness of implementation. In our analysis alternative S2 generally produced stronger positive results than the other two alternatives.

We believe our analysis provides a useful step for broad scale land management planning. Complex physical and biological interactions and management alternatives can be compartmentalized into simpler, more comprehensible components. By formalizing our understanding and assumptions, we provided a framework for exploring differences in the management alternatives that is quantifiable, spatially explicit, and flexible. The BBNs can incorporate both quantitative and qualitative information, a point key to any analysis attempted at this scale. Our assumptions can be revisited and the structure of the BBNs can even be changed to determine whether critical uncertainties for management exist.

Despite these benefits the BBNs have important limitations. Uncertainty is explicit in the use of conditional probabilities. This uncertainty reflects the limitations of understanding and information but also means that important trends and differences can be obscured. Another and perhaps more important danger is that results can be misinterpreted or misused. Because much of the information represented in the networks is subjective, the outputs should be viewed only as relative trends among alternatives rather than absolute numbers or true probabilities for high habitat capacity and strong or extant populations.

As Box (1979) suggested, ‘‘all models are wrong, but some are useful.’’ Used appropriately the BBNs that we have constructed can provide insight into the potential effects and differences of management considered for the Columbia River basin. They do not describe what those effects will be.

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## Appendix A

Definitions and sources of information used to populate the conditional probability table for nodes in the BBNs used to evaluate the ICBEMP SDEIS management alternatives. All nodes and associated information were compiled for individual subwatersheds unless otherwise noted. For the sake of brevity, break points for quantitative states have not been included (e.g., ‘‘low’’ road density 0.06–0.43 km/km$^2$).

| Name | Definition and source | States |
|---|---|---|
| Road density | Miles of road per square mile of subwatershed area. Base information from landscape characterization of Jensen et al. (1997). Future conditions estimated as by Hemstrom et al. (2001). | Low<br>Medium<br>High |
| Slope steepness | Calculated from two variables that describe the proportion of area with slope gradient <10% or >50% based on digital elevation data at 1 km grid spacing. Base information from landscape characterization (Jensen et al., 1997). | Gentle<br>Medium<br>Steep |
| Road disturbance | An interpreted condition that represents the relative amount of soil disturbance resulting from the interaction of slope steepness and road density. Expert opinion. | Low<br>Medium<br>High |
| Ground disturbance index | The proportion of the subwatershed with soil disturbance associated with logging, thinning and prescribed fire on an annual basis. Current and future conditions estimated as by Hemstrom et al. (2001). | Low<br>Medium<br>High |
| Standards and guides | Degree of mitigation and restoration (e.g., riparian management areas, watershed management designation, required planning and analysis) resulting from the management direction of the alternatives. Interpretation of the SDEIS, expert opinion. | High<br>Moderate<br>Low |
| Sediment | Accelerated sediment delivery to a stream (proportion over natural) arising from road disturbance, logging, and prescribed fire activities. Expert opinion. | Low<br>Medium<br>High |
| Future grazing | Grazing causing a successional change in upland vegetation. Viewed as an index of grazing intensity throughout the subwatersheds. Current and future conditions estimated as by Hemstrom et al. (2001). | No<br>Yes |

| | | |
|---|---|---|
| Prior riparian condition | Disruption of riparian functions in the current conditions that influence aquatic habitat. Expert opinion informed by classification of ownership and management history (Lee et al., 1997). | None to light<br>Moderate to heavy |
| Riparian condition | Condition of the riparian zone and associated functions likely to influence aquatic habitat in the future. Expert opinion. | Intact<br>Moderately degraded<br>Highly degraded |
| Fire-rain | Occurrence of an uncharacteristic, severe wildfire affecting at least 20% of the subwatershed followed within 5 years by a large rain or rain-on-snow event (40-year return interval). Current and future conditions for the probability of fire estimated by Hemstrom et al. (2001). Flood probability from expert opinion. | Yes<br>No |
| Slope 2 | Proportion of subwatershed with slopes steeper than 50%. Base information from landscape characterization (Jensen et al., 1997). | Continuous variable |
| Flood | Occurrence of a flood of sufficient size to cause widespread scour, deposition, and riparian vegetation mortality, resulting in major channel and habitat reorganization through at least 30% of the channel network. Expert opinion. | Yes<br>No |
| Aquatic habitat capacity | The amount and quality, relative to potential, of aquatic habitat attributes necessary to support the salmonids that were historically present in a subwatershed. Expert opinion. | High<br>Moderate<br>Low |
| Exotic threat | Occurrence and relative status of introduced fishes that may affect native salmonids estimated from known occurrence in encompassing watersheds (fifth code HUCs) throughout the basin (Lee et al., 1997) and a subset of subwatersheds, using categorical data analysis procedures (e.g., Lee et al., 1997). | Yes<br>No |
| Anadromous loss | For subwatersheds within the historical range of anadromous salmonids, probability that anadromous salmonids no longer occur. | Yes<br>No |
| Community change | Change in the local fish community resulting from either the loss of anadromous salmonids or the establishment of an exotic and potentially disruptive species. | Yes<br>No |
| Current status | Current status for one of the six salmonid populations associated with the subwatershed (Lee et al., 1997). | Strong<br>Depressed<br>Absent |
| Potential spawn/rear | Occurrence of suitable spawning and rearing habitat for one of the six salmonids. Estimated from existing known occurrences using categorical data analysis procedures (e.g., Lee et al., 1997). Used to identify subwatersheds that do not currently support extant populations but could in the future. | Potential<br>No potential |

