Evidence-based economic analysis demonstrates that ecosystem service benefits of water hyacinth management greatly exceed research and control costs

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Invasive species management can be a victim of its own success when decades of effective control cause memories of past harm to fade and raise questions of whether programs should continue. Economic analysis can be used to assess the efficiency of investing in invasive species control by comparing ecosystem service benefits to program costs, but only if appropriate data exist. We used a case study of water hyacinth (*Eichhornia crassipes* (Mart.) Solms), a nuisance floating aquatic plant, in Louisiana to demonstrate how comprehensive record-keeping supports economic analysis. Using long-term data sets, we developed empirical and spatio-temporal simulation models of intermediate complexity to project invasive species growth for control and no-control scenarios.

For Louisiana, we estimated that peak plant cover would be 76% higher without the substantial growth rate suppression (84% reduction) that appeared due primarily to biological control agents. Our economic analysis revealed that combined biological and herbicide control programs, monitored over an unusually long time period (1975-2013), generated a benefit-cost ratio of about 34:1 derived from the relatively modest costs of $124 million ($2013) compared to the $4.2 billion ($2013) in benefits to anglers, waterfowl hunters, boating-dependent businesses, and water treatment facilities over the 38-year analysis period. This work adds to the literature by 1) providing evidence of the effectiveness of water hyacinth biological control; 2) demonstrating use of parsimonious spatio-temporal models to estimate benefits of invasive species control; and 3) incorporating activity substitution into economic benefit transfer to avoid overstating benefits. Our study suggests that robust and cost-effective economic analysis is enabled by good record keeping and generalizable models that can demonstrate management effectiveness and promote social efficiency of invasive species control.

1   **Evidence-based economic analysis demonstrates that ecosystem service**

2   **benefits of water hyacinth management greatly exceed research and control**

3   **costs**

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1   Abstract

Invasive species management can be a victim of its own success when decades of effective control cause memories of past harm to fade and raise questions of whether programs should continue. Economic analysis can be used to assess the efficiency of investing in invasive species control by comparing ecosystem service benefits to program costs, but only if appropriate data exist. We used a case study of water hyacinth (*Eichhornia crassipes* (Mart.) Solms), a nuisance floating aquatic plant, in Louisiana to demonstrate how comprehensive record-keeping supports economic analysis. Using long-term data sets, we developed empirical and spatio-temporal simulation models of intermediate complexity to project invasive species growth for control and no-control scenarios.

For Louisiana, we estimated that peak plant cover would be 76% higher without the substantial growth rate suppression (84% reduction) that appeared due primarily to biological control agents. Our economic analysis revealed that combined biological and herbicide control programs, monitored over an unusually long time period (1975-2013), generated a benefit-cost ratio of about 34:1 derived from the relatively modest costs of $124 million ($2013) compared to the $4.2 billion ($2013) in benefits to anglers, waterfowl hunters, boating-dependent businesses, and water treatment facilities over the 38-year analysis period. This work adds to the literature by 1) providing evidence of the effectiveness of water hyacinth biological control; 2) demonstrating use of parsimonious spatio-temporal models to estimate benefits of invasive species control; and 3) incorporating activity substitution into economic benefit transfer to avoid overstating benefits. Our study suggests that robust and cost-effective economic analysis is enabled by good record keeping and generalizable models that can demonstrate management effectiveness and promote social efficiency of invasive species control.

2   Why conduct economic analysis of invasive species?

Robust demonstration of invasive species management program benefits may be crucial to maintaining programs with tight budgets over the long term, particularly if programs are successful at reducing harms, thereby reducing the apparent urgency of action. Yet, such analyses are not possible if agencies do not collect the appropriate data and information. Economic analyses require robust cause-and-effect relationships be established between management actions and environmental changes in order to design control strategies to maximize net benefits (Shackelford et al., 2013) and avoid spending when success rates are low (Wainger et al., 2010). However, the necessary data are often incomplete, largely because conditions prior to control and the effectiveness of control are not fully documented or records are not maintained for long enough.

Control of harmful non-native invasive species is the type of management decision that can benefit from a thorough economic analysis comparing benefits and costs for several reasons. First, efficient spending is needed since there are always more threats to species and ecosystems than resources to confront them (e.g., Wilcove et al., 1998). Second, the ongoing debate about whether we are vilifying invasive species without cause, and thereby wasting resources on their control (Lodge & Shrader-Frechette, 2003; Shackelford et al., 2013), can often be resolved through economic analyses. While much of the economic literature examines potential optimal control, retrospective analyses of actions are useful because they reveal the harms averted through diligent management and the value of ongoing management or prevention.

3   What information supports a solid economic analysis?

For economic evaluations to enable efficient allocation of scarce invasive control resources, values (or indicators of value) must measure outcomes for which people would be willing to pay or otherwise trade off other goods and services. This basic concept, which is fundamental to economic theory, is often obfuscated by studies that quantify benefits of invasive species control in monetary terms that are not measuring economic benefits. In particular, many studies have equated reduced costs of control with benefits (e.g., Sinden & Griffith, 2007). However, changes in costs of invasive species control do not directly demonstrate that the spending was in the public interest.

66   The economic measure of benefits, *utility*, encompasses all tangible and intangible effects on

67   well-being, not only financial effects (Freeman, Herriges & Kling, 2014). As a result, a wide

68   array of environmental changes will be relevant to cost benefit or cost-effectiveness analysis, if

69   they can be connected to human concerns. For example, many studies have documented peoples'

70   willingness to pay to retain rare species (Richardson & Loomis, 2009). These *nonuse* values

71   represent intangible benefits derived from stewardship of the environment. For aquatic species,

72   previously measured benefits include these nonuse values and use values from recreational and

73   commercial fishing, energy supply, water supply, agriculture, industry, tourism, property value

74   support, and flood damages avoided (Lovell, Stone & Fernandez, 2006).

75   Even with the best data collection, cost-benefit analysis of proposed invasive species

76   management requires modeling to fill in for unobservable data. A substantial challenge to

77   analyzing benefits of past control is developing the *counterfactual* or without-action scenario that

78   is needed to understand how management changes outcomes. Data-rich case studies reveal the

79   methods that can be used to dynamically integrate socio-ecological systems and quantify benefits

80   of invasive species control (as described in Olson, 2006). However, sophisticated models are

81   time-consuming or impossible to build for many data-poor case studies and their use is far from

82   routine for evaluating management options.

