Canadian Journal of Fisheries and Aquatic Sciences
 Journal canadien des sciences halieutiques et aquatiques

**Evaluating Tradeoffs in Bull Trout Reintroduction Strategies
 Using Structured Decision Making**

Journal:	<i>Canadian Journal of Fisheries and Aquatic Sciences</i>
Manuscript ID	cjfas-2016-0516.R1
Manuscript Type:	Article
Date Submitted by the Author:	14-Mar-2017
Complete List of Authors:	Brignon, William; Columbia River Fish and Wildlife Conservation Office, USFWS ; United States Geological Survey, Oregon Cooperative Fish and Wildlife Research Unit; Oregon State University, Department of Fisheries and Wildlife Peterson, James; Oregon State University, Department of Fisheries and Wildlife Dunham, Jason; U.S. Geological Survey Schaller, Howard; U.S. Fish and Wildlife Service, Pacific Region Fisheries and Aquatic Conservation Schreck, Carl; U.S. Geological Survey, Oregon Cooperative Fish and Wildlife Research Unit
Please Select from this Special Issues list if applicable:	
Keyword:	CONSERVATION < General, ENDANGERED SPECIES < General

SCHOLARONE™
 Manuscripts

1 Evaluating Tradeoffs in Bull Trout Reintroduction Strategies Using Structured Decision
2 Making
3
4 William R. Brignon (Corresponding author)
5 bill_brignon@fws.gov
6 United States Fish and Wildlife Service, Columbia River Fish and Wildlife Conservation
7 Office, 1211 SE Cardinal Court, Suite 100, Vancouver, Washington 98683, USA
8 and
9 Oregon Cooperative Fish and Wildlife Research Unit, United States Geological Survey,
10 Department of Fisheries and Wildlife, Oregon State University, 104 Nash Hall, Corvallis,
11 Oregon 97331-3803, USA
12
13 James T. Peterson
14 jt.peterson@oregonstate.edu
15 Oregon Cooperative Fish and Wildlife Research Unit, United States Geological Survey,
16 Department of Fisheries and Wildlife, Oregon State University, 104 Nash Hall, Corvallis,
17 Oregon 97331-3803, USA
18
19 Jason B. Dunham
20 jdunham@usgs.gov
21 U.S. Geological Survey, Forest and Rangeland Ecosystem Science Center, 3200
22 Southwest Jefferson Way, Corvallis, Oregon 97331, USA
23

24 Howard A. Schaller

25 howard_schaller@fws.gov

26 United States Fish and Wildlife Service, Fish and Aquatic Conservation Program, Pacific

27 Regional Office, 911 NE 1th Avenue, Portland 97232, USA

28

29 Carl B. Schreck

30 Carl.schreck@oregonsate.edu

31 Oregon Cooperative Fish and Wildlife Research Unit, United States Geological Survey,

32 Department of Fisheries and Wildlife, Oregon State University, 104 Nash Hall, Corvallis,

33 Oregon 97331-3803, USA

Draft

34 **ABSTRACT**

35 Structured decision making allows reintroduction decisions to be made despite
36 uncertainty by linking reintroduction goals with alternative management actions through
37 predictive models of ecological processes. We developed a decision model to evaluate
38 the tradeoffs between six bull trout *Salvelinus confluentus* reintroduction decisions with
39 the goal of maximizing the number of adults in the recipient population without reducing
40 the donor population to an unacceptable level. Sensitivity analyses suggested the
41 decision identity and outcome were most influenced by survival parameters that result in
42 increased adult abundance in the recipient population, increased juvenile survival in the
43 donor and recipient populations, adult fecundity rates and sex ratio. The decision was
44 least sensitive to survival parameters associated with the captive-reared population, the
45 effect of naivety on released individuals, and juvenile carrying capacity of the
46 reintroduced population. The model and sensitivity analyses can serve as the foundation
47 for formal adaptive management and improved effectiveness, efficiency, and
48 transparency of bull trout reintroduction decisions.

49 INTRODUCTION

50 Bull trout *Salvelinus confluentus* require cold water for spawning and early
51 rearing (Buchanan and Gregory 1997), and if climate change predictions are accurate and
52 habitat degradation and fragmentation continue to limit connectivity, then populations are
53 expected to decline even further (Rieman et al. 1997). The Pacific Northwest warmed
54 0.8°C in the twentieth century, and this trend is expected to continue and magnify (IPCC
55 2007; Mote and Salathé 2010). Rieman et al. (2007) evaluated bull trout response to a
56 range of predicted climate warming scenarios for the Columbia River Basin and
57 estimated a loss ranging from 18–92% of suitable natal habitat. The range of this
58 estimate suggests substantial uncertainty when considering the effects of climate change
59 on bull trout populations (Wenger et al. 2014). It is thought that warming trends will
60 restrict populations to small isolated patches in headwater streams, further limiting
61 population connectivity (Rieman et al. 2007) and increasing extinction risk (Rieman and
62 McIntyre 1995). Large populations residing in less degraded habitat will increase in
63 importance as source populations for recolonization and reintroduction efforts (Dunham
64 and Rieman 1999).

65 There are numerous options to deal with the harmful effects of climate change on
66 bull trout populations. Isaak et al. (2010) suggested minimizing disturbances in riparian
67 habitat (i.e., grazing, road building, and timber harvest) to buffer stream temperatures
68 from additional warming. Unoccupied habitat can be made available for natural
69 recolonization or reintroduction (Dunham and Rieman 1999) by removing or
70 reengineering manmade structures that limit habitat connectivity. Connectivity is the
71 mechanism that promotes complex life histories exhibited by bull trout where resident

72 populations supply individuals to migratory populations and vice versa (Rieman and
73 McIntyre 1993). Connectivity between suitable habitats is required to promote these life
74 history strategies and long term persistence of the population. Connecting multiple small
75 isolated populations that result from a warming climate will create functional
76 metapopulations that provide a greater value to conservation (Dunham and Rieman
77 1999). In areas where reclaiming and reconnecting suitable habitat does not result in
78 natural recolonization managers may decide to implement a reintroduction program.

79 The Bull Trout Recovery Plan (USFWS 2015) identifies translocation and
80 controlled propagation as potential tools to produce individuals for reintroduction
81 programs. A translocation strategy consists of capturing individuals from a donor
82 population and directly transporting and releasing them into a recipient habitat.
83 Controlled propagation can be divided into two strategies; captive-rearing, and artificial
84 production. A captive-rearing strategy involves capturing, transporting and rearing wild
85 individuals in a controlled environment to be released at a later date when their potential
86 for survival has been improved. Artificial production differs from captive rearing in that
87 wild fish are brought into captivity and spawned. The resulting progeny are reared in
88 captivity to maturity and subsequently spawned or released into the wild as part of a
89 reintroduction program (Hard et al. 1992).

90 Each strategy comes with its own advantages, disadvantages, and uncertainties.
91 Translocation is the least expensive of the three strategies and comes with a variety of
92 benefits. Natural fish behaviors, genetic diversity, and ecological diversity will not be
93 affected by the captive environment (e.g., Flagg and Nash 1999; Brown and Day 2002;
94 Fraser 2008; Naish et al. 2008). For these reasons translocation is the first option among

95 many managers and has produced some success with releases of bull trout into the
96 Willamette and Clackamas rivers in Oregon (UWBTWG 2010; Barrows et al. 2016).
97 Disadvantages to translocation include high risk to donor populations due to the number
98 of individuals needed to create a viable population (Shivley et al. 2007), the potential for
99 local adaptation of donor populations hindering performance in the recipient system, and
100 limited ability for fish health screening. Relative to translocation, captive rearing and
101 artificial production can provide more individuals for reintroduction while lessening the
102 demographic impact to donor populations (Shivley et al. 2007). These strategies also
103 provide a greater opportunity to monitor fish health. However, there is a suite of recent
104 literature summarizing the effects of captivity on salmonids (e.g., Flagg and Nash 1999;
105 Brown and Day 2002; Fraser 2008; Naish et al. 2008). Impacts to stress (Dickens et al.
106 2010), behavior (Berejikian et al. 2001), morphology (Taylor 1986) and concerns over
107 genetic diversity (Fraser 2008) are inevitable in captive rearing and artificial production
108 programs. In general, the costs and benefits of a captive-rearing strategy fall in-between
109 a translocation and an artificial production strategy. Managers opt to use artificial
110 production only when natural populations are extremely low, all other options have been
111 exhausted, and the extinction risk outweighs the negative impacts of artificial production
112 (Hard et al. 1992). Selecting the best strategy to benefit bull trout conservation will
113 depend on levels of uncertainty, the understanding of how the biological system works,
114 the amount of risk managers are willing to incur, and the probability of achieving the
115 desired outcome (e.g., increased abundance, distribution, or both).

116 A better understanding of the tradeoffs between alternative bull trout
117 reintroduction decisions is needed and can be accomplished through structured decision

118 making. Structured decision making is a process that promotes informed decision
119 making and transparency by linking explicit quantifiable objectives to management
120 alternatives (i.e., reintroduction strategies) through predictive models (Conroy and
121 Peterson 2013; Clemen 1996). These predictive models can represent competing
122 hypotheses that describe the understanding of how ecological processes may respond to a
123 particular decision and the structure of those processes. However, most decisions in
124 natural resources are confounded by uncertainty. Rather than ignore uncertainty,
125 structured decision making incorporates it into the decision process. Natural resource
126 decisions reoccur in time and space and new information gained through monitoring will
127 minimize uncertainty (i.e., learning, adaptive management). Given the inherent
128 uncertainty, need for transparency, and importance of natural resource conservation and
129 recovery, the Department of the Interior and the U.S. Fish and Wildlife Service have
130 adopted structured decision making to guide habitat and species conservation into the
131 future (i.e., Strategic Habitat Conservation; USFWS 2008; Williams et al. 2009).

132 The process of structured decision making can inform future bull trout
133 reintroductions and would be a useful tool for managers given the threatened status of the
134 species and uncertainty of reintroduction strategies (USFWS 2015). To this end, we
135 employed structured decision making to evaluate the tradeoffs of alternative bull trout
136 reintroduction decisions. We identified model objectives, alternative reintroduction
137 decisions, and presented the model structure. A series of sensitivity analyses were
138 conducted to determine the relative uncertainty of decision model parameters. We then
139 used the model and stochastic dynamic programming to develop a state dependent policy
140 that managers can use to guide bull trout reintroductions given any combination of

141 available donor and recipient population sizes (i.e., states). Finally, we used forward
142 simulation to compare the quasi-extinction probability of donor and recipient populations
143 under different management alternatives and environmental stochasticity.

