1 **Managing recreational fisheries from a multi-species perspective: Leveraging species**

2 **interactions and accounting for positive feedback loops to maintain desired ecosystem**

3 **states**

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16 **Author contributions**

17 All authors have approved the submitted manuscript and agreed to be listed as such. C.D. and

18 C.L.N. formulated the theoretical framework. S.J., C.T.S., and G.S. helped C.D. and C.L.N. to

19 develop the model and the manuscript.

20 **Keywords:** Regime Shift, Alternative Stable States, Recreational Fisheries, Safe-Operating

21 Space

22 **Abstract**

23 Historically, recreational fisheries have been managed through a single-species framework where

24 fish species are considered in isolation. This single-species framework can lead to unintended

25 consequences for fisheries, potentially resulting in fishery collapses or regime shift. A common

26 factor leading to regime shift and preventing population recovery are interspecific interactions

27 between the collapsed species and their competitors. Increasingly, ecosystem-based management

28 has been advocated in recreational fisheries where users have a diverse set of goals, however, in

29 practice, ecosystem-based management can be difficult in systems subject to non-linearity. We

30 use a modeled recreational fishery to describe how interactions between two harvested species

31 can drive shifts in the stable state of the system, and seek to understand how interactions can be

32 leveraged for efficient management. These experiments explore 1) the effectiveness of single-

33 species management actions at maintaining the desired stable state as compared to multi-species

34 management 2) the diversity of decision making paths that can lead to positive outcomes when

35 leveraging certain interspecific interactions, and 3) how interspecific interactions can be

36 leveraged to maintain a system in a safe-operating-space despite stable state drivers outside of a

37 managers control. This model demonstrates how interspecific interactions within a system can

38 lead to non-linear outcomes, and when these interactions are unaccounted for, result in regime

39 shifts. Accounting for interspecific action allows decision makers to meet their goals through a

40 diverse and cost-effective combination of direct (i.e., managing the focal species through

41 stocking and harvest limitation) and indirect (i.e., managing the competitor) means.

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Introduction

45   Interspecific interactions are often not heavily considered in natural resource

46   management in order to simplify complex management problems (Walters and Martell 2004;

47   Baum and Worm 2009). This simplification may lead to decisions that in hindsight prove to be

48   ineffective or even detrimental (Walters et al. 2000; Myers et al. 2007; Hansen et al. 2015; Sass

49   and Shaw 2020). For example, Atlantic Cod (*Gadus morhua*) stocks in the Baltic Sea collapsed

50   in the late 1980's and the population failed to recover during the 20 years after collapse despite

51   strong restrictions on harvest, suggesting a shift in stable state (ICES 2006). Subsequent research

52   identified ecological interactions with Sprat (*Sprattus sprattus*) as the factor limiting cod

53   recovery and called for a more holistic plan that accounts for these interspecific interactions in

54   the cod recovery strategy (Casini et al. 2009). These unexpected outcomes have occurred at least

55   in part because earlier management decisions were unable to account for interactions between

56   multiple species and life stages (Walters et al. 2000; Jackson et al. 2001; Hansen et al. 2017;

57   Embke et al. 2019; Hutchings 2000). Although challenging, ecosystem-based management (i.e.,

58   holistic approaches to manage natural resources that includes accounting for interspecific

59   interactions and human decision making) can help decision makers avoid unexpected, and often

60   undesirable, outcomes (Pine et al. 2009).

61   While in many instances, ecosystem-based management may result in unexpected

62   outcomes with a wide variety of potentially negative effects, here, we focus on unexpected

63   regime shifts that push the system to an undesirable alternative stable state. Regime shifts have

64   been well documented in aquatic systems and are often exceedingly difficult to reverse once they

65   have occurred (Carpenter and Kinne 2003). Regime shifts represent an abrupt change in

66   ecosystem configuration that can be self-reinforcing (Carpenter and Kinne 2003). Complex intra-

67   and interspecific interactions in aquatic systems can result in positive feedback loops that allow a

68   stable state to reinforce itself such that efforts by decision makers to change the stable state may

69   not have their intended effects. Walters and Kitchell (2001) described how positive feedback

70   loops due to cultivation effects could create two alternative stable states in a "trophic triangle"

71   food web consisting of adult and juvenile stages of a top predator and a forage species. Further,

72   size- and resource-dependent individual growth can result in depensatory population growth, also

73   known as an 'Allee Effect' (De Roos and Persson 2002). Under low exploitation, in this model,

74   the top predator is abundant and able to cultivate conditions to increase survival of its juveniles

75   by preying on the predators of its juveniles, namely the forage species. Alternatively, the forage

76   species may dominate when exploitation of the top predator is high (as is the case in many

77   fisheries), allowing the forage species to cultivate conditions for itself through predation on

78   juveniles of the top predator. Depensatory growth effects on the predator population further limit

79   its ability to rebound and become abundant again.

80   In addition to interactions with non-targeted species as in the simple trophic triangle

81   models described above, exploited populations are often embedded in a larger community in

82   which harvest of multiple species takes place (Hansen et al. 2015). The tradeoffs between

83   competing management goals for several co-occurring exploited species in recreational fisheries

84   are often not considered; however, some notable exceptions do exist in commercial fisheries (e,g,

85   Essington et al. 2015; Oken and Essington 2016). Essington et al. (2015) used competing

86   objectives for a predator fishery (Atlantic cod, *Gadus morhua*) and a forage species fishery

87   (Atlantic herring, *Clupea harengus*) and showed how ecological interactions between the two

88   and the market price of each species could be combined to determine the appropriate level of

89   mortality for each species given specific management goals (maximizing combined profit of both

90   species at equilibrium). Decision makers accounting for interspecific interactions could leverage

91   these dynamics to creatively achieve their goals and thereby reduce unexpected outcomes and

92   decrease the probability of catastrophic regime shifts.

93   Recreational fisheries are ideal study systems to examine how decision makers could take

94   advantage of interspecific interactions to solve complex problems through an ecosystem-based or

95   adaptive management approach. Inland recreational fisheries are specifically amenable to these

96   approaches because of their well-defined boundaries and widespread occurrence across the

97   landscape, which allows for replication and comparison (Walters 1986). An adaptive approach

98   using experimental management actions across many independent systems could allow decision

99   makers to generate new knowledge about how to creatively manage these systems. Of course

100   large-scale experimental and adaptive management approaches are not always feasible and other

101   approaches, such as model-based experiments, may be a useful first step toward this goal.

102   Management goals, in systems that have the potential to exhibit multiple stable states,

103   often aim to maintain fisheries in the 'desired' stable state or the state in which the species most

104   desired by anglers dominates the system. Although these desired stable states exist within a

105   broader safe operating space, certain management interventions could result in shifts into

106   'undesired' states – or states outside a safe operating space. In these systems certain drivers of

107   regime shifts may be outside a decision maker's control such as slow moving changes in climate.

