Research Square

Preprints are preliminary reports that have not undergone peer review.
They should not be considered conclusive, used to inform clinical practice,
or referenced by the media as validated information.

Ecological Drivers and population trend of Yuma Ridgway's Rail (Rallus obsoletus yumanensis) breeding in the Cienega de Santa Clara, Sonora, Mexico

Stefanny VillaGomez-Palma
Pronatura Noroeste
Osvel Hinojosa-Huerta
Cornell University Cornell Laboratory of Ornithology
Timothy D. Meehan
National Audubon Society
Jorge Cortes-Ramos
SECIHTI-CICESE-UALP
Eduardo Palacios
epalacio@cicese.edu.mx

Centro de Investigación Científica y de Educación Superior de Ensenada, Baja California Unidad Académica La Paz https://orcid.org/0000-0003-4557-2427

Research Article

Keywords:

Posted Date: July 2nd, 2026

DOI: https://doi.org/10.21203/rs.3.rs-9794290/v1

License: This work is licensed under a Creative Commons Attribution 4.0 International License.
Read Full License

Abstract

The Yuma Ridgway's Rail (*Rallus obsoletus yumanensis*) is an endangered marsh bird inhabiting the remaining wetlands of the Colorado River Delta, with the largest population in the Cienega de Santa Clara, Sonora, Mexico. This elusive species faces threats from habitat loss due to land-use changes and river regulations in the southwestern U.S and northwestern Mexico. To protect marsh-dependent species in heavily managed wetlands and preserve quality habitats, it is essential to identify ecological factors that influence their abundance. This study examined how ecological drivers and passive management practices influenced population trends of the Yuma Ridgway's Rail (yRIRA) in the Cienega de Santa Clara from 2013 to 2022, with the goal to guide conservation efforts in this evolving wetland. We conducted a long-term assessment of breeding abundance of yRIRAs in the Cienega using hierarchical distance sampling (HDS) models to account for imperfect detection and evaluate the influence of habitat composition, hydrological conditions, vegetation productivity, disturbance, and interannual variability. Abundance was positively associated with emergent vegetation cover, vegetation greenness (NDVI), surface water availability (NDWI), and negatively associated with greater water depth and dominance by *Juncus cooperi* and *Phragmites australis*. Model selection strongly supported year effects, vegetation productivity, and surface water availability as primary drivers of population dynamics. Our results indicate that suitable breeding habitat relies on complex emergent vegetation and consistent surface water. They emphasize the need for adaptive, binational water management to preserve the Cienega de Santa Clara's ecological health and support the endangered marsh specialist.

Introduction

The Yuma Ridgway's Rail (*Rallus obsoletus yumanensis*) (hereafter, yRIRA) is a marsh bird federally listed as endangered in both the USA and Mexico (Conway & Eddleman 2000; SEMARNAT 2025). It is also a priority species for conservation by the Upper Gulf of California and Colorado River Delta Biosphere Reserve in Mexico (CONANP, 2009). Historically, yRIRA was common along marshlands along the lower Colorado River and its delta. Currently, it inhabits wetland fragments in the Lower Colorado River Basin (Eddleman & Conway 2020), with the largest population (around 6,000 individuals) located in the Cienega de Santa Clara. The Cienega is a crucial habitat for marsh birds within the Colorado River basin and northwestern Mexico, making it one of the most vital wetlands in the region (Hinojosa-Huerta et al. 2001; Hinojosa-Huerta et al. 2008).

Population growth in this subspecies has been linked to the expansion of cattail after fires, while declines are associated with aging cattail and marsh drying (Conway et al. 2010; Hinojosa-Huerta et al. 2013). The coverage, distribution, and structure of emergent vegetation in the Cienega respond to changes in the hydrological system (Lomeli-Banda et al. 2021) which, in turn, influence the distribution and abundance of marsh birds and other wildlife. The habitat dynamics are closely tied to ecological and hydrological processes, as well as management actions in the Cienega de Santa Clara, including vegetation greenness cycles, uncontrolled wildfires, sedimentation, maintenance dredging at the wetland

input canal, shifts in water flow caused by these dredging activities, and fluctuations in the brackish agricultural return flows that sustain this wetland.

Our goals in this study were to determine the long-term abundance trend of yRIRA in the Cienega de Santa Clara and to assess how hydrological variability and ecosystem dynamics influence their population fluctuation. Effective conservation and recovery efforts for Yuma Ridgway's Rail and other marsh bird populations depend on identifying critical habitat features and understanding the ecological factors behind population changes.

