

Architecture and behavior of IF3 population persistence models

Matthew Dare

USDA Forest Service, Rocky Mountain Research Station, 322 East Front Street, Suite 401,
Boise, Idaho, 83702, USA
email: mdare@fs.fed.us

Charlie Luce

USDA Forest Service, Rocky Mountain Research Station, 322 East Front Street, Suite 401,
Boise, Idaho, 83702, USA

Paul Hessburg

U.S.D.A. Forest Service, Pacific Northwest Research Station, 1133 N. Western Ave.,
Wenatchee, Washington 98801, USA

Bruce Rieman

U.S.D.A. Forest Service, Rocky Mountain Research Station, P.O. Box 1541, Seeley Lake,
Montana 59868, USA

Anne Black

U.S.D.A Forest, Service, Rocky Mountain Research Station, 790 East Beckwith Ave., Missoula,
Montana 59801, USA

Carol Miller

U.S.D.A Forest, Service, Rocky Mountain Research Station, 790 East Beckwith Ave., Missoula,
Montana 59801, USA

Introduction

The models we used to estimate post-fire bull trout population persistence in the SFBR are Bayesian Belief Networks (BBN) constructed in Netica (Norsys, 1998). We developed BBNs to predict the probability of persistence for bull trout in individual stream networks or patches. In a BBN the inter-relationships of a suite of causal variables generate predictions about a variable of interest, in this case patch-scale persistence probability. Within a network, the state of a node is conditional upon the state of its parental nodes. Uncertainty in the relationship is represented within a conditional probability table (CPT) for each “child” node. Within the persistence models potential threats to post-fire patch persistence are broadly categorized into anthropogenic threats (e.g. fragmentation by barriers and chronic sediment inputs from roads) and fire-related threats (e.g. fire induced changes to riparian vegetation and upslope areas resulting in debris flows).

Several management scenarios were evaluated using two models that differed only in the assumptions about wildfire size and severity. Model 1, incorporated probability distributions for fire patch sizes and fire severities supported by potential vegetation groups within habitat networks. The intent was to evaluate the post-fire persistence probability of bull trout exposed to fires consistent with historical probability distributions of fire patch size and severity.

In the second model we omitted nodes pertaining to vegetation composition and fire patch size and directly set fire sizes and severities. The intent was to evaluate the post-fire persistence probability of bull trout exposed to very large, high-severity fires that may become increasingly common in western forest landscapes (Westerling et al., 2006).

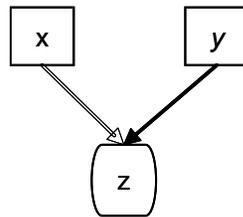
An advantage of a BBN in decision-support modeling is the ability to handle qualitative and quantitative information in the development of CPTs. The conditional probability tables for

many of the nodes were generated directly using GIS data, or were the direct product of parent nodes via simple mathematical operations or functions. For variables where expert opinion factored into node construction additional justification is provided. The following section defines and describes each node included in the persistence models.

Conditional probability table calculations

The Bayesian model we used, Netica, takes probability distributions of inputs in classes and estimates the probability of being in output classes based on a conditional probability table. Each combination of potential inputs results in a probability for each potential outcome.

A simple relationship with two inputs x and y and one output z can be represented using an influence diagram.



An individual entry in the conditional probability table for z, p_{ijk} is defined as

$$p_{ijk} = p(z_i) | x_j \ \& \ y_k, \tag{D.1}$$

where i is the i^{th} class of z, j the j^{th} class of x, and k the k^{th} class of y. The p_{ijk} can be estimated using a variety of methods ranging from professional judgment (e.g., Pollino et al., 2007), to bootstrapped samples, to formulae. Because the states for variables and parameters in the model are carried in classes, we need to account for within class variability when using formulae to estimate the p_{ijk} . Consider for example z, x, and y continuous variables and

$$z = f(x,y), \tag{D.2}$$

then we need to consider the probability of various values of z knowing only constrained ranges for x and y and their distribution within that class:

$$p_{ijk} = p(z_{imin} < z=f(x,y) < z_{imax}) \mid x_{jmin} < x < x_{jmax} \text{ and } y_{kmin} < y < y_{kmax}, \quad (D.3)$$

