Using historical dip net data to infer absence of flatwoods salamanders in stochastic environments

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Local extinction and undetected presence are two very different biological phenomena, but they can be challenging to differentiate. Stochastic environments hamper the development of standardized monitoring schemes for wildlife, and make it more challenging to plan and evaluate the success of conservation efforts. To avoid reintroductions of species at risk that could jeopardize extant populations, managers attempting translocation events require a higher level of confidence that a failure to confirm presence represents a true absence. For many pond breeding amphibians, monitoring of the breeding population occurs indirectly through larval surveys. Larval development and successful recruitment only occurs after a sequence of appropriate environmental conditions, thus it is possible for a breeding population of adults to exist at a site but for detectability of the species to be functionally zero. We investigate how annual variability in detection influences long-term monitoring efforts of Reticulated Flatwoods Salamanders (*Ambystoma bishopi*) breeding in 29 wetlands in Florida. Using 10 years of historic dip net data, we simulate plausible monitoring scenarios that incorporate environmental stochasticity into estimates of detection probability. We found that annual variation in environmental conditions precluded a high degree of certainty in predicting site status for low-intensity monitoring schemes. Uncertainty was partly alleviated by increasing survey effort, but even at the highest level of sampling intensity assessed (four surveys per year), multiple years of monitoring are required to confidently determine presence/absence at a site. Combined with assessments of habitat quality and landscape connectivity, our results can be used to identify sites suitable for reintroduction efforts. Our methodologies can be generally applied to increase the effectiveness of surveys for diverse organisms for which annual variability in detectability is known.

USING HISTORICAL DIP NET DATA TO INFER ABSENCE OF FLATWOODS SALAMANDERS IN STOCHASTIC ENVIRONMENTS

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17 **Abstract**

18 Local extinction and undetected presence are two very different biological phenomena, but they

19 can be challenging to differentiate. Stochastic environments hamper the development of

20 standardized monitoring schemes for wildlife, and make it more challenging to plan and evaluate

21 the success of conservation efforts. To avoid reintroductions of species at risk that could

22 jeopardize extant populations, managers attempting translocation events require a higher level of

23 confidence that a failure to confirm presence represents a true absence. For many pond breeding

24 amphibians, monitoring of the breeding population occurs indirectly through larval surveys.

25 Larval development and successful recruitment only occurs after a sequence of appropriate

26 environmental conditions, thus it is possible for a breeding population of adults to exist at a site

27 but for detectability of the species to be functionally zero. We investigate how annual variability

28 in detection influences long-term monitoring efforts of Reticulated Flatwoods Salamanders

29 (*Ambystoma bishopi*) breeding in 29 wetlands in Florida. Using 10 years of historic dip net data,

30 we simulate plausible monitoring scenarios that incorporate environmental stochasticity into

31 estimates of detection probability. We found that annual variation in environmental conditions

32 precluded a high degree of certainty in predicting site status for low-intensity monitoring

33 schemes. Uncertainty was partly alleviated by increasing survey effort, but even at the highest

34 level of sampling intensity assessed (four surveys per year), multiple years of monitoring are

35 required to confidently determine presence/absence at a site. Combined with assessments of

36 habitat quality and landscape connectivity, our results can be used to identify sites suitable for

37 reintroduction efforts. Our methodologies can be generally applied to increase the effectiveness

38 of surveys for diverse organisms for which annual variability in detectability is known.