| | | |
|---|---|---|
| Refounding/support | Relative potential for genetic and demographic support via dispersal from surrounding subwatersheds. Expert opinion informed by the number of known or predicted subwatersheds supporting populations in the encompassing subbasin. | High<br>Moderate<br>Low |
| Connectivity | Accessibility and suitability for dispersal of the mainstem river connecting subwatersheds throughout the encompassing subbasin. Expert opinion informed by the subbasin categories defined in Lee et al. (1997). | Yes<br>No |
| Recovery potential | Potential for a population to recover from local extinction or a severe bottleneck. Expert opinion. | High<br>Moderate<br>Low |
| Corridor conditions | Condition of the migratory corridor represented by the number of mainstem dams encountered by migratory juvenile and adult anadromous salmonids. Expert opinion. | High<br>Moderate<br>Low |
| Ocean conditions | Relative productivity of the ocean environment in the area used by Columbia River anadromous salmonids. This node was uninformed in our analyses (i.e., each state was equally likely). | High<br>Low |
| Migrant survival | Survival of anadromous salmonids from outmigrating smolt to returning adult. Expert opinion. | High<br>Moderate<br>Low |
| Biological potential | Potential for the population to grow if unconstrained by the condition of habitat in the subwatershed. Expert opinion. | High<br>Moderate<br>Low<br>None |
| Future population status | Status of the population 50 years from the point of evaluation characterized in the input variables. Expert opinion estimated independently for each species. | Strong<br>Depressed<br>Absent |

## References

Allendorf, F.W., Bayles, D.W., Bottom, D.L., Currens, K.P., Frissell, C.A., Hankin, D., Lichatowich, J.A., Nehlsen, W., Trotter, P.C., Williams, T.H., 1997. Prioritizing Pacific salmon stocks for conservation. Conserv. Biol. 11 (1), 140–152.

Beechie, T., Bolton, S., 1999. An approach to restoring salmonid habitat-forming processes in Pacific Northwest watersheds. Fisheries 24 (4), 6–15.

Box, G.E.P., 1979. Robustness is the strategy of scientific model building. In: Launer, R.L., Wilkinson, G.N. (Eds.), Robustness in Statistics. Academic Press, San Diego, CA, p. 202.

Dunham, J.B., Rieman, B.E., 1999. Metapopulation structure of bull trout: influences of physical, biotic, and geometrical landscape characteristics. Ecol. Appl. 9 (2), 642–655.

Dunning, J.B., Danielson, B.J., Pulliam, H.R., 1992. Ecological processes that affect populations in complex landscapes. Oikos 65 (1), 169–175.

Hemstrom, M.A., Hann, W.J., Gravenmier, R.A., Korol, J.J., 2000. Draft landscape effects analysis of the SDEIS alternatives. In: Science Advisory Group. Draft March 2000. Science Advisory Group Effects Analysis for the SDEIS Alternatives. USDA, FS, USDI, BLM, Interior Columbia Basin Ecosystem Management Project, Boise, ID.

Hemstrom, M.A., Korol, J.J., Hann, W.J., 2001. Trends in terrestrial plant communities and landscape health indicate the effects of alternative management strategies in the interior Columbia River basin. For. Ecol. Manage. 153, 105–125.

Jensen, M., Goodman, I., Brewer, K., 1997. Chapter 2: Biophysical Environments of the Basin. In: Quigley, T.M., Arbelbide, S.J. (Tech. Eds.), An Assessment of the Ecosystem Components of the Interior Columbia Basin and Portions of the Klamath and Great Basins, Vol. I. General Technical Report PNW-GTR-405. US Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.

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.

Lee, D.C., Rieman, B.E., 1997. Population viability assessment of salmonids by using probabilistic networks. N. Am. J. Fish. Manage. 17, 1144–1157.