Methods

Study area

The Cienega de Santa Clara is located in the southeastern part of the Colorado River Delta in Sonora, Mexico, within the Upper Gulf of California and Colorado River Delta Biosphere Reserve, a federally protected natural area (Fig. 1). The climate is arid and semi-warm, with rainfall occurring in summer and winter, and a total annual precipitation of less than 100 mm (CONABIO 1998). Water depth of the Cienega de Santa Clara ranges from 15 to 30 cm in the shallowest areas along its edges to between 70 and 110 cm in the deepest parts of the wetland. The southern section consists of muddy floodplains influenced by the tides of the Upper Gulf of California (Zengel et al. 1995). Out of its total area (18,000–20,000 ha), 36% is marsh habitat, while the rest is open water lagoons and mudflats (Carrillo–Guerrero et al. 2013). The discharge of brackish agricultural return flows from the Welton-Mohawk Irrigation District in Arizona (around 120 million cubic meters per year) triggered the accidental restoration of the Cienega and has been the main source of water for this wetland since 1976 (Glenn et al. 1996). This water is not included in Mexico's allocation of Colorado River water and is not yet legally assigned for the Cienega (Cohen et al. 2001).

Data collection

We conducted two annual marsh bird surveys during the breeding season in accordance with the Standardized North American Marsh Birds Monitoring Protocols established by Conway in 2002. This extensive program spanned 19 years, from 2003 to 2022. The focus of these surveys included several key species, specifically the California Black Rail (*Laterallus jamaicensis coturniculus*), Least Bittern (*Ixobrychus exilis*), Sora (*Porzana carolina*), Virginia Rail (*Rallus limicola*), Yuma Ridgway’s Rail (*Rallus obsoletus yumanensis*), and American Bittern (*Botaurus lentiginosus*). The methodology employed in these surveys involved structured call-response surveys utilizing variable-distance point counts. Each survey was conducted within a 100-meter radius, with points spaced at least 200 meters apart to ensure data accuracy and minimize potential bias in bird counts. Monitoring encompassed 26 transects randomly selected in the marshland within the Cienega (Fig. 1), resulting in a total of 130 counting stations within these transects. These routes were established by Hinojosa-Huerta et al. in 1999 and

were positioned along the edge of the marsh-highland interface and at the boundary between the marsh and open water.

In each count station, pre-recorded vocalizations were used to stimulate responses from the target marsh bird species. Each survey commenced with a five-minute period of silence to allow the natural environment to stabilize. Following this, we initiated the playback of pre-recorded calls from the six focal species in a specific sequence: California Black Rail, Least Bittern, Sora, Virginia Rail, Yuma Ridgway's Rail, and American Bittern. Each vocalization was played for 30 seconds, followed by an additional 30 seconds of silence. This arrangement was aimed to maximize initial responses from less dominant species, as suggested by Conway (2002). Typically, the initial survey visits occurred at the beginning of the breeding season, defined as March 15 to April 15, followed by a second round of visits occurring from May 15 to June 15. Data collection sessions commenced at dawn and continued until approximately 10:30 a.m. Data collection was performed from either canoes or kayaks navigating within the wetland, or by foot along the shoreline adjacent to the marsh.

At each survey station, we recorded relevant data, which included geographic coordinates obtained via GPS, the number of marsh bird detections and their respective distances, as well as the start time and date of observations. Furthermore, we documented critical habitat characteristics such as vegetation type, percentage cover, presence and depth of surface water, with measurements taken using a tape measure.

The water flows of the Wellton-Mohawk Canal were determined from measurements taken at the Southern International Boundary and provided by the International Boundary and Water Commission (IBWC). These measurements were based on streamflow gauging, and continuous recording was using a limnimeter and a Parshall flume, located 24 meters upstream of the Southern International Boundary Line, 168 meters east of the Colorado River, and 2.9 kilometers west of San Luis, Arizona. The data were recorded hourly, covering the study period from 2013 to 2022.

Environmental data were sourced from the National Meteorological Service (SMN 2025), collected at the automated station in San Luis Río Colorado, Sonora, about 42 km from the Cienega de Santa Clara. The station recorded data every 10 minutes. For this study, we used data from 2013 to 2022, focusing solely on ambient temperature.

To correlate bird populations with habitat changes, we quantified the proportions of the main vegetation types at each station within a 50-meter buffer. We recorded vegetation data at the species level, since some marsh birds prefer specific emergent plants (Conway 2002). For each station, we visually estimated the percentage coverage of vegetation stratum, surface water, and bare ground based on presence and coverage, totaling 100%. When possible, we identified plants to the species level and recorded their coverage to ensure 100% species composition per sampling unit. We measured the surface water depth with a tape when water was present.

