

Effects of an Extreme Flood Event on Federally Endangered Diamond Darter Abundances

AUSTIN A. RIZZO¹, CHRISTOPHER T. ROTA AND PATRICIA A. THOMPSON

Division of Forestry and Natural Resources, West Virginia University, 322 Percival Hall, Morgantown 26506

DONALD J. BROWN

Division of Forestry and Natural Resources, West Virginia University, 322 Percival Hall, Morgantown 26506;
U.S. Forest Service, Northern Research Station, PO Box 404, Parsons 26287

AND

STUART A. WELSH

U.S. Geological Survey, West Virginia Cooperative Fish and Wildlife Research Unit, PO Box 6125, Morgantown 26506

ABSTRACT.—Extreme flood events can substantially affect riverine systems, modifying in-stream habitat and influencing fish assemblages and densities. Rare species are especially vulnerable to these disturbance events because of their small population size and often reduced phenotypic heterogeneity. In June 2016 the lower Elk River in West Virginia experienced severe flooding, resulting in a peak discharge that exceeded the 0.005 annual exceedance probability (>200 y flood) in the main stem. We obtained pre-flood and post-flood population count data and estimated abundances for one cohort of the federally endangered Diamond Darter (*Crystallaria cincotta*) at 15 sites. While both the total count data and total estimated abundance decreased following the flood, our analyses did not indicate the extreme flood event strongly impacted Diamond Darter abundance. This indicates individuals are able to withstand high velocities and resist displacement or mortality. In addition site-level abundances were estimated at three sentinel sites during 2015 and 2016 using a multinomial *N*-mixture model that accounted for variation in detectability resulting from water temperature. Mean estimated abundance varied among the three sites and between the 2 y. Our results suggest there is substantial variation in year-class strength between the two cohorts we sampled. It is suggested that survey efforts at established sentinel sites be continued on an annual basis in order to help determine factors influencing year-class strength.

INTRODUCTION

Disturbance events can drastically affect ecological systems, disrupting the physical environment and associated ecosystem, community, and population structure of organisms inhabiting these areas (Resh *et al.*, 1988; Scheffer *et al.*, 2001). While disturbance events are natural in many systems, extreme disturbance events can have profound impacts. An extreme disturbance event is considered to be any discrete event that is characterized by frequency or magnitude outside of a predictable range that disrupts the structure of abiotic and biotic components of an ecosystem (Turner and Dale, 1998). In many ecosystems the organisms present have evolved under the constraints imposed by the frequency and magnitude of disturbance events (Williams, 1998; Lytle and Poff, 2004). Climate change research predicts disturbance events caused by weather extremes may increase in magnitude

¹ Corresponding Author: Telephone: (315) 276-6289; FAX: (304)-293-4826; e-mail: aarizzo@mix.wvu.edu

and frequency in some regions of the world (Aldous *et al.*, 2011; IPCC, 2013). Research suggests changes in the nature of disturbance regimes could push some ecosystems beyond a threshold of dynamic equilibrium (Scheffer *et al.*, 2001; Jentsch *et al.*, 2007; Death *et al.*, 2015).

Extreme flood events (*i.e.*, flows with a 1% annual exceedance probability [AEP]) can greatly modify instream habitat (Death *et al.*, 2015). The severity of such events on river geomorphology is largely contingent upon the nature of the stream channel form, magnitude of the flood, and the ability of floodwaters to become laterally expansive into floodplains and backwater areas (Lytle and Poff, 2004; Death *et al.*, 2015). Major flood events can influence fish assemblages and fish densities, although the exact nature of the relationship between a flood and these features varies (Aldous *et al.*, 2011). For example Collins *et al.* (1981) witnessed the elimination of an endangered fish population resulting from an extreme flood event in a desert stream, and Carlson *et al.* (2016) found numerous fish species exhibited greater relative abundance after a flood. Research has shown that while some severe floods can negatively affect a lotic ecosystem, others have been found to have negligible effects, or even result in long-term benefits to an ecosystem (Williams, 1998; Death *et al.*, 2015). In a study conducted by Death *et al.* (2015), researchers found some extreme floods negatively affect river systems by aiding in the dispersal of invasive species and mobilizing sediment and toxic chemicals; however, extreme floods also were shown to benefit some riverine biota by increasing habitat complexity and the floodplain area. A flood's effect on a fish population largely depends on the timing of the event relative to the ontogenetic stage of present cohorts, frequency and predictability of the event, magnitude and duration of the event, and the availability of refugia (Fausch *et al.*, 2001; Lytle and Poff, 2004; Death *et al.*, 2015; George *et al.*, 2015). Additionally, research has suggested aquatic systems already severely stressed by anthropogenic factors (*e.g.*, high nutrient and contaminant loads, increased sedimentation, invasive species) may be more prone to negative effects (*e.g.*, decreased diversity and abundance) by severe flood events (Staudt *et al.*, 2013; Death *et al.*, 2015).