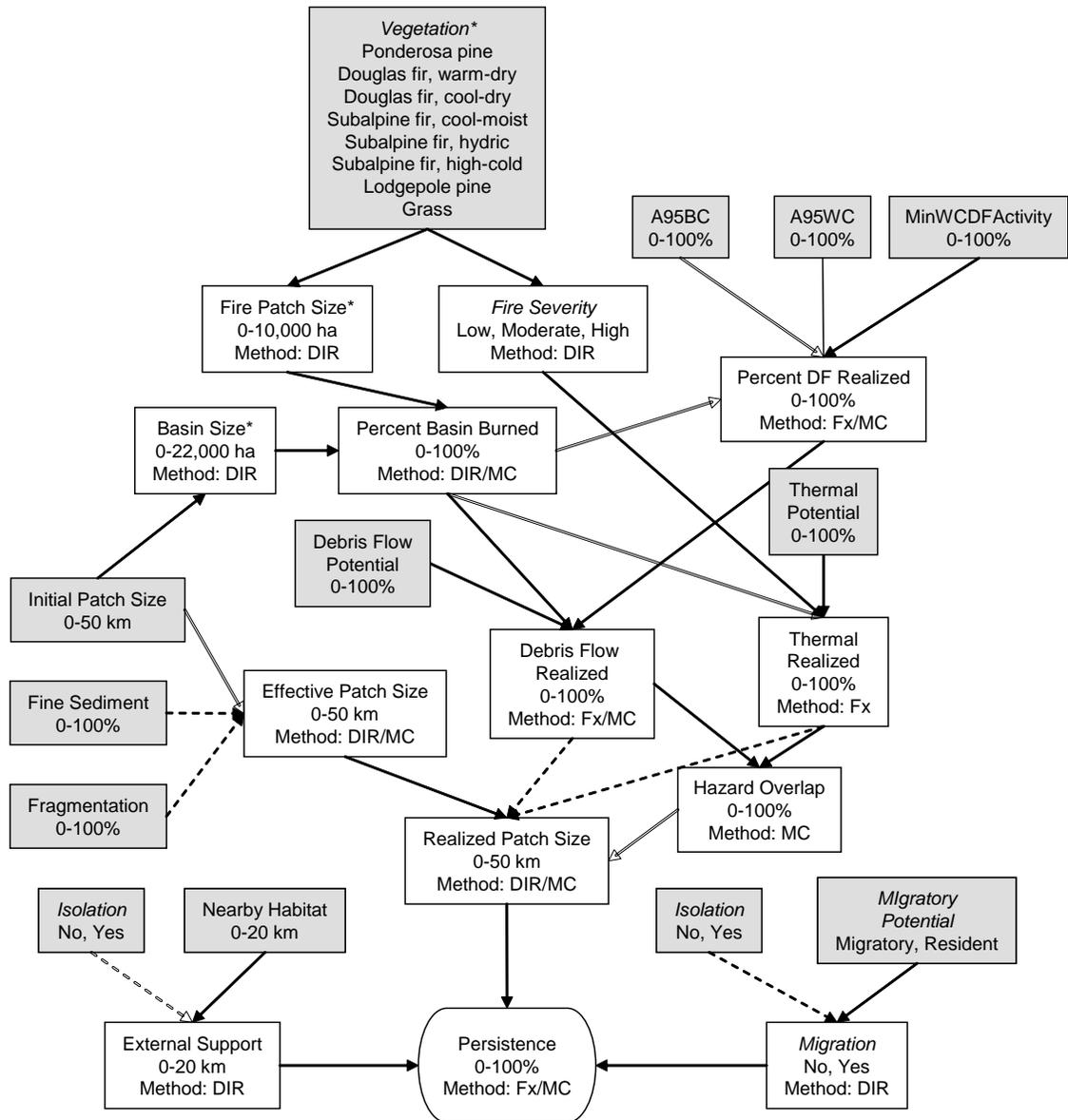
which can be evaluated numerically using Monte Carlo methods. In such an approach a large number of random samples of x and y are generated and z values are computed. The count of z values in each class divided by the total number of samples generated gives p_i . The random samples of x and y should come from an appropriate distribution, for example a simple assumption is that x and y are uniformly distributed on the class range and

$$f(y_i) = 1/(y_{imax}-y_{imin}) \text{ for } y_{imin} < y < y_{imax}, \quad (D.4)$$

$$f(x_i) = 1/(x_{imax}-x_{imin}) \text{ for } x_{imin} < x < x_{imax}, \quad (D.5)$$

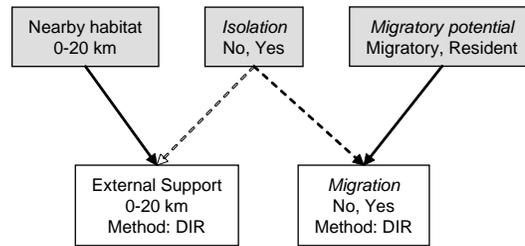
Other distributions within classes may be generated as well when using Monte Carlo simulations. We note for each parameter/variable the distribution from which it is sampled. For each simulation we increased the number of trials until variation in probabilities was less than 0.01, the number of trials necessary to achieve this precision varied among simulations.

In the following figures grey boxes represent model inputs derived from GIS and field measurements. Solid lines depict positive relationships; dashed lines depict negative relationships. Categorical variables are italicized. States (categorical) or ranges (continuous) appear below node titles. Conditional probability tables for each node were derived using one of the following methods: DIR – direct or empirical relationship between child and parent nodes; Fx – probabilities for node states derived from mathematical function(s); MC – uncertainty among node states modeled using Monte Carlo techniques.



Node Definitions

Life history



Migratory potential (MP)

MP represents the capacity of a bull trout population within a habitat patch to support resident and migratory life-histories, as represented by the amount of seasonal movement possible. MP is a two-state node including *migratory* and *resident* states. Without information to the contrary, we assumed all occupied bull trout habitat patches had the potential to produce both resident and migratory life-history forms and MP was *migratory*. For isolated patches MP was set to *resident*.

Populations containing more than a single life-history pattern are probably more robust to shifts in environmental conditions that favor one strategy over another through time (Northcote, 1992; Rieman and Clayton, 1997; Rieman and Dunham, 2000). We invoked this interpretation of life history diversity as the conceptual basis for the effect of MP on persistence (P). For two habitat networks of equal size, the network supporting both resident and migratory life histories will have a higher probability for P than its counterpart containing only resident individuals.

Isolation (I)

An isolated habitat patch is separated from the surrounding watershed by a fish-passage barrier located downstream of the habitat patch. Within the model I had two states: *no* and *yes*, where isolated habitat patches received the node state *yes*. We used a GIS layer defining the location of stream culverts and other human-made barriers provided by National Forest personnel to identify isolated habitat patches. Within the model the effect of isolation was to reduce nearby habitat (NH) to 0 and limit migratory potential (MP) to “resident”.

