

## RESEARCH ARTICLE

***Rallus elegans* (King Rail) occupancy is stable, but habitat is in short supply in the Arkansas Delta**

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**ABSTRACT**

Secretive marsh birds experienced precipitous declines due to losses in wetland habitat across North America. However, from 1998 to 2004, wetland area increased, and between 2009 and 2019, the extent of emergent wetlands did not significantly decrease. This raises the question: how did secretive marsh birds, which are tied to emergent wetlands, respond to this period of relative stability in wetland area? Here, we use *Rallus elegans* (King Rail) occurrence data collected in 2005, 2006, 2012, and 2017 in the Arkansas Delta in the United States to test the hypothesis that a period of stability in wetland vegetation is linked to stable *Rallus elegans* occupancy. Specifically, we sought to (1) quantify relationships between *R. elegans* occupancy and land cover, (2) quantify temporal trends in *R. elegans* occupancy, and (3) characterize changes in land cover types linked to *R. elegans* occupancy. We developed spatial multi-season occupancy models, and our top model contained the categorical year effect but showed no trend in *R. elegans* occupancy between 2005 and 2017, instead showing high interannual variation in occupancy. We found strong associations between *R. elegans* occupancy and emergent wetlands (positive) and elevation (negative). From 2005 to 2017 in the ~3.7 million ha Arkansas Delta, rice crop cover decreased by 294,750 ha, emergent wetlands increased by 6,719 ha, and all other cover types increased or decreased by <7,500 ha. Thus, although emergent wetlands did increase, the total area of emergent wetlands in 2016 (22,262 ha) comprised <1% of the Arkansas Delta. Our results support the hypothesis that *R. elegans* occupancy stabilized during a time when the

main habitat type *R. elegans* depend on, emergent wetlands, also stabilized in spatial extent. There is an opportunity to turn the tide in *R. elegans* declines by conserving extant emergent wetlands and creating new emergent wetlands.

**Keywords:** Bayesian, emergent wetland, Mississippi Alluvial Valley, multi-season occupancy model, *Rallus elegans*, secretive marsh bird

### How to Cite

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### LAY SUMMARY

- Wetland habitats and the birds that rely on them have been declining across the globe.
- In the Mississippi Alluvial Valley region of Arkansas, USA, wetland habitats had a period of stability, but it was not known how bird populations responded to this.
- We used bird survey data from 2005, 2006, 2012 and 2017 to explore how a secretive marsh bird, *Rallus elegans* (King Rail), responded to this period of stability.
- *Rallus elegans* occupancy remained stable from 2005 to 2017, and *Rallus elegans* presence was strongly positively associated with emergent wetlands.
- We show the tide could turn for *Rallus elegans* conservation. By protecting or restoring wetland habitats that already exist, *Rallus elegans* populations may remain stable and could even increase.

### La ocupación de *Rallus elegans* es estable, pero el hábitat es escaso en el delta de Arkansas

### RESUMEN

Las aves de pantano esquivas han experimentado declives marcados debido a la pérdida de hábitat de humedales en toda Norteamérica. Sin embargo, de 1998 a 2004, el área de humedales aumentó, y entre 2009 y 2019 la extensión de humedales emergentes no disminuyó significativamente. Esto plantea la pregunta: ¿cómo respondieron las aves de pantano esquivas, que dependen de los humedales emergentes, a este período de relativa estabilidad en el área de humedales? Aquí utilizamos datos de ocurrencia de *Rallus elegans* recolectados en 2005, 2006, 2012 y 2017 en el delta de Arkansas, EEUU, para probar la hipótesis de que un período de estabilidad en la vegetación de humedales está vinculado a la estabilidad en la ocupación de esta especie. Específicamente, buscamos (1) cuantificar las relaciones entre la ocupación de *R. elegans* y la cobertura del suelo, (2) cuantificar las tendencias temporales en la ocupación de *R. elegans* y (3) caracterizar los cambios en los tipos de cobertura asociados a la ocupación de *R. elegans*. Desarrollamos modelos espaciales de ocupación multi-estacionales, y nuestro mejor modelo incluyó el efecto categórico de año, pero no mostró tendencia en la ocupación de *R. elegans* entre 2005 y 2017, sino una alta variación interanual en la ocupación. Encontramos asociaciones fuertes entre la ocupación de *R. elegans* y los humedales emergentes (positiva) y la altitud (negativa). De 2005 a 2017, en los ~3,7 millones de hectáreas del delta de Arkansas, la cobertura de cultivos de arroz disminuyó en 294.750 ha, los humedales emergentes aumentaron en 6.719 ha, y todos los demás tipos de cobertura aumentaron o disminuyeron en <7.500 ha. Así,

aunque los humedales emergentes aumentaron, el área total de humedales emergentes en 2016 (22.262 ha) representó <1% del delta de Arkansas. Nuestros resultados apoyan la hipótesis de que la ocupación de *R. elegans* se estabilizó durante un período en que el principal tipo de hábitat del que depende, los humedales emergentes, también se estabilizó en su extensión espacial. Existe una oportunidad de revertir el declive de *R. elegans* mediante la conservación de los humedales emergentes existentes y la creación de nuevos humedales emergentes.