108   Indeed, decision makers are limited in the ways in which they can influence recreational fisheries

109   (i.e., fishing regulations, stocking, effort limitation, habitat alteration, valuation) and fisheries

110   users have diverse goals. Given the complexity of recreational fisheries, understanding and

111   leveraging ecological interactions may allow decision makers to make the most of the limited

112 tools at their disposal to keep systems within a safe operating space despite slow moving changes

113 outside their control (e.g., changes to fecundity, loss of refuge, etc.), and to meet the diverse

114 goals of recreational fishery users (Carpenter et al. 2017).

115   Here, we use a model of a recreational fishery with two managed species to explore

116 whether decision makers can leverage interspecific interactions to achieve desired outcomes. The

117 hypothetical decision makers' motivation in our model experiments is to promote stable states in

118 which the desired species dominates, resulting in higher economic benefits and user satisfaction.

119 Our model, like all models, makes simplifying assumptions to balance tractability with realism.

120 We use a relatively simple fishery model that allows for the interaction and harvest of two

121 species. With this model, we compare and contrast outcomes of management actions under

122 scenarios when interspecific interactions are considered and when they are not. We hypothesize

123 that management activities taking species interactions into account will be more successful at

124 keeping a system in a 'desired' state. As such, we predict that consideration of these interspecific

125 interactions and the resulting non-linear dynamics will lead to more positive, predictable, and

126 desired outcomes. We use a series of modeling experiments to understand the influence of

127 simultaneous management of two species in a recreational fishery, the increased diversity of

128 management options when accounting for species interactions, and the effects of slow change on

129 the stable state of the system.

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131 **Methods**

132   Given the long time scales over which management decisions and fisheries dynamics

133 operate, alternative avenues of inference such as long-term observations, comparative surveys, or

134   experiments are often not feasible for addressing questions about alternate stable states in

135   fisheries (Carpenter 1998). Instead, we use a dynamic model to explore the implications and

136   opportunities of a systems-based approach to managing regime shifts in recreational fisheries.

137   Our theoretical framework allowed us to conduct a series of modeling experiments exploring the

138   benefits of considering interspecific interactions and state-maintaining positive feedbacks in

139   management, and potential opportunities afforded by an ecosystem-based management approach.

140   For example, we used our model to explore management outcomes for scenarios in which the

141   hypothetical decision maker either ignores or accounts for interspecific interactions.

142   ***Model***

143   For our modeling experiments, we adapted a stage-structured food web model previously

144   used to explore alternative stable states in lake ecosystems (Carpenter and Brock 2004; Roth et

145   al. 2007; Carpenter et al. 2008; Biggs et al. 2009). The original model used a trophic triangle

146   structure that included interactions among the juvenile and adult stages of a harvested sport fish

147   and a single-stage planktivorous fish not subject to harvest. We modified this model to include

148   two stage-structured fish populations that are both harvested. Our model contains basic foraging

149   arena dynamics in which juveniles move between the foraging arena and refuge (Walters and

150   Juanes 1993; Walters and Martell 2004; Ahrens et al. 2012). In our model predation on juveniles

151   occurs in the foraging arena where adults can prey on their own juveniles and on juveniles of the

152   competing species. The survival and fecundity of the two species are identical, while the

153   competition coefficients are not. Juveniles of both species have equal effects on each other, but

154   adults have asymmetrical effects on the juveniles of the opposite species (Table 1). Unless noted,

155   all parameters are constant through time.

156   *Adult Dynamics*

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$$\frac{dA_1}{dt} = s_1 J_1 - m_1 A_1 - q E_1 A_1 \qquad \text{Eq. 1}$$

158
$$\frac{dA_2}{dt} = s_2 J_2 - m_2 A_2 - q E_2 A_2 \qquad \text{Eq. 2}$$

159   Adults of Species 1 or Species 2 are produced through the maturation of juveniles at a

160   constant rate s_1 or s_2. Adults undergo natural mortality, m_1 and m_2, and are harvested at rates

161    $q E_1$ and $q E_2$. Harvest rate can be either constant or vary as a function of time.

162   *Juvenile Dynamics*

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$$\frac{dJ_1}{dt} = a_1 A_1 e^{-b_1 A_1} + k_1 - \frac{c_{J_1,A_1} v_1 J_1 A_1}{h_1 + v_1 + c_{J_1,A_1}} - \frac{c_{J_1,A_2} v_1 J_1 A_2}{h_1 + v_1 + c_{J_1,A_2} A_2} - c_{J_1,J_2} J_1 J_2 - s_1 J_1 \qquad \text{Eq. 3}$$

164
$$\frac{dJ_2}{dt} = a_2 A_2 e^{-b_2 A_2} + k_2 - \frac{c_{J_2,A_2} v_2 J_2 A_2}{h_2 + v_2 + c_{J_2,A_2}} - \frac{c_{J_2,A_1} v_2 J_2 A_1}{h_2 + v_2 + c_{J_2,A_1} A_1} - c_{J_2,J_1} J_2 J_1 - s_2 J_2 \qquad \text{Eq. 4}$$

165   Juveniles are produced through Beverton-Holt stock-recruitment relationships (Beverton

166   and Holt 1957). Additionally, stocking of juveniles can be imposed through k_1 and k_2. Juveniles

167   are removed from the population in one of three ways. The strength of each mortality source is

168   represented by the parameter c, which can be thought of in general terms as the 'effect' of one

169   species/life stage on another. First, juvenile mortality can occur through cannibalism, for

170   example $c_{J_1,A_1} J_1 A_1$ (read this as 'the effect of A_1 on J_1'), which is dependent on refuge dynamics.

171   Second, juveniles can be removed through predation by adults of the opposite species, $c_{J_1,A_1} J_1 A_2$

172   and $c_{J_1,A_1} J_2 A_1$. These dynamics are dependent on refuge and are controlled by two rates, h, the

173   rate at which juveniles leave refuge and enter the foraging arena and v, the rate at which they

174   leave the forage arena and enter refuge. Refuge availability is commonly assumed to decrease

175   predation risk (Walters and Juanes 1993; Ahrens et al. 2012; but see Ziegler et al. 2018). Last,

176   juvenile mortality is imposed through direct competition with juveniles of the opposite species,

177 such as might occur through competition for resources or direct predation. This competition

178 occurs independent of refuge dynamics; all juveniles compete in all areas. We assumed that

179 juveniles of both species occupy the same refuge and same foraging arena. Remaining juveniles

180 not claimed by the three sources of mortality mature to adults at the rate (sJ).