Nearby habitat (NH)

NH is the stream length (km) of occupied bull trout habitat patches within 10 stream-kilometers of the downstream end of a habitat patch. We identified occupied habitat patches within 10 stream kilometers of each habitat patch and used the stream length of these habitat patches as an estimate of NH.

The mobility of bull trout is well documented (Swanberg, 1997; Bahr and Shrimpton, 2004; Brenkman and Corbett, 2005) and inter-network distances in the SFBR are substantially less than distances over which juveniles of this species have been observed to move (Monnot et al., 2008). Genetic research on another mobile salmonid, Chinook salmon *Oncorhynchus tshawytscha*, suggested dispersal distances were typically less than 10 km (Neville et al., 2006). These data were cited by Isaak et al. (2007), in their analysis of inter-network distance on use of spawning habitat by Chinook salmon. Following the logic of Isaak et al. (2007) we used 10-km as the maximum inter-network distance for estimating NH for bull trout in the SFBR.

External support (ES)

ES is nearby habitat (NH) accounting for the effect isolation (I) has on the movement of migratory bull trout between neighboring habitat patches.

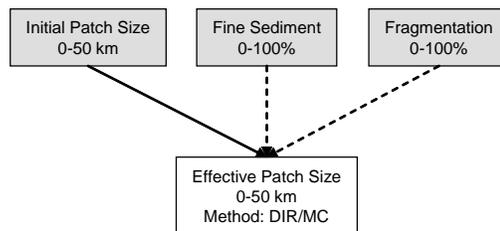
$$ES = \begin{cases} NH, & \text{if } I = no \\ 0, & \text{if } I = yes \end{cases}, \quad (D.6)$$

where NH is the stream length of occupied bull trout habitat within 10 stream km of the habitat patch (see NH above).

Migration (M)

M represents the expression of a migratory life history within a bull trout population. M has two states: *yes* – migratory individuals are present in a population, and *no* – the population lacks a migratory component. Within the model M is constrained only by isolation (I): isolated populations are composed entirely of non-migratory, or resident bull trout and the resulting node state is *no*. The node state for non-isolated populations was *yes*.

Human influence



Initial patch size (IPS)

IPS is the stream length (km) within a habitat patch capable of supporting bull trout reproduction. Bull trout habitat patches in the SFBR were initially identified by Rieman and MacIntyre (1993) and occupancy status was verified by subsequent field studies (Rieman and

MacIntyre, 1995; Dunham and Rieman, 1999). We used Terrain Analysis Using Digital Elevation Models (TauDEM; Tarboton, 2004), an extension of ArcGIS to identify stream segments within occupied bull trout habitat patches that had downstream contributing areas greater than 400 ha. Stream channels with contributing areas less than 400 ha were eliminated from consideration because they are considered too small to support spawning and rearing by bull trout (Rieman and MacIntyre, 1995). The length (km) of remaining stream channels was the IPS of a habitat patch. IPS ranged from 0 to 50 km encompassing the size range of occupied habitat patches in the SFBR.

Fragmentation (F)

F is the proportion of a habitat patch isolated by fish-passage barriers located **within** the habitat patch. We quantified the effect of fragmentation as reducing the network size to the longest group of contiguous stream segments within the network boundary. The formula for calculating F was

$$F = L_f/L_t, \quad (D.7)$$

where L_f is the total stream length in a habitat patch that is isolated by internal barriers and L_t is the total stream length within a habitat patch. F is expressed as a proportion (0-0.1, 0.1-0.2, ..., 1.0) of total stream length within a habitat patch..

Sediment (S)

S represents the proportion of stream length within a habitat patch associated with catchment-scale road density greater than 1 km·km⁻². We calculated road density in the

contributing watershed of each stream segment in the SFBR in a GIS. The formula for calculating S was

$$S = L_s/L_t, \quad (\text{D.8})$$

where L_s is the total stream length in a habitat patch associated with catchments having road density greater than $1 \text{ km}\cdot\text{km}^{-2}$ and L_t is the total stream length within a habitat patch. S was expressed as a proportion (0-0.1, 0.1-0.2, ..., 1.0) of the total stream length within a habitat patch.

Because the exact nature of the relationship between road density and the amount of fine sediment delivered to stream channels is unknown, previous studies have used a minimum road density to identify stream channels or basins impacted by sediments. Cederholm et al. (1981) suggested stream channels in basins having road density greater than $2.5 \text{ km}\cdot\text{km}^{-2}$ would be negatively impacted. We used a more conservative threshold of $1 \text{ km}\cdot\text{km}^{-2}$ based on the work of Lee et al. (1997) and Thompson and Lee (2000). In the former study road density was classified as “high” at levels greater than $1.7 \text{ mi}\cdot\text{mi}^{-2}$ or $1.04 \text{ km}\cdot\text{km}^{-2}$. The authors of the latter study suggested fine sediments would negatively impact fish habitat when basin-scale road densities exceeded $1 \text{ km}\cdot\text{km}^{-2}$.