**Palabras clave:** bayesiano, humedal emergente, valle aluvial del Misisipi, modelo de ocupación multi-estacional, *Rallus elegans*, ave de pantano esquivia

## INTRODUCTION

Most secretive marsh bird populations (e.g., Rallids, bitterns, night-herons and grebes) in North America are declining across their ranges, and one of the strongest drivers of these declines has historically been emergent wetland habitat loss (Malone et al. 2023). Wetlands are among the most threatened and rapidly declining habitats globally due to anthropogenic drivers of change, such as urbanization, conversion of land to agriculture, and increasing demands for freshwater consumption (USFWS 2024). Since the mid-20th century, wetlands in North America have declined by at least 35% (Davidson 2014), and the vast majority of wetland habitat loss occurred between the 1950s and 1997. Given that secretive marsh bird species require emergent wetlands with robust herbaceous vegetation throughout their life cycle (Malone et al. 2023), concurrent secretive marsh bird declines are unsurprising (Rosenburg et al. 2019). However, wetland area slightly increased across North America in 1998–2004 (Dahl 2005), and although total wetland area decreased again from 2005 to 2019, freshwater emergent wetland area did not significantly change between 2009 and 2019 (USFWS 2024).

This period from 1998 to 2019, in which emergent wetland area remained stable, may have contributed to stabilizing secretive marsh bird populations in some locations. There is evidence to support this hypothesis: where wetland area has increased, site occupancy and abundance of secretive marsh birds appear to rebound quickly (Hargrove and Osborne 2023). For example, 50% of secretive marsh bird species increased in abundance between 2011 and 2021 due to increased water levels and expanding wetlands in the Great Lakes region of the United States (Tozer et al. 2024). In contrast, there are also overarching trends showing declines in waterbird abundance (Rosenberg et al. 2019), and Stevens and Conway (2020) found negative trends in multiple secretive marsh bird species' occupancy probabilities between 1999 and 2012. Given these divergent responses, there is a need to understand the response of secretive marsh birds to the recent relative stability in emergent wetland area.

A prime example of declines in secretive marsh bird species is *Rallus elegans* (King Rail), a large member of the Rallidae family. They inhabit coastal and inland marshes of the United States, from the coast of the Gulf of Mexico, where they reside year-round, to their breeding grounds in estuarine habitats of the Great Lakes (Brewer et al 2023). They require wetland habitats with dense emergent vegetation and minimal woody cover for breeding, brooding, and protection from predation (Darrah and Krementz 2009; Brewer et al 2023). They are a hunted species in most of their range (in 13 states from Rhode Island along the coast southward and westward to Texas along the coast, except for in New York and New Jersey; Meanly 1969, Louisiana Department of Wildlife and Fisheries 2025, Virginia Department of Wildlife Resources 2025). The loss of suitable nesting and breeding habitat (Herring et al. 2021) has contributed to the overall population decline of *R. elegans* over the years (Budd and

Krementz 2011, Krementz 2017). Consequently, *R. elegans* is listed as a priority conservation species in nearly all the states in which it occurs (Bradshaw et al. 2020).

Given that populations of *R. elegans* depend on emergent wetlands, quantifying (1) trends in *R. elegans* populations and (2) changes in habitat required by *R. elegans* during a period of relative stability in wetland extents (1998–2019) could test the hypothesis that stability in wetland extent is linked to population stability of secretive marsh birds. While quantifying population trends, such as nest success, survival, or abundance would be ideal for testing this hypothesis, a more accessible data source is occupancy. Occupancy typically is positively related to abundance (Gaston 2000), and occupancy is often used as a proxy for abundance when studying rare or elusive species (Fuller et al. 2022). While habitat quality influences species abundance and occupancy (Mortelliti et al. 2010), habitat size or spatial extent also strongly predict occupancy and abundance (Griffen et al. 2008; Hodgson et al. 2011).

Here, we use *R. elegans* occurrence data and land use/land cover data spanning 2005–2017 in a former stronghold of *R. elegans* range, the Mississippi Alluvial Valley of Arkansas, USA (hereafter, “the Arkansas Delta”), to test the hypothesis that a period of relative stability in wetland habitat spatial extents is linked to stabilization of *R. elegans* occupancy. Specifically, we seek to (1) quantify relationships between *R. elegans* occupancy and land cover, (2) quantify temporal trends in *R. elegans* occupancy, and (3) characterize changes in land cover types linked to *R. elegans* occupancy. We will consider our hypothesis supported if the *R. elegans* temporal occupancy trend does not exhibit a decline between 2005 and 2017 and if land cover types that are strongly positively associated with *R. elegans* occupancy do not decline from 2005 to 2017.