144 **METHODS**

145 *Valuation of decision outcome*

146 Fundamental objectives of a reintroduction are to build a viable recipient
147 population, and do so without endangering the donor population (IUCN/SSC 2013).
148 Accurate estimates of abundance and demographics are needed to determine the
149 probability that a population will persist (Caswell 2001; Morris and Doak 2002).
150 However, obtaining accurate estimates of demography is difficult and expensive, relative
151 to estimation of adult abundance. Redd surveys are widely used to estimate adult bull
152 trout abundance. However, these estimates may be biased by observer experience,
153 sampling effort, and habitat variability. These biases may result in abundance estimates
154 that are typically less than the true population size (Al-Chokhachy et al. 2005; Mulfeld et
155 al. 2006) which suggests that basing the decision value on abundance estimates derived
156 from readily available redd count data will result in conservative abundance estimates
157 and limit the need for additional monitoring resulting in a user-friendly model. We
158 calculated the value of a decision outcome (i.e., the utility) as the number of adults in the
159 recipient population up to a maximum of 300 adults, with the assumption that the
160 marginal gain from additional adults would not outweigh the cost and effort of additional
161 releases. We assumed that any removal of individuals would reduce adult abundance in
162 the donor population by some level and used a state-based approach to apply a penalty if
163 the donor population was reduced to an unacceptable state (Table 1). The value of the

164 decision (i.e., utility) was multiplied by zero if abundance in the ending donor population
165 state was less than abundance in the starting donor population state (Table 1). For
166 example, if the recipient population is estimated to be 80 adults after a reintroduction and
167 the donor population is not reduced to an undesirable state the value of the division is 80
168 $\times 1 = 80$. Whereas, if the donor populations is reduced to an undesirable state the value
169 of the decision is $80 \times 0 = 0$.

170 *Decision alternatives*

171 A reintroduction decision for bull trout consists of two components. First, a
172 reintroduction strategy (i.e., translocation, captive rearing, and artificial production) is
173 selected. Then, depending on the strategy selected, the life stage(s) and number of
174 individuals needed for the reintroduction effort can be determined. We evaluated six
175 reintroduction decision alternatives (Table 2). The first alternative is to do nothing.
176 Alternatives 2 through 4 are structured similar to the Clackamas River reintroduction in
177 which 30 adults, 30 subadults, and 1000 juveniles were targeted for transfer (USFWS
178 2011). We used these numbers for the basis of alternative 4 in which 1000 juveniles, 60
179 sub adults, and 40 adults were translocated. Also, we evaluated a 1000 juvenile-only
180 translocation (alternative 2) and a 60 adult-only translocation (alternative 3) using similar
181 numbers of fish. The fifth alternative was a captive-rearing strategy to collect 20,000
182 eyed embryos by hydraulic redd sampling (McNeil 1964; Berejikian et al. 2011) and
183 incubate and rear those individuals for one year before releasing them into the recipient
184 habitat (alternative 5). The sixth alternative is an artificial production strategy of
185 collecting 30 small adults and 30 large adults, artificially spawning those individuals in
186 captivity and releasing the resulting progeny after one year of rearing. We assumed that

187 managers would be able to collect the exact number of individuals for the planned
188 reintroduction (i.e., perfect controllability; Conroy and Peterson 2013) and each decision
189 represents what is feasible from either a logistical perspective (e.g., collection, rearing)
190 and/or reasonable risk to a donor population.

191 Each decision alternative consisted of five annual collections and subsequent
192 releases of individuals. Do-nothing and translocation strategies encompassed five years,
193 whereas the captive-rearing strategy and artificial production strategy encompassed six or
194 seven years, respectively. These timesteps were a function of the additional time it takes
195 to incubate and rear animals in captivity before release. A five to seven year timestep
196 coincides with the age at maturity in bull trout (4 to 7 years; McPhail and Murray 1979;
197 Frailey and Shepard 1989; Johnston 2005) and allows the youngest life stages an
198 opportunity to become mature adults at the end of the time horizon. The Final Bull Trout
199 Recovery Plan is to be updated approximately every five years to reflect current
200 information (USFWS 2015) and this time horizon allows new information gained from
201 monitoring efforts to be incorporated into recovery plan revisions.

202 *Model structure and parameters*

203 The foundation of the model were three stage-based, post-breeding, Leslie matrix
204 models (Caswell 2001) that represent the donor, captive, and recipient populations. Once
205 a decision is made, individuals are collected, moved between populations, and released
206 before future populations are projected. Populations were projected for seven years using
207 matrix multiplication (Caswell 2001). A seven year projection provided all individuals
208 collected for an artificial production strategy to be released before evaluating a decision
209 outcome. We incorporated demographic stochasticity after matrix population projections

where the number of individuals surviving was drawn from a binomial distribution where n_i is the number of individuals in life stage i and p is the stage base survival parameter. Similarly, fecundity was drawn from a Poisson distribution where λ was equal to F_i (Conroy and Peterson 2013).

The donor model consisted of five life stages adopted from Schaller et al. (2014); embryos, juveniles, subadults, small adults, and large adults. The transition matrix of the donor population (d) takes the form:

$$\mathbf{A}_d = \begin{pmatrix} 0 & 0 & 0 & F_4 & F_5 \\ G_1 & 0 & 0 & 0 & 0 \\ 0 & G_2 & P_3 & 0 & 0 \\ 0 & 0 & G_3 & P_4 & 0 \\ 0 & 0 & 0 & G_4 & P_5 \end{pmatrix},$$

where F_i is the fecundity of stage i , G_i is the probability of surviving and transitioning to stage $i + 1$, and P_i is the probability of surviving and staying in stage i . Small adult fecundity (F_4) is calculated as:

$$F_4 = m_4 \times \text{sexr} \times B_4 \times (G_4 + P_4),$$

and large adult fecundity (F_5) is calculated as:

$$F_5 = m_5 \times \text{sexr} \times B_5 \times P_5,$$

where m_i is the number of eggs produced by a female in stage i , sexr is the sex ratio of the adult population, and B_i is the probability of a female in stage i spawning. We chose to incorporate sex ratio in the population models rather than build a female-only matrix model (Caswell 2001) to evaluate the uncertainty associated with sex ratio of a reintroduction program. We incorporated density dependence in survival at the juvenile stage (G_1) of the population as:

$$G_1 = \text{egg.max.S} \times \left(1 - \exp \left[\frac{-\text{juvi.CC}}{\text{egg.max.S} \times N.\text{juvi}} \right] \right),$$

231 where *egg.max.S* is the maximum embryo survival rate at low densities, *juvi.CC* is the
 232 juvenile carrying capacity, and *N.juvi* is the number of juveniles in the population (Lee
 233 and Rieman 1997).

234 Animals released into new environments, captive or natural, take time to
 235 acclimate and individuals that fail to do so may exhibit reduced survival (Dickens et al.
 236 2010). Therefore, the transition matrix for the captive population consisted of ten life
 237 stages; a naive (*n*) component of five life stages that transition into the experienced (*e*)
 238 component of the population. Animals that are collected and transferred into captivity
 239 remained in the naive group for one timestep before transitioning into the experienced
 240 group. The transition matrix of the captive population (*c*) takes the form:

$$241 \quad \mathbf{A}_c = \begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & nF_4 & nF_5 & 0 & 0 & 0 & eF_4 & eF_5 \\ nG_1 & 0 & 0 & 0 & 0 & eG_1 & 0 & 0 & 0 & 0 \\ nR_1 & nG_2 & nP_3 & 0 & 0 & eR_1 & eG_2 & eP_3 & 0 & 0 \\ 0 & 0 & nG_3 & nP_4 & 0 & 0 & 0 & eG_3 & eP_4 & 0 \\ 0 & 0 & 0 & nG_4 & nP_5 & 0 & 0 & 0 & eG_4 & eP_5 \end{pmatrix},$$

242 where stage based fecundity (F_i), growth (G_i), and survival probabilities (P_i) are as
 243 described for the donor population. However, each parameter in the naive component (*n*)
 244 of the population matrix was multiplied by an additional variable to account for the
 245 reduced survival of a naive individual (*naive*). For example, if the probability of survival
 246 for an experienced fish is 0.80 and *naive* is 0.50, then the probability of survival for a
 247 naive fish is equal to 0.80 x 0.50, or 0.40. In addition, we added a rapid growth
 248 parameter (R_1) that represents the probability that an embryo may survive and advance to
 249 the subadult life stage when provided a controlled captive environment and consistent

250 feed availability (Fredenberg et al. 1995; Fredenberg 1998). We assumed that space and
 251 resources in captivity were not limiting and therefore density dependence would not
 252 affect the captive population.

253 The transition matrix of the recipient population (r) was similar to the captive
 254 population in that we included a naive (n) and experienced (e) component. The transition
 255 matrix takes the form:

$$256 \quad \mathbf{A}_r = \begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & nF_4 & nF_5 & 0 & 0 & 0 & eF_4 & eF_5 \\ nG_1 & 0 & 0 & 0 & 0 & eG_1 & 0 & 0 & 0 & 0 \\ 0 & nG_2 & nP_3 & 0 & 0 & 0 & eG_2 & eP_3 & 0 & 0 \\ 0 & 0 & nG_3 & nP_4 & 0 & 0 & 0 & eG_3 & eP_4 & 0 \\ 0 & 0 & 0 & nG_4 & nP_5 & 0 & 0 & 0 & eG_4 & eP_5 \end{pmatrix},$$

257 where stage based fecundity (F_i), growth (G_i), survival probabilities (P_i), and reduced
 258 survival for naive individuals (*naive*) are as described for the donor and captive
 259 populations. We again accounted for density dependence using the equation from Lee
 260 and Rieman (1997) with a carrying capacity (*juvi.CC*) and maximum embryo survival
 261 (*egg.max.S*) parameter that are specific to the recipient population, and where the *N.juvi*
 262 parameter was equal to the summation of juvenile individuals in both the naive and
 263 experienced components of the population. In addition to applying a survival reduction
 264 for a fish's naivety, we also applied a survival reduction at the first timestep depending
 265 on which reintroduction strategy was used. This reduction represented the negative
 266 effects of handling and captivity and we assumed that survival of a translocated fish
 267 (*trans.reduc*) would be greater than that of a captive-reared (*cap.reduc*) fish, which
 268 would be greater than that of an artificially produced fish (*artprod.reduc*).