Rare species are especially vulnerable to stochastic disturbance events because of their small population size and often reduced phenotypic heterogeneity (Caughley, 1994; Hellmair and Kinziger, 2014). Research has shown rare species are at an elevated risk of extinction when disturbance events occur that are outside of the evolutionary experience of that species, especially when these disturbance events occur at a frequency that outpaces the adaptive capabilities of the species (Williams, 1998; Brook *et al.*, 2008). For many species that have become extinct, the final quietus is often unrelated to those processes that initially triggered their population decline. Instead, the cause of extinction is often associated with stochastic events, such as genetic, demographic, catastrophic, and/or environmental events (Lande, 1993; Brook *et al.*, 2008).

The Diamond Darter is federally endangered and known to occur within the lower 50 river kilometers (rkm) of the Elk River in West Virginia. Museum specimens indicate the Diamond Darter was once distributed throughout the Ohio River Basin, but it is now likely extirpated from the Muskingum River, the Ohio River, the Green River, and the Cumberland River drainage (Welsh and Wood, 2008; Welsh *et al.*, 2009; Welsh *et al.*, 2014). The Diamond Darter obtained its endangered status because of its small range and population size, and continued anthropogenic threats to its habitat (USFWS, 2013).

In this study we investigated changes in Diamond Darter relative abundance after a high magnitude flood event in the Elk River, West Virginia. Diamond Darter surveys were conducted at 15 sites within the lower 50 rkm of the Elk River in 2016, to evaluate

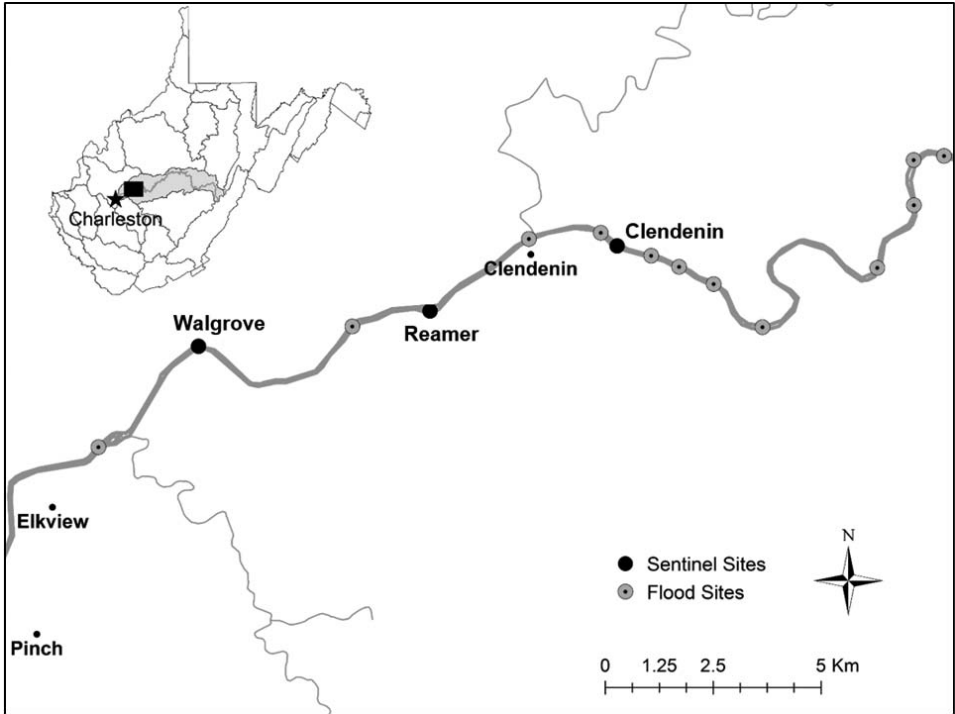


FIG. 1.—Map showing locations of 15 Diamond Darter *Crystallaria cincotta* study sites on the Elk River, West Virginia. The sites were used to investigate the potential influence of an extreme flood event on abundance (flood sites + sentinel sites), and between-year changes in abundance (sentinel sites). Sentinel site names (Clendenin, Reamer, and Walgrove) are included in the map

distribution and abundances of this species within the river. In addition we assessed changes in mean estimated abundances at three sentinel sites between the 2015 and 2016 sampling seasons. Collectively, these analyses provide useful insights into the potential short-term effects of major flood events on Diamond Darter abundance and provide valuable information that will be useful for future population monitoring efforts.