Effective patch size (EPS)

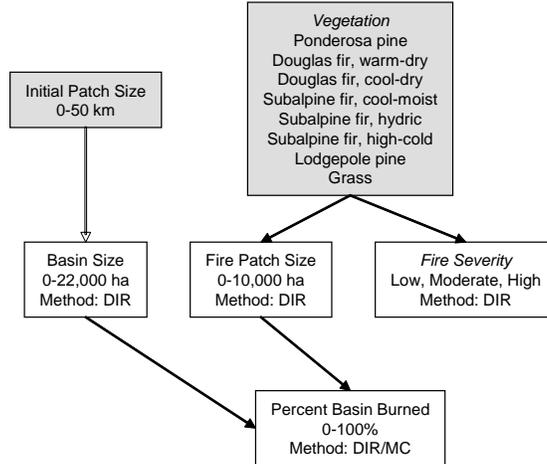
EPS is the stream length (km) within a habitat patch remaining after subtracting the stream length affected by fine-sediment inputs and fragmentation from internal fish-passage barriers. The formula for EPS was

$$EPS = IPS \times S \times F, \quad (\text{D.9})$$

where IPS is initial patch size, S is the proportion of stream length in a patch affected by fine sediment, and F is the proportion of stream length within a habitat patch not fragmented by

internal barriers. For example, a 10-km habitat network having 1 km of stream habitat impacted by sediment and 1 km of stream segment isolated by an internal barrier would have an EPS of 8 km.

Fire



Initial patch size (IPS)

See IPS description above.

Basin size (BS)

BS is the area (ha) of a habitat patch. Stream length (km) was the primary metric of bull trout habitat patch size; however, we used regression analysis to convert stream length to area in order to model potential effects of wildfire to habitat patches. The relationship between length (km) and basin size (ha; *Area* in the following equations) for bull trout habitat patches was:

$$Area = 532.2(Length) - 1328.5 \tag{D.10}$$

Based on this equation, every kilometer of stream within a habitat patch has, on average, 532.2 ha of contributing basin. Its relationship with network length explained 97% of the variation in BS.

Vegetation (V)

V is the proportional composition of potential vegetation groups within an occupied bull trout habitat patch. We evaluated vegetation composition based on the Boise National Forest's potential vegetation group (PVG) classification for the SFBR. A potential vegetation group represents the suite of plant species likely to constitute the climax vegetation community at a site. A site's PVG is a function of its soil type, aspect, microclimate, etc. (Hessburg et al., 2000). The following are brief descriptions of PVGs present in the SFBR based on those available in the Boise National Forest Revised Forest Plan (Appendix A):

[//www.fs.fed.us/r4/sawtooth/arevision/boiseplan.htm](http://www.fs.fed.us/r4/sawtooth/arevision/boiseplan.htm). The PVG name is based on the 'climax' vegetation species. The descriptions detail dominant overstory species currently present on the sites, as well as diagnostic understory or patch structure conditions.

Ponderosa pine: the warmest and driest forested vegetation class. "Park-like" stands of ponderosa pine (*Pinus ponderosa*) with low understory cover dominate. Douglas fir (*Pseudotsuga menziesii*) present at higher elevations.

Warm, dry Douglas fir: low- to mid-elevation mixture of Douglas fir and ponderosa pine. Understory vegetation is a mix of grasses and shrubs.

Cool, moist Douglas fir: relatively rare group is found adjacent to subalpine fir stands. Douglas fir dominates with an understory primarily consisting of shrubs.

Cool, dry Douglas fir: Douglas fir dominates with lodgepole pine (*Pinus contorta*) and aspen (*Populus* spp.) present under particular conditions. Understory is typically grass but some shrubs are present.

Warm, dry subalpine fir: Douglas fir is the most common cover type with a variety of other tree species present depending on elevation and microclimate. Shrubs dominate understory.

Common at higher elevations.

Hydric subalpine fir: lodgepole pine, Engelmann spruce (*Picea engelmanni*), and subalpine fir (*Abies lasiocarpa*) most common species. Group typically found in wet areas. Understory vegetation primarily grass species requiring seasonal pulses of water.

Lodgepole pine: lodgepole pine dominates with other species sparsely distributed throughout range of this group, which is common at higher elevations. Understory is primarily grasses with some shrubs under right conditions.

High elevation subalpine fir: vegetation group present at highest elevations. Engelmann spruce and subalpine fir most common. Understory typically grasses.

Water: lakes and water bodies.

Rock and barren ground: non-vegetated cover type.

Grass and shrublands: non-forest vegetation group including various grass species and several varieties of sagebrush.

The persistence models we used were designed to evaluate wildfire-related effects at the habitat patch scale, we therefore estimated the PVG composition of each habitat patch within a GIS. Hessburg et al. (2007) derived probability distributions for fire patch size and fire severity of PVGs using data collected in eastern Washington. We crosswalked the PVGs used in that study with those in the SFBR in order to develop condition probability tables for the nodes Fire patch size and Fire severity in Model 1. Node states for Fire patch size and Fire severity in Model 1 were determined by the PVG composition of a habitat patch.

Fire patch size (FPS)

FPS is the area (ha) of canopy opening created by a wildfire. We used data from Hessburg et al. (2007) to develop a probability distribution of patch sizes for PVGs present in the SFBR.

Medium and large-sized fires generally result in a mosaic of dead, under-burned, and unburned areas in forest stands (Eberhart and Woodard, 1987; Agee, 1998). Post-fire disturbances that affect streams, such as debris flows and thermal changes, are associated with patches of high-severity fire which kill trees and understory vegetation (Istanbulluoglu et al., 2002; Dunham et al., 2007). In forest types supporting low and mixed-severity fire regimes (e.g. ponderosa pine *Pinus ponderosa*) the area affected by high-severity fire may be substantially less than area burned. We used FPS rather than total fire area because we assumed post-fire disturbance would be spatially coincident with patches of high-severity fire inside fire perimeters.