269 Until recently, a comprehensive set of demographic parameters for all bull trout
270 lifestages was unavailable. To fill this data gap Schaller et al. (2014) estimated survival
271 rates for a bull trout metapopulation using ten years of mark-recapture data. We used
272 similar values to parameterize the Leslie matrixes (Table 3). Estimates of the survival
273 reduction for naive fish and reintroduction strategy reduction do not exist. As such, we
274 set the survival reduction for naive fish at 0.50 with the understanding that the variable
275 would range from 0.10 to 0.99 during sensitivity analysis which would provide a better
276 understanding of the effect of this variable on the decision outcome. After conversations
277 with biologists working with bull trout, we assumed that handling stress during capture,
278 transport, and release would result in delayed mortality and reduce the survival of
279 translocated fish by 0.70. In addition to the effect of capture, transport, and release, we
280 assumed that deleterious effect of captivity would result in a reduced survival of captive-
281 reared fish by 0.30. We applied a survival reduction of 0.20 to artificially produced fish
282 due to the combined effects of capture, transport, release, captivity, and artificial
283 spawning practices that lack natural mate selection processes (Table 3).

284 *Decision model sensitivity analysis*

285 Before determining the optimal decision we evaluated the characteristics of the
286 decision model with sensitivity analyses, response profiles, and indifference curves
287 (Clemen 1996; Conroy and Peterson 2013). These analyses were used to identify model
288 parameters that have the greatest effect on the value of the decision and the resulting
289 decision alternative, as well as ensure the model is functioning as expected. One-way
290 sensitivity analysis was conducted by holding all model parameters, including starting
291 donor population abundance, at a mean value and incrementally varying each parameter

292 between a minimum and maximum range (Table 3, Table 4). The output of the one-way
293 sensitivity analysis was used to create a tornado diagram of all parameters (Figure 1) and
294 indifference curves for each parameter (Figure 2). To evaluate interactions between
295 parameters, we conducted a two-way sensitivity analysis in which two model parameters
296 were varied simultaneously while the others were kept at mean values. We then created
297 contour plots of the expected outcome and response profiles of the identity of the optimal
298 decision (Figure 2; Conroy and Peterson 2013). We ran 1000 simulations for all
299 sensitivity analyses and used the mean utility value.

300 *Optimal state-based decisions*

301 The goal of optimization is to identify the decision alternative that maximizes or
302 minimizes the outcome as quantified by the utility. We applied stochastic dynamic
303 programming (SDP) to determine the optimal management decision (Lubow 1996; Rout
304 2009; Chadès et al. 2014) across a combination of donor and reintroduced population
305 states. Stochastic dynamic programming is used to solve Markov decision problems and
306 uses backward induction to identify the best decision under uncertainty by maximizing
307 the sum of future expected rewards, in this case, recipient adult abundance. This
308 technique is appropriate for management decisions that are confounded with uncertainty
309 and stochasticity, and consist of a finite set of potential states where sequential decisions
310 are being made (Bellman 1957; Rout et al. 2009). Stochastic dynamic programming has
311 been applied to maximize harvest of ducks (Anderson 1975) and sandhill cranes (Gerber
312 2015), exploitation of fisheries (Walters 1975; Walters and Hilborn 1976), and to
313 optimize the translocation of threatened species (Lubow 1996; Rout et al. 2009).

314 We generated 24 starting populations of bull trout with large adult abundances
315 ranging from 0 to 307 individuals. The stable age distribution from the donor transition
316 matrix was used to determine the abundance of all younger lifestages. This resulted in a
317 maximum of 1499 mature adults (small and large adults) in the largest starting population
318 (Table 4). Using these populations as starting abundances for the donor and recipient
319 population, we created every possible combination of populations that may be observed
320 in real world reintroduction scenarios, resulting in 576 (24 x 24) potential starting
321 population sizes. These populations were replicated 100 times to account for
322 demographic stochasticity during the optimization. For each of these starting
323 populations, the utility and ending population size of the donor and recipient populations
324 were calculated for all six reintroduction decisions using the mean parameter values
325 (Table 3). We conducted 10,000 simulations of this process and the utility and ending
326 populations used to create a transition probability matrix for each decision that describes
327 the probability of transitioning from a starting population state to an ending population
328 state and the associated return (i.e., the decision outcome or utility). The transition
329 probability tables and return values were put in the Markov decision process policy
330 iteration algorithm implemented in R package *MDPtoolbox* (Chadès et al. 2014) with a
331 discount factor of 0.9999. The algorithm uses backward induction outputs a table of state
332 dependent management policies for all possible starting donor or recipient population
333 states. The adult populations were discretized into five abundance states; 0 to 50, 50 to
334 100, 100 to 250, 250 to 500, and greater than 500 individuals (Table 5).

335 *Leslie matrix sensitivity and elasticity*

336 We considered the implications of demography with sensitivity and elasticity
337 analyses of the population growth rate (λ) to changes in transition matrix elements
338 (Caswell 2001; Wilson 2003) for the donor, recipient, and captive populations.
339 Sensitivity and elasticity estimates were calculated with the *vitalsens* function in the
340 popbio package in R which calculates sensitivities and elasticities of lambda to lower-
341 level vital rates (e.g., *sexr*, *B_i*, *m_i*, *d.juvi.CC*; Stubben and Milligan 2007). All parameters
342 were set at mean values (Table 3) for these analyses and juvenile abundance (*N.juvi*) was
343 set at 15,000 individuals to determine the sensitivity and elasticity of lower-level vital
344 rates contained in the density dependent equation (i.e., *egg.max.s, juvi.cc*) of *G_I* for the
345 donor and recipient populations.

346 *Management scenarios*

347 A viable population is one that has a low probability of going extinct within a
348 specific timeframe (i.e., quasi-extinction probability; Morris and Doak 2002). To better
349 understand the viability of populations after a management action, we used forward
350 simulation to compare quasi-extinction probabilities of the donor and recipient
351 populations under seven different management scenarios. Six of the scenarios consisted
352 of simulating each reintroduction decision in the set (Table 2) for one decision timestep
353 (i.e., 7 years) and using forward simulation to estimate the donor and recipient population
354 abundances after 50 years (i.e., one-and-done scenario). The final scenario consisted of
355 using the optimal state dependent policy (Table 5) to simulate recurring reintroduction
356 decisions over a 50 year time period (i.e., the SDP scenario). With this scenario, current
357 adult abundance of the donor and recipient populations are estimated via monitoring, and
358 the state dependent policy (Table 5) identifies the optimal management decision to be

359 implemented for the next timestep (i.e., 7 years). Once a timestep is complete, the state
360 dependent policy informs the subsequent management action, and so on for a total of 50
361 years, or seven consecutive management decisions.

362 All scenarios were run with four starting adult abundance states for the donor
363 population; 100, 150, 300, and 500 adults. Similarly, we also ran all scenarios with an
364 annual disturbance probability of 0.05 to evaluate the resiliency of the populations under
365 environmental stochasticity. If a disturbance did occur the population abundance of each
366 life stage was reduced by 25%. Populations were simulated 1000 times for each decision,
367 starting donor population state, and with and without environmental stochasticity, and the
368 mean adult abundances were plotted (Figure 4, Figure 5) and used to determine the quasi-
369 extinction probability of the average population. Quasi-extinction probabilities were
370 determined with the count-based extinction time cumulative distribution function in R
371 package *popbio* (Table 6; Stubben and Milligan 2007; Morris and Doak 2002).

372 Estimated mean population size (μ) and sample variance (σ^2) was calculated from
373 year 8 to 50, Current population size (N_c) was equal to adult population abundance at
374 year 8, the quasi-extinction threshold (N_e) was set at 50 adults, and the latest time to
375 calculate extinction probability (t_{max}) was equal to 50 years. Beginning the quasi-
376 extinction estimates at year 8 allows all scenarios to complete one decision timestep.

377 RESULTS

378 *Optimal state-based decisions*

379 The state dependent policy included four of the six decision sets that were
380 evaluated. All translocation decisions (alternatives 2, 3, and 4) and the do nothing
381 decision (alternative 1) were included in the state dependent policy table (Table 5).

Captive-rearing of embryos (alternative 5) and artificial production (alternative 6) decisions were never the optimal alternative. In situations where adult abundance in the donor population was below 100 individuals or the recipient population was above 500 individuals the optimal strategy was to do-nothing (alternative 1). Any scenario where the recipient adult abundance was less than 50 individuals, like one would expect to encounter at the first timestep of a reintroduction program, the optimal decision changed four times depending on donor population abundance. When donor abundance was greater than 100 individuals the optimal decision shifted from doing nothing (alternative 1) to translocating 1000 juveniles (alternative 2), and between 300 to 500 donor adults the optimal decision shifted to translocation of all life stages (alternative 4). When the donor population was large (i.e., greater than 500 adults), translocating adults only (alternative 3) became the optimal decision. As a reintroduction program continues, the decisions prescribed in the policy table reflect the value in continued supplementation of the recipient population to reduce extirpation risks while balancing the risk of crashing the donor population. This tradeoff process continued until the recipient population was above 300 individuals at which point the marginal gain of additional adults was negligible, as defined by the structure of the utility function, resulting in the optimal decision of doing nothing.

Decision model sensitivity analysis

The tornado diagram created from the one-way sensitivity analysis (Figure 1) suggested that the decision outcome was most influenced by survival parameters that resulted in increased adult abundance in the recipient population ($r.P_5$, $r.G_2$, $r.G_3$), and maximum egg survival in the donor and recipient populations ($d.egg.max.s$, $r.egg.max.s$).

405 The decision outcome was also sensitive to small and large adult fecundity rates and sex
406 ratio (m_i , $sexr$). The outcome was least sensitive to survival parameters associated with
407 the captive population ($c.P_3$, $c.P_4$, $c.G_2$, $c.G_3$, $c.G_4$), the survival reduction of naive
408 reintroduced individuals ($r.naive$), and juvenile carrying capacity of the reintroduced
409 population ($r.juvi.CC$).