39

Introduction

40

41 Local extinction and undetected presence are two very different phenomena, but they can be

42 challenging to differentiate. There has been much attention devoted to detection probability, and

43 especially to the challenges raised when working with rare and cryptic species in stochastic

44 environments (Lebreton et al., 1992; MacKenzie et al., 2002; Royle, Nichols & Kéry, 2005;

45 Pacifici, Dorazio & Conroy, 2012; Kellner and Swihart, 2014; Moore et al., 2014; Specht et al.,

46 2017; Folt et al., 2019). The detectability of many species varies with environmental conditions;

47 annual variation in weather can strongly affect the optimal time of year to survey, and the overall

48 likelihood of detecting the target species (Field, Tyre & Possingham, 2005; Jackson et al., 2006;

49 McConville et al., 2009; Rizzo et al., 2017; Shaffer, Roloff & Campa 2019). If the factors

50 influencing detectability are not known, one can never be confident that failure to detect

51 constitutes a true absence, and the development of standardized monitoring schemes will be

52 severely hampered (Penteriani et al., 2005; Bried & Pellet, 2012; Gervasi et al., 2014; Bellier,

53 Kéry & Schaub, 2016; Crone, 2016).

54 Species with complex life cycles pose an additional challenge because different life-stages may

55 require different sampling methodologies, or may only be available for capture at particular

56 times of the year. Many pond-breeding amphibians are fossorial for most of the year, migrating

57 to and from breeding sites annually in response to seasonal cues. For such species, it is nearly

58 impossible to avoid monitoring techniques that rely on specific environmental conditions to be

59 effective. Factors known to directly influence detectability are often unknown, difficult to

60 assess, or poorly understood. Thus in many instances, monitoring schemes cannot be

61 strategically tailored to maximize success with any degree of precision.

62   Amphibian monitoring typically occurs at breeding wetlands, either through drift-fences that

63   intercept arriving individuals, dip-net surveys for aquatic larvae, or egg-mass counts (Wells,

64   2007). Owing to the high labor cost of continually running a drift-fence however, monitoring of

65   adult populations is often sporadic or entirely absent. Population assessments based solely on

66   dip-net data are common, yet fraught with uncertainty (Penteriani et al., 2005; Bried & Pellet,

67   2012; Gervasi et al., 2014; Bellier, Kéry & Schaub, 2016; Crone, 2016). Because larval

68   development occurs only in years with suitable weather and hydrological conditions, it is

69   possible for a breeding population of adults to occur but for the detection probability of larvae to

70   be zero. Failure to detect may simply represent a bust recruitment year, offering no information

71   as to the health of the population. In some cases it should be easy enough to realize that the

72   species would be undetectable in a given year (extreme drought), but certain patterns of pond

73   filling and drying and refilling, or flooding carrying in fish predators, may also cause complete

74   egg or larval mortality without being obvious (Semlitsch, 1987; Dodd, 1993; Taylor, Scott &

75   Gibbons, 2006).

76   In some cases, the repercussions of failing to detect an organism are considerably more severe

77   than incorrectly documenting a presence, i.e., false negatives are more problematic than false

78   positives (Hauser & McCarthy, 2009; Charney, Kubel & Eiseman, 2015; Veale & Russello,

79   2016). Conservation reintroductions are a useful example of the risk imbalance between type-1

80   and type-2 errors. Failure to detect a target organism may result in reintroductions that could

81   jeopardize extant populations (Kéry & Schmidt, 2008). Introducing animals to an extant

82   population can have positive (Hedrick & Frederickson, 2010) or negative (Laikre et al., 2010)

83   impacts on genetic viability, and as such a clear distinction must be made between establishing

84   new populations and augmenting existing ones. Identifying suitable, unoccupied sites for

85   establishing new populations can be hampered by environmental stochasticity. Similarly,

86   evaluating the relative success of reintroduction efforts is made challenging by low or uncertain

87   detectability brought on by environmental variation. Thus there is a clear need for approaches

88   that can confidently predict site status and determine the fate of individuals following

89   reintroduction under a range of environmental conditions.