Fire severity (FS)

FS is a measure of the effect of wildfire on a forest canopy (Agee, 1998) but may reflect the effects on soils (e.g. extent of hydrophobicity) and other watershed characteristics as well (Luce 2005). We used information from Hessburg et al. (2007) to populate a CPT of FS for each potential vegetation group in the SFBR. For example, based on analysis of the data discussed in Hessburg et al. (2007) we calculated that historically, on average, 49, 20, and 31% of burned patches in the ponderosa pine PVG burned at low, mixed, and high-severity, respectively. We populated the CPT for other PVGs in this manner.

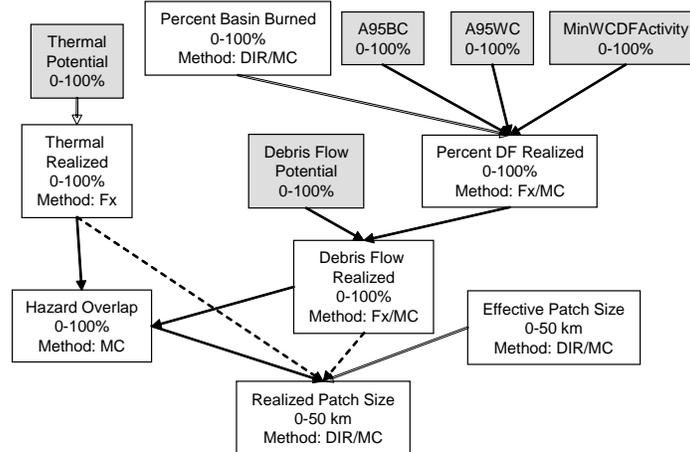
Percent basin burned (PBB)

PBB is the proportion of the area of a habitat patch affected by a wildfire event. PBB represents a probability distribution of the area of high severity fire within a habitat network based on its size and the distribution of fire sizes and severities typical of its terrestrial vegetation composition. We developed this probability distribution by Monte Carlo simulation involving randomly sized overlapping circles representing *Basin size* (BS) and *Fire patch size* (FPS). In order to facilitate the simulation BS and FPS were modeled as circles and the area of overlap for each trial was calculated using the following formula:

$$\gamma_{ij} = r_i^2(\Theta_i - \sin \Theta_i) + r_j^2(\Theta_j - \sin \Theta_j), \quad (\text{D.11})$$

where γ_{ij} is the overlapping area of circles i and j , r_i is the radius of circle i , and Θ_i is the angle between the centerpoint of circle i and the points its perimeter intersects the perimeter of circle j . A detailed explanation of this formula is available in Aikio (2004). For each trial, BS and FPS were randomly chosen based on a uniform distribution within their respective size classes within the model. Each simulation involved 24,000 trials.

Post-fire disturbance



A95, best case and A95, worst case (A95B and A95W)

A95 is the proportion of the area of a bull trout habitat patch that must burn in order to activate 95 percent of potential debris flows within the patch. We used best-case (A95B) and worst case (A95W) A95s, along with DFWC to generate probabilities for the proportion of potential debris flow activity resulting from wildfire within a habitat patch. We used a GIS to quantify the relationship between wildfire and debris flow activity within a habitat patch by simulating fires of increasing size (1-km incremental increases in fire radius) with an origination point located at 1) the downstream end of a habitat patch where a large proportion of a patch would have to burn to activate all of the potential debris flows (best-case scenario); and 2) a point in the upstream portion of the basin where fire size necessary to activate all potential debris flows was minimized (worst-case scenario). Origination points for the best-case and worst-case scenarios were identified by visual inspection within a GIS. A potential initiating segment was considered activated when the fire perimeter overlapped its catchment. For each scenario we counted the number of debris flow initiating segments activated as fire size increased. The proportion of potential debris flow activity realized by each fire size was calculated by dividing

the total length of stream segments activated by each fire size by the total length of stream segments that could support debris flow activity within each habitat patch.

We used logistic regression to relate the proportion of basin area experiencing wildfire to the proportion of potential debris flow activity realized. Separate regression analyses were conducted for the best-case and worst-case scenarios for each habitat patch. Regression analyses were conducted using PROC LOGISTIC in SAS (version 9.2, SAS Institute, 2007). We used this equation to interpolate best-case and worst-case A95 for each habitat patch. We verified predicted values for A95 by comparing model predictions with raw data collected during the GIS analysis (see above). In some cases, we adjusted predicted A95 values to more closely match values derived from direct observation within the GIS.

A95 is meant to represent the topography or physical arrangement of stream channels within a habitat patch. It has been suggested that increasing topological complexity provides resilience to stream fish populations exposed to disturbance (Guy et al., 2008). For small or simple basins where the arrangement of stream channels is relatively linear, A95 is near 0.0, meaning a small amount of fire activity could trigger a large proportion of potential debris flow activity. As basin size and complexity increase the arrangement of stream channels decreases in linearity and individual debris flows affect large proportions of a habitat patch. In this case A95 is near 1.0. In the following table patches are arranged from smallest to largest with corresponding best case and worst case A95 values.

Table 1

A95 values for bull trout habitat patches in the SFBR. Habitat patches are arranged from smallest (patch 1) to largest (patch 9).

Patch	A95B	A95W
1	0.745	0.020
2	0.210	0.020
3	0.745	0.500
4	0.574	0.555
5	0.737	0.439
7	0.551	0.542
8	0.950	0.930
9	0.950	0.900

Minimum worst case debris flow activity (DFWC)

DFWC represents the amount of potential debris flow activity that would be realized if wildfire of any size was to occur within a habitat patch. Values for this variable were derived from a logistic regression analysis relating the proportional area of a habitat patch affected by wildfire to the proportion of potential debris flow activity realized (see A95 above). The DFWC for a habitat patch is the exponent of the intercept of that patch’s logistic regression equation. We took the exponent of the intercepts in order to constrain the range of possible values for DFWC between 0 and 1. We validated regression model predictions by comparing estimated intercepts with the raw data used to generate regression equations. In several cases predicted intercepts were dramatically different than “observed” values obtained directly from the data. In these cases we adjusted the value of DFWC to more closely approximate the values for each

habitat patch estimated within a GIS. Values for this variable ranged from 0.2 to 0.94. DFWC along with A95 (A95B, A95W) was used to estimate percent debris flow realized (PDFR).