METHODS

STUDY AREA

The Elk River runs through the Appalachian Plateau in central West Virginia, following a 290 rkm course from its headwaters in Pocahontas County to its confluence with the Kanawha River near the city of Charleston (Fig. 1). The 4000 km² watershed is 90% forested, river geology largely consisting of sandstone and shale bedrock (Strager, 2008). The Elk River is impounded at Sutton Dam (6 km² reservoir), which is about 190 rkm upstream of the mouth of the river. Below the dam, the river is characterized as low gradient, with long deep pools separated by short riffle areas (Welsh *et al.*, 2013). The Elk River is currently on West Virginia's Clean Water Act section 303(d) list because of impaired waters, due largely to violations of fecal coliform levels and dissolved metals (Strager, 2008). In addition the Elk

River has experienced a number of anthropogenic disturbance events in the last decade (Bahadur and Samuels, 2014; USCSB, 2016). While the Elk River has experienced a number of different stressors, it is still regarded as being one of the most diverse rivers in West Virginia, with approximately 100 fish species and 30 mussel species. In addition the Elk River has scored high on the West Virginia Stream Condition Index (WVSCI), which uses benthic macroinvertebrate metrics (WVDEP, 1997). This high benthic macroinvertebrate score indicates the water quality within the Elk River is clean enough to support a diverse and abundant community of benthos, which is generally a reliable indicator of aquatic ecosystem health (WVDEP, 1997). Our study area includes the lower 50 rkm of the Elk River where the Diamond Darter is known to occur (Fig. 1).

Approximately 1 wk after our initial searches (late June 2016), the region experienced severe flooding from a series of thunderstorms that resulted in a rainfall recurrence interval of less than once every 1000 yr in several areas of the Elk River watershed (Di Liberto, 2016). The peak flow at the Queen Shoals U.S. Geological Survey (USGS) stream gauge, immediately upstream of our study sites, was between 0.005 and 0.002 annual exceedance probability (AEP), which corresponded to a flood recurrence interval of 200 and 500 yr, respectively (Wiley and Atkins, 2010).

DATA COLLECTION

Sites were surveyed using a search method that employs the use of spotlighting at night with flashlights within wadeable sections of the river (Welsh *et al.*, 2013). This method has proven to be effective only in glide habitats (areas immediately upstream of riffles). Glides are shallow enough to wade transects and have a smooth water surface, which allows the spotlighter to see through the water column to the substrate (Welsh *et al.*, 2013). As a result of water level fluctuations in the river, the number of transects required to search a site varied by sampling night. Consequently, there was no standard number of transects sampled per sampling occasion. Counts of Diamond Darters were summed across all transects at a site to get a total count during a sampling occasion (Rizzo *et al.*, 2017a).

Fifteen known-occupied glide areas (sites) were sampled (Fig. 1; Table 1) to investigate single-year pre-flood vs. post-flood relative abundance. Sampling events included three searches conducted during 2016; one in mid-June (prior to the flood), one in July (after the water levels and turbidity had returned to pre-flood levels), and one in September (to determine if any lag-effects of the flood occurred throughout the season).

We used three of the 15 sites, which were each sampled 15 times between the months of May–October in 2015 and 2016 to investigate multi-year changes in abundance. Periods of high river discharge or high water turbidity prevented sampling during the late fall and early spring, and periodically throughout the sampling season of both years. Only surveys conducted following the flood event were included in the analyses for the 2016 sampling season. This was done to ensure that the 2016 sampling season could be considered a ‘closed-season’ where no mass migration or mortality occurred (Donovan and Hines, 2007). Water temperature was found to influence individual Diamond Darter detectability in a previous study conducted in 2015 (Rizzo *et al.*, 2017a). Therefore, water temperature was recorded during each sampling event to use as a detection covariate.

DATA ANALYSES

Pre-flood vs. post-flood relative abundance.—A paired randomization test with 10,000 iterations was used to determine if raw count numbers at each site differed between pre- and post-

TABLE 1.—Study Site Coordinates

| Site number | Site name | x | y | Sentinel site |
|-------------|------------------|-----------|------------|---------------|
| 1 | Northern | 38.50684 | -81.2404 | |
| 2 | Sign | 38.505981 | -81.248472 | |
| 3 | Stuart | 38.496545 | -81.248359 | |
| 4 | Porter Creek | 38.483412 | -81.258078 | |
| 5 | Queen Shoals | 38.47096 | -81.28854 | |
| 6 | Church | 38.47993 | -81.30156 | |
| 7 | Morris Creek | 38.483503 | -81.310782 | |
| 8 | Gross's | 38.485797 | -81.318197 | |
| 9 | Clendenin | 38.487817 | -81.327258 | X |
| 10 | Carlos's Landing | 38.49057 | -81.33163 | |
| 11 | Bob Evans | 38.489228 | -81.350723 | |
| 12 | Reamer | 38.474105 | -81.376919 | X |
| 13 | Gristmill Lane | 38.470799 | -81.397556 | |
| 15 | Walgrove | 38.466494 | -81.438404 | X |
| 16 | BoyScout Camp | 38.445335 | -81.464889 | |

flood searches at the 15 sites. The P-value obtained from this test represents the proportion of trials resulting in a count difference between sampling events as great or greater than the one obtained (Sokal and Rohlf, 1995). This test was performed using count data corrected for incomplete detection based on the relationship between water temperature and detection probability (p). Detection probability is the probability of detecting an individual organism during a sampling event (Royle, 2004). Detection probabilities were determined from replicated count surveys conducted in 2015 and 2016, and therefore represent a modification to the p function in Rizzo *et al.* (2017a). Count data were corrected by dividing the raw count of Diamond Darters found at a site on a particular night by the p associated with the known water temperature value recorded during that sampling event.