410 The response profiles from the one-way sensitivity analysis showed that the
411 optimal decision changed as a function of the strategy specific survival reduction (Figure
412 2a, b, c). For example, if a translocated fish survived at 60% or less than a non-
413 translocated fish, then the optimal decision was artificial production (alternative 6).
414 Whereas, at above a 60% survival reduction the optimal decision was to translocate
415 adults only (alternative 3). When the survival reduction for the translocation strategy
416 ($trans.reduc$) was at 60%, the decision maker would be indifferent to artificial production
417 or adult translocation, because either option was expected to produce the same outcome
418 (Figure 2a). Similarly, the optimal decision was to captive rear embryos (alternative 5)
419 when the captive-rearing survival reduction ($cap.reduc$) was above 40% (Figure 2b), and
420 artificial production (alternative 6) was the optimal decision when the survival reduction
421 for artificial production ($artprop.reduc$) was greater than 20% (Figure 2c). The survival
422 reduction for fish that are naive to captivity ($c.naive$) also influenced which decision was
423 optimal. As more fish survived in captivity more fish are available for release into the
424 recipient habitat, thereby resulting in more recipient adults; the primary component of the
425 decision utility. When naive captive fish survived at a rate of 50% or more than an
426 experienced captive fish, artificial production and then captive-rearing of embryos
427 became the optimal and second best decision, respectively (Figure 2d).

Response profiles suggested that the optimal reintroduction decision changed 4 times as a function of the adult abundance in the donor population; below 25 adults the optimal decision was to do nothing (alternative 1), between 25 and 250 adults the optimal decision was to translocate only juveniles (alternative 2), between 250 and 400 adults translocating all life stages (alternative 4) was the optimal decision, and above 400 adults translocating adults only (alternative 3) is the optimal decision (Figure 2e). Understandably so, parameters that drive donor adult abundance ($d.egg.max.s$, $d.juvi.CC$, $d.G_b$, $d.P_i$) exhibited very similar patterns as Figure 2e. There were other parameters that when varied over their estimated range did not change the optimal decision. The optimal decision was static to changes in growth and survival of captive individuals ($c.G_b$, $c.P_i$), probability of large adults spawning (B_5), juvenile carrying capacity in the recipient population ($r.juvi.CC$), and the survival reduction of naive reintroduced fish ($r.naive$; Figure 2f).

Two-way sensitivity analysis with all combinations of model parameters resulted in more than 1000 contour and response profile plots. We present contour and response profile plots of three parameter combinations that were influential in the decision context and represent the types of results one can expect from two-way sensitivity analyses (Figure 3). A comparison of the juvenile density dependent parameters ($d.egg.max.s$, $d.juvi.CC$) for the donor population indicated that when above a maximum egg survival of 0.14 the juvenile carrying capacity had a greater influence on the expected value of the decision, this was indicated by the expected value of the decision changing more across the range of carrying capacity values identified by the close contour lines (Figure 3a). There were four alternatives that resulted in the optimal outcome, depending on the value

451 of the juvenile carrying capacity and maximum egg survival of the donor population
452 (Figure 3b). When juvenile carrying capacity in the donor population was less than
453 ~5500 individuals, the optimal strategy was to do nothing (alternative 1), which most
454 likely avoided an unacceptable reduction in donor population abundance. Whereas,
455 translocating adults (alternative 3) was the optimal decision when both density dependent
456 parameters were in the upper end of the ranges that were evaluated (Figure 3b) and
457 resulted in a decision outcome of greater than 60 adults in the recipient population at the
458 end of one decision timestep (i.e., 7 years; Figure 3a).

459 The survival reduction of captive-reared (*cap.reduc*) and translocated
460 (*trans.reduc*) fish were parameters of interest given the limited information regarding the
461 effects of captive-rearing and direct translocation of bull trout. Two-way sensitivity
462 analysis suggested a strong interaction between these parameters when *cap.reduc* ranged
463 from 0.83 to 0.99 and *trans.reduc* ranged from 0.13 to 0.23; this was evident in the lack
464 of parallel contours between these ranges (Figure 3c). The optimal decision was to
465 captive rear embryos (alternative 5) when both parameters were the lower limits of their
466 range, whereas the optimal decision was to translocate adults (alternative 3) when
467 *cap.reduc* was less than 0.75 and *trans.reduc* was greater than 0.14. As *cap.reduc* nears
468 1.00, which indicated little to no effects of captivity on a fish's survival after release,
469 artificial production (alternative 6) was the optimal decision regardless of the value of
470 *trans.reduc* (Figure 3d).

471 The final combination of variables presented for the two-way sensitivity analysis
472 is survival of large adults in the recipient population ($r.P_5$) and adult abundance in the
473 donor population. One-way sensitivity analyses suggested that the decision outcome

(i.e., utility; Figure 1) and the decision identity (Figure 2e) were sensitive to these variables and collectively, these parameters are major components of the model, in that the utility was a function of recipient and donor adult abundances. When $r.P_5$ was below 0.14, the parameter had little influence on the value (Figure 3e) or identity (Figure 3f) of the decision. When adults in the donor population were less than 450 individuals, the value of the decision, regardless of the optimal strategy, resulted in 40 or less adults in the recipient population. Doing nothing (alternative 1) was the optimal strategy when donor adult abundance was less than 25 individuals and translocating 1000 juveniles (alternative 2) was the optimal strategy until donor adult abundance was greater than 300 individuals (Figure 3f). When donor adult abundance was approximately 390 individuals, the model indicated that translocation of all life stages (alternative 4) was the optimal decision, suggesting that at this abundance level, the donor population could sustain the removal of adults without falling below a point at which the donor penalty would be applied. When donor adult abundance was above 400 and $r.P_5$ was less than 0.14, artificial production (alternative 6) produced the greatest value resulting in between 40 and 60 adults in the recipient population. However, if $r.P_5$ was greater than 0.14 the value of the decision increased at a much faster rate, indicative of the narrow spaces between contour lines on this portion of the plot. This suggests that accurate estimates of adult survival in the recipient population ($r.P_5$) are important and any conservation actions that improve $r.P_5$ could result in a more than doubling of the decision value.

Matrix model sensitivity and elasticity analysis

Setting the vital rates for the donor and recipient populations at similar levels resulted in the sensitivities and elasticities for these Leslie matrices being similar (Table

7). The sensitivity and elasticity analysis of matrix model elements suggest that small changes in G_2 , G_3 , G_4 , P_5 , $sexr$, and $egg.max.s$ will have the greatest effect on λ of the donor and recipient populations, and small changes in G_3 , G_4 , R_I and $sexr$ will have the greatest effect on λ in the captive population. Surviving and persisting in the subadult lifestages (P_3) had a minimal effect on λ for all populations. The juvenile carrying capacity ($juvi.cc$) and number of juveniles ($N.juvi$) in the density dependence equation had a minimal effect on λ of the donor and recipient populations, with an increase in $N.juvi$ resulting in a reduction in λ . Neither the survival reduction of naive individuals (*naive*) or the survival reduction associated with a reintroduction strategy (*strat.reduc*) had any effect on λ .

Management scenarios

The simulation of management scenarios suggests that regardless of the starting adult abundance in the donor population the state dependent policy (SDP) scenario is better at balancing the risk to the donor population and the benefit to the recipient population. This is evident in that under the SDP scenario the adult donor population never drops below the quasi-extinction threshold (Ne) of 50 individuals (Figure 4) and the probability of quasi-extinction is less than 0.05, regardless of the starting donor adult abundance (Table 6). Similar results are seen with the one-and-done scenario of decisions 1 and 2, but these decisions fail to generate a recipient population of adults as large as or greater than the SDP scenario. Each of these three scenarios fail to generate a recipient population with a quasi-extinction probability of less than 1.00, with the exception of the SDP scenario and a large starting donor population of 500 individuals which has a quasi-extinction probability of 0.08 (Table 6). The one-and-done scenario

520 with decisions 3 through 6, consistently produce recipient populations as large as or
521 larger than the SDP scenario. However, these one-and-done scenarios also result in
522 quasi-extinction on the donor population when the adult donor population is small
523 (Figure 4; Table 6). When simulating a 5% chance of environmental stochasticity that
524 reduced the population abundance by 25%, all management scenarios resulted in quasi-
525 extinction of the donor and recipient population. However, despite the pending quasi-
526 extinction after 50 years, the SDP scenario continues to balance the risk to the donor
527 population while attempting to create a recipient population (Figure 5).

528 **DISCUSSION**

529 Structured decision making is a process that links alternative management actions
530 to quantifiable objectives through predictive models (Conroy and Peterson 2014). We
531 have presented an example of how the structured decision making process can be used to
532 evaluate the tradeoffs of alternative reintroduction decisions with the objective of
533 maximizing the number of adults in the recipient population without reducing the donor
534 population to an unacceptable level. The structure and inputs of this model are intended
535 to be flexible and can be adjusted to suit the reintroduction strategy, numbers of animals
536 for removal, and population specific vital rates for any bull trout donor population or
537 suitable recipient habitat throughout the historical range of the species. As the outcome
538 of ongoing and future reintroductions are realized through effective monitoring programs
539 the structural and demographic uncertainty of the model can be reduced, resulting in a
540 better understanding of system dynamics (i.e., learning). The process of incorporating
541 new information into subsequent decisions is adaptive management which is a special
542 case of structured decision making (Williams et al. 2009; Conroy and Peterson 2013).

Monitoring may consist of a combination of state variables (e.g., population abundance, patch occupancy) and vital rates (e.g., fecundity, survival, emigration). State variables describe the health and status of the system and are necessary when using a state dependent policy to guide management decisions. Vital rates describe the parameter rates that result in changes to the state variables (Nichols and Armstrong 2012) and monitoring of vital rates is necessary for the learning component of adaptive resource management. Even though demography is difficult and expensive to collect, it should not be ignored (Caswell 2001). The only comprehensive set of vital rates, that we found for all bull trout life stages was provided by Schaller et al. (2014). These estimates provided helpful starting values for the model though model parameters can be updated as monitoring informs future reintroductions. The bull trout decision model consisted of 34 demographic parameters making up three discrete populations (i.e., donor, captive, and recipient) and it is unlikely that resources would be available to monitor all demographic rates for each population. The results of the decision model sensitivity analysis suggests that additional resources available for monitoring the reintroduction effort are best focused on survival and growth parameters of the adult component in the recipient population. This will provide the greatest benefit in reducing uncertainty and better estimating the effect of these parameters on the decision outcome (Williams et al. 2009; Conroy and Peterson 2013). Monitoring the state of the donor and recipient populations ensures that they are not reduced to an undesirable state, is necessary if recurrent decisions are to be made, and if managers intend to identify the optimal decision given the state dependent policy (Table 5). Also, monitoring of vital rates and state variables is

565 implicit in fish culture programs with fish size, mortality, behavior, and population size
566 being documented as a part of basic animal husbandry.