## METHODS

### Study Area

We set this study in the Lower Mississippi Alluvial Valley ecoregion that lies within Arkansas, USA, and is ~3.7 million ha in extent (Omernick and Griffith 2014, Arkansas GIS Office 2024). The Mississippi River Valley bottomland habitats support one of the largest wetlands in the continent, providing migration corridors to 40% of the continent’s waterfowl and other important bird species (Lehnen and Krementz 2005, Herbert et al. 2021). Historically, much of the ecoregion consisted of mosaics of emergent wetlands and bottomland hardwoods (Winters 1939). Much of the ecoregion in Arkansas was converted to rice (*Oryza sativa*) farms with the arrival of European colonizers (Dethloff 1970). Rice continues to be a major crop, but corn (*Zea mays*) and soybeans (*Glycine max*) are also common (rice, corn, and soybean harvest in 2022 was 455,621 ha, 293,073 ha, and 1,316,686 ha, respectively; USDA NASS 2024). Hereafter, we refer to our study area as the “Arkansas Delta.”

### Data Collection

#### [LEVEL HEADING 3] *Rallus elegans* occurrence data

We collated *R. elegans* occurrence data collected in 2005, 2006, 2012, and 2017 from studies performed by Budd and Krementz (2011), Budd and Rowe (2013), and Krementz (2017) (Figure 1). Data were collected using the Standardized North American Marsh Bird Monitoring Protocol (Conway 2011). Surveys were conducted starting in April each year and continued until the end of June or early July. At each survey location, the observer began with a 5-min quiet period when vocalizing birds were detected and their presence/absence were recorded during five 1-min

intervals. Once complete, surveyors broadcasted 30 s of pre-recorded audio of breeding vocalizations and calls for *R. elegans*, followed by a 30-s period of silence. Subsequent 30-s periods of vocalization followed by 30-s periods of silence were then repeated to survey for other secretive marsh birds. We recorded all other species found in the wetlands using the call sequence described by Conway (2011) from the least to most intrusive species. We followed this sequence to minimize the chances that *R. elegans* calls will draw in other co-species thereby affecting their probability of detection. Portable sound devices were used to broadcast vocalizations at 80–90 dB 1 m from the devices. Survey locations were spaced  $\geq 200$  m apart to avoid double-counting individuals (Conway 2011). Surveys were completed around sunrise and sunset. During morning surveys, surveyors began 30 min prior to sunrise and continued until 2 hr after sunrise. During evening surveys, surveyors began 2 hr before sunset and continued until 30 min after sunset. Surveys were not conducted in case of dense fog, heavy rain or winds exceeding  $19 \text{ km hr}^{-1}$ , as these conditions would reduce detection probability and location of birds (Conway 2011). Survey points were revisited 1–22 times within a season (Supplementary Material Table S1).

### [LEVEL HEADING 3] Land cover covariates

We extracted land cover covariates based on known *R. elegans* habitat needs (Darrah and Krementz 2009, Bolenbaugh et al. 2011, Bolenbaugh et al. 2012, Glisson et al. 2015). From the National Land Cover Database ( $30 \times 30$  m resolution; Homer et al. 2012), we extracted emergent wetland, open water, woody wetland, forest (all types), and developed (all types) cover. Because *R. elegans* are known to avoid forested and developed lands (USGS 2015), we condensed all forest cover types into a single “forest” cover type, and we condensed all developed cover types into a single “developed” cover type. We did not include the “cultivated” land cover type from the National Land Cover Database because it was strongly negatively correlated with woody wetlands ( $r = -0.86$ ) and because we included rice crop cover (as discussed below). During the time of this analysis, the National Land Cover Database was only available for certain years; therefore, we used the years 2004, 2006, 2011, and 2016, as they most closely aligned with the *R. elegans* occurrence years. From the USDA National Agricultural Statistics Service crop layer ( $30 \times 30$  m resolution; USDA NASS 2024), we extracted rice, corn, and soybean cover for the years 2005, 2006, 2012, and 2017. We obtained 10-m resolution elevation data from the U.S. Geological Survey (USGS 2015). Finally, because *R. elegans* occurrence was collected within 400-m buffers around survey locations, we calculated percent cover for each land cover covariate around each survey location (Bolenbaugh et al. 2012). We used Google Earth Engine to extract and calculate areal coverage ( $\text{km}^2$ ) for all land cover covariates (Gorelick et al. 2017).

## Analysis

### [LEVEL HEADING 3] Quantifying land cover associations and temporal occupancy trend

We used a Bayesian multi-season spatial occupancy model to assess the relationship between *R. elegans* occupancy and selected landcover covariates. To develop this model, we used the *stPGOcc* function within the R package *spOccupancy* (Doser et al. 2022, Doser and Stoudt 2024). In this modeling approach, we used an unstructured temporal random effect, which does not assume temporal correlation between years (Doser and Stoudt 2024), and we used a spatial random effect to account for spatial autocorrelation.