DFWC and A95 are meant to describe the size and topography of habitat patches. Theoretically, large and complex basins would be more resilient to wildfire activity and the corresponding proportion of potential debris flow activity realized by a small fire would be near 0.0. Small and simple basins, by contrast, could experience a relatively large amount of debris flow activity following a small wildfire. In such cases, values for DFWC approach 1.0 as basin size and complexity decrease.

Percent debris flow realized (PDFR)

PDFR represents the proportion of the total potential debris flow activity in a habitat patch that would be realized by a wildfire of a certain size occurring inside the boundary of the habitat patch. PDFR is a function of the topography of a habitat patch and the proportion of patch area that experiences wildfire. Conditional probabilities for PDFR for each habitat patch in the SFBR were derived using a Monte Carlo simulation that includes best case and worst case A95 (A95B, A95W), minimum worst case debris flow activity (DFWC), and percent basin burned (PBB). Values for A95B, A95W, and DFWC were partitioned into four bins: 0-0.25, 0.26-0.50, 0.51-0.75, and 0.76-1.00 and used as inputs to the Monte Carlo simulation. The last input, PBB was partitioned into 11 bins: 0-0.10, 0.1-0.2, ..., 0.9-1.0, and 1.0. Every combination of inputs was evaluated in order to produce the condition probability table for PDFR.

Random values for A95B, A95W, and DFWC were selected using a uniform distribution. These values were used to calculate best case and worst case slope coefficients for logistic

curves describing the relationship between PDFR and PBB. The best-case slope was calculated using this formula:

$$BI_{bc} = \frac{\ln\left(\frac{(0.95 \times DFBC - 0.95)}{(0.95 \times DFBC - DFBC)}\right)}{A95B}, \quad (D.12)$$

where BI_{bc} is the best-case slope, $DFBC$ is the minimum best case debris flow activity, and $A95B$ is the proportion of habitat patch area that must be burned to activate 95% of the potential debris flow activity under the best-case scenario. For best case trials $A95B$ was set to 0.01.

Worst-case slope values were calculated using the following formula:

$$BI_{wc} = \frac{\ln\left(\frac{(0.95 \times DFWC - 0.95)}{(0.95 \times DFWC - DFWC)}\right)}{A95W}, \quad (D.13)$$

where BI_{wc} is the worst-case slope, $DFWC$ is the minimum worst case debris flow activity, and $A95W$ is the proportion of habitat patch area that must be burned to activate 95% of the potential debris flow activity under the worst-case scenario. Values for $DFWC$ within a trial were randomly selected from within the assigned bin of this variable for each trial.

A best-case and worst-case estimate of PDFR was calculated for each trial based on the values for B1 and PBB. PDFR under the best-case scenario was calculated using the following formula:

$$PDFR_{bc} = 1 \times DFBC \times \frac{e^{(BI_{bc} \times PBB)}}{1 + DFBC \times \left((e^{(BI_{bc} \times PBB)}) - 1 \right)}, \quad (D.14)$$

where $PDFR_{bc}$ is the PDFR under the best case scenario. $DFBC$ was 0.01 for all trials in the best-case scenario. PDFR for the worst case scenario was calculated using the following formula:

$$PDFR_{wc} = 1 \times DFWC \times \frac{e^{(B1_{wc} \times PBB)}}{1 + DFWC \times (e^{(B1_{wc} \times PBB)} - 1)}, \quad (D.15)$$

where $PDFR_{wc}$ is the PDFR under the worst case scenario. The value for PDFR for each trial was randomly selected from within the range encompassed by the $PDFR_{bc}$ and $PDFR_{wc}$.

Condition probabilities were based on 10,000 trials for each combination of $A95_{bc}$, $A95_{wc}$, DFWC, and PBB. Values for PDFR were partitioned into four bins: 0-0.25, 0.26-0.50, 0.51-0.75, and 0.76-1.0. PDFR and *Debris flow potential* (DFP) were used to calculate conditional probabilities for *Debris flow realized*.

Debris flow potential (DFP)

DFP is the proportion of stream length within a habitat patch that is vulnerable to post-fire debris flows. Within the model DFP ranged from 0 to 100% and was divided into six bins: 0%, 0-25%, 26-50%, 51-75%, 76-99%, and 100%.

Thermal potential (TP)

TP is the proportion of stream length within a habitat patch that is vulnerable to post-fire stream temperature increases above 18 °C. Temperatures above 18 °C are considered unsuitable for spawning and rearing by bull trout. Within the model TP ranged from 0 to 100 percent and was divided into six bins: 0%, 1-25%, 26-50%, 51-75%, 76-99%, and 100%.