567 Parameters of the captive population (i.e., *c.P3*, *c.P4*, *c.G2*, *c.G3*, *c.G4*), survival
568 reduction of naive reintroduced individuals (*r.naive*), and juvenile carrying capacity of
569 the reintroduced population (*r.juvi.CC*) were identified by the one-way sensitivity
570 analysis as having had the least influence on the decision outcome. This can be explained
571 in that these parameters do not directly affect abundances in the recipient or donor
572 population; the influential components of the utility function. These parameters are only
573 one component of an equation, combined with other lower-level vital rates (e.g.,
574 *cap.reduc*, *c.naive*, *B_i*, *sexr*, *m_i*, *egg.max.s*, *N.juvi*), that was used to calculate stage-
575 specific matrix elements. While one-way sensitivity analysis indicated these parameters
576 have little influence on the decision outcome, two-way sensitivity analysis suggests that
577 the influence of these parameters is contingent on level of other lower-level vital rates
578 (e.g., *cap.reduc*, *c.naive*, *B_i*, *sexr*, *m_i*, *egg.max.s*, *N.juvi*) used to calculate the stage-
579 specific matrix elements.

580 There are few examples of bull trout being reared or artificially spawned in
581 captivity and survival of animals in these programs is highly variable, resulting in
582 uncertainty. Survival and productivity in captivity will be a function of rearing
583 temperature, life stage, nutrition, disease, and probability of successful fertilization
584 (Fredenberg et al. 1995; Fredenberg 1998; N. Zymonas, Oregon Department of Fish and
585 Wildlife, personal communication, 2015). This uncertainty is compounded by the
586 unknown survival of artificially produced, captive-reared, and translocated bull trout
587 upon release into the recipient habitat. We examined these uncertainties by applying a

588 reduction in survival as a function of the reintroduction strategy (i.e., *trans.reduc*,
589 *cap.reduc*, *artprod.reduc*) and naivety (Table 3). The response profiles and two-way
590 sensitivity analysis suggests that the optimal decision is sensitive to these parameters and
591 future bull trout reintroduction programs would benefit from a better understanding of the
592 effects of reintroduction strategy and the concept of naivety on the survival of animals at
593 release (McCarthy et al. 2012) and ultimately the utility and identity of optimal
594 reintroduction decisions.

595 We evaluated six alternative management actions or decisions that could be taken
596 to reintroduce bull trout. These decisions reflect past and ongoing reintroduction
597 decisions (Buchanan et al. 1997; Soorae 2011; USFWS and ODFW 2011). Because a
598 reintroduction decision is a combination of a reintroduction strategy, life stage(s) and
599 number of animals, there are endless combinations of alternative decisions that can be
600 evaluated. The model was developed to be flexible and the inputs for life stage and
601 numbers of animals to be removed can easily be adjusted to evaluate the tradeoffs of
602 alternative translocation, captive-rearing, or artificial production decisions for site and
603 population specific reintroduction scenarios. The risk of incorporating flexibility in the
604 model is that determining the optimal decision could become an exercise in “dredging” or
605 “fishing” until the desired result is reached. Ideally the decision set will be developed
606 with stakeholder input and represent what is feasible and reasonable from a biological,
607 logistic, and financial point-of-view.

608 Optimization techniques require multiple objectives to be combined into one
609 value with an objective or utility function, however, the relative weights or values of each
610 attribute may differ between stakeholders. It is likely in reintroduction decisions that

611 donor populations and recipient habitats are managed by different stakeholder groups.
612 Some groups may be more risk adverse where other groups are more liberal regarding
613 risk. In these situations the penalty for reducing the donor population (Table 1) is a
614 composite of the risk aversion of the stakeholder group. Incorporating stakeholder input
615 in identifying the fundamental objectives improves buy-in with the decision process and
616 helps transparently convey how and why decisions are made.

617 A state dependent policy (Table 5) is a useful tool for reintroduction programs. A
618 reintroduction program is only considered a reintroduction in the first timestep of the
619 decision when abundance in the recipient population is zero. Managers and decision
620 makers can use the state dependent policy to identify the optimal decision at subsequent
621 timesteps given the current state of the donor and recipient populations. Ideally, a
622 reintroduction program would accomplish its goals after one timestep; however, this is an
623 unlikely outcome. Forward simulating a reintroduced population using one-and-done
624 management scenarios suggested that, in the best case, the recipient population would
625 consist of less than 80 adults after 50 years in the absence of environmental stochasticity
626 (Figure 4). Whereas, simulation of a reintroduction with the state dependent policy
627 resulted in more than 100 adults after 50 years. This is a function of selecting the optimal
628 decision given the current state of the donor and recipient populations which balanced the
629 risk to the donor population and the benefit to the recipient population. In doing so, the
630 quasi-extinction probability of the donor and recipient populations is minimized while
631 allowing for some level of reintroduction to occur (Table 6). The effective population
632 size of bull trout to minimize inbreeding depression is 50 adults and more than 500 adults
633 are needed to maintain adaptive genetic variation (Rieman and Allendorf 2001). Also,

634 the goal of the ongoing Clackamas River reintroduction is to reestablish a viable
635 population of 300 to 500 adults within 20 years (USFWS and ODFW 2011).
636 Collectively, this suggests that any reintroduction program for bull trout will require
637 more than one decision timestep to avoid the genetic effects of small population sizes and
638 to build a viable population that is resistant to demographic and environmental
639 stochasticity.

640 Before beginning a reintroduction a feasibility assessment should be completed to
641 determine the suitability of recipient habitat and donor populations (Dunham et al. 2011).
642 Components of a habitat assessment may explore historical occupancy, current species
643 presence, availability of spawning and rearing areas, mitigation of past, present, and
644 future threats, and likelihood of natural recolonization. In terms of the donor population,
645 an assessment may consider the potential for an evolutionary match to the recipient
646 habitat and strength of the donor population to withstand removal of individuals (see
647 Dunham et al. (2011) for example applied to Bull Trout). In our model we assumed that
648 suitable recipient habitat was identified and we considered the impact to the donor
649 population via a penalty applied to the decision utility when the donor population was
650 reduced below an unacceptable level. Two forms of uncertainty that are prevalent in
651 natural resource decisions are partial observability and partial controllability. Partial
652 observability is uncertainty associated with the ability to perfectly “see” nature and can
653 be incorporated into the decision process as statistical distributions (Conroy and Peterson
654 2013). For example, partial observability may influence bull trout reintroduction
655 decisions when redd surveys are used to estimate adult abundance. Partial controllability
656 is uncertainty in that the intended management action is not completely under control.

657 For example, it would be more likely to collect the number of individuals intended for a
658 reintroduction when the donor population is large. We chose not to include partial
659 observability in the model because redd surveys typically underestimate the number of
660 adults in a population (Al-Chokhachy et al. 2005; Mulfeld et al. 2006) resulting in more
661 conservative estimates of adult abundance. Also, it is difficult to include partial
662 observability and partial controllability when using stochastic dynamic programming to
663 determine the optimal decision and there are no software programs that can be used to
664 solve a partially observable Markov decision process (Conroy and Peterson 2013). As
665 reintroductions are implemented and initial estimates of partial controllability and partial
666 observability are generated via monitoring, consideration of these forms of uncertainty
667 can be incorporated into the model to inform future decisions (i.e., adaptive
668 management).

669 Three features must be present for the formal process of adaptive resource
670 management to occur: 1) recurrent decisions that allow for learning to influence
671 subsequent timesteps, 2) alternative models representing hypothesis of ecological
672 dynamics that identify the optimal decision based on prediction of the outcome at a future
673 state, and 3) a monitoring program that generates data describing the outcome, which is
674 used to update subsequent decisions (i.e., learning; Williams et al. 2009; Conroy and
675 Peterson 2013). One of the first attempts at bull trout reintroduction occurred in the
676 McCloud River, CA and was conducted as a practice in trial and error. An artificial
677 production program began in 1989 and over 60 adults were collected from the Klamath
678 basin. High mortality in captivity resulted in less than 300 fingerling bull trout for the
679 reintroduction and after five years of monitoring the reintroduction was considered a

680 failure. There was only one release of individuals due to a reduction in abundance and
681 distribution of the donor stock (Buchanan et al. 1997). More recent reintroductions
682 consist of an “ad-hoc” adaptive management approach. Between 1997 and 2005, more
683 than 10,000 fry were translocated from the McKenzie River, OR to the Willamette River,
684 OR resulting in less than 15 redds observed. Due to the limited success of the program,
685 translocation of fry was discontinued and the reintroduction program has shifted to using
686 a captive-rearing strategy since 2007. The wild fry are now reared in captivity for up to
687 eight months in the hopes that larger fish will exhibit better survival to spawning
688 (UWBTWG 2010; Soorae 2011). A reintroduction to the Clackamas River has been
689 ongoing since 2011 with five years of releases consisting of more than 1700 juveniles,
690 250 subadults, and 75 adults being translocated from the Metolius River Basin, OR.
691 Monitoring has identified mature adults attempting to spawn, however natural production
692 has yet to be documented (Barrows et al. 2015). The implementation plan for the
693 reintroduction suggests that in 2017 the monitoring data will be reviewed and a change in
694 life stages or release locations to more favorable scenarios may occur (USFWS and
695 ODFW 2011). These “ad-hoc” approaches to adaptive management are better than trial
696 and error but lack predictive models that are a key component of formal adaptive
697 management which is a special case of structure decision making (Fischman and Ruhl
698 2015).