We included 4 years of *R. elegans* occurrence data (2005, 2006, 2012, and 2017) as the response variable. For the detection process of the model, we set Julian day and time-of-day as fixed effects, and we allowed the intercept to vary by observer. For the occupancy process, we created 14 candidate models based on a priori hypotheses of factors influencing *R. elegans* occupancy. Prior to creating models, we calculated Pearson's correlation coefficient to check for potential covariates for pairwise correlations, and we removed covariates where  $|r| > 0.60$ . Consequently, the covariates we included as fixed effects were a categorical variable for year, elevation (m), and area (km<sup>2</sup>) covered in forest, emergent wetlands, woody wetlands, developed land, open water, and rice fields. Because elevation is known to have a strong effect on *R. elegans* occupancy (Clauser and McRae 2016, Krementz et al. 2016), we included elevation in all models, and our null model included only elevation. For all continuous covariates, we subtracted the mean and divided by the standard deviation prior to analysis to improve model interpretability.

We compared our candidate models with widely applicable information criterion (WAIC; Watanabe 2013). For each candidate model, we ran 3 chains with 200,000 iterations, 60,000 burn-in iterations, and a thinning rate of 50. We checked all models for convergence (Rhat < 1.1; Brooks and Gelman 1998), and we ran posterior predictive checks (Freeman-Tukey statistic) on the top-ranked model to determine model fit and Bayesian p-values. All data and reproducible code to conduct analyses are available in Roberts (2025).

### [LEVEL HEADING 3] Characterize changes in land cover types associated with *Rallus elegans* occupancy

To characterize changes in land cover types linked to *R. elegans* occupancy, we compared the area of each land cover type (Emergent Wetlands, Woody Wetlands, Forest, Developed, Rice Crop, and Open Water) between 2005 and 2017 across the entirety of the Arkansas Delta ecoregion. Specifically, we calculated the total area of each land cover type and subtracted the area of each type in 2017 (2016 for National Land Cover Dataset (NLCD) data) from the area of each type in 2005 (2004 for NLCD data).

## RESULTS

Across all years (2005, 2006, 2012, and 2017), observers recorded *R. elegans* 56 times. The number of *R. elegans* observations declined over time, which also corresponded with declines in the number of surveys and revisits (Roberts 2025; Supplementary Material Table S1). The mean distance between *R. elegans* records was 93 km (SD = 85 km). The distribution of occupancy covariates was fairly consistent across years (Supplementary Material Table S2).

### Model Selection and Performance

The global model containing all covariates was the top model, and models containing the temporal trend (categorical “year” factor) outcompeted all models without the temporal trend (Table 1). The null model containing the temporal trend ranked below all other models containing the temporal trend, but there was considerable model uncertainty ( $\Delta\text{WAIC} < 2$ ) for all other models. The top model had an overall Bayesian  $p$ -value = 0.25, indicating adequate fit (Hobbs and Hooten 2015). Individual year Bayesian  $p$ -values varied: 2005 and 2012 had the highest  $p$ -values (Bayesian  $p$ -values = 0.40 for both) and 2006 and 2017 had the lowest (Bayesian  $p$ -values = 0.12 and 0.10, respectively; Table 1).

## Land Cover Associations

We found *R. elegans* occupancy was strongly related (95% credible intervals did not overlap zero) to only 2 covariates: *R. elegans* occupancy increased as emergent wetland area increased, and *R. elegans* occupancy decreased as elevation increased (Figure 2; Supplementary Material Table S3). However, we did find weak negative relationships (80% credible intervals did not overlap zero) between *R. elegans* occupancy and both woody wetlands and developed areas. Forest, open water, and rice all had ambiguous relationships with *R. elegans* occupancy (50% credible intervals overlapped zero). Other than emergent wetlands, rice was the only covariate to have a positive mean coefficient estimate.

## Temporal Occupancy Trend

*Rallus elegans* occupancy did not change between 2005 and 2017 (Figures 2 and 3; Supplementary Material Table S3). Although the mean coefficient estimates for the categorical year were negative, the 80% credible interval for the year 2012 overlapped zero, indicating an absence of monotonic trend (Figure 2; Supplementary Material Table S3). Predicting average yearly occupancy probability while holding land cover covariates constant (median = developed, rice, forest, woody wetlands; 75<sup>th</sup> quantile = emergent wetlands; 25<sup>th</sup> quantile = elevation) revealed high interannual variation and high overlap in both 95% and 80% credible intervals—indicating no clear evidence for a temporal trend (Figure 3). And although occupancy probability in 2017 was lower than in 2012, the probability in 2017 was higher than in 2006 (Figure 3).