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196   Table 1. Model parameterization

Term	Definitions	Value/Range
s_1	survival, species 1 juvenile	0.1
m_1	natural mortality, species 1 adult	0.1
c_{J_1,A_1}	cannibalism, species 1	0.001
c_{J_1,A_2}	predation by species 2 on species 1	0.05
c_{J_1,J_2}	juvenile competition	0.003
v_1	rate at which species 1 juveniles enter foraging arena	1
h_1	rate at which species 1 juveniles leave foraging arena	8
k_1	stocking, species 1	0-4000
qE_1	harvest rate, species 1	0-8
s_2	survival, species 2 juvenile	0.1
m_2	natural mortality, species 2 adult	0.1
c_{J_2,A_2}	cannibalism, species 2	0.001
c_{J_2,A_1}	predation by species 1 on species 2	0.03
c_{J_2,J_1}	juvenile competition	0.003
v_2	rate at which species 2 juveniles enter foraging arena	1

h_2	rate at which species 2 juveniles leave foraging arena	8
k_2	stocking, species 2	0-4000
qE_2	harvest rate, species 2	0-8

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212 *Model Experiments*

213   Recreational fisheries are complex systems in which human influences and ecological

214 interactions feed back on each other to make applied management of any species challenging. A

215 key challenge facing many decision makers is how to maintain or improve abundances of highly

216 valued and exploited species in the face of competition with other less valued and exploited

217 species. Our modeling experiments were designed to mimic this situation. Species 1 in our

218 models represents the initially dominant, highly valued and highly exploited species that

219 managers are seeking to maintain, while species 2 represents a less valued and less exploited

220 species. When managing the focal species (species 1) alone, decision makers have control over

221 harvest limits and stocking rates for the focal species only; as opposed to controlling harvest and

222 stocking for both species respectively when managing from a whole-ecosystem perspective.

223 Given the hypothetical goals of decision makers in our simulations, species 2 management will

224 always take the form of setting harvest limits and not include stocking.

225   After exploring its basic dynamics and validating that the model produced alternate stable

226 states, we focused on three different modeling experiments that reflect scenarios commonly

227 encountered by fisheries decision makers (Table S1).

228   In our first experiment (Leveraging Interactions Experiment), we sought to understand

229 the implications of active management of only one species (species 1) *versus* both species

230 (species 1 and 2) on the "desired state" of the system. In this experiment, we sought to explore

231 the influence of leveraging ecological interactions to achieve decision maker's goals related to

232 retaining a desired stable state. In this experiment, while species 2 is not the focal species,

233 harvest of species 2 is likely to occur, however, in the first part of the experiment (species 1

234 active management), species 2 is managed for species 2 alone (no interaction considered). In the

235 second part, species 2 is managed explicitly with these ecological interactions in mind.

236   Our second modeling experiment focused on the diversity of management options

237 available to decision makers when accounting for interspecific interactions (Alternative

238 Approaches Experiment). Here, we sought to understand the different paths decision makers may

239 take to the same outcome through managing one or both species using available management

240 tools (i.e., stocking and harvest regulation).

241   Finally, we explored the influences of slow changes (i.e., those outside of managerial

242 control) in adult fecundity and the resultant effects on stable states (Safe Operating Space

243 Experiment). This experiment used a safe-operating space approach in which decision makers

244 use the tools at their disposal, including leveraging species interactions, to keep the system in a

245 desired stable state despite slow moving changes outside of managerial control (Carpenter et al.

246 2017). We explored a scenario in which slow changes to fecundity of species 1 may drive an

247 eventual shift in stable state from species 1 to species 2, mimicking recruitment declines that

248 have been observed in fisheries for a wide array of reasons (Walters and Martell 2004; Lynch et

249 al. 2016). Similar slow-moving changes may occur in other parameters such as juvenile refuge

250 availability (though in some cases managers may be able to control this variable) or angler

251 preferences. Species interaction strengths, mortality, survival, and habitat availability were held

252 constant across simulations (Table 1).

253   Our theoretical model was loosely based on interactions between walleye (*Sander*

254 *vitreus*) and largemouth bass (*Micropterus salmoides*) in North American north-temperate lakes.

255 However, we have focused on the conceptual interactions rather than quantitative resemblance to

256 that system in order to maximize generality of our model. As such, we only interpret the

257   qualitative dynamics of model simulations. For example, in order to define dominance and the

258   presence of a system in the desired or undesired state we chose a threshold of one population's

259   abundance exceeding that of its competitor by a minimum of 100 individuals. Model simulations

260   were performed in R using RStudio and the deSolve package (RStudio Team; Soetaert et al.

261   2010; R Core Team 2020).

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263   **Results**

264   *Model Validation*

265   Our model of a two-species recreational fishery exhibited alternative stable states. Initial

266   species abundances and species harvest rates determined stable state outcomes (Fig. 1). For

267   example, a moderate harvest rate on species 1 resulted in scenarios in which species 1 dominates

268   over species 2 or vice versa depending on initial system state. Equilibrium abundances of the two

269   species were influenced both by their initial abundances and by the harvest rate on species 1.

270   When species 1 was initially dominant, there was a much wider range of harvest rates over

271   which species 1 remained dominant. Harvesting in the model decreased the target species'

272   abundance and eventually led to a regime shift.

273   Alternative stable states persisted across all but the most extreme values for species

274   interaction strengths, mortality, survival, and fecundity (Fig. S1). However, the harvest rates at

275   which the system transitioned differed with variation in parameter values (Fig. S1). Increasing

276   juvenile survival to adulthood (s_1) for example, shifts the threshold higher indicating that the

277   range of harvests over which alternative stable states occur is increased (Fig. S1). The model was

278   most sensitive to changes in competition amongst juveniles, and to Beverton-Holt stock-

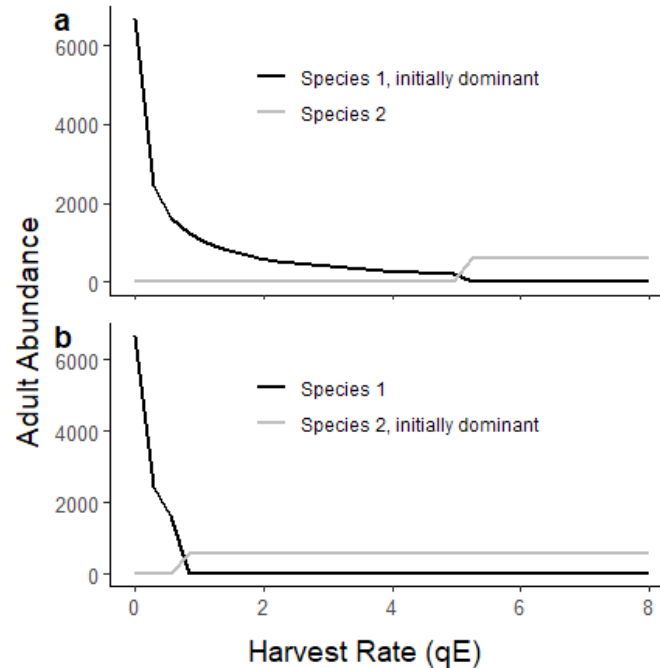279   recruitment parameters $(c_{J_1,J_2},\ c_{J_2,J_1}, a, b)$.