90   We have monitored hydrology and worked with partners to restore suitable vegetation in

91   wetlands across Eglin Air Force Base, in hopes of restoring Reticulated Flatwoods Salamanders

92   (*Ambystoma bishopi*) to a greater proportion of their former range. Procedures for allowing an

93   existing population to expand into improved habitat, augmenting an existing population, or

94   conducting a re-introduction are very different (and for a protected species would require

95   different levels of approval from regulatory agencies). Assessing whether a translocation action

96   would result in augmentation vs. re-introduction depends on knowing whether there is an extant

97   population or not. Here we investigate how annual variability in detection influences long-term

98   monitoring efforts. Using historic dipnet data we simulate a range of scenarios that incorporate

99   environmental stochasticity into estimates of detection probability.and evaluate the efficacy of

100 different monitoring schemes. In the face of environmental uncertainty, identifying potential

101 reintroduction sites requires high survey effort over a period of years to confidently declare

102 absence. Our procedure can be adapted for any species when there are data available for

103 detection probability over a series of years that encompass a meaningful range of environmental

104 conditions.

105

106 **Materials & Methods**

107 Study site and species.—Our study was conducted on Eglin Air Force Base (Eglin), Okaloosa

108 and Santa Rosa counties, Florida, USA from 2009 to 2017. Eglin is a large military installation

109 (188,459 ha) primarily consisting of actively managed longleaf pine-dominated sandhills

110 (approximately 145,000 ha) interspersed with treeless open test ranges, pine plantations, and

111 mesic flatwoods. The study focused on 29 ephemeral wetlands that have been monitored

112 intensively for Reticulated Flatwoods Salamander larvae since 2009. Reticulated Flatwoods

113 Salamanders are mole salamanders in the family Ambystomatidae and are federally listed as

114 endangered, with restoration efforts underway and reintroduction efforts currently being planned.

115 Individuals migrate in the fall to dry basins, and deposit eggs terrestrially (Palis 1997). Embryos

116 develop so that if winter rains inundate the ponds after several weeks or months, the eggs hatch

117 into aquatic larvae. Wetlands were surveyed for salamander larvae using standard dipnet

118 sampling techniques (Bishop et al., 2006; Heyer et al., 1994; Wilkinson, 2015) between 2009

119 and 2018. Surveyed wetlands range in size from 0.1 ha to 20.9 ha and consist of shallow

120 depressions which fill in the autumn and typically remain inundated throughout the spring

121 (Chandler et al., 2016). To account for imperfect detection, at least three surveys were conducted

122 annually, spaced one per calendar month. Surveys were conducted during the months of January

123 until May, and survey effort was standardized to 30 min for each occasion (see Brooks et al.,

124 2019 for full description of field methods).

125

126 Statistical approach.—Using posterior distributions of year-specific detection probabilities

127 derived from a recent occupancy study (Brooks et al., 2019), we investigated the impact of

128 environmental stochasticity on long-term monitoring of Reticulated Flatwoods Salamanders.

129 Over the study period, detection probabilities conditional on a site being occupied ranged from

130   0.02 to 0.88 for a single survey. We adopted a bootstrapping approach, whereby ten years were

131   randomly drawn from the dataset with replacement to construct a decade-long series of potential

132   detection probabilities. We iterated this procedure 1000 times to generate a sample of

133   hypothetical time-series, reflecting our uncertainty in detection given annual environmental

134   fluctuations. A decade was chosen partly to reflect the study duration from which the original

135   data were drawn, and also to approximate the longevity of the species (Brooks 2020), i.e. if our

136   aim is to determine whether Reticulated Flatwoods Salamanders have been extirpated from a

137   site, we would need to continue monitoring until we were confident that no breeding adults

138   remained on the landscape.

139   We first calculated the detection probabilities and associated uncertainty with increasing survey

140   effort (up to 10 surveys) conducted within a single year. We combined the estimates from all

141   years for which detection was available, corresponding to a situation in which the relationship

142   between environmental conditions and detectability are unknown. We then estimated the

143   likelihood of detecting larvae given a site is occupied under four levels of survey effort.