83   To ease some of the analytic burdens of measuring economic benefits, many government

84   agencies use economic benefit transfer models for routinely estimating program benefits (Wilson

85   & Hoehn, 2006). Economic benefit transfer is the process of transferring values that have been

86   empirically estimated for one or more locations to unstudied, or transfer, locations (Johnston et

87   al., 2015). Although this approach is subject to error, the accuracy is generally increased by

88   using studies that are closely matched to the transfer site (Plummer, 2009) or using functional

89   models to adjust values based on the social, economic, and ecological conditions of the site

90   (Johnston & Wainger, 2015).

91   Even with benefit transfer, substantial work can be required to apply the technique to quantify

92   how environmental changes impact human concerns (Johnston & Wainger, 2015; Mazzotta et

93   al., 2015). Modeling complexity can be compounded when temporal and spatial detail is required

94   to accurately assess impacts. For example, the degree of overlap in time between aquatic plant

95   growth and seasonal boat-dependent activities can determine degree of harm (e.g., Adams &

96   Lee, 2007). Similarly, a spatial framework may be needed to accurately model effects of

97   proposed management, such as using barriers to prevent range expansion (Sharov, 2004; e.g.,

98   Rahel & Olden, 2008).

99   An often overlooked component of system response in benefit transfer is the degree to which

100 people adapt to change, which if not considered, can inflate value estimates. For example, Keller

101 et al. (2008) valued the benefits of preventing invasion of rusty crayfish into lakes as the

102 *elimination* of spending by anglers seeking panfish, since these fish would be extirpated by the

103 invasion. Yet, it is plausible that at least some anglers would switch to alternative lakes or

104 alternative species and enjoy comparable or modestly reduced benefits. Assuming all benefits are

105 lost from an environmental change is common in benefit transfer because data for generalizing

106 how people adapt are poor. Site-specific studies with detailed data can be used to estimate

107 substitution of sites or other adaptations.

108 Although it is always tempting to add model complexity to better capture human-environment

109 interactions and reduce error, complexity also tends to reduce the transferability of methods and

110 findings and increases data requirements. In the analysis that follows, we sought to create

111 parsimonious models to support routine economic analyses of invasive effects (e.g., see Robinet

112 et al., 2012). Yet, we also wanted to include relevant spatial, temporal and behavioral detail to

113 reduce error of estimates. To achieve these goals, our modeling approach combined economic

114 benefit transfer with moderately detailed ecological models. We selected a case study of

115 biological control of an invasive aquatic weed (water hyacinth or *Eichhornia crassipes* (Mart.)

116 Solms) in Louisiana due to the rich data sources available, but show that even this data-rich case

117 lacks some critical data for measuring outcomes that are primary program goals.

118 Our work adds to the existing literature by 1) providing evidence of the effectiveness of water

119 hyacinth biological control; 2) demonstrating use of spatially and temporally detailed models to

120 estimate ecosystem service benefits of invasive control; and 3) incorporating substitutability of

121 the most valuable ecosystem services to avoid overstating benefits. In addition, we aim to

122 motivate thorough data collection and thereby improve the data landscape for economic studies

123 by highlighting how even small additions to monitoring data can enable economic analysis.

124 **3.1 Case Study**

125 In Louisiana, water hyacinth (*Eichhornia crassipes* (Mart.) Solms) has been a problem for over a

126 century. Water hyacinth is a widespread invasive floating plant that is often considered one of

127 the world's worst nuisance plants due to its aggressive growth and ability to form large

128 interlocking mats that impede navigation and fishing and impair water quality, among other

129 harms (Howard & Harley, 1997; Center, 2004; Villamagna & Murphy, 2010). Water hyacinth

130 was first introduced into the U.S. in New Orleans, Louisiana in 1884 as an ornamental plant

131 (Wunderlich, 1962). Just 15 years later, it had become such an impediment to navigation in the

132 southeastern states that the U.S. Congress authorized the U.S. Army Corps of Engineers

133 (USACE) to clear infestations in navigable waters (Sanders, Theriot & Perfetti, 1985).

134 A primary concern in Louisiana is that water hyacinth disrupts recreational and commercial

135 boating by limiting access and increasing costs of navigation and maintenance. Small to medium

136 size boats with propeller engines aim to navigate around mats since floating mats clog propellers

137 and water-cooling systems and obscure navigational hazards (South Atlantic Division Field

138 Committee, 1948; Obeid, 1975; Alimi & Akinyemiju, 1991). In addition, dense infestations

139 block wharves, piers and ramps (Howard & Harley, 1997). About 80% of Louisiana recreational

140 boaters surveyed in 2009 moderately or strongly supported "improving aquatic weed control"

141 (Isaacs & Lavergne, 2010).

142 A wide array of additional harms has also been documented in Louisiana or similar settings.

143 Despite being a detriment to recreational fishing access, a substantial literature documents both

144 harms and benefits to fish communities from water hyacinth with effects apparently dependent

145 on the fish species present and plant density (Villamagna & Murphy, 2010). Water hyacinth

146 impairs waterfowl habitat by reducing open water area and depleting food sources (Gowanloch,

147 1944). Effects on infrastructure include clogged water intakes of power plants, water treatment

148 plants, and other industries (USAID, 1971; Mailu, 2001; California Department of Parks and

149 Recreation, Division of Boating and Waterways, 2013) and reduced drinking water quality

150 (Mailu, 2001; UNEP Global Environmental Alert Service, 2013). Further, water hyacinth can

151 block streams and drainage canals, reducing their discharge capacity and increasing flooding risk

152 (Gowanloch & Bajkov, 1948).

3.1.1 History of water hyacinth control in Louisiana

Mechanical removal and herbicide application were the primary treatment strategies of water hyacinth infestations in Louisiana until the early 1970s, when biological control agents were introduced. Biological control, the intentional introduction of a host-specific organism to suppress a target species, has been used as part of the strategy to manage water hyacinth infestations in many southern states and in a number of countries since the 1970s (Center, 2004). Four insect biological control agents have been introduced in Louisiana, *Neochetina eichhorniae* and *N. bruchi* from 1974-1977, *Niphograpta albiguttalis* from 1979-1981, and *Megamelus scutellaris* from 2010-2016. *N. eichhorniae*, or the mottled water hyacinth weevil, was the most successful at establishing and dispersing throughout the state (Coombs, 2004; Grodowitz et al., 2014). *Neochetina* spp. reduce water hyacinth vigor through larval and adult feeding which reduces production, fertility and spread, and increases susceptibility to herbicides (Goyer & Stark, 1984; Gettys et al., 2014; Jones et al., 2017).