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281   *Fig. 1. Model exhibits alternative stable states. The model is run to equilibrium over a range of*

282   *harvest rates for species 1, while harvest of species 2 is held constant ($qE_2=2$). Panels show*

283   *equilibrium results when (a) species 1 is initially dominant or (a) species 2 is initially dominant.*

284   *The region of the x-axis between the thresholds in panels a and b represents the range of harvest*

285   *rates over which alternative outcomes can occur; over this range of harvest rates it is the initial*

286   *abundances, not the harvest rates, that determine equilibrium abundances.*

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288   *Modeling Experiment 1: Leveraging Interactions*

289   Managing both species simultaneously produced different outcomes than single species

290   management (Fig. 2). In the Leveraging Interactions Experiment, species 1 was the initially

291   dominant species. This dominance tended to be maintained for simulations where stocking rates

292   were sufficiently high and/or harvest levels were sufficiently low. The combination of stocking

293   and harvest rates for species one that represented a boundary condition for maintenance of

294   Species 1 dominance can be seen as the dashed isocline in Fig. 2. When the decision maker uses

295   species 2 harvest rate as an additional tool for maintaining the desired state, the range of species

296   1 harvest and stocking rates that produce the desired outcome is greatly expanded (solid isocline

297   in Fig. 2). More harvest can be tolerated for species 1 while requiring less stocking when harvest

298   is increased for species 2. Similar analyses were also conducted in a modeling scenario in which

299   the undesirable species (species 2) was initially dominant and the management goal was to

300   change the stable state to favor species 1 (Fig. S2). The dynamics in this scenario mirror those

301   presented in Fig. 2, but because of the initial dominance of species 2, the magnitude of

302   management action (stocking or harvest) needed to change the stable state towards species 1 was
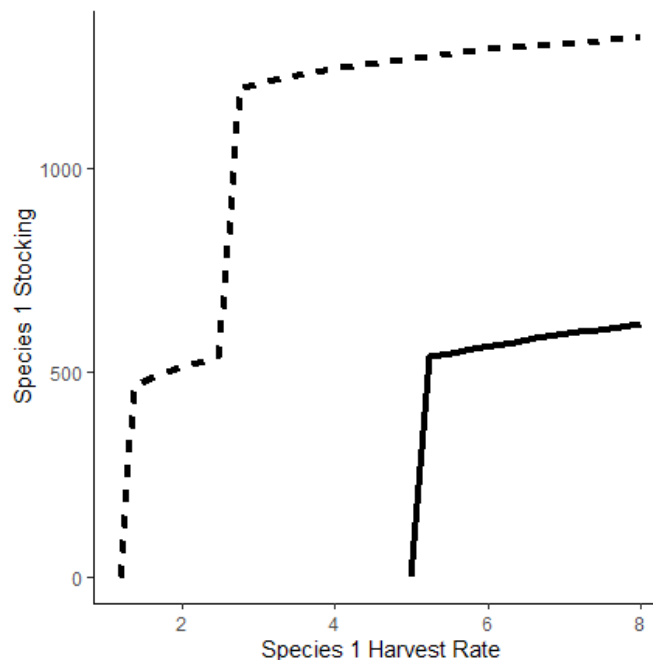
303   much higher.

304

305   *Fig. 2. Leveraging interspecific interactions expands the range of management options that can*

306   *achieve the management goal. In this experiment, species 1 is initially dominant and the goal is*

307   *to maintain that dominance. The dashed and solid lines are isoclines separating combinations of*

308   *species 1 harvesting and stocking rates that yield species 1 dominance (above each line) from*

309   *combinations that yield species 2 dominance (below each line). Dominance is only assigned*

310   *when abundances differ by >100 individuals. If the manager only controls species 1 harvesting*

311   *and stocking rates (dashed line), the ranges of those rates that produce the desired outcome of*

312   *species 1 dominance are relatively limited (above dashed line). If the manager also imposes*

313   *harvest on species 2 at a rate of 2 (solid line), the range of species 1 harvest and stocking rates*

314   *that produce the desired outcome is greatly expanded (above solid line).*

315

316   *Modeling Experiment 2: Alternative Approaches*

317   The Alternative Approaches Experiment showed that species 1 dominance can be

318   maintained through diverse management actions when accounting for interspecific interactions.

319   Combining direct management action (i.e., stocking) with indirect management action (i.e.,

320   managing a competitor) creates a diverse set of options for the decision maker allowing them to

321   tradeoff less direct management action for more indirect action or vice versa (Fig. 3). The trade-

322   off between stocking and harvest of the competitor was consistent across different levels of

323   harvest on the desired species; only the magnitude of management action necessary changed. At

324   low levels of species 2 harvest, more stocking was required to maintain the stable state of the

325   system in which species 1 remained dominant. Decision makers can decrease stocking effort by

326   encouraging harvest of species 2 in order to maintain the stable state of a system. These

327   dynamics were also explored for a scenario in which the decision maker aimed to change the

328   stable state from species 2 dominance toward species 1 (Fig. S3). Because of the initial

329   dominance of species 2 in this scenario, the magnitude of management action (stocking or

330   harvest) needed to change the stable state towards species 1 was higher to account for initial

331   dominance of species 2.
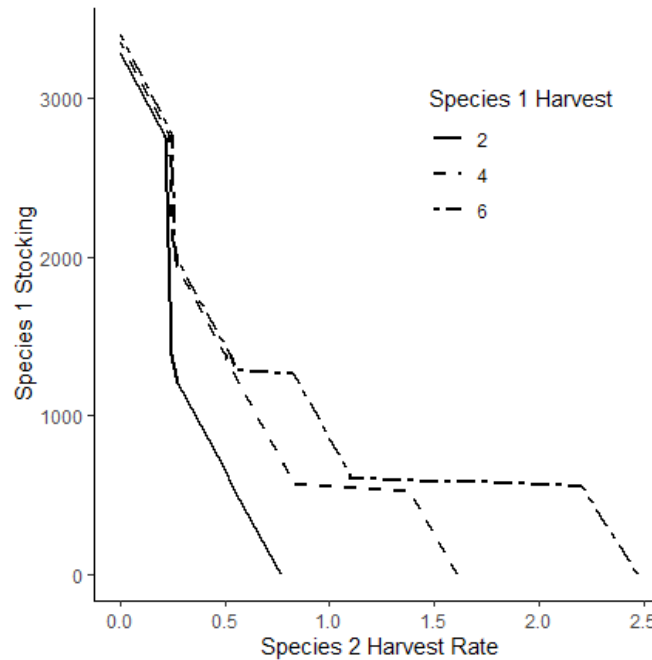
332

333   *Fig. 3. Stocking of species 1 and harvest of species 2 can, on their own, maintain the desired*

334   *stable state of a system (species 1 dominance) in the Alternative Approaches Experiment.*

335   *Tradeoff between stocking and competitor harvest are presented for various levels of harvest on*

336   *species 1 (solid and dashed lines). Areas above/to the right of the lines represent positive*

337   *outcomes (species 1 dominance), areas below/to the left represent regime shifts to species 2.*

338   *Dominance is only assigned when abundances differ by >100 individuals. The negative*

339   *relationship between stocking species 1 and harvesting species 2 allows managers to achieve*

340   *similar outcomes through implementation of either strategy or a combination of both.*

341

342   *Modeling Experiment 3: Safe Operating Space*

343   Investigation of slow change towards a threshold in the system revealed the effectiveness

344   of management intervention for the prevention of shifts to alternate stable states (Fig. 4a).