We used the Normalized Difference Vegetation Index (NDVI) and the Normalized Difference Water Index (NDWI) as integrative proxies for primary productivity and hydrological dynamics across the landscape, capturing spatial and temporal processes that local vegetation measurements alone may not fully reflect. NDVI is an index of greenness calculated from near-infrared and red reflectance and is a common proxy for measuring greenness and habitat quality (Tucker 1979; Nieto et al. 2015; Fairbairn et al. 2025). The Normalized Difference Water Index (NDWI) is derived from near-infrared (NIR) and shortwave infrared (SWIR) reflectance and is a strong remote sensing indicator of surface water presence (Gao 1996). Both indices were calculated on a monthly and yearly basis using the Google Earth Engine platform (Gorelick et al. 2017), with Landsat-8 Collection 1 Tier 1 (32-Day NDVI Composite). By combining NDVI and NDWI, we measured temporal changes in greenness and hydrological conditions, providing important indicators of habitat suitability and environmental change in the study area.

To analyze the fire events, we first used the NASA Fire Information for Resource Management System (FIRMS) to identify periods before and after the fires in the Cienega. Next, we employed Landsat 8–9 OLI/TIRS surface reflectance imagery, which includes atmospheric correction via the Landsat Surface Reflectance Code (LaSRC) (Eric Vermote et al. 2016). We also applied the Normalized Burned Area Index to evaluate severity and set thresholds for the affected areas, utilizing the Google Earth Engine platform (Gorelick et al. 2017). We used key value groups provided by the United States Geological Survey, calculating ratios of NIR to SWIR values in the traditional way as a reference point for interpretation. The calculated burned extent was exported to ArcGIS, where a map of the affected area was created using spectral signatures. Afterwards, a supervised classification was performed, enabling us to determine the values for each station.

Data Analysis

We estimated population density using hierarchical distance sampling models implemented in the R statistical computing software version 4.5.2 (R Core Team 2025) using the *gdistsamp* function in the unmarked package (Kellner et al. 2023; Fiske & Chandler 2011). This framework distinguishes ecological from observational processes by estimating latent abundance, temporary detection availability, and detection probability as a function of distance. Detection distances were divided into four intervals (0–100 m, 101–200 m, 201–300 m, and 301–400 m). An upper truncation distance of 300 m was selected after assessing the observed distance distribution to include 95% of detections. This reduces the influence of sparse long-distance detections and enhances numerical stability. Count data were organized into a site × year matrix, with primary sampling units and repeat visits as secondary occasions, structured by visit and distance class.

Environmental and vegetation covariates measured at the site–year level served as predictors of abundance. These included spectral vegetation indices (NDVI and NDWI), surface water coverage, water depth, burn severity, cover of emergent vegetation, and the percent cover of dominant emergent plant species like *Phragmites australis*, *Juncus cooperi*, *Typha domingensis*, *Scirpus americanus*, and *Cyperus odoratus*. All continuous covariates were standardized to a mean of zero and a standard deviation of one

before analysis to facilitate interpretation of effect sizes and improve model convergence. To model temporary availability for detection, we included Julian date and decimal hour as visit-level covariates. Both linear and quadratic terms for time of day were used to account for potential nonlinear variations in availability throughout the survey period.

Missing values in site-level covariates were addressed using Multiple Imputation by Chained Equations (MICE) (Buuren & Groothuis-Oudshoorn 2011) with predictive mean matching. A customized predictor matrix was created to control the imputation process, allowing NDVI and NDWI to act as general predictors while avoiding highly collinear variables, such as flow-related metrics, from predicting each other. Site identity was included as a categorical predictor. A ridge penalty was applied to reduce potential multicollinearity during imputation. The distributions of imputed values were visually compared with observed data to ensure that the imputation maintained the variables' underlying structure.

Before fitting hierarchical distance sampling models, we initially screened variables for the abundance component. We used the maximum observed count across two visits per site–year as a proxy response and fitted a negative binomial generalized linear model with all candidate site-level covariates. All model subsets were generated and ranked according to Akaike ‘s Information Criteria (AIC). Summed Akaike weights for each covariate were computed, and those with weights of 0.50 or higher were selected for the hierarchical abundance model. This method helped reduce the number of predictors and prevent overfitting in the framework.

The hierarchical distance sampling model was fitted using a negative binomial mixture distribution to account for overdispersion in the abundance of data. In this formulation, abundance was modeled as a function of selected environmental covariates, while availability was modeled as a function of Julian date and time of day. Detection probability was modeled as a function of distance using an intercept detection component. Models were compared using AIC. Goodness-of-fit of the final selected model was evaluated using a parametric bootstrap with 100 simulations performed. For each bootstrap, we computed the sum of squared errors, Pearson’s chi-square statistic, and the Freeman–Tukey statistic to evaluate the fit between observed and predicted counts.