A program coordinated between Louisiana Department of Wildlife and Fisheries (LDWF), USACE, and the US Department of Agriculture (USDA) was successful in dispersing *Neochetina* spp. within the state from 1975-77 (Manning 1979). *Neochetina* spp. remain widespread in Louisiana, despite the lack of new releases since the 1980s, although local abundance varies considerably (N. Harms, personal observation, 2017). Variability in weevil populations is likely due to spatially variable habitat conditions including water quality, especially available nutrients, nearby insecticide use and climate. The biological control agent, *M. scutellaris,* was recently released in Louisiana, but colonization and effectiveness at controlling water hyacinth is unknown. No other biological control agents for water hyacinth were being released as of 2017, and herbicide control is the primary active management tool. Approximately 16,600 ha of water hyacinth control were treated annually from 2000-2013 (Alexander Perret, LDWF, pers. comm., January 22, 2015).

4 Methods

Using data from multiple government agencies and private businesses, we developed an integrated set of models and analyses to conduct a cost-benefit analysis of invasive water hyacinth control in Louisiana. Of the ecosystem services identified through literature review and interviews with government officials or business owners, four had sufficient data and evidence of

183   responsiveness to water hyacinth cover for quantitative analysis: 1) recreational fishing from

184   boats, 2) recreational waterfowl hunting from boats, 3) boat-dependent businesses (marinas,

185   tourism), and 4) drinking water supply. Other services were identified that could not be evaluated

186   due to lack of data to quantify harms in the absence of invasive control including: commercial

187   fishing, commercial navigation, flood risk mitigation, hydroelectric production, and nonuse

188   values for species.

189   Ecosystem service benefits were measured as the difference between conditions with and without

190   water hyacinth treatment. Four types of models or analyses made up our integrated suite of tools

191   used to estimate biophysical conditions and benefits (Figure 1): 1) water hyacinth intrinsic

192   growth rate model; 2) water hyacinth spatio-temporal spread model; 3) damage functions relating

193   water hyacinth cover to biophysical changes; and 4) economic benefit transfer analyses.

194   Extensive data were collected to model the with-treatment scenario and historic conditions or

195   data from unmanaged infestations were used to model the counterfactual scenario. Costs were

196   derived from state and federal government records of research and implementation activities for

197   biological control and herbicides. All analyses and models used the time period of 1975-2013 in

198   the state of Louisiana.

199   **4.1   Intrinsic growth rate model (treatment effectiveness)**

200   Water hyacinth distribution data collected in the spring and fall each year (usually April and

201   November) from 1975 to 2013 (Louisiana Department of Wildlife and Fisheries, 2014) enabled

202   us to model plant growth rate response to biological control. We fit predicted to observed cover

203   data using a logistic growth model that controlled for winter severity by including a variable of

204   days with minimum temperature at or below freezing (Nesslage et al., 2016). The model fit was

205   substantially improved by incorporating a time-varying intrinsic growth rate for water hyacinth

206   such that the growth rate declined through time. Results from this model informed the spatio-

207   temporal spread model, as described below.

208   **4.2   Spatio-temporal spread model**

209   Because many benefits and harms of ecosystem changes depend on where and when people are

210   using the ecosystem, we developed a spatially and temporally detailed model of occurrence by

211   adapting an existing parsimonious model, which had been tested on six invasive species,

212   including water hyacinth (Model D, Robinet et al., 2012). The spatio-temporal model included

213 three submodels: a) habitat suitability model, b) logistic growth model, and c) kernel density

214 function to spread plants across the landscape. The first model was used to parameterize spatial

215 differences in growth and carrying capacity and the latter two models were run in sequence, at

216 each time step to evaluate biomass and spread.

217 We modified the Robinet et al. (2012) model by incorporating a time-varying growth parameter

218 in the logistic growth model (to reflect biological control effects), developing a sparse kernel

219 density function adapted to work over large regions, and fit multiple empirical model parameters

220 to Louisiana data (Appendix A; Supplement S1). For example, we estimated water hyacinth

221 carrying capacity per landscape grid cell by fitting ratios of observed levels of maximum cover

222 (Manning, 1979) to model-generated estimates of habitat suitability (Sutherst, Maywald &

223 Kriticos, 2007, CLIMEX niche maps, D. Kriticos pers. comm.). We also developed kernel

224 density function parameters by calibrating a simulation model to historic data on water hyacinth

225 coverage by major watershed. Cover data from the period prior to widespread biological control

226 release (1975-1978) provided information on conditions prior to biological control that were

227 used to calibrate the model for the counterfactual scenario.

228 **4.3 Damage functions for ecosystem services**

229 We created a general damage function to relate percent cover to loss of benefits and

230 parameterized the function for each ecosystem service. The damage function generates a

231 proportion (unitless value) that is multiplied by the total potential benefits by location and year,

232 in the absence of water hyacinth. The function documents the degree to which boat travel time

233 increases as a function of percent cover of water hyacinth, up to the point at which a waterway

234 becomes impassible.

235 The damage function was parameterized using findings from landscape ecology since

236 recreational boats navigate around water hyacinth mats in a manner similar to animals navigating

237 a fragmented landscape (Obeid, 1975; Alimi & Akinyemiju, 1991). Research suggests that the

238 path length that an organism requires to move through a landscape (i.e., connectivity) increases

239 when unsuitable land cover reaches 15-25% of the landscape (With, Gardner & Turner, 1997).

240 Further, movement can become impossible when unsuitable cover reaches 70-90% of the

241 landscape (Mönkkönen & Reunanen, 1999). Based on these findings, we selected 20% and 80%

242 as the thresholds to represent the water hyacinth cover that would begin to impede and then

243   prevent navigation (Figure 2).[1] The linear segment of the function was drawn to connect these

244   two points and reflected increasing path length as suitable cover (i.e., open water) was lost (after

245   With, Gardner & Turner, 1997). Selected boat operators were consulted by phone and all

246   concurred that our estimates were realistic.