## Characterize Changes in Land Cover Types Associated with *Rallus elegans* Occupancy

From 2005 to 2017 in the ~3.7 million ha (37,000 km<sup>2</sup>) Arkansas Delta, rice crop cover decreased by 294,750 ha (-40%; Table 2), emergent wetlands increased by 6,719 ha, and all other cover types increased or decreased by <7,500 ha (Table 2). Overall, emergent wetlands were the smallest cover type by area by an order of magnitude relative to all cover types except forest. Critically, although the area of emergent wetlands did increase, its total area in 2016 (22,262 ha) comprised <1% of the Arkansas Delta.

## DISCUSSION

We found *R. elegans* occupancy was stable between 2005 and 2017. This relatively stable occupancy was associated with minimal changes in emergent wetland area in the Arkansas Delta, as well as minimal changes in emergent wetland area across the United States. The only exception was that rice crop cover, which was historically associated with *R. elegans* breeding habitat, declined by 40%. Despite this stability, emergent wetlands—and *R. elegans*—remained rare in the Arkansas Delta at both the start and end of the study period. Overall, our findings supported our hypothesis that a period of relative stability in wetland habitat spatial extents is linked to stabilization of *R. elegans* occupancy.

Although long-term studies of *R. elegans* occupancy are rare, the stable occupancy trend that we observed in the Arkansas Delta both contrasts and comports with trends in other secretive marsh bird surveys. From 1996 to 2013, occupancy of 9 out of 15 (60%) of the marsh breeding bird species significantly declined, and most of these declining species exclusively used emergent marsh habitat (Tozer et al. 2016). These surveys, and detection surveys conducted in 2017, included different species of rails, specifically *R. limicola* (Virginia Rail), *Porzana carolina* (Sora), *Fulica americana* (American Coot), and *Gallinula galeata* (Common Gallinule), but did not include *R. elegans* (Tozer et al 2017). In spite of this general decrease in marsh bird



occupancy, the abundance of secretive species increased from 2011 to 2021 but again did not include *R. elegans* (Tozer et al. (2024). For *R. elegans* specifically, Stevens and Conway (2020) found range-wide occupancy declined slightly between 1999 and 2012; however, the model that included a temporal trend was not the top-ranked model in their model selection analysis. Combined with our results, these studies suggest there is, at best, evidence for a weak decline in *R. elegans* during a period that extended from prior to our study (2005) until midway through our study (2006 to 2012), and overall, secretive marsh birds did not have a monolithic trend.

Of all variables assessed, *R. elegans* occupancy was only positively associated with emergent wetlands and rice fields. The strong positive association between *R. elegans* occupancy and emergent wetlands is not surprising: it is mirrored across the species range (Meanley, 1969; Price et al. 1995; Alderfer, 2014; Glisson et al. 2015). From the 1950s to 1998, the extent of emergent wetlands declined across the *R. elegans* range, and this is linked to the range-wide declines in *R. elegans* populations during that same time (Budd and Krementz, 2011; Krementz, 2017; Stevens and Conway 2020). Nevertheless, even though rice fields and the ditches at their margins were historically well-used habitat by *R. elegans* in the southeastern United States (Pickens and King, 2012), we found only a weak positive relationship between *R. elegans* occupancy and rice fields. *Rallus elegans* have been documented to forage and nest in flooded rice fields during the breeding season (Meanley 1969; Hohman et al. 1994). But changes in rice farming practices, such as shorter rice field inundation periods and no longer allowing ditches to become weedy and simulate emergent wetlands, have likely reduced the benefits of rice fields for *R. elegans* (Hunter et al. 2006, Pierluissi and King 2008). However, a recent study found that in the Arkansas Delta, *R. elegans* made short-term dispersals to rice fields (Schmit 2024).

Although *R. elegans* are known to avoid tree cover, we found only weak negative associations between *R. elegans* occupancy and tree-dominated cover types. An explanation for this could be the study design: survey sites were selected based on wetland presence, and any forested or developed land included in our analysis would simply be the landscape matrix surrounding the wetland (Budd and Krementz 2011; Supplementary Material Table S2). Previous studies have found strong negative associations between *R. elegans* and canopy cover (Darrah and Krementz 2011, Pickens and King 2012). However, a meta-analysis of secretive marsh bird habitat associations in the Mississippi Flyway showed that forest and (non-specific) woody wetlands have only weak negative associations with secret marsh birds (Malone et al. 2021). Additionally, a multi-scale analysis of *R. elegans* habitat associations showed that *R. elegans* exhibit the weakest negative association with forested land cover at a 400-m buffer—which is the buffer size we used in our analysis (Glisson et al. 2015).