699 In 1995, the Director of the U.S. Fish and Wildlife Service approved a plan to use
700 structured decision making with adaptive resource management to guide hunting
701 regulations of the 6 to 11 million mallard *Anas platyrhynchos* ducks breeding in North
702 America (Nichols et al. 2015). Similarities between the decision context for mallard

703 harvest and bull trout reintroduction include species that inhabit multiple federal, state,
704 and agency jurisdictions, are of interest to user groups and conservationists, and actively
705 migrate between areas of overlapping jurisdiction. Anytime management decisions
706 include multiple stakeholders there can, and most often will be, disagreement about what
707 is the optimal decision. Disagreements are typically a function of institutional inertia, the
708 belief in process complexity, competing value systems, and hidden objectives (Conroy
709 and Peterson 2013; Nichols et al. 2015). Over the last 20 years of using adaptive
710 resource management to set hunting regulations mallard populations have fluctuated near
711 the desired abundance levels, optimal harvest opportunities have been provided, the
712 model has been updated and refined with new information. Maybe most important, there
713 has been a reduction in the contention between stakeholders because they can see their
714 belief in system dynamics incorporated in the decision process each year (Nichols et al.
715 2015); all characteristics of adaptive management that could benefit future bull trout
716 reintroduction efforts.

717 The U.S. Fish and Wildlife Service adopted the process of adaptive resource
718 management to guide effective conservation actions into the future (i.e., Strategic Habitat
719 Conservation; USFWS 2008) recognizing the success of the mallard harvest program and
720 that natural resource decisions will become more complex as human populations grow
721 and continue to impact animals and their habitats. Effective and efficient conservation
722 will be promoted with a clear connection between management actions and goals, and a
723 combination of institutional memory and ongoing research. The bull trout recovery plan
724 (USFWS 2015) represents the current understanding of abundance and distribution of the
725 species and outlines the management goals and recovery actions to delist the species.

726 Linking recovery plan objectives to management actions with predictive models, similar
727 to the one presented here, will build the foundation for formal adaptive management and
728 improved effectiveness, efficiency, and transparency of management decisions. Ongoing
729 bull trout reintroductions and research will reduce uncertainty and new information can
730 be incorporated into this decision model to guide future reintroduction decisions and
731 maximize the benefit from limited resources available for bull trout recovery.

732 **ACKNOWLEDGMENTS**

733 We would like to thank Dr. Robert Mason and five anonymous reviewers for
734 thoughtful reviews of the manuscript. References to trade names do not imply
735 endorsement by the U.S. Government. The findings and conclusions in this manuscript
736 are those of the author and do not necessarily represent the views of the U.S. Fish and
737 Wildlife Service.

738 **REFERENCES**

- 739 Al-Chokhachy, R., Budy, P., and Schaller, H. 2005. Understanding the significance of
740 redd counts: a comparison between two methods for estimating the abundance of
741 and monitoring bull trout populations. *North American Journal of Fisheries*
742 *Management* 25:1505-1512.
- 743 Anderson, D.R. 1975. Optimal exploitation strategies for an animal population in a
744 Markovian environment: a theory and an example. *Ecology* 56:1281-1297.
- 745 Barrows, M., Koch, R.C., Johnson, J., Koski M.L., and Bailey, E. 2016. Clackamas River
746 bull trout reintroduction project, 2015 annual report. U.S. Fish and Wildlife
747 Service, Columbia River Fisheries Program Office, Vancouver, WA and Oregon
748 Department of Fish and Wildlife, Corvallis, OR, USA.
- 749 Bellman, R. 1957. *Dynamic programming*. Princeton University Press, Princeton, NJ,
750 USA.
- 751 Berejikian, B.A., Tezak, E.P., Riley, S.C., and LaRae, A.L. 2001. Competitive ability and
752 social behaviour of juvenile steelhead reared in enriched and conventional
753 hatchery tanks and a stream environment. *Journal of Fish Biology* 59:1600-1613.
- 754 Berejikian, B.A., Gable, J., and Vidergar D. 2011. Effectiveness and trade-offs associated
755 with hydraulic egg collections from natural salmon and steelhead redds for
756 conservation hatchery programs. *Transactions of the American Fisheries Society*
757 140:549-556.
- 758 Brown, C., and Day, R.L. 2002. The future of stock enhancements: lessons for hatchery
759 practice from conservation biology. *Fish and Fisheries* 3:79-94.

- 760 Buchanan, D.V., and Gregory, S.V. 1997. Development of water temperature standards
761 to protect and restore habitat for bull trout and other cold water species in Oregon.
762 Pages 119-126 in MacKay, W.C., Brewin, M.K., and Motina M., editors. Friends
763 of the Bull Trout Conference Proceedings. Bull Trout Task Force, Trout
764 Unlimited, Calgary, AB, Canada.
- 765 Buchanan, D.V., Hanson, M.L., and Hooton R.M. 1997. Status of Oregon's Bull Trout:
766 distribution, life history, limiting factors, management considerations, and status.
767 Technical Report to Bonneville Power Administration, Project 199505400,
768 Portland, OR, USA.
- 769 Caswell, H. 2001. Matrix population models, 2nd edition. Sinauer Associates,
770 Sunderland, MA, USA.
- 771 Chadès, I., Chapron, G., Cros, M.J., Garcia, F., and Sabbadin, R. 2014. MDPtoolbox: a
772 multi-platform toolbox to solve stochastic dynamic programming problems.
773 *Ecography* 37:916-920.
- 774 Clemen, R.T. 1996. Making hard decisions: an introduction to decision analysis. 2nd
775 edition. Duxbury Press, Belmont, CA, USA.
- 776 Conroy, M.J., and Peterson, J.T. 2013. Decision making in natural resource management:
777 a structured, adaptive approach. Wiley-Blackwell, New York, NY, USA.
- 778 Dickens, M.J., Delehanty, D.J., and Romero, L.M. 2010. Stress: an inevitable component
779 of animal translocation. *Biological Conservation*. 143:1329-1341.

- 780 Dunham, J., K. Gallo, D. Shively, C. Allen, and B. Goehring. 2011. Assessing the
781 feasibility of native fish reintroductions: a framework applied to threatened bull
782 trout. *North American Journal of Fisheries Management* 31:106-115.
- 783 Dunham, J.B., and Rieman, B.E. 1999. Metapopulation structure of bull trout: influences
784 of physical, biotic, and geometrical landscape characteristics. *Ecological*
785 *Applications* 9:642-655.
- 786 Fischman, R.L., and Ruhl, J.B. 2015. Judging adaptive management practices of US
787 agencies. *Conservation Biology* 30:268-275.
- 788 Flagg, T.A., and Nash, C.F. (editors). 1999. A conceptual framework for conservation
789 hatchery strategies for Pacific salmonids. U.S. Department of Commerce,
790 National Oceanic and Atmospheric Administration technical memorandum
791 NMFS-NWFSC-38. National Marine Fisheries Service, Northwest Fisheries
792 Science Center, Seattle, WA, USA.
- 793 Fraley, J.J., and Shepard, B.B. 1989. Life history, ecology and population status of
794 migratory bull trout (*Salvelinus confluentus*) in the Flathead Lake river system,
795 Montana. *Northwest Science* 63:133- 143.
- 796 Fraser, D.J. 2008. How well can captive breeding programs conserve biodiversity? A
797 review of salmonids. *Evolutionary Applications* 1:535-586.
- 798 Fredenberg, W. 1998. Experimental bull trout hatchery. Progress report two:
799 experimental broodstock development, 1995–1997. U.S. Fish and Wildlife
800 Service, Creston Fish and Wildlife Center, Kalispell, MT, USA.

- 801 Fredenberg, W., Dwyer, P., and Barrows, R. 1995. Experimental bull trout hatchery.
802 Progress report, 1993–1994. U.S. Fish and Wildlife Service, Creston Fish and
803 Wildlife Center, Kalispell, MT, USA.
- 804 Gerber, B.D. 2015. Sandhill crane population monitoring, modeling, and harvest decision
805 making. Dissertation, Colorado State University, Fort Collins, CO, USA.
- 806 Gregory, R., and Long, G. 2009. Using structured decision making to help implement a
807 precautionary approach to endangered species management. *Risk Analysis* 29:
808 518-532.
- 809 Hard, J.J., Jones, R.P., Delarm, M.R., and Waples, R.S. 1992. Pacific salmon and
810 artificial propagation under the Endangered Species Act. U.S. Department of
811 Commerce. National Oceanic and Atmospheric Administration, National Marine
812 Fisheries Service, Northwest Fisheries Science Center, Seattle, WA, USA.
- 813 IPCC (Intergovernmental Panel on Climate Change). 2007. Climate Change 2007: The
814 physical science basis. Contribution of Working Group I to the Fourth
815 Assessment Report of the Intergovernmental Panel on Climate Change. Solomon,
816 S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B, Tignor, M., and
817 Miller, H.L., eds. Cambridge University Press, Cambridge, United Kingdom and
818 New York, NY, USA.
- 819 Isaak, D.J., Luce, C., Rieman, B.E., Nagel, D., Peterson, E., Horan, D., Parkes, S., and
820 Chandler, G. 2010. Effects of climate change and wildfires on stream
821 temperatures and salmonid thermal habitats across a mountain river network.
822 *Ecological Applications* 20:1350-1371.

- 823 IUCN/SSC (International Union for Conservation of Nature/Species Survival
824 Commission. 2013. Guidelines for reintroductions and other conservation
825 translocations. Gland, Switzerland.
- 826 Johnston, F.D. 2005. Demographic and life-history responses of an over-exploited bull
827 trout (*Salvelinus confluentus*) population to zero harvest regulations. Thesis.
828 University of Calgary, Calgary, AB, Canada.
- 829 Lee, D.C., and Rieman, B.E. 1997. Population viability assessment of salmonids by using
830 probabilistic networks. *North American Journal of Fisheries Management*
831 17:1144-1157.
- 832 Lubow, B.C. 1996. Optimal translocation strategies for enhancing stochastic
833 metapopulation viability. *Ecological Applications* 6:1268-1280.
- 834 Maguire, L.A. 1988. Black footed ferret recovery in Montana: a decision analysis.
835 *Wildlife Society Bulletin* 16:111-120.
- 836 McCarthy, M.A., Armstrong, D.P., and Runge, M.C. 2012. Adaptive management of
837 reintroduction. Pages 256-289 in Ewen, J.G., Armstrong, D.P., Parker, K.A., and
838 Seddon, P.J., eds. *Reintroduction biology: integrating science and management*.
839 Wiley-Blackwell.
- 840 McNeil, W.J. 1964. A method of measuring mortality of pink salmon eggs and larvae.
841 *Fishery Bulletin*. 63:575-588.