247   **4.4   Economic valuation methods**

248   All ecosystem service benefits were estimated through benefit transfer or costs avoided methods

249   that required multiplying a unit value by the number of affected entities (Table 1). In benefit

250   transfer, the final value was estimated as the total consumer surplus, which is a measure of well-

251   being derived from the difference between what a consumer would have been willing to pay to

252   enjoy a service and what was actually paid (Rosenberger & Loomis, 2001; Freeman, Herriges &

253   Kling, 2014). Costs avoided were measured as direct expenses avoided.

254   **4.4.1   Benefit transfer methods**

255   Using an existing database (Rosenberger, 2011), we estimated a unit value per recreational user

256   day as the average of 19 studies, selected because they represented freshwater fishing or hunting

257   from boats in southern states. We converted values to 2013 dollars using the consumer price

258   index (Bureau of Labor Statistics (BLS), 2013). We estimated an average user day consumer

259   surplus of \$55.90/day for fishing and \$47.46/day for hunting. The hunting consumer surplus was

260   made a function of existing area suitable for hunting per parish, following methods described in

261   Wainger et al. (2013).

262   Total user days per year were estimated from license sales data (Louisiana Department of

263   Wildlife and Fisheries, 2015) and survey data for fishing and hunting. To estimate freshwater

264   fishing days, two surveys were cross-referenced (Ogunyinka & Lavergne, 2009; US Fish and

265   Wildlife Service, 2013) and only 20% of waterfowl hunting days were used from the national

266   survey (US Fish and Wildlife Service, 2013) to represent freshwater activity, as based on a

267   Louisiana survey (Laborde & Rohwer, 2010). The ratio of permits sold to user days was

268   calculated for the year when both data were available and then applied to convert historic data on

269   licenses to total annual user days for all years.

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[1] Some types of boats that use air propulsion rather than propeller engines are able to navigate water hyacinth mats without problems. No data were available to quantify the proportion of such boats in use, however, anecdotal reports suggest that these are not in widespread use for fishing, hunting and boat tourism.

270   The affected users by location and time step were determined by comparing spatio-temporal

271   model projections of water hyacinth cover with estimated spatial and temporal usage patterns.

272   Recreational fishing days were distributed to each parish using an angler survey (Ogunyinka &

273   Lavergne, 2009) and divided into two seasons using monthly crappie fishing effort data, which

274   were the most complete of available data and representative of multiple fisheries (A. Perret pers.

275   comm. and Louisiana Department of Wildlife & Fisheries, 2013). Based on the survey, we

276   estimated that about 65% of fishing effort occurred December - April, when biomass of water

277   hyacinth was estimated to be at low winter-spring levels and 35% occurred the rest of the year

278   when biomass was at or near peak. Hunting days were assigned to parishes based on the

279   percentage of survey respondents that identified a given parish as their favorite (Laborde &

280   Rohwer, 2010) and distributed evenly across the months in which hunting was permitted.

281   Waterfowl hunting effort was split 50-50 between low and high biomass seasons.

282   The final benefit calculation by recreational service, spatial region and season was the product of

283   user day value and user days, modified by the percent of total benefits per water hyacinth cover,

284   as provided by the damage function (Figure 2). The benefits per service per year (V_y) were a

285   weighted sum of the two seasons, as given by:

286
$$V_y = \Sigma_{s=(1,2)} a_s \Sigma_{r=(1,\ldots,n)} (ud_{yr})(b(c_s)) \qquad \text{(Equation 1)}$$

287   where y = year; s = season (1 = high biomass, 2 = low biomass); r = region (1,…,n) which

288   represents sub-areas of parishes defined by discrete bins of percent cover; a_s = proportion of

289   annual recreational activity allocated to season; u = value per user day without water hyacinth;

290   d_{yr} = annual user days per year and region. The function $b(c_s)$ is the damage function that

291   provides the proportion of benefits delivered per water hyacinth cover per season (c_s).

292   Cumulative benefits were the sum of the 38 years of record (1975-2013).

293   In the without control scenario, cover becomes dense and widespread in some areas, resulting in

294   most or all fishing days lost. It is likely that under such extreme conditions some anglers would

295   find alternative recreation activities, rather than lose all recreation benefits. A recent survey of

296   freshwater anglers in Australia suggested that 59% of anglers would be willing to substitute a

297   different outdoor recreation, if they could not go fishing (Sutton & Oh, 2015). To account for

298   peoples' willingness to substitute another activity for fishing, we assumed a linear increase in

299 substitution between 20% and 80% water hyacinth cover, up to a maximum of 59% substitution.

300 We handled substitution differently for hunting by reducing consumer surplus per additional

301 hunting day as huntable area increased. Additional methods are provided in Supplement S2.

4.4.2 Avoided cost methods

303 Costs avoided were judged appropriate to use because businesses that were affected by water

304 hyacinth were expending resources in management. Marina operators respond to presence of

305 water hyacinth by using boats to mechanically break up mats. The reported costs of control (fuel,

306 labor and equipment) ranged from $13,000 to $23,000 per marina per year, based on interviews

307 with a small set of operators. Approximately 400 marinas on non-estuarine brackish water

308 (Louisiana Oil Spill Coordinator's Office, 2004) were estimated from GIS analysis to be

309 vulnerable to invasion.

310 Water treatment plant operators, similarly, used boats to mechanically break up water hyacinth

311 mats that clogged intake pipes. The average annual cost of mechanical breakup (including fuel,

312 labor, and equipment) was reported as $2,300 (2013$) or a range of $1,100-$3,800. We

313 identified 77 vulnerable sites using a GIS analysis of surface water intake locations (Louisiana

314 Department of Health and Hospitals, 2006).

315 Avoided costs were scaled to the density of water hyacinth in the vicinity of the marina or

316 treatment plant using the damage function (Figure 2), parameterized for each service

317 (Supplement S3). For both entities, the damage function was scaled by using the average annual

318 cost at 80-100% cover and assuming spending was 0 at 0-20% cover. Cover was assessed in the

319 immediate vicinity (grid cell) of the marina or water treatment plant.

4.5 Cost of water hyacinth research and control

321 We estimated the total costs of water hyacinth management as the sum of treatment (mechanical,

322 herbicide and biological control), research and program costs for water hyacinth from 1975-

323 2013. Ongoing treatment costs in Louisiana have been borne by the state and the USACE.