144   Scenarios ranged from one to four surveys per year and extrapolated out to 10 years of

145   continuous monitoring for each of the 1000 hypothetical detection histories. Cumulative

146   detection probabilities for multiple surveys within years were calculated by taking the exponent

147   of the probability of failing to detect an animal, $(1 - p)^n$, where p is the detection probability for

148   a given year and n is the number of surveys. Cumulative detection probabilities across years are

149   simply one minus the product of each year-specific estimate. This approach requires that a site's

150   occupancy status does not change over the course of monitoring, but given the high site fidelity

151   of this species (Brooks et al. 2019), we consider there to be a negligible probability of emigration

152   from the focal site over that time (i.e. no turnover). From the 1000 scenarios we derived 90%

153   credible intervals for detection probabilities to quantify the uncertainty in site status under

154   different levels of sampling intensity and duration. All analyses were conducted in R version

155   4.10 (R Core Team 2021). All of the data and code used to perform the analysis and generate the

156   figures are provided as supplementary material.

157

158   **Results**

159   We found that in the face of environmental uncertainty, increasing the number of intensive, 30-

160   minute surveys in a single year could never achieve detection probabilities above 95% with

161   confidence (Fig. 1). Unsurprisingly, given such tremendous annual variability in detection and

162   no knowledge of the environmental conditions in a particular year, an assessment of

163   presence/absence cannot be made without multiple years of surveying. Although median

164   detection probability reached 99% after six surveys, the lower bound for detection probability

165   remained below 90% even after 10 surveys (Fig. 1), because during drought years the ability to

166   detect larvae can become next to impossible. In contrast the upper confidence limit for detection

167   approaches unity after only three surveys, reflecting years where conditions for detecting larvae

168   are optimal.

169   We further found that even when multiple years of surveys are conducted, a high degree of

170   survey effort within years is necessary to achieve high detection rates (Fig. 2). If sites are

171   sampled only once per year, surveys must be carried out for five years on average to obtain a

172   detection probability above 95%, and seven years to obtain a detection probability above 99%

173   (Fig. 2). Again, given uncertainty in the relationship between environmental conditions and

174   detectability, there are wide confidence bands on these estimates, whereby 99% probabilities

175   could be achieved in three years or could take a decade, depending on the specific years in which

176   sampling takes place. If sites are surveyed twice per year detection probabilities above 95 % and

177   99% can be achieved sooner, at three and five years respectively, and the associated uncertainty

178   in when thresholds are reached is reduced (Fig. 2). Three surveys per year achieved a detection

179   probability above 95% after only three years of monitoring and 99% after four years (Fig. 2). If

180   four surveys are conducted each year, 95% and 99% thresholds are reached after two and three

181   years respectively, and uncertainty in detection is completely ameliorated following four years of

182   monitoring (Fig. 2).

183

184   **Discussion**

185   Environmental stochasticity strongly impacts long-term monitoring efforts for Reticulated

186   Flatwoods Salamanders. Even after a decade of sampling, the occupancy status of Reticulated

187   Flatwoods Salamanders cannot be confidently determined if only one survey is conducted each

188   year (Fig. 1). Increasing the number of surveys within a year greatly reduces the uncertainty in

189   cumulative detection, but even under the highest sampling intensity, at least 5 years are required

190   before sites can confidently be declared unoccupied (Fig. 2).

191   Amphibian populations are frequently monitored through dipnet surveys that target larvae

192   following breeding. For Reticulated Flatwoods Salamanders, researchers have surveyed using

193   dipnets two or three times per year (Palis, 1997; Bishop et al., 2006), and we followed protocols

194   described by Gorman, Bishop & Haas (2009). Standardized monitoring schemes are attractive

195   because they are easy to employ and produce data in a structure desirable for statistical analysis.