Multi-year changes in abundance.— N -mixture models (Royle, 2004) were used to estimate site-level abundances at our three sentinel sites, which we implemented in the ‘unmarked’ (Fiske and Chandler, 2011) and ‘AICcmovg’ (Mazerolle, 2016) packages in program R (R Developmental Core Team, 2017). N -mixture models are a class of hierarchical models that use spatially and temporally replicated count data to model abundance (N) and p . Abundance at a site is typically considered a latent variable because of imperfect detection and therefore must be modeled using count data and p (Royle, 2004; Fiske and Chandler, 2011). As a result of variable temporal conditions (*e.g.*, time of day, weather conditions) and/or site-specific conditions (*e.g.*, type of habitat), there is often extrinsic heterogeneity found in individual detection probabilities among sampling events. N -mixture models can account for this heterogeneity by including relevant p covariates (Veech *et al.*, 2016).

We used single-season N -mixture models to estimate N each year (Kéry and Royle, 2016). Because of our limited dataset, we chose to treat each site-year combination as a separate site, therefore increasing our sample size ($n = 6$) for model fitting. This ‘stacked’ site-year analysis, using a single-season model, has been shown to be a more effective method than a multi-season model when using N -mixture modeling on smaller datasets (Yamaura *et al.*, 2011). By structuring our data in this site-year ‘stacked’ format, it essentially creates an open model in which N [site,year] is completely independent of N [site, year + 1]. Demographic closure was assumed for each ‘stacked’ site based on analysis of age class size structure throughout the season (Rizzo *et al.*, 2017b). Based on a previous study conducted by Rizzo *et*

TABLE 2.—Four candidate models for explaining variation in abundance and detection probability during surveys of the endangered Diamond Darter *Crystallaria cincotta*. For each model, the Quasi-Akaike Information Criterion adjusted for sample size (QAIC_c), the delta QAIC_c, QAIC_c weight (w_i), the number of parameters (K), and the maximum log likelihood ($-2 \log(L)$) are given. The abundance variables evaluated include SITE, YEAR, and SITE + YEAR

| Covariates | | K | QAIC _c | Δ QAIC _c | w_i | $-2 \log(L)$ |
|-------------|-------------------------------|---|-------------------|----------------------------|-------|--------------|
| Abundance | Detection | | | | | |
| SITE + YEAR | (WTEMP + WTEMP ²) | 7 | 511.6 | 0.00 | 1.00 | -276.778 |
| SITE | (WTEMP + WTEMP ²) | 6 | 528.3 | 16.75 | 0.00 | -300.152 |
| ~ | (WTEMP + WTEMP ²) | 4 | 719.5 | 207.91 | 0.00 | -335.734 |
| YEAR | (WTEMP + WTEMP ²) | 5 | Inf | Inf | 0.00 | -310.020 |

al. (2017b), researchers determined that only one age class of Diamond Darters was being sampled using the spotlight sampling technique in glide habitat. Additionally, adjusted abundance estimates (*i.e.*, counts/ p) were similar throughout the sampling season, lending support to our assumption that this age class within our populations was closed.

A total of four *a priori* models were constructed to examine among site and between year variation (Table 2). We included water temperature as a p covariate in all four models (Rizzo, 2017). The number of models included was partially restricted by the small sample size. Model complexity was minimized due to small sample size. A Poisson distribution was chosen for the state process after comparing Akaike Information Criterion (AIC_c) values and residual diagnostic plots for three distributions (*i.e.*, Poisson, zero-inflated Poisson, and negative binomial; Kéry and Royle, 2016).

Once the error distribution of the latent abundance process was determined (*e.g.*, Poisson, Zero-inflated Poisson, Negative Binomial), candidate models were ranked according to QAIC_c, corrected for small sample size. The model with the lowest QAIC_c was considered to be the most parsimonious among the collection of candidate models and all models within seven QAIC_c values of the minimum were considered to have some support (Burnham and Anderson, 2002). Abundances were estimated from the best-fitting model using model coefficients. Model goodness-of-fit was evaluated using a parametric bootstrap of the Pearson chi-square statistic (Mazerolle, 2016). The goodness-of-fit test indicated the data were overdispersed and we accounted for this by inflating the estimated standard errors and 95% confidence intervals (CI) based on the \hat{c} value (*i.e.*, 3.61), using the function ‘modavgPred’ in the software package ‘AICcmodavg’ (Mazerolle, 2016). Confidence intervals (CI) were estimated by first multiplying the variance/covariance matrix by the \hat{c} value, extracting the adjusted standard deviation, and then taking each point estimate $\pm 1.96 \times$ standard error. The importance of abundance and detection variables of the best model was evaluated by examining 95% confidence intervals.