To evaluate the effects of individual covariates on density, we generated model-based predictions across the observed range of each focal covariate while holding all other predictors at their mean standardized values. Predicted densities and 95% confidence intervals were obtained using the predict function for the abundance component and visualized to illustrate functional relationships. Annual density estimates were derived by fixing environmental covariates at their mean values and varying year as a categorical predictor. Total annual abundance was calculated by multiplying the predicted density by the total marshland study area (5,800 ha), and percent change between years was calculated as the proportional difference in the predicted abundance values.

Finally, we evaluated associations between annual density estimates and regional climate variables, including minimum and maximum temperatures and water flow-related metrics. Annual averages of regional covariates were calculated and standardized before analysis. To account for uncertainty from

the hierarchical distance-sampling model in this secondary analysis, we employed a parametric simulation method. Specifically, we generated 3,000 simulated annual density values by sampling from a normal distribution with the estimated mean and standard error of annual density. For each simulated dataset, we fitted a linear model to relate the simulated density to a standardized regional covariate and recorded the slope coefficient. This resulted in an empirical distribution of slopes, from which we used the 2.5% and 97.5% quantiles to determine 95% confidence intervals. This approach explicitly accounts for uncertainty in density estimates when evaluating their relationships with regional ecological factors.

Results

Temporal variation in density and abundance

Annual density estimates exhibited considerable variability throughout the study period, reaching a peak in 2018, followed by a sharp decline. The lowest densities were recorded in 2020 and 2021, with a partial recovery observed in 2022. When scaled to the entire 5,800-hectare marsh land study area, estimated abundance decreased from approximately 5,913 individuals (95% CI: 4,903–7,276) in 2013 to 3,123 individuals (95% CI: 2,501–3,900) in 2022, representing roughly a 47% reduction over the study period. The most pronounced declines occurred between 2020 and 2021. Confidence intervals were generally narrow, especially during years of lower abundance, indicating reasonably precise estimates (see Fig. 2).

The hierarchical distance sampling model incorporating environmental covariates outperformed the intercept-only model based on AIC (AIC = 10847.5; 1300 sites), indicating that variation in density was explained by habitat and environmental predictors. A negative binomial mixture was used to account for overdispersion in counts, and the dispersion parameter was positive and highly significant. Parametric bootstrap goodness-of-fit tests showed mixed results: SSE and Pearson chi-square statistics suggested overdispersion ($P < 0.01$), whereas the Freeman–Tukey statistic, which is less sensitive to extreme values, indicated adequate model fit ($P = 0.099$).

Detection and availability were modeled separately. The estimated availability probability across visits was 0.39, indicating that individuals were not always detectable even when present at a site. Availability varied significantly with Julian date and time of day, highlighting a nonlinear (quadratic) relationship with survey hour (Fig. 3b). Availability changed throughout the day, following a parabolic curve, with higher availability in the early hours and before sunset, and lower availability at certain intermediate times. Availability declined as the breeding season progressed, suggesting that individuals were more vocal or detectable earlier in the season (Fig. 3a). Detection probability declined with distance, as indicated by the fitted detection function (supplemental material), with an estimated effective detection radius of approximately 91 meters.

Effects of habitat and environmental covariates

In the preliminary covariate screening for distance models, the negative binomial GLM showed that maximum counts were significantly related to several habitat, hydrological, and temporal covariates. The intercept was positive and highly significant ($\beta = 0.6532 \pm 0.0822$ SE, $p < 0.001$), indicating the expected log-abundance under baseline conditions.

Among environmental predictors, burn severity showed a positive and significant association with abundance ($\beta = 0.0699 \pm 0.0268$, $p = 0.009$), indicating higher counts in areas affected by greater fire impact. Water depth was negatively related to abundance ($\beta = -0.0640 \pm 0.0300$, $p = 0.033$), while the percent cover of emergent vegetation had a strong positive effect ($\beta = 0.1289 \pm 0.0305$, $p < 0.001$), emphasizing the importance of structurally complex marsh habitats. Vegetation greenness (NDVI) and surface water availability (NDWI) also showed positive associations with abundance ($\beta = 0.1302 \pm 0.0308$, $p < 0.001$; $\beta = 0.0970 \pm 0.0296$, $p = 0.001$), indicating higher counts in more productive and wetter areas. Conversely, the cover of *Juncus cooperi* and *Phragmites australis* was negatively associated with abundance ($\beta = -0.0680 \pm 0.0318$, $p = 0.033$; $\beta = -0.1501 \pm 0.0349$, $p < 0.001$), with the latter showing one of the strongest vegetation-related effects in the model (see Tables S1 and S2 in the supplemental material).