345   Management actions implemented singularly may delay a transition through either harvesting

346   species 2 (Fig. 4b) or stocking species 1 (Fig. 4c). In combination however, managing both

347   species (through stocking of species 1 and harvest of species 2) prevented a regime shift

348   altogether (Fig. 4d). A combination of strategies still led to a decrease in species 1 abundance but

349   avoided a compensatory increase in species 2, thereby effectively maintaining conditions for

350   species 1 even as a driver outside managerial control pushes the system towards species 2

351   dominance. When no management action is taken, the stable state changes after 40 years (Fig.

352   4a). Minimal harvesting (qE_2=0.5) alone was able to delay the transition by 111 years (Fig 4b).

353   Adding 500 juveniles annually through stocking delayed the change in stable state by 18 years
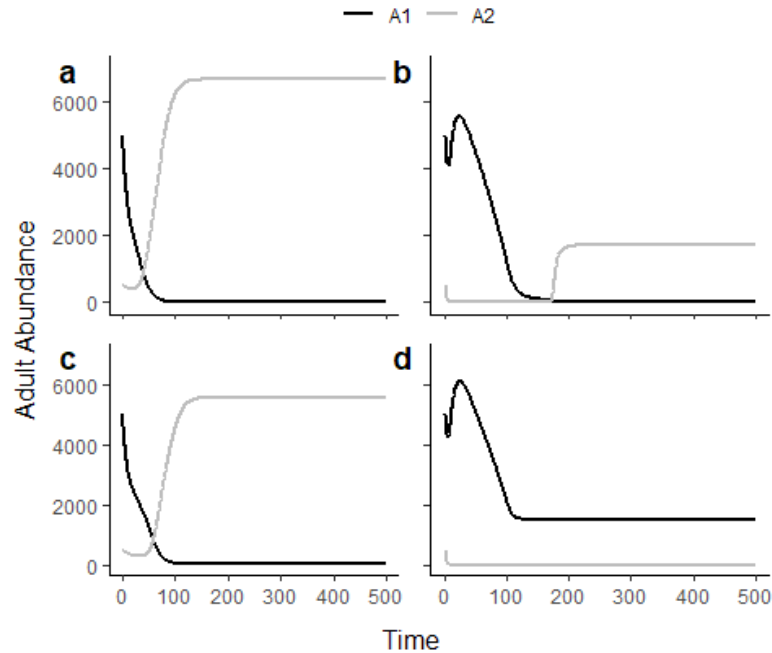
354   (Fig. 4c).

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Fig. 4. Delaying transitions in the Safe Operating Space Experiment. Slow declines in

recruitment represents a factor outside of managerial control, which will inevitably change the

stable state from sp1 dominated to sp2 (no action; panel a). The change in system state can be

delayed through either harvest of species 2 (panel b), or stocking of the desired species (panel c),

or perhaps prevented altogether by stocking and harvesting (panel d).

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Discussion

363

Sudden, unexpected regime shifts represent a growing threat to aquatic systems as human

influences grow and erode ecosystem resilience (De Roos and Persson 2002; Carpenter and

Kinne 2003; Persson et al. 2007). We demonstrated how management interventions could be

used to maintain stable states of a system through careful consideration of human influences and

368   interspecific interactions as drivers of regime shifts within a system. When a single species

369   management approach is infeasible or unable to achieve the desired stable state, our relatively

370   simple model of a multi-species recreational fishery demonstrated how species interactions could

371   allow a decision maker to creatively manage a system to reach desired outcomes. Although

372   interspecific interactions have long been known to exert influences on a system, here we showed

373   how direct management can use those interactions in order to influence fishery outcomes,

374   including changing the stable state of a system. Although our model is a simplification of a

375   complex system, it demonstrated the need to incorporate drivers of stable state dynamics into the

376   management of these important resources.

377   Fisheries have often been managed through a single species lens (Hjermann et al. 2004;

378   Walters et al. 2005; Carpenter et al. 2017); however, this practice has not always resulted in

379   positive outcomes. Our results, and the research of others, have demonstrated why positive

380   feedback loops, which are often unaccounted for, sometimes produce unexpected outcomes in

381   the eyes of decision makers (Tonn et al. 1992; Pine et al. 2009; Solomon et al. 2020). In our

382   model, the key feedback loop was through juvenile competition and predation by adults. When

383   maintaining the abundance of species 1, the decision maker's ultimate goal is to conserve or

384   increase the number of species 1 juveniles maturing to adulthood. Although this result can be

385   achieved directly through stocking, this may not be the most effective management strategy;

386   limitations to stocking include density-dependent mortality and high costs associated with adding

387   individuals (Cowx 1999). Furthermore, our model demonstrated how stocking can be rendered

388   ineffective when a portion of the stocked fish will feed species 2, thereby promoting species 2

389   abundance and beginning a feedback loop wherein their own juveniles increase in abundance

390   (Fig 4c). Thus, the magnitude of stocking that was necessary to maintain the system was greatly

391   increased when solely stocking was implemented, which may not be feasible given other

392   limitations associated with stocking (Fig. 2). Alternatively, if fishing mortality was increased on

393   species 2, with or without stocking, species 1 juvenile survival increased as predation pressure

394   was alleviated, allowing species 1 adults to maintain dominance (Figs. 2 &3). An understanding

395   of how ecological interactions create positive feedback loops (e.g., Pine et al. 2009) that result in

396   stable ecosystem states can allow decision makers to make choices that leverage these feedback

397   loops to increase the probability of maintaining the desired stable state.

398   Decision makers are limited by political, monetary, mechanical, and technological

399   constraints when confronting complex management problems. Most commonly, fishery decision

400   makers turn to one of four different tools for preventing or mitigating the negative influences of

401   humans on a system: (1) stocking (e.g., Cowx 1994); (2) harvest regulation (e.g., length and bag

402   limits; Post et al., 2003); (3) habitat modification (Jennings et al. 1999; Sass et al. 2017); and/or

403   (4) fishery closure (either temporary or permanent). Although each of these management

404   interventions has a history of success in certain circumstances, management responses in

405   complex systems (beyond single species) are not always straightforward. Often, these actions

406   produce no response or a counterintuitive response when species interactions are not

407   acknowledged (Fig. 2). For example, stocking of lake trout (*Salvelinus namaycush*) in Lake

408   Granby, Colorado resulted in declines in Kokanee salmon (*Oncorhynchus nerka*) and other

409   meso-predator species (Johnson and Martinez 1995). However, by investigating feedbacks in

410   these interactions, we provide a strategy for using those tools already available in innovative

411   ways to produce positive fishery outcomes. Not only must a decision maker consider direct and

412   indirect management, but timing of management interventions and lags in implementation have

413   also been shown to influence the outcome of action (Biggs et al. 2009; Martin et al. 2020).