196   Such boilerplate schemes however, fail to account for year-to-year fluctuations in survey

197   conditions which prevents a high degree of certainty in predicting site status for low-intensity

198   monitoring schemes. To improve confidence in our own data, we have added a fourth sampling

199   event per season to our protocol, if no flatwoods salamanders were detected after the first three,

200   and have shifted from attempting to sample 25 to 50% of ponds each year to attempting to

201   sample all ponds annually. This will give us the best chance of having sufficient data over a 10-

202   year period to have confidence in our estimates of occupancy, given there will likely be a few

203   years in each decade when drought makes sampling impossible and likely one or two years when

204   financial constraints allow us to only sample 50-75% of ponds. Because reticulated flatwoods

205   salamanders have a maximum adult lifespan of around 10-12 years, to ensure that the local

206   breeding population has been extirpated, it is necessary to confirming lack of breeding under

207   suitable environmental conditions over at least a 10-year period. In fact, on our study site, we

208   have observed periods of 8 or 9 years of sampling with no detections and then detected larvae

209   again, in ponds that are far (1000 m or more) from other occupied ponds.

210   The complex life-cycle of Reticulated Flatwoods Salamanders severely inhibits a reliable

211   assessment of population viability from aquatic surveys alone. Zero detection can occur if 1) the

212   ponds are already filled with water when adults arrive, resulting in eggs being laid outside the

213   pond basin and therefore never being inundated, 2) the pond never fills or fills too late for

214   hatching to occur, 3) the pond dries before larvae can be detected, 4) ponds are overrun with

215   predators. Only when complex facets of a species' life history and ecology are well understood

216   can a flexible monitoring scheme be employed that incorporates the likelihood of encountering

217   an individual in the timing and/or intensity of surveys. Anecdotally, in our own study we have

218   observed a 9-year gap between detections, so could have considered certain populations to have

219   been extirpated if we were not aware of these issues. We had two ponds where flatwoods

220   salamander larvae were detected in 2010, but no detections were made for the next 9 years

221   despite annual sampling. Severe summer drying followed by abundant winter rainfall in 2019-

222   2020 created ideal conditions for larval development and we detected larvae at these two ponds

223   again in the spring of 2020.

224   Until recently, employing a dynamic sampling approach would make subsequent analysis and

225   detecting long-term trends more challenging, but modern statistical techniques can readily

226   accommodate such imbalanced data. Charney, Kubel & Eiseman (2015) describe a method of

227   adaptive sampling that focuses available resources to sample ambystomatid larvae only in years

228   when they are most likely to be detectable, based on conditions in bellwether ponds. Because

229   regional surveys showed high correlation between reproduction within bellwether ponds and

230   other sites, differences in larval abundance were used to determine whether a year was likely to

231   be appropriate for wide scale sampling (N Charney, pers. comm.). However, if breeding

232   attempts are not highly synchronized across the landscape, if bellwether ponds are unavailable,

233   or if climatic conditions that create "good" breeding years are unknown, the approach can be

234   challenging to implement (Pacifici, Dorazio & Conroy, 2012; Shaffer, Roloff & Campa 2019).

235   Although confidence that a species is truly absent if it's not detected can be improved if survey

236   effort is increased, the likelihood will also strongly depend on the prior probability of presence

237   (Wintle et al., 2012). This prior could depend on the time since the species was last observed,

238   perceived site suitability, presence of other species at this site or the focal species at neighboring

239   sites on the landscape. Any reintroduction decisions should take into account habitat

240   connectivity, potential future metapopulation dynamics, and long distance dispersal mechanisms.

241   The marked uncertainty in environmental fluctuations and their impact on detection rates

242   necessitate intensive survey effort over many years before a site can be considered for

243   translocated animals.

244   For other aquatic organisms for which detection is challenging, environmental DNA (eDNA) has

245   been a powerful tool. The annual fluctuations that make detection through dipnetting

246   challenging however, will likely influence eDNA results in the same way. If the ponds are dry,

247   no method that depends on the presence of aquatic larvae will be able to detect them. When

248   conditions are suitable, even using modified techniques of taking frequent local samples and

249   processing larger volumes of water, our colleagues found that eDNA was comparable to our

250   dipnetting techniques, with both detecting salamanders in most ponds, but each failing to detect

251   salamanders in about the same number of occupied sites (Goldberg et al. 2018).