Model selection using multi-model inference supported the importance of year, NDVI, and *Phragmites australis* (summed weights = 1.00), followed closely by emergent vegetation (0.99) and NDWI (0.98). Burn severity (0.92), depth (0.85), and *Juncus cooperi* (0.82) also demonstrated strong support, whereas other vegetation types had weaker evidence of association. Overall, observed abundance was positively associated with vegetation productivity, surface water availability, emergent vegetation cover, and burn severity, and negatively associated with water depth and dominance by certain plant species. The data showed that temporal variation was significant, with notable declines in recent years highlighting the combined effects of habitat structure, hydrological conditions, and interannual dynamics on maximum counts (see Tables S1 and S2 in the supplemental material).

Hierarchical Distance Sampling (HDS)

In contrast, hierarchical distance sampling (HDS) models, which explicitly accounted for both detection probability and temporary availability, revealed that density was significantly associated with most habitat and hydrological variables but not with burn severity. Indicators of water availability consistently showed positive effects: both NDWI and the proportion of open water were significantly and positively associated with density, suggesting that wetter sites with more surface water support higher rail densities. Vegetation productivity, measured by NDVI, also had a positive association, indicating that more productive habitats further improved suitability. Emergent vegetation cover correlated positively with density, while water depth had a negative relationship, implying that shallower waters are more favorable than deeper ones.

Similarly, the cover of *Phragmites australis* and *Juncus cooperi* was negatively related to density, consistent with reduced habitat suitability when these species dominate. However, burn severity did not

significantly influence density after accounting for detection processes. Predicted density responses generally followed a linear pattern across the observed covariate ranges, with confidence intervals consistently supporting relationships with NDVI, NDWI, emergent vegetation, depth, and invasive plant cover (Figs. 4 and 5)

Simulation-based analyses of climate and flow variables

Temperature variables showed weak or inconsistent effects. The distribution of simulated slope coefficients for maximum temperature was centered near zero (median ≈ − 0.001; 95% CI: −0.062 to 0.060), indicating no clear linear association with density. The minimum temperature showed a weak negative trend (median ≈ − 0.039; 95% CI: −0.093 to 0.015), but the confidence interval included zero, suggesting no statistically significant effect (Fig. 6)

Water flow variables exhibited clear seasonal patterns in relation to density (Fig. 7). During summer, flow showed a strong positive correlation, with a median of about 0.199 and a 95% credible interval from 0.148 to 0.252, indicating a significant positive effect. Fall flow also tended to be positive (median approximately 0.107; 95% CI: −0.041 to 0.263), although the interval crossed zero, suggesting only moderate evidence of a positive relationship. Spring flow showed a slight positive trend (median roughly 0.037; 95% CI: −0.034 to 0.112), with the interval including zero. Conversely, winter flow was negatively correlated with density (median about − 0.128; 95% CI: −0.181 to − 0.075).

Discussion

This study offers a detailed evaluation of the ecological and hydrological factors affecting yRIRA populations at the Cienega. It underscores the importance of consistent water inflows, especially avoiding long water shortages during key periods such as spring and summer, and supports vegetation recovery through well-timed management actions, including prescribed burns when suitable. Several limitations need acknowledgment, like the absence of prey availability data, which limits understanding of the mechanistic pathways underlying population changes, and the use of remotely sensed indices that, while effective for capturing broad habitat patterns, may miss finer structural features vital to rail ecology. Despite these challenges, the findings provide useful insights for adaptive management and emphasize the urgent need for coordinated international conservation efforts to protect the Cienega and secure the long-term survival of the Yuma Ridgway’s Rail in the region.

Our long-term assessment of yRIRA breeding populations in the Cienega de Santa Clara reveals significant year-to-year variability and an overall reduction in numbers from 2013 to 2022. Estimated abundance declined by around 47%, with the most notable decreases occurring during 2020–2021 and only a partial rebound in 2022. These changing trends emphasize the population's vulnerability to annual environmental changes and habitat conditions.

Consistent with previous research in the lower Colorado River basin (Smith 1975; Anderson and Ohmart 1985; Conway et al. 1993) our models indicate that habitat structure and hydrological conditions are key factors influencing yRIRA distribution and density. We found that emergent vegetation cover, vegetation greenness (NDVI), and surface water availability (NDWI and open water proportion) all have positive correlations with density, even after accounting for detectability. In contrast, deeper water and dominance by cattail and bulrush are negatively associated. These findings align with earlier studies (Eddelman 1989; Conway et al. 2010) highlighting yRIRA's preference for structurally complex wetlands and their avoidance of deep water or uniform tall, dense vegetation stands.