414   Consideration of alternative management strategies, such as leveraging ecological

415   interactions, can aid decision makers in reinforcing the desired stable state of a system. Although

416   the limited set of options available to decision makers may be ineffective or even detrimental

417   when implemented without consideration of species interactions, these interactions can be

418   leveraged to create more avenues for maintenance of a stable state. For example, stocking has the

419   potential to be ineffective at maintaining the stable state of a system (Fig. 4c). Here, we highlight

420   how interspecific interactions can be a reason for stocking ineffectiveness at times. Our model

421   showed that alternatives such as harvest controls of the target species or management of a

422   competitor species can often be more effective than stocking in the production of favorable

423   outcomes (Fig.s 2 & 3). Although there are other drivers that influence the effectiveness of

424   stocking in a system (e.g., habitat loss, climate change, genetics; Lorenzen 2014; Hansen et al.

425   2015; Ziegler et al. 2017; Tingley III et al. 2019), our research emphasizes the critical need to

426   integrate species interactions into management scenarios. This idea is not entirely novel; indeed,

427   invasive species management has long included introducing 'biocontrol' agents into a system in

428   an effort to reduce invasive species abundance (Secord 2003; Hoddle 2004; Roth et al. 2010). A

429   key distinction between our multi-species fishery and invasive species management was the use

430   of existing ecological interactions between the species already present, as opposed to the

431   introduction of a novel agent.

432   Certain drivers of regime shifts in ecosystems may be outside of managerial control, such

433   as slow moving changes in recruitment as a result of climate change (Hansen et al. 2017).

434   Increasing consideration of these drivers has resulted in the emergence of a safe operating space

435   concept, increasing the call for adapting management to respond to ecological variables and

436   complexity in the system (Carpenter et al. 2017; Hansen et al. 2019). Although safe operating

437   space concepts allow for the management of complexity, we highlight managing such

438   complexity explicitly through consideration of interspecific interactions. Tradeoffs are likely to

439   arise between directly managing a species or indirectly managing that species through its

440   competitor; however, a better understanding of those interactions is likely to increase predictive

441   ability when proposing alternative management options. Incorporation of the feedbacks and

442   complexity contained within interspecific ecological interactions can provide decision makers

443   with a new dimension to maintain a system in a safe operating space, even in the face of other

444   slowly changing variables.

445   Our two-species model, although relatively simple, illustrates the need to incorporate

446   ecological interactions in fisheries management within complex fishery systems. Human

447   influences on ecosystems will continue to increase (Sih et al. 2011), and understanding species

448   interactions can help to creatively manage these systems given the constraints that decision

449   makers face. Although our model added a layer of complexity not usually considered in most

450   fisheries management models (Solomon et al. 2020), we acknowledge that there is still

451   significant complexity inherent in these systems that was not captured here. Further exploration

452   of this complexity will allow the integration of multiple ecological and social interactions into

453   fisheries management, as well as provide decision makers with the tools necessary to sustainably

454   manage fisheries in the most efficient ways possible (Solomon et al. 2020). Future research

455   incorporating cultivation-depensation effects of species interactions, or other ecological

456   interactions that induce alternate stable states, may provide empirical evidence supporting the

457   importance of considering these dynamics in managing complex systems.

458   Another layer of complexity to consider is the social component of fisheries (Solomon et

459   al. 2020). In contrast to commercial fisheries in which users aim to maximize profit, recreational

460   fishery users vary along multiple axes of species preference, catch rate, fish size, location,

461   valuation, utility, avidity, and harvest-orientation (e.g., Johnston et al. 2010; Beardmore et al.

462   2015; Arlinghaus et al. 2017). Users place differing levels of importance on each of these aspects

463   of the fishing experience, leading to divergent, and in some cases, competing desires by fishery

464   users and ultimately complex management problems. For example, anglers may choose to

465   voluntarily catch and release certain species (Gaeta et al. 2015; Sass and Shaw 2020). When

466   managers try to promote harvest of a given species (e.g., through liberalized bag and length

467   limits), anglers may simply choose to continue releasing their catch, rendering this management

468   strategy ineffective (Miranda et al. 2017; Sass and Shaw 2020). Management goals are often

469   focused on maintaining a system in a 'desired' stable state; however, what is 'desired' is

470   determined by human desires and may conflict with overall sustainability of the resource.

471   Appropriate responses or management to changing demands from stakeholders in their system

472   will be reliant on a foundational understanding of ecological interactions (specifically through

473   cultivation-depensation mechanisms) and the resultant shifts towards a more 'desirable' stable

474   state. Ultimately, sustainability of the resource in the long-term should trump human desires.

475   Sustainable resource use and meeting human desires need not be in conflict, and by accounting

476   for interspecific interactions, decision makers may be able to achieve both goals.

477   Integration of ecological dynamics, interspecific interactions, and potential regime shifts

478   into ecosystem-based freshwater fisheries management may increase a decision maker's ability

479   to maintain systems in a desired stable state and reduce the likelihood of unexpected or

480   undesirable outcomes while using standard interventions and reducing overall costs.

481   Experimental reductions in competitor abundance coupled with various stocking regimes is just

482   one example of how our modeling results could be used to design an adaptive management

483   experiment that generates new knowledge about creatively managing fisheries. In practice,

484   adaptive management is challenging to implement and often fails; however, the causes of these

485   failures have been well studied and allow for a way forward towards success (Walters 1998,

486   2007; Allen and Gunderson 2011). The wide breadth of knowledge accumulated can play an

487   integral role in building resilient fisheries. By taking a more ecosystem-oriented view of

488   management, outcomes can be improved, and areas can be identified for further exploration

489   when actions produce unexpected outcomes.

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491   **References**

492   Ahrens, R. N. M., C. J. Walters, and V. Christensen. 2012. Foraging arena theory: Foraging

493   arena theory. Fish and Fisheries **13**: 41–59. doi:10.1111/j.1467-2979.2011.00432.x

494   Allen, C. R., and L. H. Gunderson. 2011. Pathology and failure in the design and implementation

495   of adaptive management. Journal of Environmental Management **92**: 1379–1384.

496   doi:https://doi.org/10.1016/j.jenvman.2010.10.063

497   Arlinghaus, R., J. Alós, B. Beardmore, and others. 2017. Understanding and Managing

498   Freshwater Recreational Fisheries as Complex Adaptive Social-Ecological Systems.

499   Reviews in Fisheries Science & Aquaculture **25**: 1–41.

500   doi:10.1080/23308249.2016.1209160

501   Baum, J. K., and B. Worm. 2009. Cascading top-down effects of changing oceanic predator

502   abundances. Journal of Animal Ecology **78**: 699–714. doi:10.1111/j.1365-

503   2656.2009.01531.x

504   Beardmore, B., L. M. Hunt, W. Haider, M. Dorow, and R. Arlinghaus. 2015. Effectively

505   managing angler satisfaction in recreational fisheries requires understanding the fish

506   species and the anglers C. Ramcharan [ed.]. Can. J. Fish. Aquat. Sci. **72**: 500–513.