One important caveat of our study is that we cannot account for emigration or immigration of *R. elegans* in the Arkansas Delta, but these types of movements could affect the apparent relative stability we observed. For example, during our study, *R. elegans* could have emigrated into the Arkansas Delta from areas with poorer habitat, which could lead to higher apparent occupancy. In the Arkansas Delta, there is contrasting evidence that migratory or recolonization behaviors could be influencing the stability in occupancy. At a restored emergent wetland within the Arkansas Delta, *R. elegans* quickly recolonized and began breeding (Hargrove and Osborne 2023), which was the first record of *R. elegans* breeding in Arkansas since 2006. Schmit (2024) found that only 2 out of 23 birds in an Arkansas Delta wildlife management area exhibited migratory behavior, suggesting that migration may not be an important factor driving occupancy stability. But *R. elegans* can also quickly recolonize restored emergent wetlands (Hargrove and Osborne 2023), indicating emigration could be influencing



occupancy trends if there is more or better emergent wetland habitat in the Arkansas Delta. Whether Arkansas Delta emergent wetlands (both condition and extent) are faring better or worse than emergent wetlands in any other area in the Mississippi Flyway are beyond the scope of this study, but this could be an interesting future comparison.

### **Management Implications**

There appears to be an opportunity for turning the tide in *R. elegans* declines in the Arkansas Delta. Our study suggests there is a stable, if rare, source of *R. elegans* in the Arkansas Delta. Additionally, *R. elegans* can rapidly recolonize restored emergent wetlands (Hargrove and Osborne 2023). For example, in 2017, the creation of Freddie Black Choctaw Island West Wildlife Management Area (WMA) and restoration from a rice field to an emergent wetland produced the first recorded *R. elegans* breeding in Arkansas since 2006 (Hargrove and Osborne 2023). Since 2017, *R. elegans* have continued to breed and thrive at Choctaw West WMA (Schmidt 2024). Thus, conserving extant emergent wetlands and creating more emergent wetlands could contribute to keeping *R. elegans* populations stable—and possibly contribute to population increases. However, creating high-quality emergent wetlands with robust emergent vegetation could be critical if population increases are the goal (Valente et al. 2011). Capitalizing on this period of relative stability could stand to benefit *R. elegans* populations in the Arkansas Delta and beyond.

### **Supplementary material**

Supplementary material is available at *Ornithological Applications* online.

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### **Ethics statement**

Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government. The findings and conclusions in this article are those of the authors and do not necessarily represent the views of the U.S. Fish and Wildlife Service.

### **Conflict of interest statement**

Authors declare no conflict of interest.

### **Author contributions**

Caleb P. Roberts, Auriel M. V. Fournier, David Krementz, Michael Budd, and Karen Rowe conceived the idea, design, experiment (supervised research, formulated question or hypothesis). David Krementz and Michael Budd performed the experiments (collected data, conducted the research). Caleb P. Roberts, Auriel M. V. Fournier, Jessica Novobilsky, Paul Akpejelu, Lauren L. Berry, Michael C. Ferrara, Lindsey A. P. LaBrie, Lauren Luther, Vinita Karki, Michael Shaw,

and Kenneth Wilson wrote the paper (or substantially edited the paper). Caleb P. Roberts, Auriel M. V. Fournier, David Kremetz, and Michael Budd developed or designed methods. Caleb P. Roberts analyzed the data.

### Data availability

Analyses reported in this article can be reproduced using the data provided by Roberts (2025).

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**Table 1.** Model selection results for estimating *Rallus elegans* (King Rail) occupancy trends in the Arkansas Delta (Mississippi Alluvial Valley of Arkansas, United States) from 2005 to 2017. Candidate models were Bayesian multi-season, single-species occupancy models. Columns indicate the ecological process (occupancy) covariates, the expected log point-wise predictive density (elpd), effective number of parameters (pD), and widely applicable information criterion (WAIC).

Model	elpd	pD	WAIC
~YearFactor + Elevation + Rice + EmergentWetlands + OpenWater + Forest + Developed + WoodyWetlands	-334.15	29.41	727.12
~YearFactor + Elevation + EmergentWetlands + WoodyWetlands + OpenWater	-334.30	29.92	728.44
~YearFactor + Elevation + EmergentWetlands + Rice	-334.73	29.94	729.35
~YearFactor + Elevation + EmergentWetlands + Forest	-334.99	29.75	729.49
~YearFactor + Elevation + Developed + Forest	-335.06	29.69	729.50
~YearFactor + Elevation + EmergentWetlands	-335.01	29.75	729.52
~YearFactor + Elevation	-335.18	29.69	729.74
~Elevation + Rice + EmergentWetlands + OpenWater + Forest + Developed + WoodyWetlands	-335.45	29.96	730.82
~Elevation + EmergentWetlands + WoodyWetlands + OpenWater	-335.60	30.00	731.20
~Elevation + EmergentWetlands + Rice	-336.19	30.02	732.43
~Elevation	-336.27	29.98	732.50
~Elevation + EmergentWetlands	-336.26	30.04	732.60
~Elevation + Developed + Forest	-336.26	30.14	732.79
~Elevation + EmergentWetlands + Forest	-336.32	30.17	732.97

**Table 2.** Total area per land cover class in the Arkansas portion of the Mississippi Alluvial Valley ecoregion, United States (Arkansas Delta; ~3.7 million ha total area). All areas are in hectares. Rice cover comes from the National Agriculture Statistical Service Crop Layer dataset (NASS), and all other cover types come from the National Land Cover Dataset (NLCD). NASS data are from 2005 and 2017, and NLCD data are from 2004 and 2016. The final “Change” row shows the difference in hectares between 2004/2005 and 2016/2017. Negative values indicate decrease in the respective cover type.