RESULTS

N-mixture model.—A model including site + year as abundance covariates had substantial support (QAIC_c = 511.6, w_i = 1.00; Table 2). Water temperatures near 22 C resulted in the highest detection probability (Fig. 2). Detection probability when surveying at the optimal temperature was approximately 13.5% greater than when surveying at 14 C and 30 C.

Pre-flood vs. post-flood relative abundance.—We documented 36 Diamond Darters during pre-flood searches and 21 Diamond Darters during post-flood searches at our 15 study sites.

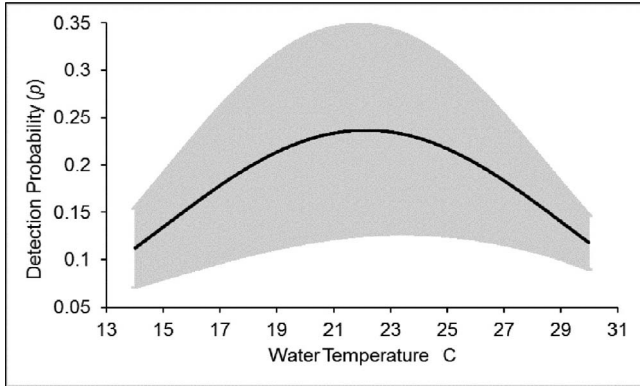


FIG. 2.—Relationship between water temperature and per-individual detection probability (p) of the endangered Diamond Darter *Crystallaria cincotta* based on 90 surveys conducted at 3 sites in the Elk River, West Virginia during 2015 and 2016. The gray band encompasses the 95% confidence interval

Uncorrected count data did not differ between pre-flood and post-flood searches ($P=0.296$). Correcting count data based on p indicated total pre-flood abundance was 250 (95% CI = 175.0–435.5), and total post-flood abundance was 190 (95% CI = 131.4–339.9).

Multi-year changes in abundance.—Diamond Darter estimated abundances varied between 2015 and 2016 sampling seasons. We detected Diamond Darters during 74 of the 90 total surveys in 2015 and 2016, resulting in a naive species detection probability of 0.82. Counts at the three study sites ranged from 0–42 Diamond Darters detected during a single survey event in 2015, and 0–8 Diamond Darters detected during a single survey event in 2016. Mean abundances estimates were substantially lower at all sites in 2016, but 95% CI's slightly overlapped (Table 2). Predicted abundance varied more among sites than it did between years for a given site. The Clendenin site had the highest abundance during both years, followed by the Reamer site, and then the Walgrove site (Table 3).

DISCUSSION

Our study indicates there was not a substantial short-term negative response in Diamond Darter abundance to an extreme flood event in the Elk River, West Virginia. Previous research on this species suggests that only one age class of Diamond Darters is being detected in glide habitat of the river (Rizzo *et al.*, 2017b). Consequently, our analysis regarding the short-term effects of the flood on this species is restricted to one cohort of individuals. Although our study is limited in scope, our data provide evidence that individuals are able to withstand high velocities and resist displacement or mortality. Our results are consistent with other studies that have shown many riverine fish species are well adapted to large flood events and are able to resist short-term direct effects of these events (George *et al.*, 2015; Carlson *et al.*, 2016). For many rare species, these types of severe disturbances or unusual events can prove catastrophic (Collins *et al.*, 1981). While it is unclear how other ontogenetic stages of the Diamond Darter were affected by this event, this study does give hope to the resilience of this rare species.

Although we did not find an effect of the flood event on abundance, we did find large variability in estimated abundances between the 2 y at our three sentinel sites. Because our analysis is restricted to a single cohort of individuals, our results suggest that there are

TABLE 3.—Model coefficients and Standard Error values for our best-fitting model. Abundance coefficients are on the log-scale and detection coefficients are on the logit-scale. Predicted abundances and 95% confidence intervals between the 2015 and 2016 sampling seasons at all three sites were also include based on our best-fitting model

| Abundance covariates | Detection covariates | |
|-------------------------------|-------------------------------|-----------------|
| Site + year | (WTEMP + WTEMP ²) | |
| | Estimate | SE |
| Clendenin_Y2015 (Intercept) | 4.121 | 0.357 |
| Reamer_Y2015 | -0.189 | 0.991 |
| Walgrove_Y2015 | -1.666 | 0.325 |
| Y2016 | -1.257 | 0.221 |
| Detection Model (logit-scale) | Estimate | SE |
| (Intercept) | -8.484 | 0.958 |
| WTEMP | 0.669 | 0.116 |
| WTEMP ² | -0.0152 | 0.003 |
| Site | 2015 λ | 2016 λ |
| Clendenin | 61.6 [16.3–233.0] | 17.5 [4.2–72.4] |
| Reamer | 51.0 [13.2–196.4] | 14.5 [3.5–60.7] |
| Walgrove | 11.6 [2.9–47.3] | 3.3 [0.7–14.9] |