Smith (1975) and Anderson & Ohmart (1985) documented higher densities of yRIRA and related species in marshes with moderate coverage of cattail (*Typha* sp.) and bulrush (*Schoenoplectus* sp.), environments that provide abundant nesting and foraging opportunities. Our results aligned with this pattern: areas with more emergent plants and increased greenness exhibit higher densities, indicating that mid-successional, productive vegetation stands are especially ideal (Figs. 3a, 3b). Likewise, Conway et al. (1993) observed that yRIRA tended to avoid habitats dominated by saltcedar (*Tamarix* sp.) and arrowweed (*Pluchea* sp.), as well as large open water areas. This matches our findings that higher dominance by *Phragmites australis* and *Juncus cooperi*, both of which are less structurally diverse, correlates with lower densities. Saltcedar and arrowweed are shrubs, not emergent plants, emphasizing yRIRA's preference for structurally complex wetland habitats.

Despite the above, our results differ from those of a study conducted in the Lower Colorado by Conway et al. (2010). That study suggested post-fire regeneration could temporarily enhance habitat conditions when hydrological disturbances are minimal. They recognized fire as a significant factor influencing marsh dynamics in the Lower Colorado River Basin, acting as a secondary disturbance when natural flood pulses from the river are absent. Fire can eliminate accumulated litter, increase light availability, and stimulate the growth of emergent vegetation, promoting early successional stages and structural heterogeneity, conditions linked to better habitat suitability for Yuma Ridgway's Rail. In our analysis, the observed positive fire impact on maximum counts in the negative binomial GLM might be influenced by increased detectability after fire, rather than true demographic responses. This effect was not observed in hierarchical distance sampling (HDS) models that incorporated detection probability and temporary availability, highlighting the complexity of separating demographic effects from detectability. Post-fire environments often have less dense vegetation and altered structures, which can improve visibility and vocal detectability of rails, potentially inflating raw count estimates. After accounting for detection bias, the evidence for a direct positive effect of fire on density became weaker in our study.

Another plausible explanation is that the number of fire events during the study period was relatively limited and spatially clustered, thereby reducing statistical power to detect strong population-level responses. As a result, their demographic effects may have been localized or short-lived, making them difficult to capture in annual landscape-level density estimates. Furthermore, the advantages of fire probably rely on maintaining optimal hydrological conditions at the wetland: without sufficient water availability after a fire, vegetation may not recover enough to create the complex emergent habitat that

yRIRA prefers. Therefore, fire alone may not ensure better habitat; rather, its ecological benefits depend on the right seasonal inflows working together.

The role of water depth in our study aligned with earlier research emphasizing hydrological factors as key influences on marsh bird habitat use (Eddleman 1989; Conway et al. 2010). Generally, deeper water was associated with lower rail densities, probably because greater depth limits the availability of shallow foraging areas and suitable nesting sites. Meanwhile, decreases in water flow can raise wetland salinity, affecting vegetation extent and species distribution, as noted by Zengel et al. (1995) and García-Hernández et al. (2013), and may also reduce prey biomass under hypersaline conditions.

Although short periods without inflow might not significantly impact population estimates (Hinojosa-Huerta et al., 2013), our results indicate that seasonal flow patterns are crucial. The timing of inflows appears more influential than total annual discharge, with breeding-season flows showing positive relationships and wintering flows showing negative relationships. Increased flows during spring and summer are associated with higher densities, likely because they help sustain productive shallow-water habitats and promote emergent vegetation, consistent with findings from the lower Colorado River basin linking spring pulse flows to improved habitat quality (Conway et al. 1993). Conversely, winter flows are negatively correlated with density, implying that excessive water levels outside the breeding season could deepen marsh areas beyond optimal conditions. Nonetheless, maintaining some winter discharge remains important to prevent hypersalinity and to support nearby mudflat habitats, which are essential for species such as shorebirds (Mexicano et al. 2013; Gomez-Sapiens et al. 2013).

Taken together, our findings support and advance the regional literature on the ecological drivers of Yuma Ridgway's Rail populations. Like earlier work, we confirm the importance of emergent vegetation and hydrologically dynamic habitats for yRIRA. We also demonstrate more clearly how seasonal timing of hydrological inputs interacts with vegetation structure to influence density. Variations in how fire is involved and its nuanced links to water depth emphasize the need for tailored, system-specific management strategies.