	Year	Rice (ha)	Open Water (ha)	Woody Wetlands (ha)	Forest (ha)	Emergent Wetlands (ha)	Developed (ha)
	2004/2005	719,285	138,614	665,471	72,441	15,543	166,264
	2016/2017	424,535	131,221	661,975	70,869	22,262	173,673
<b>Change</b>		-294,750	-7,393	-3,496	-1,571	6,719	7,410

**Figure 1.** A map of the study region, the Lower Mississippi Alluvial Valley ecoregion, that lies within Arkansas, USA. Marsh bird survey points are depicted with red dots and the inset map shows the location of Arkansas in the United States. Map generated using ArcGIS Pro Version 3.2.0 (ESRI 2023).

**Figure 2.** Coefficient estimates for *Rallus elegans* (King Rail) occupancy predictors in the Arkansas Delta (Mississippi Alluvial Valley of Arkansas, United States) from 2005 to 2017 derived from spatial multi-season occupancy model. Predictors were land cover types, elevation,



and time (year). Black dots represent the mean estimate. From thickest to thinnest, error bars represent 50%, 80%, and 95% credible intervals, respectively.

**Figure 3.** Predicted occupancy probability of *Rallus elegans* (King Rail) in the Arkansas Delta (Mississippi Alluvial Valley of Arkansas, United States) in the years 2005, 2006, 2012, and 2017 derived from spatial multi-season occupancy model. Black dots represent the mean estimate. From thickest to thinnest, error bars represent 80% and 95% credible intervals, respectively. Land cover covariates were set to median values, except for elevation (25<sup>th</sup> quantile) and emergent wetlands (75<sup>th</sup> quantile; Supplementary Material Table S2). This was done to make contrasts between years easier to visualize.