324 Research investments were made through the USACE Aquatic Plant Control Research Program

325 (APCRP) on biological, herbicidal and integrated control to manage water hyacinth in the US

326 (1975-1989). US Department of Agriculture, Agricultural Research Service (USDA-ARS)

327 contributed to biological control agent development via efforts such as overseas exploration and

328 host-specificity studies. The USACE Large Scale Operational Management Test (LSOMT)

329   program (in place 1976-1980) primarily funded release and monitoring of biological control

330   agents in Louisiana (Sanders & Theriot, 1986). Costs reflected government management

331   activities conducted on public waters and not activities on privately owned lands. Program

332   management costs were included in most data sources but we added a 5% management cost to

333   federal programs when such costs were not provided. Further information is in supplement S4.

334   Historical (1975-2013) management costs for Louisiana were consolidated from annual reports

335   and spending data from the LDWF, removal of aquatic growth (RAG) programs of the USACE

336   New Orleans District (USACE-MVN) and the USACE Mobile district (USACE-SAM), and the

337   USACE Aquatic Plant Control Research Program (USACE-APCRP). When data were

338   unavailable in some years, we estimated missing values by interpolation, so as not to

339   underestimate spending levels. All values were adjusted for inflation (Bureau of Labor Statistics

340   (BLS), 2013) and reported in 2013 dollars.

341   Three different approaches were used to estimate costs of developing biological control agents

342   for water hyacinth. For each of the two initial *Neochetina* spp. agents, we apportioned national

343   program research spending (Andres 1977) to Louisiana using the percentage of water hyacinth

344   habitat contained in that state (23% or 2.3 million ha) relative to all US states, including Hawaii

345   (Wainger et al. 2016). Reported costs were not available for development of two other agents,

346   *Niphograpta albiguttalis* or *Megamelus scutellaris*, so we conservatively estimated costs

347   associated with *N. albiguttalis* as the same as *Neochetina* spp., and estimated *M. scutellaris* costs

348   using information provided by principal investigators involved with the project (Dr. Phillip

349   Tipping, personal communication).

350   5   Results

351   5.1   Spatio-temporal spread modeling

352   The results of the spatio-temporal modeling showed substantial differences between the with-

353   and without-control scenarios. The differences in the percentage of water area invaded by water

354   hyacinth were estimated to be 57% higher in the spring and 76% higher in fall 2013 (Figure 3)

355   without treatment. The dramatic decline in coverage with control appeared to be supported by an

356   84% decline in growth rates over the study period (1976-2003) (Nesslage et al., 2016). The

357 spatial results reflected the north-south gradient in habitat suitability, since abundance was

358 greater in the more suitable (warmer) southern areas (Figure 4).

359 Results strongly suggested that biological control had reduced water hyacinth cover because of

360 the observed decrease in growth rate over time and because we found no statistically significant

361 difference in herbicide use between years with high and low extent of water hyacinth (Exact

362 Wilcoxen Rank Test, Wilcox W=51, p = 0.14; high extent defined as >400,000 acres statewide),

363 suggesting that herbicide treatment was not responsible for overall historic declines. However,

364 increased efficiency of herbicide treatment may partially explain a lack of correlation between

365 treatment and subsequent declines in cover. We also noted that growth decreased, despite

366 documented increases in average air temperatures over the study period that would tend to

367 increase the growth rate of this tropical plant (Wilson, Holst & Rees, 2005).

368 **5.2 Economic benefits**

369 The total value of the four ecosystem service benefits over the 38-year analysis period was

370 estimated as $4.2 Billion (Table 2). The vast majority of these benefits were from preserving

371 recreational freshwater fishing, which would be substantially impacted during times of peak

372 water hyacinth cover. For 2013, the last year of analysis, the annual benefits were $195 Million.

373 The correction for activity substitution had a large effect on values, reducing fishing benefits

374 from $418 M to $172 M in year 2013. All reported values are 2013 dollars, unless otherwise

375 specified.

376 **5.3 Cost analysis**

377 Costs of research and programmatic expenses of water hyacinth biological control were

378 estimated to total $29.5 Million across multiple government agencies (Table 3) or $8.4 Million

379 for Louisiana alone. Among the responsible agencies, a total of approximately $115 M was spent

380 on herbicide application of water hyacinth between 1975 and 2013. With that cost added, the

381 total spending for Louisiana was $124 M, reflecting the large proportion of herbicide control

382 costs in management spending.

383 Missing data create some uncertainty in these values. Some biological control development costs

384 from the 1970s are lacking due to missing data from large scale rearing and large equipment used

385   in releasing agents. Herbicide costs were modestly overestimated (5-10%) because, in some

386   years, program funds were targeted to other plants.

387   **5.4   Cost benefit analysis**

388   When aggregate benefits (Table 2, *Cumulative Benefits*) were divided by aggregate costs (Table

389   3, *Louisiana Cost*), the benefit cost (B/C) ratio was 34:1, strongly suggesting that the benefits of

390   water hyacinth control well exceeded the program costs. If we adopt the perspective that B/C

391   ratios should be evaluated based on the present value at program initiation, all costs and benefits

392   for our case study should be discounted to a present value for 1975 (Hill and Greathead, 2000).

393   When we conducted this analysis, the B/C ratio dropped to 6.8:1 at a 3% discount rate or 2.9:1 at

394   a 7% discount rate. The ratio drops because costs accrue early while substantial benefits accrue

395   many years later. This new B/C ratio based on 1975 present values nonetheless, still suggests

396   that the program is socially efficient, since it more than pays for itself. These analyses show that

397   the B/C ratio is sensitive to perspectives built into the analysis and how future users are reflected

398   in the discount rate (See Cropper, 2013).

399   **6   Discussion**

400   The high B/C ratio that we found (34:1) suggests that the investment in developing biological

401   control agents was an efficient use of funds. The aggregate costs of $124 Million were an order

402   of magnitude smaller than the $4.2 Billion in benefits generated in the 38 years of our dataset.

403   This B/C ratio is consistent with results from other programs to control water hyacinth or similar

404   aquatic nuisance species, which have ranged globally from 2.5:1 to 124:1 (representing many

405   undiscounted values) (McConnachie et al., 2003; De Groote et al, 2003). Ratios for biological

406   control programs across many types of invasive species have ranged from 1-1000:1 (using

407   discounted values) (Hill & Greathead, 2000). While many published studies show a substantial

408   B/C ratio, Hill and Greathead (2000) suggested that even if some biocontrol had B/C ratios less

409   than one, when viewed as a portfolio of investments, the high returns to some agents makes the

410   risk of development new agents worthwhile.