252   Under the ESA, reintroductions are being considered in areas where wild populations have been

253   extirpated to aid recovery of Reticulated Flatwoods Salamanders. In order to identify potential

254   reintroduction sites, intensive, long-term sampling is necessary to provide a high level of

255   confidence that lack of detections reflected true absence. Once new populations have been

256   established, multiple survey methods should be implemented to evaluate the success of

257   reintroduction efforts (He & Gaston, 2000; Manel, Williams & Ormerod, 2001; Royle &

258   Nichols, 2003; Nielson et al., 2005; Royle, Nichols & Kéry, 2005; Cosentino, 2014). Owing to

259   density dependence in the aquatic stage, larval abundance can seldom be used to infer population

260   dynamics (Shea, Wolf & Mangel, 2006; Scherer & Tracy, 2011). Whilst confirming the return

261   of mature individuals to a breeding site, the often weak relationship between larval presence and

262   population processes prevents a robust assessment of long-term viability from dip-net data alone

263   (Strayer, 1999; Gaston et al., 2000; Zhou & Griffiths, 2007; Conlisk et al., 2009; Korfel et al.,

264   2010; Bried & Pellet, 2012).

265   This paper does not consider cost as a limitation (Field, Tyre & Possingham, 2005; Moore et al.,

266   2014). It is important to note however, that the cost of larval sampling is orders of magnitude

267   less than the cost of monitoring adults through drift-fence sampling, because the latter requires

268   installation of infrastructure, nightly monitoring at least every 24 hours for an extended period of

269   time (or repeated removal and replacement of traps over weeks or months of sampling), and a

270   large enough crew to monitor all sites at the same time. The strategies of stopping surveys after

271   detection (Regan et al., 2006; Rout, Heinze & McCarthy, 2010; Guillera-Arroita, Hauser &

272   McCarthy, 2014) or determining occupancy through eDNA may reduce monitoring costs, but do

273   not readily translate to conservation applications when other management objectives (such as

274   assessing growth rates, obtaining genetic samples from a population, etc.) necessitate captures of

275   multiple individuals.

276

277   Conclusions

278   In conclusion, annual variability in environmental conditions can severely affect the results of

279   standardized long-term amphibian monitoring efforts. Ideally, multiple sampling methods would

280   be employed to discern population trends or distinguish between local extirpation and undetected

281   presence, but this is often cost-prohibitive. Given the longevity of ambystomatid salamanders,

282   coupled with their cryptic lifestyles, a high survey effort across multiple years is required to infer

283   absence, particularly when sites are being evaluated for future reintroductions. The

284   methodologies outlined here will aid other researchers in tailoring monitoring schemes to

285   overcome the confounding effects of environmental stochasticity.

286

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292   collection.

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Figure 1

Annual detection probabilities of larval flatwoods salamanders as a function of survey effort.