Management actions in the Cienega de Santa Clara can produce contrasting ecological outcomes depending on their timing, intensity, and hydrological context. Maintenance activities such as dredging or channel clearing during late fall or early winter (October–December) can be particularly detrimental if they coincide with reduced inflows and the post-breeding period, when yRIRA populations might already be under physiological stress.

Under low-water conditions, mechanical disturbance can disrupt hydrological connectivity, accelerate vegetation senescence, promote biomass accumulation, and increase fire risk. In contrast, implementing dredging in late winter or early spring, when vegetation is largely dormant and hydrological conditions are typically more stable, may reduce these negative impacts. For this reason, management should adopt an integrated and adaptive approach that explicitly considers seasonality, vegetation age structure, and prevailing hydrological conditions. Maintaining internal water connectivity must be a

central objective of any intervention, as localized marsh drying can rapidly alter vegetation composition and diminish habitat suitability for wetland-dependent species.

From a broader conservation perspective, sustaining seasonally appropriate inflows that promote shallow, productive marsh conditions, particularly during summer and spring, is essential for maintaining yRIRA populations. Hydrological management should prioritize regimes that support a heterogeneous mosaic of emergent vegetation interspersed with shallow open water, the structural configuration most consistently associated with higher densities. Within this framework, prescribed fire may serve as a complementary tool to restore vegetation dynamics where natural disturbance regimes have been altered. Carefully planned burns, coordinated with the National Commission on Natural Protected Areas in Mexico (CONANP) under a monitored experimental design, could be implemented every 5–6 years during late winter or early spring, when yRIRA are less vulnerable, to rejuvenate emergent vegetation while minimizing demographic risks.

Importantly, hydrological manipulation and fire management must be complemented by long-term monitoring that explicitly considers detection probability. This monitoring should assess short-term habitat responses as well as longer-term population trends. Combining seasonal water management, vegetation restoration, and detection-corrected monitoring within an adaptive framework provides the most cohesive approach to maintaining habitat quality and ensuring population stability in the Cienega.

Ultimately, these results underscore the urgency of hydrologically informed, adaptive conservation strategies to reverse recent population declines and maintain ecological integrity in the Cienega de Santa Clara, while recognizing both the consistencies and complexities relative to previous studies in the lower Colorado River basin. Our findings reinforce the central role of sustained, seasonally appropriate inflows and structurally complex emergent vegetation in supporting breeding and foraging habitats for yRIRA.

The long-term viability of the Cienega remains uncertain, however, due to desalination and water reclamation initiatives in Arizona, which may represent potential future hydrological scenarios that could reduce the agricultural return flows sustaining the wetland (Pitt & Hinojosa-Huerta 2022). Because the marsh depends almost entirely on these return flows, reductions in discharge could lead to increased salinity, reduced surface water extent, vegetation shifts, and ultimately further population declines. Binational collaboration will therefore be essential to secure environmental water allocations that safeguard the hydrological and ecological health of one of the last remaining wetlands in the Colorado River Delta.

Declarations

Competing Interests

The authors have no relevant financial or non-financial interests to disclose.

Funding

This work was supported by Pronatura Noroeste. The first author (SVP) has received an scholarship support from Consejo Nacional de Humanidades, Ciencias y Tecnologías (CONAHCYT). Comisión Internacional de Límites y Aguas (CILA), Servicio Meteorológico Nacional, provided data. We also thank the support of Centro de Investigación Científica y de Educación Superior de Ensenada, Baja California (CICESE).

Author Contributions

All authors contributed to the study conception and design. Material preparation, data collection and analysis were performed by Stefanny Villagómez-Palma, Osvel Hinojosa-Huerta, Timothy D. Meehan, Jorge Cortes-Ramos, and Eduardo Palacios. The first draft of the manuscript was written by Stefanny Villagómez-Palma and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Acknowledgments

We extend our special gratitude to the field crew of Pronatura Noroeste: Juan Butrón, José Juan Butrón, Benito Brambila, Juan Carlos Medina, Miguel Cruz, and Juan Ángel Butrón. Our work has been made possible thanks to the support of Alejandra Calvo-Fonseca and Eduardo Soto-Montoya from Pronatura Noroeste.