- 842 McPhail, J.D., and Murray, C.B. 1979. The early life history and ecology of Dolly
843 Varden (*Salvelinus malma*) in the upper Arrow lakes. Department of Zoology and
844 Institute of Animal Resources. University of British Columbia, Vancouver,
845 Canada.
- 846 Moore, C.T., and Conroy, M.J. 2006. Optimal regeneration planning for old-growth
847 forest: addressing scientific uncertainty in endangered species recovery through
848 adaptive management. *Forest Science* 52:155–172.
- 849 Morris, W.F., and Doak, D.F. 2002. Quantitative conservation biology. Sinauer,
850 Sunderland, MA, USA.
- 851 Mote, P.W., and Salathé, E.P. 2010. Future climate in the Pacific Northwest. *Climatic*
852 *Change* 102:29-50.
- 853 Muhlfeld, C.C., Taper, M.L., Staples, D.F., and Shepard, B.B. 2006. Observer error
854 structure in bull trout redd counts in Montana streams: implications for inference
855 on true redd numbers. *Transactions of the American Fisheries Society* 135:643-
856 654.
- 857 Naish, K.A., Taylor, J.E., Levin P.S., Quinn T.P., Winton, J.R., Huppert, J., and Hilborn,
858 R. 2008. An evaluation of the effects of conservation and fishery enhancement
859 hatcheries on wild populations of salmon. *Advances in Marine Biology* 53:61–
860 194.
- 861 Nichols, J.D., and Armstrong, D.P. 2012. Monitoring for reintroductions. Pages 223-255
862 *in* Ewen, J.G., Armstrong, D.P., Parker, K.A., and Seddon, P.J., eds.
863 *Reintroduction biology: integrating science and management*. Wiley-Blackwell.

- 864 Nichols, J.D., Johnson, F.A., Williams, B.K., and Boomer, G.S. 2015. On formally
865 integrating science and policy: walking the walk. *Journal of Applied Ecology*
866 52:539-543.
- 867 Rieman, B.E., and McIntyre, J.D. 1995. Occurrence of bull trout in naturally fragmented
868 habitat patches of varied size. *Transactions of the American Fisheries Society*
869 124:285-296.
- 870 Rieman, B.E., Lee, D.C., and Thurow, R.F. 1997. Distribution, status, and likely, future
871 trends of bull trout within the Columbia River and Klamath River Basins. *North*
872 *American Journal of Fisheries Management* 17:1111-1125.
- 873 Rieman, B.E., Isaak, D., Adams, S., Horan, D., Nagel, D., Luce, C., and Meyers, D.
874 2007. Anticipated climate warming effects on bull trout habitats and populations
875 across the interior Columbia River Basin. *Transaction of the American Fisheries*
876 *Society* 136:1552-1565.
- 877 Rieman, B.E., and Allendorf, F.W. 2001. Effective population size and genetic
878 conservation criteria for bull trout. *North American Journal of Fisheries*
879 *Management* 21:756-764.
- 880 Rieman, B.E. and McIntyre, J.D. 1993. Demographic and habitat requirements for
881 conservation of bull trout. U.S. Forest Service, Intermountain Research Station
882 General Technical Report INT-302, Ogden, Utah, USA.
- 883 Rout, T.M., Hauser, C.E., and Possingham, H.P. 2009. Optimal adaptive management for
884 the translocation of a threatened species. *Ecological Applications* 19:515-526.

- 885 Schaller, H.A., Budy, P., Newlon, C., Haeseker, S.L., Harris, J.E., Barrows, M., Gallion,
886 D., Koch, R.C., Bowerman, T., Conner, M., Al-Chokhachy, R., Skalicky, J., and
887 Anglin, D. 2014. Walla Walla River bull trout ten year retrospective analysis and
888 implications for recovery planning. U.S. Fish and Wildlife Service, Columbia
889 River Fisheries Program Office, Vancouver, WA, USA.
- 890 Shively, D., Allen, C., Alsbury, T., Bergamini, B., Goehring, B., Horning, T., and
891 Strobel, B. 2007. Clackamas River Bull Trout Reintroduction Feasibility
892 Assessment. Published by USDA Forest Service, Mt. Hood National Forest; U.S.
893 Fish and Wildlife Service, Oregon State Office; and Oregon Department of Fish
894 and Wildlife, North Willamette Region, Oregon, USA.
- 895 Soorae, P.S. 2011. Global reintroduction perspectives: 2011. More case studies from
896 around the globe. Gland, Switzerland: IUCN/SSC Re-introduction Specialist
897 Group and Abu Dhabi, UAE: Environment Agency-Abu Dhabi.
- 898 Stubben, C., and Milligan, B.G. 2007. Estimating and analyzing demographic models
899 using the popbio package in R. *Journal of Statistical Software* 22:1-23.
- 900 Taylor, E.B. 1986. Differences in morphology between wild and hatchery populations of
901 juvenile Coho salmon. *The Progressive Fish Culturist*. 48:171-176.
- 902 USFWS (U.S. Fish and Wildlife Service). 2008. Strategic habitat conservation handbook:
903 A guide to implementing the technical elements of strategic habitat conservation.
904 Washington, D.C., USA.

- 905 USFWS (U.S. Fish and Wildlife Service). 2015. Recovery plan for the coterminous
906 United States population of bull trout (*Salvelinus confluentus*). Portland, Oregon,
907 USA.
- 908 USFWS and ODFW (U.S. Fish and Wildlife Service and Oregon Department of Fish and
909 Wildlife). 2011. Clackamas River bull trout reintroduction implementation,
910 monitoring, and evaluation plan. Portland, Oregon, USA.
- 911 UWBTWG (Upper Willamette Bull Trout Working Group). 2010. Upper Willamette
912 Basin Bull Trout Action Plan 2010. Oregon, USA.
- 913 VanderWerf, E.A., Groombridge, J.J., Fretz, J.S., and Swinnerton, K.J. 2006. Decision
914 analysis to guide recovery of the po‘ouli, a critically endangered Hawaiian
915 honeycreeper. *Biological conservation* 129:383-392.
- 916 Wakamiya, S.M., and Roy, C.L. 2009. Use of monitoring data and population viability
917 analysis to inform reintroduction decisions: Peregrine falcons in the Midwestern
918 United States. *Biological Conservation* 142:1767-1776.
- 919 Walters, C.J. 1975. Optimal harvest strategies for salmon in relation to environmental
920 variability and uncertain production parameters. *Journal of the Fisheries Research*
921 *Board of Canada* 32:1777-1784.
- 922 Walters, C.J., and Hilborn, R. 1976. Adaptive control of fishing systems. *Journal of the*
923 *Fisheries Research Board of Canada* 33:145-159.
- 924 Walters, C.J., Goruk, R.D., and Radford, D. 1993. Rivers Inlet sockeye salmon: an
925 experiment in adaptive management. *North American Journal of Fisheries*
926 *Management* 13:253–262.

- 927 Wenger, S.J., Som, N.A., Dauwalter, D.C., Isaak, D.J., Neville, H.M., Luce, C.H.,
928 Dunham, J.B., Young, M.K., Fausch, K.D. and Rieman, B.E. 2013. Probabilistic
929 accounting of uncertainty in forecasts of species distributions under climate
930 change. *Global Change Biology*, 19(11), pp.3343-3354.
- 931 Williams, B.K., Szaro, R.C., and Shapiro, C.D. 2009. Adaptive management: The U.S.
932 Department of the Interior technical guide. Adaptive Management Working
933 Group, U.S. Department of the Interior, Washington, DC, USA.
- 934 Wilson, P.H. 2003. Using population projection matrices to evaluate recovery strategies
935 for Snake River spring and summer Chinook salmon. *Conservation Biology*
936 17:782-794.

Draft

937 Table 1. Matrix of state dependent penalties applied to the decision utility for reducing
938 the adult abundance of the donor population below a desired state. If the donor
939 population was reduced to an undesirable state the value of the decision was 0. If the
940 donor population was not reduced to an undesirable state the value of the decision was
941 the number of adults in the recipient population, up to a maximum of 300.

Ending abundance	Starting abundance				
	0 - 50	50 - 100	100 - 250	250 - 500	> 500
0 - 50	0	0	0	0	0
50 - 100	1	1	0	0	0
100 - 250	1	1	1	0	0
250 - 500	1	1	1	1	1
> 500	1	1	1	1	1

942 Table 2. Six alternative bull trout reintroduction decisions evaluated in the decision model. A decision consisted of two components;
 943 a reintroduction strategy and the number of individuals by lifestage to remove from a donor population.

Alternative	Strategy	Embryo	Juvenile	Subadults	Small adults	Large adults
1	Do nothing	0	0	0	0	0
2	Translocation	0	1000	0	0	0
3	Translocation	0	0	0	30	30
4	Translocation	0	1000	60	20	20
5	Captive-rearing	20,000	0	0	0	0
6	Artificial production	0	0	0	30	30

944 Table 3. Lower-level vital rates of populations used in Leslie projection matrices and in
945 sensitivity analyses for the bull trout reintroduction decision model. The mean value was
946 varied by a coefficient of variation (CV) to determine the minimum (min) and maximum
947 (max) range of that value for sensitivity analyses.