Lee, D.C., Sedell, J.R., Rieman, B.E., Thurow, R.F., Williams, J.E., 1997. Broadscale assessment of aquatic species and habitats. In: Quigley, T.M., Arbelbide, S.J. (Tech. Eds.), An Assessment of Ecosystem Components in the Interior Columbia Basin and Portions of the Klamath and Great Basins. General Technical Report PNW-405. US Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR, pp. 1057–1496.

Li, H.W., Moyle, P.B., 1981. Ecological analysis of species introductions into aquatic systems. Trans. Am. Fish. Soc. 110, 772–782.

Marcot, B.G., Holthausen, R.S., Raphael, M.G., Rowland, M.M., Wisdom, M.J., 2001. Using Bayesian belief networks to evaluate wildlife population viability under land management alternatives from an environmental impact statement. For. Ecol. Manage. 153, 29–42.

Maxwell, J.R., Edwards, C.J., Jensen, M.E., Paustian, S.T., Parrott, H., Hill, D.M., 1995. A hierarchical framework of aquatic ecological units in North America (Nearctic Zone). General Technical Report NC-1761. US Department of Agriculture, Forest Service, North Central Forest Experiment Station, St. Paul, MN.

Meehan, W.R. (Ed.), 1991. Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats. Amer. Fisheries Soc. Spec. Pub. No. 19, Bethesda, MD.

Moyle, P.B., Sato, G.M., 1991. On the design of preserves to protect native fishes. In: Minckley, W.L., Deacon, J.E. (Eds.), The Battle Against Extinction: Native Fish Management in the American West. University of Arizona Press, Tucson, AZ, pp. 155–173.

Moyle, P.B., Li, H.W., Barton, B.A., 1986. The Frankenstein effect: impact of introduced fishes on native fishes in North America. In: Stroud, R.H. (Ed.), Fish Culture in Fisheries Management. Amer. Fisheries Soc., Bethesda, MD, pp. 415–426.

Naiman, R.J. (Ed.), 1992. Watershed Management — Balancing Sustainability and Environmental Change. Springer, New York.

NRC (National Research Council), 1996. Upstream: salmon and society in the Pacific Northwest. Washington, DC.

Quigley, T.M., Arbelbide, S.J. (Tech. Eds.), 1997. An Assessment of Ecosystem Components for the Interior Columbia Basin and Portions of the Klamath and Great Basins. General Technical Report PNW-GTR-405. US Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.

Reckhow, K.H., 1999. Water quality prediction and probability network models. Can. J. Fish. Aquat. Sci. 56 (7), 1150–1158.

Reeves, G.H., Sedell, J.R., 1992. An ecosystem approach to the conservation and management of freshwater habitat for anadromous salmonids in the Pacific Northwest. In: Transactions of the 57th North American Wildlife and Natural Research Conference, Vol. 57, pp. 408–415.

Rieman, B.E., Hessburg, P.F., Lee, D.C., Thurow, R.F., Sedell, J.R., 2000. Toward an integrated classification of ecosystems: defining opportunities for managing fish and forest health. Environ. Manage. 25 (4), 425–444.

Schaller, H.A., Petrosky, C.E., Langness, O.P., 1999. Contrasting patterns of productivity and survival rates for stream-type chinook salmon (*Oncorhynchus tshawytscha*) populations of the Snake and Columbia Rivers. Can. J. Fish. Aquat. Sci. 56 (6), 1031–1045.

Sedell, J.R., Lee, D.C., Rieman, B.E., Thurow, R.F., Williams, J.E., 1997. Effects of proposed alternatives on aquatic habitats and native fishes. In: Quigley, T.M., Lee, K.M., Arbelbide, S.J. (Tech. Eds.), Evaluation of EIS Alternatives by the Science Integration Team, Vol. 1. General Technical Report PNW-GTR-406. US Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR, pp. 435–535.

Swanston, D.N., 1991. Natural processes. In: Meehan, W. (Ed.), Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats. Am. Fisheries Soc. Spec. Pub. No. 19, Bethesda, MD, pp. 139–179.

USDA and USDI, 2000. Interior Columbia basin supplemental draft environmental impact statement, BLM/OR/WA/Pt-00/019 + 1792. US Department of the Interior, Bureau of Land Management, Portland, OR (irregular pagination).

Williams, R.N., Calvin, L.D., Coutant, C.C., Erho Jr., M.W., Lichatowich, J.A., Liss, W.J., McConnaha, W.E., Mundy, P.R., Stanford, J.A., Whitney, R.R., Bottom, D.L., Frissell, C.A., 1996. Return to the river: restoration of salmonid fishes in the Columbia River ecosystem, ISG 96-6. Northwest Power Planning Council, Portland, OR.

Willson, M.F., Halupka, K.C., 1995. Anadromous fish as keystone species in vertebrate communities. Conserv. Biol. 9, 489–497.