411   Although biological control can take time to show effectiveness and does not usually eradicate

412   the target species (Schooler et al. 2004), it has potentially long-lasting, self-propagating, and

413   self-sustaining benefits that offset the initial upfront investment costs. As evidence from this

414   program, releases of agents in Louisiana were discontinued in the 1980's, yet overall suppression

415   has been maintained (Nesslage et al. 2016). (Recent releases of *M. scutellaris* in Louisiana have

416   not led to confirmed establishment.) In comparison, management of water hyacinth with

417   herbicide requires continual investment.

418   The regional benefits of control can be hard to perceive given that local populations of water

419   hyacinth remain in many places and may still reach damaging levels. Ongoing herbicide

420   application, or maintenance control, is needed to keep local populations in check, a result that is

421   not uncommon with aquatic weeds that are controlled with biocontrol. However, our evidence,

422   which was developed at the regional scale, supports the idea that water hyacinth would be a

423   substantially greater problem statewide without biological control. These findings reflect

424   conditions from the peak of the damage from water hyacinth that included spatially extensive

425   and tall mats of water hyacinth, that have not been observed in 50 years or more.

426   Although measured benefits were substantial, our results likely underestimated benefits since we

427   omitted whole categories of values, particularly nonuse values, that can be comparable to use

428   values (e.g., Navrud, 2001; Johnston, Besedin & Wardwell, 2003) and have been measured for

429   aquatic invasive species cases (McIntosh, Shogren & Finnoff, 2010). Further, we were not able

430   to include harms suggested by historic information, including increased flooding risk, damage to

431   infrastructure, and disruptions to commercial navigation (as suggested by Thunberg, Pearson &

432   Milon, 1992). These events were not routinely recorded and predated the tenure of most current

433   emergency managers and harbor masters, which limited our ability to collect information needed

434   to construct models for the counterfactual scenario. In addition, we only included harms to

435   marinas on brackish waters but almost twice as many boat trips take place on freshwater systems,

436   compared to all saltwater systems (Isaacs & Lavergne, 2010). Further, annual costs to marinas

437   and water treatment facilities were based on relatively modest current infestations and may not

438   be representative of higher infestation levels. Finally, we based our analysis on benefits of agents

439   for only one state, but agents can and are being used in other areas.

440   These benefit omissions did not alter the fact that we found a high benefit-to-cost ratio, which is

441   one way for objectively evaluating optimal level of control (e.g., Leung et al., 2002) and

442   ensuring that benefits of management exceed costs and, therefore, generate net social benefits.

443   Without such analyses, decision makers are left to infer appropriate levels of spending based on

444   incomplete, and possibly biased, information. For example, past problems with biological control

445 have caused managers to be risk averse to embarking on biological control, even to the extent

446 that past successes might not have been possible under current decision rules (Hinz et al., 2014).

447 Despite its usefulness in this case, cost-benefit analysis must be used with caution since,

448 frequently, many types of environmental benefits cannot be monetized. Costs can appear to

449 exceed benefits due to data or technical limitations of economic analysis, rather than a lack of net

450 benefit. For example, if the ability of invasive species to alter system resilience to stress is poorly

451 understood, the benefits of preventative actions will be underestimated (Shackelford et al., 2013).

452 Non-monetary benefit indicators can be an alternative approach to comparing benefits within a

453 cost-effectiveness framework, when they cannot be monetized (Wainger & Mazzotta, 2011).

454 Our economic study was enabled by thorough record keeping on patterns of human use of the

455 environment. Recreational fishing and hunting effort was tracked through time in databases of

456 licenses sold and separate surveys evaluated spatial usage patterns and total use. Spatial data on

457 marina and boat launch locations, created in response to the Deepwater Horizon oil spill, enabled

458 the treatment costs avoided analysis.

459 Time series data or widespread spatial coverage that encompass both invasive species cover and

460 human responses can be used to deepen understanding of tradeoffs associated with different

461 intensities of invasive species treatment effort, relative to cover alone. For example, although we

462 were able to use literature to estimate changes in fishing effort as a function of water hyacinth

463 cover, data that encompassed both fishing effort and cover over a range of conditions would have

464 enabled direct empirical modeling of this effect. Further, data on drainage blockages and

465 upstream water hyacinth cover might have allowed other risks to be estimated to improve

466 evaluation of program success.

467 Finally, human adaptations to environmental change need to be incorporated in economic

468 analysis to accurately estimate risk. We used survey evidence to suggest how people might make

469 activity substitutions under extreme conditions of a changed environment. However, future work

470 would be strengthened by explicitly documenting any thresholds or non-linearities of human

471 responses with aquatic plant cover, including willingness to make activity substitutions, such as

472 salt-water fishing. All biophysical and socio-economic changes are ideally measured with before

473 / after and control / impact (BACI) design experiments (Underwood, 1992), which provide the

474 strongest evidence for isolating treatment effects from natural variability. However, since not all

475 case studies will have thorough studies, results from well-studied locales can be transferred to
476 less studied sites using models and foundational site-specific data.

477 Although the data in Louisiana were among the most comprehensive that we have seen, our
478 analyses have uncertainty due to some data gaps. We could not cleanly separate effects of
479 biological control and herbicides or demonstrate synergies of the two types of treatment (as
480 suggested by Van & Center, 1994; Center et al., 2002), because presence and density of control
481 agents were unknown and herbicide data were summarized at the annual, statewide scale. Also,
482 further study of species interactions that result from treatment would clarify whether treating one
483 invasive species can facilitate invasion by another (e.g., Santos et al., 2009) and undercut
484 economic gains of treatment.

485 Our study suggested that information does not have to be perfect to conduct economic analyses
486 but that robust analysis is supported by sufficient data and information to demonstrate past
487 management effectiveness or to project future success. Data gaps can be filled with after-the-fact
488 mapping using remote sensing data (Albright, Moorhouse & McNabb, 2004) and by using
489 modeling to separate causes from effects. We demonstrated the latter approach by using the
490 spring and fall water hyacinth coverage observations and historic weather records to distinguish
491 effects of overwinter mortality from treatment effects, thereby refuting the hypothesis that cold
492 winters that coincided with biological control release were solely responsible for plant declines
493 (Nesslage et al., 2016).