substantial variations in year-class strength, at least between the age classes sampled during 2015 and 2016. There are a number of reasons why year-class strength of a population of fish might vary substantially among years, including both biotic and abiotic factors (Freeman *et al.*, 1988; Cowx and Frear, 2004). High mortality, resulting from severe disturbance events that occur during vulnerable ontogenetic stages (*e.g.*, egg, larval) of a species, can lead to low recruitment and subsequent poor year-class strength. While disturbance events can result in these fluctuations, many fish populations experience variation in year-class strength as a result of environmental factors unassociated with disturbance events (Freeman *et al.*, 1998; Cowx and Frear, 2004; Koonce *et al.*, 2011). Research conducted at larger spatial scales on a number of fish species indicates synchrony in year-class strength tends to occur among populations (Cattaneo *et al.*, 2003; Bunnell *et al.*, 2010). Our data, although limited, were consistent with these findings, showing a synchronous decline in Diamond Darter abundances during the 2016 season. Variation in year-class strength in many species of fish is fairly common (Matthews, 1998). While this variation is generally inconsequential to the persistence of most taxa, species with short life spans may experience local population extirpations when there are successive years of poor year-class strength.

Consistent with Rizzo *et al.* (2017a), we found water temperature was a strong predictor of detection probability. The relationship between water temperature and detection probability is likely related to the effects of temperature on metabolic rate and subsequent feeding activity; ultimately influencing the availability of darters to be detected (Rizzo *et al.*, 2017a). We are uncertain whether these fish are emerging from under the sandy glide substrate during ideal conditions or whether they are migrating into glide habitat from deeper adjacent pool habitat. Therefore, additional research is necessary to determine where these fish seek refuge during less optimal conditions.

Our visual encounter survey design using the spotlighting technique, coupled with abundance estimation using *N*-mixture modeling, resulted in abundance estimates that appear reasonable for this rare species of darter. In addition Rizzo (2017) used simulation

experiments to validate that N -mixture models are capable of accurate abundance estimation based on the count and p distribution in this study system (Rizzo, 2017). We found large variability in mean estimated abundance among the three study sites during both years. Although the study was not designed to quantify predictors of abundance, one potential explanation is variation in microhabitat conditions at glide locations. Welsh *et al.* (2013) found Diamond Darters appeared to be associated with substrate that was primarily composed of sand, which is consistent with the higher abundances that we found at our sandier glide sites (*i.e.*, Clendenin and Reamer). Future research is needed to explicitly investigate factors (*e.g.*, substrate type, glide size, etc.) that influence Diamond Darter abundance.

Gathering information on state variables and related parameters associated with rare species is a very difficult task; however, information of this nature is also most critical for the successful conservation of these species. In recent years there have been increased efforts to gather more information on rare species in an attempt to better protect them (MacKenzie *et al.*, 2005). This effort is largely possible because of advancements in statistical analyses and study design, allowing researchers to account for detection issues. Statistical techniques like N -mixture modeling have proven to be a powerful tool for abundance and detection modeling and have helped provide important information on these poorly understood species.

The nature of the Diamond Darter and the Elk River system have limited the ability of researchers to properly sample and therefore understand the life history of this endangered fish. However, N -mixture modeling has enabled us to determine factors influencing detection and subsequently estimate abundances, giving a brief glimpse of this fish's life history characteristics. While our results indicate that Diamond Darter abundances of individuals using glide habitat were not strongly affected by this extreme disturbance event, there is still much that is unknown about this fish or what environmental factors might affect abundances. Furthermore, it is still unknown if this flood event affected other ontogenetic stages of the Diamond Darter, resulting in a delayed affect years later. Although a difficult task, this information can only be obtained by continued monitoring efforts. It is suggested survey efforts at established sentinel sites be continued on an annual basis in order to help determine factors influencing year-class strength.

Acknowledgments.—The authors wish to thank NiSource and U.S. Fish and Wildlife Service for funding. We thank J. Aldinger, B. Crabill, K. Lambert, and R. Raesly for field assistance. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government. This study was performed under the auspices of West Virginia University IACUC protocol 12-0205.

LITERATURE CITED

- ALDOUS, A., J. FITZSIMONS, B. RICHTER, AND L. BACH. 2011. Droughts, floods and freshwater ecosystems: evaluating climate change impacts and developing adaptation strategies. *Mar. Freshwater Res.*, **62**:223–231.
- BAHADUR, R. AND W. B. SAMUELS. 2014. Modeling the fate and transport of a chemical spill in the Elk River, West Virginia. *J. Environ. Eng.*, **141**. [http://dx.doi.org/10.1061/\(ASCE\)EE.1943-7870.0000930](http://dx.doi.org/10.1061/(ASCE)EE.1943-7870.0000930)
- BROOK, B. W., N. S. SODHI, AND C. J. A. BRADSHAW. 2008. Synergies among extinction drivers under global change. *Trends Ecol. Evol.*, **23**:453–460.
- BUNNELL, D. B., J. V. ADAMS, O. T. GORMAN, C. P. MADENJIAN, S. C. RILEY, E. F. ROSEMAN, AND J. S. SCHAEFFER. 2010. Population synchrony of a native fish across three Laurentian Great Lakes: Evaluating the effects of dispersal and climate. *Oecologia*, **162**:641–651.
- BURNHAM, K. P. AND D. R. ANDERSON. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer-Verlag New York, Inc., New York, New York, USA.