Population	Parameter	Value	CV	Min	Max
All	m_4	1250	0.5	427	2423
	m_5	2500	0.5	854	4846
	B_4	0.45	0.3	0.23	0.68
	B_5	0.9	0.1	0.72	0.99
	$sexr$	0.5	0.3	0.25	0.75
Donor (d)	$juvi.CC$	15,000	0.5	5124	29,076
	$egg.max.S$	0.3	0.5	0.08	0.57
	G_2	0.12	0.5	0.04	0.23
	G_3	0.15	0.5	0.05	0.29
	G_4	0.15	0.5	0.05	0.33
	P_3	0.05	0.5	0.02	0.10
	P_4	0.1	0.5	0.03	0.19
	P_5	0.35	0.5	0.09	0.66
Captive (c)	G_1	0.4	0.5	0.10	0.75
	G_2	0.4	0.5	0.10	0.75
	G_3	0.2	0.5	0.06	0.39
	G_4	0.15	0.5	0.05	0.29
	P_3	0.2	0.5	0.06	0.39
	P_4	0.5	0.5	0.10	0.90
	P_5	0.75	0.5	0.00	1.00
	R_1	0.1	0.5	0.03	0.19
	$naive$	0.5	0.7	0.01	0.99
Recipient (r)	$juvi.CC$	15,000	0.5	5124	29,076
	$egg.max.s$	0.3	0.5	0.08	0.57
	G_2	0.12	0.5	0.05	0.29
	G_3	0.15	0.5	0.06	0.39
	G_4	0.15	0.5	0.06	0.39
	P_3	0.05	0.5	0.03	0.19
	P_4	0.1	0.5	0.05	0.29
	P_5	0.35	0.5	0.10	0.83
	$naive$	0.5	0.7	0.01	0.99
	$trans.reduc$	0.7	0.5	0.45	0.90
	$cap.reduc$	0.3	0.5	0.21	0.40
	$artprod.reduc$	0.2	0.5	0.14	0.27

948 m_i = number of eggs per female at stage i 950 $sexr$ = sex ratio

949 B_i = proportion of females spawning 951 $juvi.CC$ = juvenile carrying capacity

- 952 $egg.max.s$ = maximum egg survival
- 953 G_i = probability of surviving and growing from
- 954 stage i to $i + 1$
- 955 P_i = probability of surviving and persisting in
- 956 stage i
- 957 R_i = probability of growth from embryo to
- 958 subadult in captive population
- 959 $naive$ = reduction in survival during the first year
- 960 in captivity or after release
- 961 $trans.reduc$ = reduction in survival of
- 962 translocated fish
- 963 $cap.reduc$ = reduction in survival of captive-
- 964 reared fish
- 965 $artprod.reduc$ = reduction in survival of
- 966 artificially produced fish

Draft

967 Table 4. Abundances by life stage used to create every possible combination of donor
968 and recipient populations that may be observed at the beginning of a decision timestep in
969 real-world reintroduction scenarios. This resulted in 576 potential combinations of donor
970 and recipient populations. These numbers were calculated with large adult abundance
971 ranging from 0 to 307 individuals. The stable age distribution from the donor transition
972 matrix was used to determine the abundance of all younger lifestages, with 1499 adults in
973 the largest population.

Embryo	Juvenile	Subadult	Small adult	Large adult
0	0	0	0	0
3409	449	118	19	5
6136	809	212	35	9
6817	899	236	39	10
9544	1258	330	54	14
12,271	1618	424	70	18
13,635	1797	471	78	20
19,771	2606	683	113	29
25,224	3325	872	144	37
27,951	3684	966	159	41
33,405	4403	1154	190	49
38,859	5122	1343	221	57
41,586	5482	1437	237	61
54,539	7189	1884	311	80
66,811	8807	2308	380	98
69,538	9166	2403	396	102
87,263	11,502	3015	497	128
104,988	13,839	3627	598	154
111,806	14,737	3863	637	164
118,623	15,636	4099	676	174
125,440	16,535	4334	714	184
139,757	18,422	4829	796	205
174,526	23,005	6030	994	256
209,295	27,588	7231	1192	307

974 Table 5. State dependent policy identifying the optimal bull trout reintroduction decision as a function of adult abundance
 975 (i.e., the summation of large adult and small adult abundances) in donor and recipient populations. Alternative one is to do
 976 nothing, alternative two is to translocate 1000 juveniles, alternative three is to translocate 60 adults, alternative four is to
 977 translocate 1000 juveniles and 40 adults, and alternatives five and six are not represented. This state dependent policy
 978 represents the optimal decision calculated from the mean value of lower-level vital rates.

Donor abundance	Recipient abundance							
	< 50	50 - 100	100 - 200	200 - 300	300 - 500	500 - 800	800 - 1000	> 1000
< 50	1	1	1	1	1	1	1	1
50 - 100	1	1	1	1	1	1	1	1
100 - 200	2	1	1	1	1	1	1	1
200 - 300	2	1	1	1	1	1	1	1
300 - 500	4	2	2	2	1	1	1	1
500 - 800	3	3	3	3	4	1	1	1
800 - 1000	3	3	3	3	3	1	1	1
> 1000	3	3	3	3	3	1	1	1

Table 6. Quasi-extinction probability of donor and recipient populations under seven simulated management alternatives. Six alternatives consisted of simulating each reintroduction decision in the decision set for one timestep then forward simulating to estimate the donor and recipient population abundances after 50 years (i.e., one-and-done alternatives). The final alternative consisted of using the state dependent policy (SDP) to determine recurring reintroduction decisions for a 50 year time period. Quasi-extinction probabilities are reported for four starting adult abundance states of the donor population; 100, 150, 300, and 500 adults.

Population	Decision	Starting donor adult abundance			
		100	150	300	500
Donor	1	0	0	0	0
	2	0	0	0	0
	3	1	1	0	0
	4	1	1	0	0
	5	1	1	0	0
	6	1	1	0	0
	SDP	0.02	0.05	0	0
Recipient	1	1	1	1	1
	2	1	1	1	1
	3	1	1	0.01	0.04
	4	1	1	0.09	0.17
	5	1	1	1	1
	6	1	1	0.12	0.11
	SDP	1	1	1	0.08

Table 7. Estimates of sensitivity and elasticity of lower-level vital rates used to populate the Leslie matrices representing the donor, captive, and recipient populations in the bull trout reintroduction decision model. Parameters that were not included in a matrix model for a particular population are represented with a dash.

Parameter ^a	Sensitivity			Elasticity		
	Donor	Captive	Recipient	Donor	Captive	Recipient
<i>G1</i>	--	--	0.506	--	--	0.120
<i>G2</i>	1.649	1.649	0.506	0.199	0.199	0.120
<i>G3</i>	1.319	1.319	2.085	0.199	0.199	0.246
<i>G4</i>	1.090	1.090	2.096	0.164	0.164	0.186
<i>P3</i>	0.209	0.209	0.254	0.010	0.010	0.007
<i>P4</i>	0.565	0.565	1.288	0.057	0.057	0.076
<i>P5</i>	0.492	0.492	0.577	0.173	0.173	0.119
<i>R1</i>	--	--	2.145	--	--	0.127
<i>m4</i>	0	0	0	0.086	0.086	0.151
<i>m5</i>	0	0	0	0.112	0.112	0.095
<i>B4</i>	0.191	0.191	0.570	0.086	0.086	0.151
<i>B5</i>	0.124	0.124	0.178	0.112	0.112	0.095
<i>sexr</i>	0.396	0.396	0.834	0.199	0.199	0.246
<i>egg.max.s</i>	0.578	0.578	--	0.174	0.174	--
<i>juvi.CC</i>	0	0	--	0.024	0.024	--
<i>N.juvi</i>	0	0	--	-0.024	-0.024	--
<i>naive</i>	--	0	0	--	0	0
<i>strat.reduc</i>	--	0	--	--	0	--

^a G_i = probability of surviving and growing from stage i to $i + 1$

P_i = probability of surviving and persisting in stage i to $i + 1$

R_i = probability of rapid growth from embryo to subadult stage in a captive population

m_i = number of eggs per female at stage i

B_i = proportion of females spawning

sexr = sex ratio

egg.max.s = maximum egg survival

juvi.CC = juvenile carrying capacity

N.juvi = number of juveniles in the population

naive = reduction in survival during the first year in captivity or after release

strat.reduc = reduction in survival that represent the reintroduction strategy

1000

1001 Figure 1. Tornado diagram from one-way sensitivity analysis with decision model
1002 parameters listed from most influential (top) to least influential (bottom) for a bull trout
1003 reintroduction decision. Donor (d), captive (c), and recipient (r) population specific
1004 parameters are represented by a prefix and parameter definitions are listed in the text and
1005 Table 2. The x-axis represents the number of adults in the recipient population after one
1006 timestep.

1007

1008 Figure 2. Response profiles of adult abundance in the recipient population to varying
 1009 levels of (a) the effect of translocation on post-release survival (*trans.reduc*), (b) the
 1010 effect of captive-rearing on post-release survival (*cap.reduc*), (c) the effect of artificial
 1011 production on post-release survival (*artprod.reduc*), (d) the effect of naivety to survival
 1012 in the captive population (*c.naive*), (e) the adult abundance of the donor population at the

1013 beginning of the decision timestep, and (f) the effect of naivety to survival in the recipient
1014 population (*r.naive*).

Draft

1015

1016 Figure 3. Contour plots of recipient adult abundance (a,c,e) and response profiles of the
 1017 identity of the optimal decision (b,d,f) at varying levels of the parameters in the density
 1018 dependent equation for the donor population (a,b; $d.egg.max.s$, $d.juvi.CC$), the effect of
 1019 captivity and translocation on post-release survival (c,d; $cap.reduc$, $trans.reduc$), and
 1020 large adult survival in the recipient population ($r.P_5$) and donor adult abundance at the
 1021 first decision timestep (e,f).

1022

1023 Figure 4. Population projections of the mean donor and recipient populations under
1024 seven simulated management alternatives. Six alternatives consisted of simulating each
1025 reintroduction decision in the set for one decision timestep then forward simulating to
1026 estimate the donor and recipient population abundances after 50 years (i.e., one-and-done
1027 alternatives). The final alternative consisted of using the state dependent policy (SDP) to
1028 determine recurring reintroduction decisions for a 50 year time period. Projections are

1029 presented for four starting adult abundance states of the donor population, from top to
1030 bottom, 100, 150, 300, and 500 adults.

Draft

1031
1032 Figure 5. Population projections of the mean donor and recipient populations under
1033 seven simulated management alternatives and with environmental stochasticity. Six
1034 alternatives consisted of simulating each reintroduction decision in the set for one
1035 decision timestep then forward simulating to estimate the donor and recipient population
1036 abundances after 50 years (i.e., one-and-done alternatives). The final alternative
1037 consisted of using the state dependent policy (SDP) to determine recurring reintroduction
1038 decisions for a 50 year time period. Projections are presented for four starting adult

1039 abundance states of the donor population, from top to bottom, 100, 150, 300, and 500
1040 adults. The effect of environmental stochasticity was included with a 0.05 probability of
1041 a disturbance occurring in any year. If a disturbance did occur the population abundance
1042 of each life stage was reduced by 25%.

Draft