Debris flow realized (DFR)

DFR is the proportion of stream length (km) within a habitat patch identified as vulnerable to debris flows that actually experiences a post-fire debris flow. We characterized uncertainty associated with estimating values for DFR using a Monte Carlo simulation. Within the

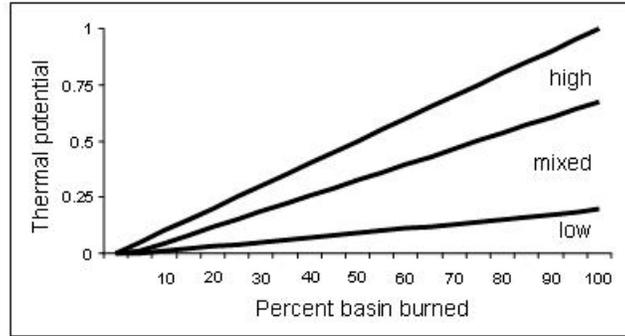
simulation the proportion (0, 0.01-0.25, 0.26-0.50, 0.51-0.75, 0.76-0.99, 1.00) of DFR is a product of debris flow potential (DFP) and percent debris flow realized (PDFR):

$$DFR = DFP \times PDFR. \quad (D.16)$$

Each trial involved a randomly selected value for DFP and PDFR, both of which were partitioned into bins (0, 0.01-0.25, 0.26-0.50, 0.51-0.75, 0.76-0.99, 1.00) in order to generate a condition probability table. For each combination of DFP and PDFR there were 10,000 trials. Bins for these two variables represent ranges of proportions of the total stream length within a habitat patch.

Thermal realized (TR)

TR is the proportion of stream length within a habitat patch rendered thermally unsuitable for bull trout following a wildfire. We used a Monte Carlo simulation to incorporate uncertainty into an estimate of the proportion of each habitat network that would be affected by critical post-fire stream temperature elevations based on the percent basin burned (PBB) and fire severity (FS). We used published boundaries of canopy loss associated with low, mixed, and high severity fires (Agee, 1993; Hessburg et al., 2007) to delineate possible values of TR for fires of low, mixed, and high severity. The assumption of this approach was the amount of canopy loss resulting from wildfire is proportional to the amount of suitable stream habitat that would be rendered unsuitable due to post-fire temperature changes. Across fire sizes, low, mixed, and high severity fires realized 0-20%, 21-70%, and 71-100% of thermal potential (TP). We used these severity limits in the construction of the simulation. The following figure illustrates the zones from which the simulation drew values for TR based on a habitat network's TP and PBB. Uncertainty regarding the amount of realized TP increased as the area of disturbance increased, recognizing the variability of realized disturbance inside a fire perimeter (Agee 1998).



Each trial of the Monte Carlo began with the selection of a random value of PBB and TP. The value for PBB was used to calculate six values based on equations for the three lines shown in the figure above. Line equations were estimated using SAS (version 9.2, SAS Institute 2007). For each fire severity (low, mixed, high) a minimum and maximum disturbance value was calculated using the following formulas:

$$TR = \begin{cases} 0, & \text{for Low Severity minimum} \\ 0.2 \times PBB - 0.009, & \text{for Low severity maximum} \\ 0.2 \times PBB - 0.009, & \text{for Mixed severity minimum} \\ 0.69 \times PBB - 0.02, & \text{for Mixed severity maximum} \\ 0.69 \times PBB - 0.02, & \text{for High severity minimum} \\ PBB, & \text{for High severity maximum} \end{cases}, \quad (D.17)$$

A value for TR for each FS was calculated within each trial by multiplying TP by a randomly selected value between the minimum and maximum values calculated using Equation 17. There were 5,000 trials for each combination of TP and PBB, and FS, within which values for each variable were selected using a uniform distribution. TR was expressed as a proportion of stream length within a habitat patch and divided among five bins: 0-0.25, 0.26-0.50, 0.51-0.75, 0.76-0.99, and 1.00.

Hazard overlap (HO)

Our analysis of debris flow potential (DFP) and thermal potential (TP) revealed many stream segments in the SFBR were vulnerable to both post-fire hazards. Regardless of the mechanism, once a stream segment was rendered unsuitable for bull trout it could not be further perturbed by the realization of an additional hazard within the model. Failure to account for this would have led to an overestimation of the effect of wildfire-related hazards and biased model predictions. We identified the proportion of stream segments within each habitat patch vulnerable to both post-fire threats within a GIS and we devised a simple Monte Carlo simulation to impart uncertainty to estimates of HO. The simulation involved random combinations of DFP and TP, both expressed as proportions and divided into six bins: 0, 0.01-0.25, 0.26-0.5, 0.51-0.75, 0.76-0.99, 0.99-1.0. Minimum and maximum values within each bin were inputs to the simulation. Minimum overlap was calculated using the following formula:

$$HO_{\min} = \begin{cases} 0, & \text{where } DFP_{\min} + TP_{\min} \leq 0 \\ DFP_{\min} + TP_{\min} - 1, & \text{where } DFP_{\min} + TP_{\min} \geq 1 \end{cases}, \quad (\text{D.18})$$

where HO_{\min} is the minimum hazard overlap expressed as a proportion of total stream length within a habitat patch, DFP_{\min} is the proportion of stream length in a habitat patch vulnerable to debris flows, and TP_{\min} is the proportion of stream length in a habitat patch vulnerable to post-fire temperature increases. Maximum threat overlap was the smaller of the two values for DFP_{\max} and TP_{\max} . Each simulation included 2,500 trials.

Effective patch size (EPS)

See description of EPS above.