494 **7   Conclusions**

495 Our cost benefit analysis revealed that water hyacinth control, largely the result of biological
496 control, has generated benefits in Louisiana well in excess of research, development and
497 implementation costs. Using the four ecosystem services with the best data, we estimated a B/C
498 ratio of 34:1, due to all forms of control. This high ratio for aquatic plant control is consistent
499 with other literature that safe and effective biological control agents generate net benefits
500 (Lovell, Stone & Fernandez, 2006) and can dramatically lower management costs of control
501 (Hinz et al., 2014).

502 An additional implication of this work is that sound economic analysis of EGS relies on a
503 substantial amount of data to quantify the many cause-and-effect relationships that link actions to

504 benefits. Benefit assessments are supported by three fundamental types of observations 1)

505 management effectiveness, 2) ecological outcome changes, and 3) social and economic changes.

506 For natural resource managers, the suggestion is that, if they want to be able to demonstrate

507 economic benefits of programs, they not only track changes in invasive cover (with and without

508 management), but also collect data reflecting potential or observed effects on human activities

509 and use of the environment. For aquatic invasive species, questions to address through data

510 collection include: Does water quality become a problem for aquatic life? Do game animals

511 disappear? Is outdoor recreation participation reduced? Not only must data be collected, but

512 critical data sources must be accessible to researchers over long time periods.

513 We also suggest that costs of economic analyses can be reduced by using generalizable analytic

514 tools, such as the spatio-temporal spread model on which we built. Complex models that require

515 many years of site-specific research to parameterize may not support cost-effective treatment

516 action (Simberloff, 2003). Further, generalizable models may be superior to highly specified

517 models because their projections of future outcomes can be more robust to potential variability

518 (discussed further in Robinet et al., 2012).

519 Our results for water hyacinth in Louisiana are likely to be generalizable to other cases where a

520 safe and effective biological control agent has the potential to suppress an aquatic invasive

521 species that interferes with recreation, boat-dependent businesses and/or water supply. The high

522 B/C ratio that we found suggests that our estimates are robust to the minor cost data omissions

523 that we noted and would only increase if we had been able to add more ecosystem services.

524 Overall, we find that biological control has the potential to deliver a long-term stream of

525 ecosystems services that can justify initial research investments.

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Ecosystem services analyzed and associated biophysical and benefit metrics

1   **Table 1. Ecosystem services analyzed and associated biophysical and benefit metrics**

Service	Biophysical Change (captured in damage functions)	Unit Value	Affected entities	Benefit Metric
Recreational Fishing	- Change in boat access	- Consumer surplus per fishing day	- Total user days per season per year (low and high biomass seasons)	- Consumer surplus value of all fishing days adjusted for substitution effects
Recreational Waterfowl Hunting	- Change in boat access	- Consumer surplus per hunting day	- Total user days per season per year (low and high biomass seasons)	- Consumer surplus value of all hunting days
Commercial Tourism	- Marinas and boat launches blocked	- Average response cost per marina per year	- Number of vulnerable marinas (brackish water only; freshwater unavailable)	- Maintenance costs avoided (mechanically breaking up mats)
Drinking Water	- Number of water intakes physically blocked by water hyacinth	- Average response cost per treatment plant per year	- Number of water supply treatment plants	- Maintenance costs avoided (mechanically breaking up mats)

Ecosystem Service Benefits results (maximum annual and total over 38 years)

1   **Table 2. Ecosystem service benefits results (maximum annual and total over 38 years)**

Ecosystem Service	Maximum Potential Affected Users/Entities	Annual Benefits of Control in 2013 – Final Year of Analysis (M$2013)	1975-2013 Cumulative Benefits (M$2013)
Recreational freshwater fishing (With activity substitution)*	583,480 anglers	$ 172	$ 3,880
Recreational freshwater fishing (No activity substitution)	*583,480 anglers*	*$ 418*	*$ 9,450*
Recreational waterfowl hunting	19,400 waterfowl hunters	$ 7.3	$ 164
Commercial boat tourism	400 marinas (South Louisiana only)	$ 15.1	$ 199
Drinking water supply	77 drinking water intakes	$ 0.24	$ 3.00
TOTAL (with substitution)		$ 612.5	$ 4,242

2   *Substitution refers to adjustment made to reflect anglers choosing an alternative form of

3   recreation if freshwater fishing is unavailable. See section 3.4. (benefit transfer methods) for

4   details.

Spending on water hyacinth management, research and development from 1975-2013 in Louisiana