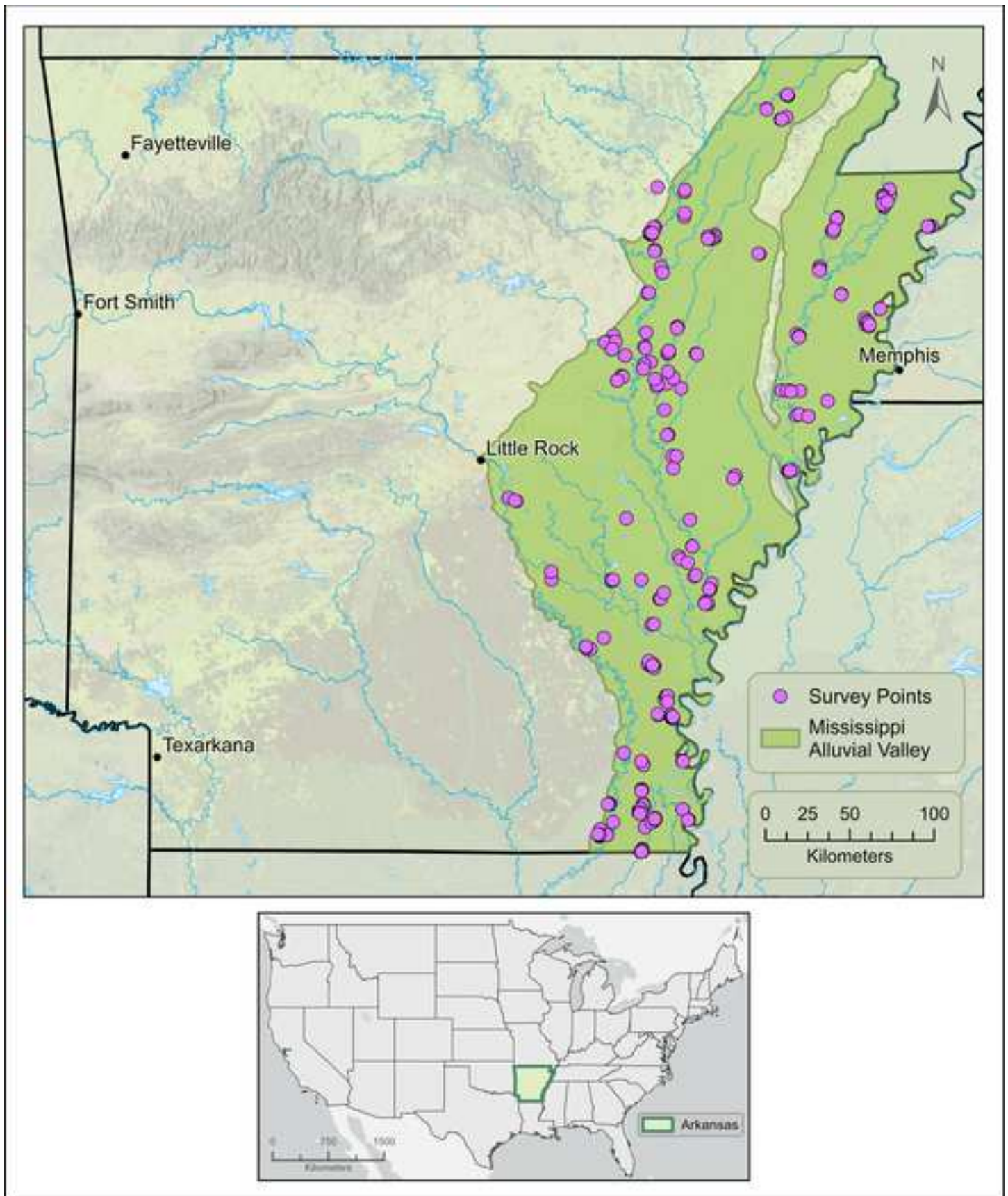


Figure 2

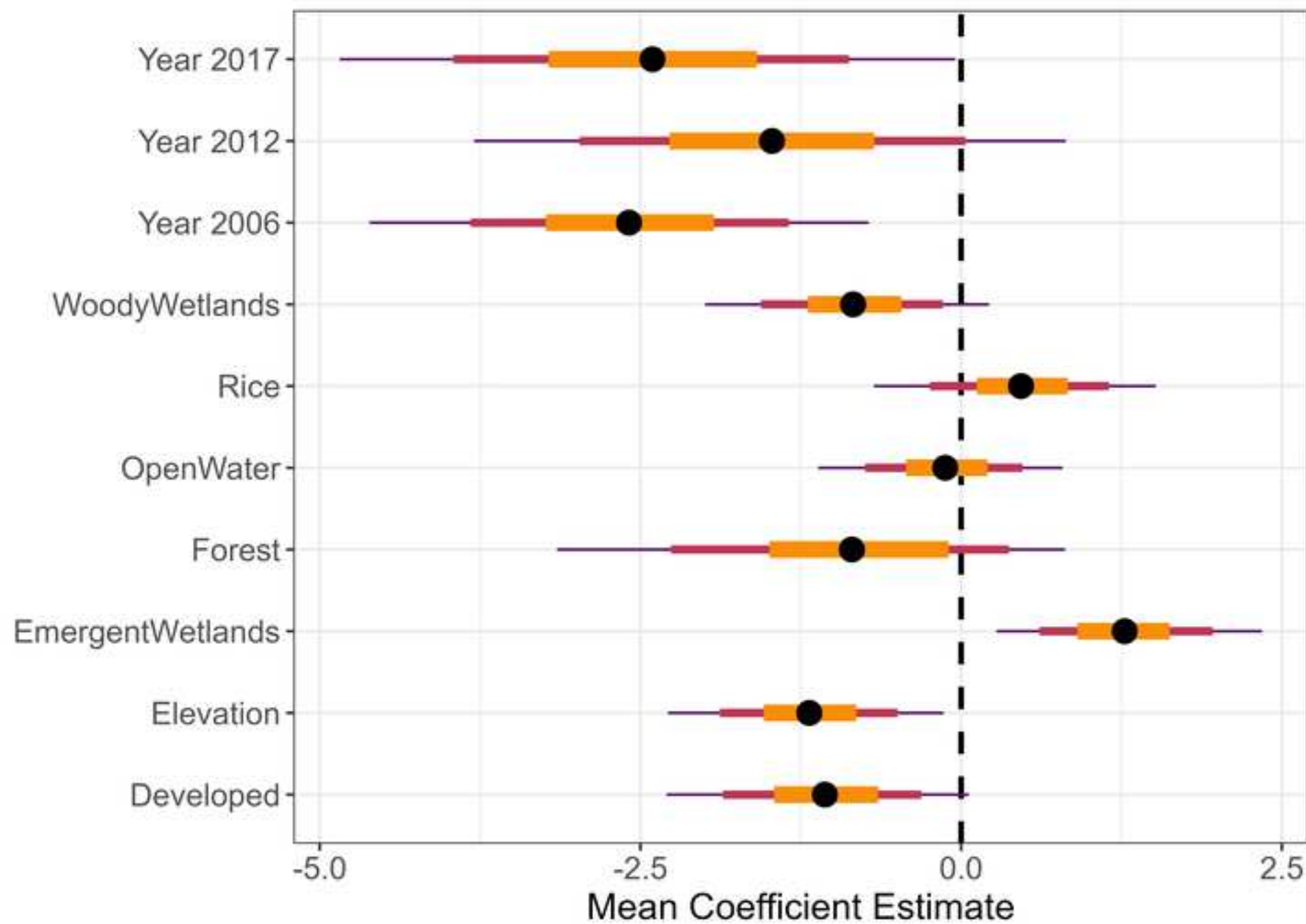


Figure 3