Realized patch size (RPS)

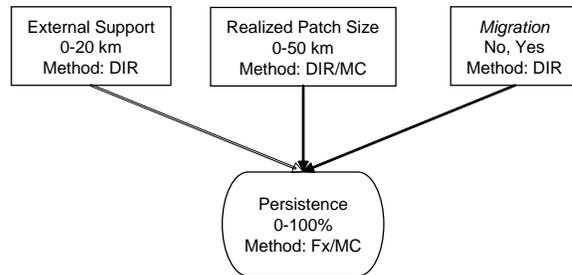
RPS is the stream length (km) within a habitat patch available for spawning and rearing by bull trout following wildfire and fire-related debris flows and stream temperature increases. Within the model RPS is a function of effective patch size (EPS), debris flow realized (DFR), thermal realized (TR), and hazard overlap (HO). The formula for calculating RPS is

$$RPS = EPS - (EPS \times DFR) - (EPS \times TR) + (EPS \times HO), \tag{D.19}$$

where RPS and EPS are expressed in km and DFR, TR, and HO are proportions.

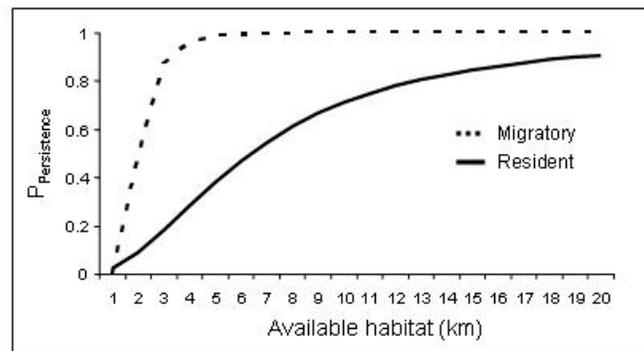
Each trial of the Monte Carlo was based on four inputs: EPS (km), DFR, TR, and HO. Values for each were selected using a uniform distribution. RPS (km) was calculated using these inputs and equation 9. EPS (km) ranged from 0 to 50 km and was divided among 11 bins: 0 km, 0-5 km, 5-10 km, ..., 45-50 km. DFR, TP, and HO ranged from 0 to 1.0 and were divided among six bins: 0, 0.01-0.25, 0.26-0.5, 0.51-0.75, 0.76-0.99, 0.99-1.0. RPS ranged from 0 to 50 km and was divided among 11 bins: 0 km, 0-5 km, 5-10 km, ..., 45-50 km. There 5,000 trials in each simulation.

Persistence (P)



P is the probability a bull trout habitat patch will continue to support spawning and rearing by bull trout following a wildfire event and associated disturbances. We conceptualized

the probability of persistence for a habitat network as a function of realized patch size (RPS), nearby habitat (NH), and the presence of a migratory life history in the bull trout population (Migration (M)). We estimated P for habitat networks using a Monte Carlo simulation assuming larger, more well-connected networks supporting both resident and migratory populations would be more likely to persist following disturbance than networks lacking these features.



In this figure, available habitat is the sum of RPS and NH of a habitat network. Increases in RPS, NH, or both, increase the persistence probability for both life histories. The relative gains in persistence probability for increased habitat size or connectivity are much higher for migratory populations where individuals from multiple habitat networks interact regularly.

Size of a resident population and probability of persistence is largely a function of network size, with larger networks supporting larger resident populations and having a greater probability of long-term persistence (MacArthur and Wilson, 1963; Hilderbrand and Kershner, 2000). For resident bull trout, persistence is a function of the ability of a habitat patch to absorb disturbances minimizing the chance the entire population is exposed to a single event.

Therefore, increasing patch area results in relatively linear increases in persistence probability.

The largest continuous patch of fire disturbance identified by Hessburg et al. (2007) was less than 5,000 ha. A high-severity fire of this size is statistically unlikely; however, six of the nine extant bull trout spawning patches in the SFBR are smaller than 5,500 ha with stream lengths of

3.7 to 15.5 km. Despite a detailed understanding of the threats posed by high-severity fire within these spawning patches there is still a large amount of uncertainty associated with the realization of these threats (Rhodes and Baker, 2007). Therefore, we conservatively set the asymptote for the resident-only persistence curve at 20 stream km, corresponding to an area of approximately 10,000 ha. The resident life-history form does not interact with surrounding populations; therefore, they may be prone to small population effects, such as in-breeding depression (Fahrig, 2002; Fausch et al., 2006; Peterson et al., 2008). The asymptote for the resident-only persistence curve is 90%, representing a 10% “penalty” associated with the absence of a migratory component in the population.

Each trial of the Monte Carlo simulation was based on randomly selected values of RPS and NH. Values for these variables ranged from 0-50 km for RPS and 0-20 km for NH, each variable into 5-km bins. We used the following functions to calculate persistence probabilities for migratory and resident populations. The formula for calculating persistence for populations with migratory individuals was

$$P_{mig} = \frac{1}{1 + e^{(0.1 - (1.4 \times \ln(l_{rps} + l_{nh})))}} \quad (D.20)$$

where P_{mig} is the probability of post-fire persistence following wildfire-related disturbance, l_{rps} is RPS (km), and l_{nh} is the NH (km). For resident-only populations the formula for P was

$$P_{res} = \frac{1}{1 + e^{(3.7034 - (2 \times \ln(l_{rps} + l_{nh})))}} \quad (D.21)$$

where P_{res} is the probability of post-fire persistence following wildfire-related disturbance, l_{rps} is the RPS (km), and l_{nh} is the NH (km).

Uncertainty in P was integrated by selecting values for P using the above equations and randomly selected value for RPS and NH. Random values for RPS and NH were selected using

a uniform distribution. We developed a conditional probability table for P by comparing predicted persistence probabilities to a random number between 0 and 1. Trials where the persistence probability was greater than or equal to the random number the population persisted. There were 30,000 trials in each simulation.

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