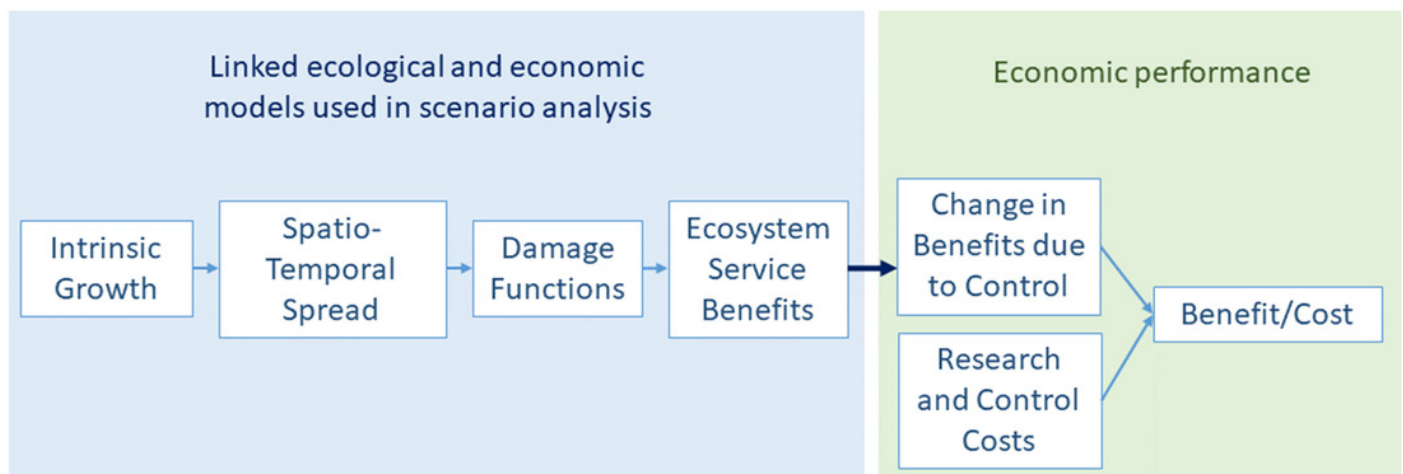
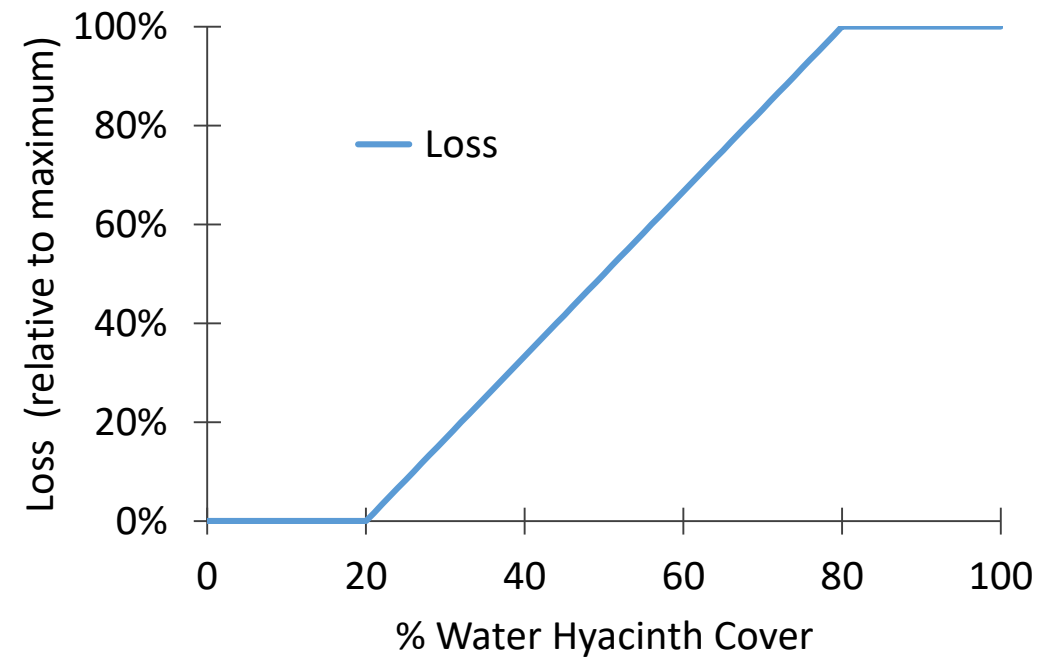
1   **Table 3. Spending on water hyacinth management, research and development from 1975-**

2   **2013 in Louisiana**

Category	Organization	Total Program Cost (M$2013)	Louisiana Cost (23% of research costs, M$2013)*	Time period
Herbicide Research (APCRP)	USACE-ERDC	$ 6.74	$ 1.55	1976-1989
Biological Control Research (APCRP)	USACE-ERDC	$ 4.95	$ 1.14	1975-2014
Biological Control Development	USDA-ARS	$ 12.76	$2.93	
Integrated Control Research (APCRP)	USACE-ERDC, MVN	$ 2.98	$ 0.69	1976-1989
Large Scale Operations Management Test (LSOMT)	USACE-ERDC, MVN	$ 2.1	$ 2.1	1975-1980
Subtotal for Research		**$29.53**	**$ 8.41**	1975-2014
Herbicide Application	USACE		$ 94.6	1975-2013
	LDWF		$ 20.7	1975-2013
Subtotal for Herbicide application			**$ 115.3**	1975-2013
TOTAL			**$ 124.36**	

3   * See Supplement S4 for further details

Figure 1

Conceptual diagram of modeling approach

General damage function used to estimate loss of ecosystem service as a function of water hyacinth density

Simulation of fall water hyacinth density through time, with and without control

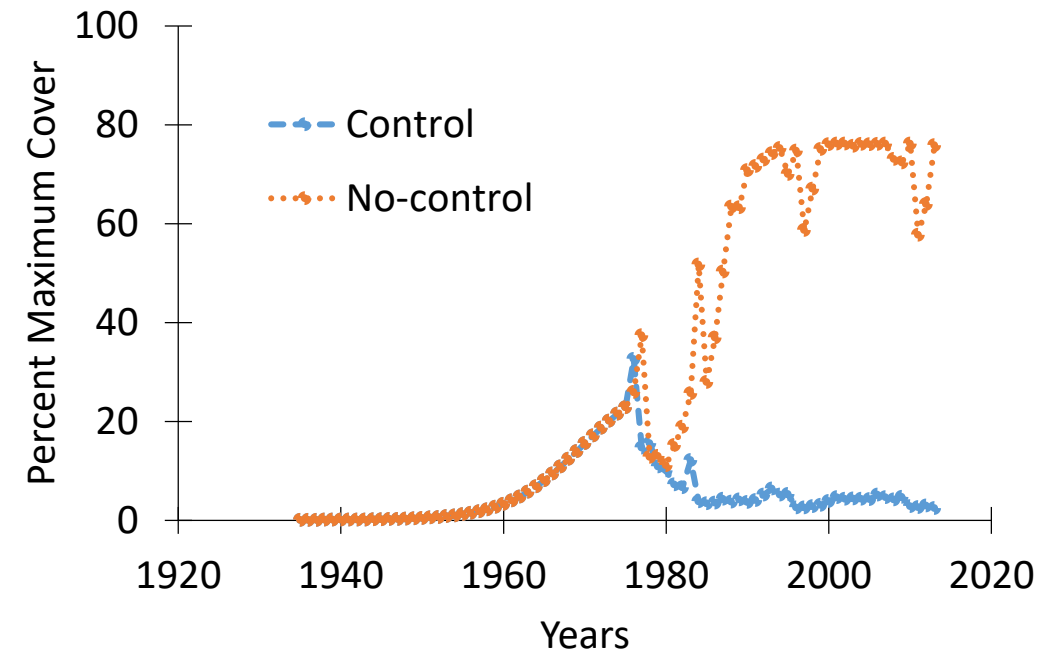



Figure 4

Water hyacinth coverage per grid cell (as % of water area) for the without and with control scenarios (Fall estimates)