- CARLSON, A. K., M. J. FINCEL, C. M. LONGHENRY, AND B. S. GRAEB. 2016. Effects of historic flooding on fishes and aquatic habitats in a Missouri River delta. *Journal of Freshwater Ecology*, **31**:271–288.
- CATTANEO, F., B. HUGUENY, AND N. LAMOUROUX. 2003. Synchrony in brown trout, *salmo trutta*, population dynamics: a ‘Moran effect’ on early-life stages. *Oikos*, **100**:43–54.
- CAUGHLEY, G. 1994. Directions in Conservation Biology. *J. Anim. Ecol.*, **63**:215–244.
- COLLINS, J. P., C. YOUNG, J. HOWELL, AND W. L. MINCKLEY. 1981. Impact of flooding in a Sonoran desert stream, including elimination of an endangered fish population (*Poeciliopsis o. occidentalis*, Poeciliidae). *Southwest. Nat.*, **26**:415–423.
- COWX, I. G. AND P. A. FREAR. 2004. Assessment of year class strength in freshwater recreational fish populations. *Fisheries Manag. Ecol.*, **11**:117–123.
- DEATH, R. G., I. C. FULLER, AND M. G. MACKLIN. 2015. Resetting the river template: the potential for climate-related extreme floods to transform river geomorphology and ecology. *Freshwater Biol.*, **60**:2477–2496.
- DI LIBERTO AND T. NOAA. “‘Thousand-year’ downpour led to deadly West Virginia floods”. Climate.gov, 8 July 2016. Web. 15 Feb. 2017. <https://www.climate.gov/news-features/event-tracker/thousand-year-downpour-led-deadly-west-virginia-floods>.
- DONOVAN, T. M. AND J. HINES. 2007. Exercises in occupancy modeling and estimation. <http://www.uvm.edu/envnr/vtcfwtu/spreadsheets/occupancy.htm>
- ETNIER, D. A. AND W. C. STARNES. 1993. *The Fishes of Tennessee*. The Univ. of Tennessee Press, Knoxville.
- FISKE, I. J. AND R. B. CHANDLER. 2011. Unmarked: An R Package for fitting hierarchical models of wildlife occurrence and abundance. *J. Stat. Softw.*, **43**:1–23.
- FREEMAN, M. C., M. K. CRAWFORD, J. C. BATTETT, D. E. FACEY, M. G. FLOOD, J. HILL, D. J. STOUDE, AND G. D. GROSSMAN. 1988. Fish assemblage stability in a southern Appalachian stream. *Can. J. Fish. Aquat. Sci.*, **45**:1949–1958.
- GEORGE, S. D., B. P. BALDIGO, A. J. SMITH, AND G. R. ROBINSON. 2015. Effects of extreme floods on trout populations and fish communities in a Catskill Mountain river. *Freshwater Biol.*, **60**:2511–2522.
- GEORGE, S. G., W. T. SLACK, AND N. H. DOUGLAS. 1996. Demography, habitat, reproduction, and sexual dimorphism of the Crystal Darter, *Crystallaria asprella* (Jordan), from south-central Arkansas. *Copeia*, **1996**:68–78.
- HELLMAIR, M. AND A. P. KINZIGER. 2014. Increased extinction potential of insular fish populations with reduced life history variation and low genetic diversity. *PLOS ONE*, **9**:e113139.
- IPCC (INTERGOVERNMENTAL PANEL ON CLIMATE CHANGE). 2013. *Climate change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report on Climate Change*. Cambridge, UK: Cambridge University Press.
- JENTSCH, A., J. KREYLING, AND C. BEIERKUHNLIN. 2007. A new generation of climate-change experiments: events, not trends. *Front. Ecol. Environ.*, **5**:365–374.
- KÉRY, M. AND J. A. ROYLE. 2016. *Applied Hierarchical Modeling in Ecology: Analysis of distribution, abundance and species richness in R and BUGS, Volume 1*. Academic Press/Elsevier Inc, San Diego, USA.
- LANDE, R. 1993. Risks of population extinction from demographic and environmental stochasticity and random catastrophes. *Am. Nat.*, **142**:911–927.
- LYTLE, D. A. AND N. L. POFF. 2004. Adaptations to natural flow regimes. *Trends Ecol. Evol.* **19**:94–100.
- MACKENZIE, D. I., J. D. NICHOLS, N. SUTTON, K. KAWANISHI, AND L. L. BAILEY. 2005. Improving inferences in population studies of rare species that are detected imperfectly. *Ecol.* **86**:1101–1113.
- MATTHEWS, W. J. 1998. *Patterns in freshwater fish ecology*. Chapman and Hall, New York, NY.
- MAZEROLLE, M. J. 2016. Package ‘AICcmoDavg’: model selection and multimodel inference based on (Q)AIC(c). R package version 2.0-4. <http://cran.r-project.org/web/packages/AICcmoDavg>
- R DEVELOPMENT CORE TEAM. 2017. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL <http://www.R-project.org>.
- RIZZO, A. 2017. *Development of techniques for assessment of population characteristics of the federally endangered Diamond Darter (PhD Dissertation)*. Retrieved from ProQuest Dissertations and Theses database. (UMI No. 10602023).