507   doi:10.1139/cjfas-2014-0177

508   Beverton, R. J., and S. J. Holt. 1957. On the dynamics of exploited fish populations, U.K.

509   Ministry of Agriculture and Fisheries.

510   Biggs, R., S. R. Carpenter, and W. A. Brock. 2009. Turning back from the brink: Detecting an

511   impending regime shift in time to avert it. PNAS **106**: 826–831.

512   doi:10.1073/pnas.0811729106

513   Carpenter, S., and W. Brock. 2004. Spatial Complexity, Resilience, and Policy Diversity: Fishing

514   on Lake-rich Landscapes. Ecology and Society **9**. doi:10.5751/ES-00622-090108

515   Carpenter, S. R. 1998. The Need for Large-Scale Experiments to Assess and Predict the

516   Response of Ecosystems to Perturbation, p. 287–312. *In* M.L. Pace and P.M. Groffman

517   [eds.], Successes, Limitations, and Frontiers in Ecosystem Science. Springer.

518   Carpenter, S. R., W. A. Brock, J. J. Cole, J. F. Kitchell, and M. L. Pace. 2008. Leading indicators

519   of trophic cascades. Ecology Letters **11**: 128–138. doi:https://doi.org/10.1111/j.1461-

520   0248.2007.01131.x

521   Carpenter, S. R., W. A. Brock, G. J. A. Hansen, and others. 2017. Defining a Safe Operating

522   Space for inland recreational fisheries. Fish Fish **18**: 1150–1160. doi:10.1111/faf.12230

523   Carpenter, S. R., and O. Kinne. 2003. Regime shifts in lake ecosystems.

524   Casini, M., J. Hjelm, J.-C. Molinero, J. Lövgren, M. Cardinale, V. Bartolino, A. Belgrano, and

525   G. Kornilovs. 2009. Trophic cascades promote threshold-like shifts in pelagic marine

526   ecosystems. PNAS **106**: 197–202. doi:10.1073/pnas.0806649105

527   Cowx, I. G. 1994. Stocking strategies. Fisheries Management and Ecology **1**: 15–30.

528   doi:https://doi.org/10.1111/j.1365-2400.1970.tb00003.x

529   Cowx, I. G. 1999. An appraisal of stocking strategies in the light of developing country

530   constraints. Fisheries Management and Ecology **6**: 21–34.

531   doi:https://doi.org/10.1046/j.1365-2400.1999.00139.x

532   De Roos, A. M., and L. Persson. 2002. Size-dependent life-history traits promote catastrophic

533   collapses of top predators. PNAS **99**: 12907–12912. doi:10.1073/pnas.192174199

534   Essington, T. E., M. L. Baskett, J. N. Sanchirico, and C. Walters. 2015. A novel model of

535   predator-prey interactions reveals the sensitivity of forage fish: piscivore fishery trade-off

536   to ecological conditions. ICES Journal of Marine Science **72**: 1349–1358.

537   doi:doi:10.1093/icesjms/fsu242

538   Gaeta, J. W., T. R. Hrabik, G. G. Sass, B. M. Roth, S. J. Gilbert, and M. J. Vander Zanden. 2015.

539   A whole-lake experiment to control invasive rainbow smelt (Actinoperygii, Osmeridae)

540   via overharvest and a food web manipulation. Hydrobiologia **746**: 433–444.

541   doi:10.1007/s10750-014-1916-3

542   Hansen, G. J. A., J. W. Gaeta, J. F. Hansen, and S. R. Carpenter. 2015. Learning to Manage and

543   Managing to Learn: Sustaining Freshwater Recreational Fisheries in a Changing

544   Environment. Fisheries **40**: 56–64. doi:10.1080/03632415.2014.996804

545   Hansen, G. J. A., J. S. Read, J. F. Hansen, and L. A. Winslow. 2017. Projected shifts in fish

546   species dominance in Wisconsin lakes under climate change. Global Change Biology **23**:

547   1463–1476. doi:10.1111/gcb.13462

548   Hansen, G. J. A., L. A. Winslow, J. S. Read, M. Treml, P. J. Schmalz, and S. R. Carpenter. 2019.

549   Water clarity and temperature effects on walleye safe harvest: an empirical test of the

550   safe operating space concept. Ecosphere **10**: e02737.

551   doi:https://doi.org/10.1002/ecs2.2737

552 Hjermann, D. Ø., G. Ottersen, and N. C. Stenseth. 2004. Competition among fishermen and fish

553   causes the collapse of Barents Sea capelin. PNAS **101**: 11679–11684.

554   doi:10.1073/pnas.0402904101

555 Hoddle, M. S. 2004. Restoring Balance: Using Exotic Species to Control Invasive Exotic

556   Species. Conservation Biology **18**: 38–49. doi:https://doi.org/10.1111/j.1523-

557   1739.2004.00249.x

558 Jennings, M. J., M. A. Bozek, G. R. Hatzenbeler, E. E. Emmons, and M. D. Staggs. 1999.

559   Cumulative Effects of Incremental Shoreline Habitat Modification on Fish Assemblages

560   in North Temperate Lakes. North American Journal of Fisheries Management **19**: 18–27.

561   doi:https://doi.org/10.1577/1548-8675(1999)019<0018:CEOISH>2.0.CO;2

562 Johnson, B. M., and P. J. Martinez. 1995. Selecting Harvest Regulations for Recreational

563   Fisheries: Opportunities for Research/Management Cooperation. Fisheries **20**: 22–29.

564   doi:https://doi.org/10.1577/1548-8446(1995)020<0022:SHRFRF>2.0.CO;2

565 Johnston, F. D., R. Arlinghaus, and U. Dieckmann. 2010. Erratum: Diversity and complexity of

566   angler behaviour drive socially optimal input and output regulations in a bioeconomic

567   recreational-fisheries model. Can. J. Fish. Aquat. Sci. **67**: 1897–1898. doi:10.1139/F10-

568   113

569 Lorenzen, K. 2014. Managing Fisheries Enhancements, p. 649–657. *In* G.G. Sass and M. Allen

570   [eds.], Foundations of Fisheries Science. American Fisheries Society.

571 Lynch, A. J., B. J. E. Myers, C. Chu, and others. 2016. Climate Change Effects on North

572   American Inland Fish Populations and Assemblages. Fisheries **41**: 346–361.

573   doi:10.1080/03632415.2016.1186016

574   Martin, R., M. Schlüter, and T. Blenckner. 2020. The importance of transient social dynamics for

575   restoring ecosystems beyond ecological tipping points. PNAS **117**: 2717–2722.

576   doi:10.1073/pnas.1817154117

577   Messing, R. H., and M. G. Wright. 2006. Biological control of invasive species: solution or

578   pollution? Frontiers in Ecology and the Environment **4**: 132–140.