The solid line indicates the median detection probability for a single year of monitoring, up to 10 surveys per year. Surveys comprise 30 minutes of dipnetting effort, typically shared between multiple surveyors, concentrated in areas of herbaceous vegetation. The dashed lines represent bootstrapped quantiles (0.25-0.75) given annual fluctuations in detectability.
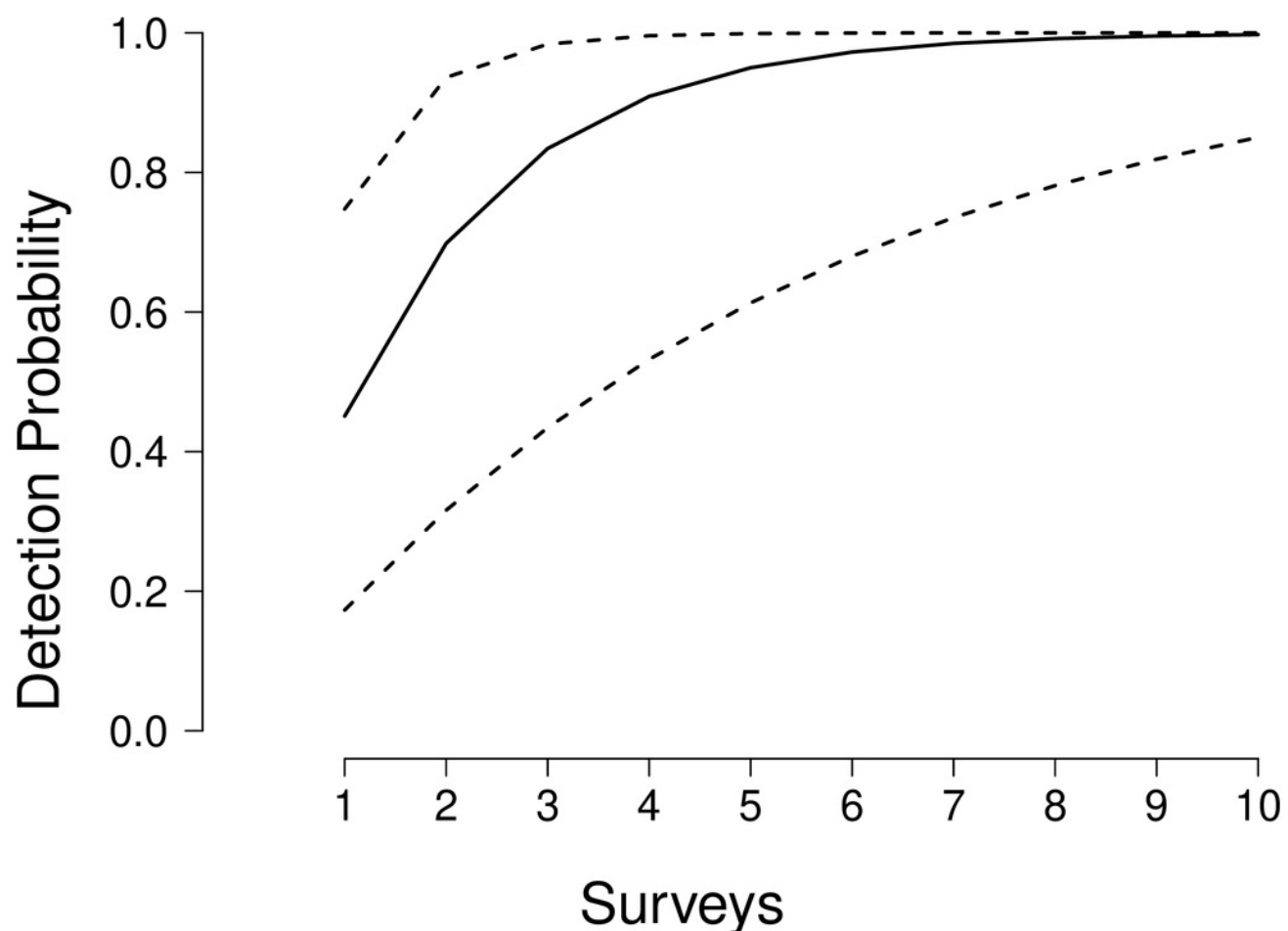
Figure 2

Cumulative detection probabilities of larval flatwoods salamanders as a function of time and survey effort.

Cumulative detection probabilities are presented for up to ten years of continuous monitoring. The four panels represent different levels of survey effort within each year (1, 2, 3, or 4 surveys per year). Surveys comprise 30 minutes of dipnetting effort, typically shared between multiple surveyors, concentrated in areas of herbaceous vegetation. The solid lines indicate median detection probabilities and the dashed lines reflect 90% quantiles (0.05-0.95) given annual fluctuations in detectability. The dot and whisker lines along the bottom of each panel indicate the mean number of years required to achieve a 99% detection probability and the associated uncertainty.