- RIZZO, A. A., D. J. BROWN, S. A. WELSH, AND P. A. THOMPSON. 2017a. Factors influencing detection of the federally endangered Diamond Darter *Crystallaria cincotta*: Implications for long-term monitoring strategies. *Am. Midl. Nat.*, **16**:051R2.
- RIZZO, A. A., S. A. WELSH, AND P. A. THOMPSON. 2017b. A paired-laser photogrammetric method for in situ length measurements of benthic fishes. *N. Am. J. Fish. Manage.*, **37**:16–22.
- ROYLE, J. A. 2004. *N*-Mixture models for estimating population size from spatially replicated counts. *Biom. J.*, **60**:108–115.
- SCHAEFFER, M., S. CARPENTER, J. A. FOLEY, C. FOLKE, AND B. WALKER. 2001. Catastrophic shifts in ecosystems. *Nature*, **413**:591–596.
- STAUDT, A., A. K. LEIDNER, J. HOWARD, K. A. BRAUMAN, J. S. DUKES, L. J. HANSEN, C. PAUKERT, J. SABO, AND L. A. SOLORZANO. 2013. The added complications of climate change: understanding and managing biodiversity and ecosystems. *Front. Ecol. Environ.*, **11**:494–501.
- STRAGER, J. M. 2008. Diamond Darter *Crystallaria cincotta* status review – threats assessment data development. Final Report Prepared for: U.S. Fish and Wildlife Service, West Virginia Field Office, Elkins, WV.
- TURNER, M. G., AND V. H. DALE. 1998. Comparing large, infrequent disturbances: What have we learned? *Ecosystems*, **1**:493–496.
- USFWS (U.S. FISH AND WILDLIFE SERVICE). 2013. Endangered and threatened wildlife and plants; endangered species status for the Diamond Darter. Federal Register, **78**:45074–45095.
- USCSB (U.S. CHEMICAL SAFETY AND HAZARD INVESTIGATION BOARD). 2016. Investigation Report: Chemical spill contaminates public water supply in Charleston, West Virginia. Report No. 2014-01-I-WV.
- VEECH, J. A., J. R. OTT, AND J. R. TROY. 2016. Intrinsic heterogeneity in detection probability and its effect on *N*-mixture models. *Methods Ecol. Evol.*, **7**:1019–1028.
- WELSH, S. A. AND R. M. WOOD. 2008. *Crystallaria cincotta*, a new species of darter (Teleostei: Percidae) from the Elk River of the Ohio River drainage, West Virginia. *Zootaxa*, **1680**:62–68.
- WELSH, S. A., R. M. WOOD, AND K. R. SHEEHAN. 2009. Threatened fishes of the world: *Crystallaria cincotta* Welsh and Wood 2008 (Percidae). *Environ. Biol. Fish.*, **84**:191–192.
- WELSH, S. A., D. M. SMITH, AND N. D. TAYLOR. 2013. Microhabitat use of the Diamond Darter. *Ecol. Freshw. Fish.*, **22**:587–595.
- WENGER, S. J. AND M. C. FREEMAN. 2008. Estimating species occurrence, abundance, and detection probability using zero-inflated distributions. *Ecology*, **89**:2953–2959.
- WILEY, J.B., AND J. T. ATKINS JR. 2010 Estimation of flood-frequency discharges for rural, unregulated streams in West Virginia: U.S. Geological Survey Scientific Investigations Report 2010–5033, 78 p.
- WILLIAMS, M. J. 1998. Patterns in Freshwater Fish Ecology. Chapman & Hall, New York, NY.
- WVDEP (WEST VIRGINIA DEPARTMENT OF ENVIRONMENTAL PROTECTION). 1997. An ecological assessment of the Elk River Watershed, West Virginia. Charleston, WV.
- YAMAURA, Y., J. A. ROYLE, K. KUBOI, T. TADA, S. IKENO, AND S. MAKINO. 2011. Modeling community dynamics based on species-level abundance models from detection/nondetection data. *J. Appl. Ecol.*, **48**:67–75.