579   doi:https://doi.org/10.1890/1540-9295(2006)004[0132:BCOISS]2.0.CO;2

580   Miranda, L. E., M. E. Colvin, A. C. Shamaskin, L. A. Bull, T. Holman, and R. Jones. 2017.

581   Length Limits Fail to Restructure a Largemouth Bass Population: A 28-Year Case

582   History. North American Journal of Fisheries Management **37**: 624–632.

583   doi:https://doi.org/10.1080/02755947.2017.1308891

584   Myers, R. A., J. K. Baum, T. D. Shepherd, S. P. Powers, and C. H. Peterson. 2007. Cascading

585   Effects of the Loss of Apex Predatory Sharks from a Coastal Ocean. Science **315**: 1846–

586   1850. doi:10.1126/science.1138657

587   Oken, K. L., and T. E. Essington. 2016. Evaluating the effect of a selective piscivore fishery on

588   rockfish recovery within marine protected areas. ICES Journal of Marine Science **73**:

589   2267–2277. doi:doi:10.1093/icesjms/fsw074 Original

590   Persson, L., P.-A. Amundsen, A. M. De Roos, A. Klemetsen, R. Knudsen, and R. Primicerio.

591   2007. Culling Prey Promotes Predator Recovery--Alternative States in a Whole-Lake

592   Experiment. Science **316**: 1743–1746. doi:10.1126/science.1141412

593   Pine, W. E., S. J. D. Martell, C. J. Walters, and J. F. Kitchell. 2009. Counterintuitive Responses

594   of Fish Populations to Management Actions. Fisheries **34**: 165–180. doi:10.1577/1548-

595   8446-34.4.165

596   R Core Team. 2020. R: A Language and Environment for Statistical Computing, R Foundation

597   for Statistical Computing.

598   Roth, B. M., T. R. Hrabik, C. T. Solomon, N. Mercado-Silva, and J. F. Kitchell. 2010. A

599   simulation of food-web interactions leading to rainbow smelt Osmerus mordax

600   dominance in Sparkling Lake, Wisconsin. Journal of Fish Biology **77**: 1379–1405.

601   doi:https://doi.org/10.1111/j.1095-8649.2010.02764.x

602   Roth, B. M., I. C. Kaplan, G. G. Sass, and others. 2007. Linking terrestrial and aquatic

603   ecosystems: The role of woody habitat in lake food webs. Ecological Modelling **203**:

604   439–452. doi:10.1016/j.ecolmodel.2006.12.005

605   RStudio Team. RStudio | Open source & professional software for data science teams.

606   Sass, G. G., A. L. Rypel, and J. D. Stafford. 2017. Inland Fisheries Habitat Management:

607   Lessons Learned from Wildlife Ecology and a Proposal for Change. Fisheries **42**: 197–

608   209. doi:https://doi.org/10.1080/03632415.2017.1276344

609   Sass, G. G., and S. L. Shaw. 2020. Catch-and-Release Influences on Inland Recreational

610   Fisheries. Reviews in Fisheries Science & Aquaculture **28**: 211–227.

611   doi:10.1080/23308249.2019.1701407

612   Secord, D. 2003. Biological control of marine invasive species: cautionary tales and land-based

613   lessons, p. 117–131. *In* J. Pederson [ed.], Marine Bioinvasions: Patterns, Processes and

614   Perspectives. Springer Netherlands.

615   Sih, A., M. C. O. Ferrari, and D. J. Harris. 2011. Evolution and behavioural responses to human-

616   induced rapid environmental change. Evolutionary Applications **4**: 367–387.

617   doi:https://doi.org/10.1111/j.1752-4571.2010.00166.x

618   Soetaert, K., T. Petzoldt, and R. W. Setzer. 2010. Solving Differential Equations in   R: Package

619   deSolve,.

620   Solomon, C. T., C. J. Dassow, C. M. Iwicki, and others. 2020. Frontiers in modelling social–

621   ecological dynamics of recreational fisheries: A review and synthesis. Fish and Fisheries

622   **21**: 973–991. doi:https://doi.org/10.1111/faf.12482

623   Tingley III, R. W., C. Paukert, G. G. Sass, P. C. Jacobson, G. J. A. Hansen, A. J. Lynch, and P.

624   D. Shannon. 2019. Adapting to climate change: guidance for the management of inland

625   glacial lake fisheries. Lake and Reservoir Management **35**: 435–452.

626   doi:10.1080/10402381.2019.1678535

627   Tonn, W. M., C. A. Paszkowski, and I. J. Holopainen. 1992. Piscivory and Recruitment:

628   Mechanisms Structuring Prey Populations in Small Lakes. Ecology **73**: 951–958.

629   doi:https://doi.org/10.2307/1940171

630   Walters, C. J. 1986. Adaptive management of renewable resources, Macmillian Publishing

631   company.

632   Walters, C. J. 1998. Improving Links Between Ecosystem Scientists and Managers, p. 272–286.

633   *In* M.L. Pace and P.M. Groffman [eds.], Successes, Limitations, and Frontiers in

634   Ecosystem Science. Springer.

635   Walters, C. J. 2007. Is Adaptive Management Helping to Solve Fisheries Problems? ambi **36**:

636   304–307. doi:10.1579/0044-7447(2007)36[304:IAMHTS]2.0.CO;2

637   Walters, C. J., V. Christensen, S. J. Martell, and J. F. Kitchell. 2005. Possible ecosystem impacts

638   of applying MSY policies from single-species assessment. ICES Journal of Marine

639   Science **62**: 558–568. doi:10.1016/j.icesjms.2004.12.005

640   Walters, C. J., and F. Juanes. 1993. Recruitment Limitation as a Consequence of Natural

641   Selection for Use of Restricted Feeding Habitats and Predation Risk Taking by Juvenile

642   Fishes. Canadian Journal of Fisheries and Aquatic Sciences. doi:10.1139/f93-229

643   Walters, C. J., and S. J. D. Martell. 2004. Fisheries ecology and management, Princeton

644   University Press.

645   Walters, C., and J. F. Kitchell. 2001. Cultivation/Depensation Effects on Juvenile Survival and

646   Recruitment: Implications for the Theory of Fishing. Can. J. Fish. Aquat. Sci. **58**: 39–50.

647   doi:10.1139/f00-160

648   Walters, C., D. Pauly, V. Christensen, and J. F. Kitchell. 2000. Representing Density Dependent

649   Consequences of Life History Strategies in Aquatic Ecosystems: EcoSim II. Ecosystems

650   **3**: 70–83. doi:10.1007/s100210000011

651   Ziegler, J. P., C. J. Dassow, S. E. Jones, A. J. Ross, and C. T. Solomon. 2018. Coarse woody

652   habitat does not predict largemouth bass young of year mortality during the open-water

653   season. Canadian Journal of Fisheries and Aquatic Sciences. doi:10.1139/cjfas-2018-

654   0050

655   Ziegler, J. P., E. J. Golebie, S. E. Jones, B. C. Weidel, and C. T. Solomon. 2017. Social-

656   ecological outcomes in recreational fisheries: the interaction of lakeshore development

657   and stocking. Ecological Applications **27**: 56–65. doi:https://doi.org/10.1002/eap.1433

658   2006. International Council for the Exploration of the Sea. Report of the Baltic Fisheries

659   Assessment Working Group **24**.

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665