Data Availability

The datasets generated during and/or analyzed during the current study are available from the first author on reasonable request

References

1. Akaike, H. (1974). A New Look at the Statistical Model Identification. In: Parzen, E., Tanabe, K., Kitagawa, G. (eds) Selected Papers of Hirotugu Akaike. Springer Series in Statistics. Springer, New York, NY. https://doi.org/10.1007/978-1-4612-1694-0_16
2. Anderson, B.W. and Ohmart, R.D. (1985). Habitat use by Clapper Rails in the lower Colorado River valley. The Condor. Volume 87(1), pages 116–126. https://doi.org/10.2307/1367143
3. Carrillo-Guerrero, Y.K., Flessa K., Hinojosa-Huerta O. and Lopez-Hoffman L. (2013). From accident to management: The Cienega de Santa Clara ecosystem. Ecological Engineering. Volume 59, pages 84-92. https://doi.org/10.1016/j.ecoleng.2013.03.003

4. Comisión Nacional de Áreas Naturales Protegidas (CONANP). (2009). Programa de Conservación y Manejo, Reserva de la Biosfera Alto Golfo de California y Delta del Río Colorado, México. México, D.F. 319 páginas.
5. CONABIO (1998). Regiones terrestres prioritarias de México. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. https://doi.org/10.5962/bhl.tittle.118644
6. Conway, C. J., Eddleman, W. R., Anderson, S. H., and Hanebury, L. R. (1993). Seasonal Changes in Yuma Clapper Rail Vocalization Rate and Habitat Use. The Journal of Wildlife Management. Volume 57(2), pages 282–290. https://doi.org/10.2307/3809425
7. Conway, C.J., W. R. Eddleman. (2000). Yuma Clapper Rail. In: Endangered animals: a reference guide to conflicting issues Greenwood Press, Westport, CT. Volume 20, pages 277–284. https://doi.org/10.1890/09-1624.1
8. Conway, C.J. (2002). Standardized Protocols for Monitoring Marsh Birds in North America. Arizona Cooperative Fish and Wildlife Research Unit, United States Geological Survey and School of Renewable Natural Resources. University of Arizona, Tucson, AZ.
9. Conway, C.J. and Gibbs, J.P. (2005). Effectiveness of call-broadcast surveys for monitoring marsh birds. The Auk 122 (1), pages 26–35. https://doi.org/10.1093/auk/122.1.26
10. Conway, C.J. (2009). Standardized North American Marsh Bird Monitoring Protocols. Version 2009-2. Wildlife Research Report #2009-2. U. S. Geological Survey, Arizona Cooperative Fish and Wildlife Research Unit, Tucson, AZ.
11. Conway, C.J., Nadeau, C.P. and Piest, L. (2010). Fire helps restore natural disturbance regime to benefit rare and endangered marsh birds endemic to the Colorado River. Ecological Applications. Volume 20, pages 2024–2035. https://doi.org/10.1890/09-1624.1
12. Eddleman, W.R. (1989). Biology of the Yuma Clapper Rail in southwestern U.S. and northwestern Mexico. Wyoming Cooperative Research Unit, University of Wyoming, Laramie, WY, pages 127.
13. Eddleman, W.R. and Conway, C. J. (1998). Clapper Rail (*Rallus longirostris*). In: Poole, A. & Gill, F. (Eds), The Birds of North America. Philadelphia, PA: The Birds of North America, Inc., pages 34.
14. Fairbairn, A. J., Katholnigg, S., Leichtle, T., Merkens, L., Schroll, L., Weisser, W. W., & Meyer, S. T. (2025). NDVI and vegetation volume as predictors of urban bird diversity. Scientific Reports, 15(1), 12863. https://doi.org/10.1038/s41598-025-96098-0
15. Gao, B. C. (1996). NDWI—A normalized difference water index for remote sensing of vegetation liquid water from space. Remote sensing of environment, 58(3), 257-266. https://doi.org/10.1016/S0034-4257(96)00067-3
16. García-Hernandez J., Flessa K., Santiago-Serrano E., Romero-Hernández S. and Zamora-Arroyo F., (2013). Salinity responses to inflow alterations in a 6500 ha *Typha* wetland. Ecological Engineering. Volume 52, pages 191-202. https://doi.org/10.1016/j.ecoleng.2012.12.102
17. Gorelick, N., Hancher, M., Dixon, M., Ilyushchenko, S., Thau, D., and Moore, R. (2017). Google Earth Engine: Planetary-scale geospatial analysis for everyone. Remote Sensing of Environment.

18. Fiske I, Chandler R. (2011). unmarked: An R Package for Fitting Hierarchical Models of Wildlife Occurrence and Abundance. Journal of Statistical Software, 43(10), 1-23. https://www.jstatsoft.org/v43/i10/
19. Kellner KF, Smith AD, Royle JA, Kery M, Belant JL, Chandler RB (2023). The unmarked R package: Twelve years of advances in occurrence and abundance modelling in ecology. Methods in Ecology and Evolution, 14(6), 1408-1415. https://doi.org/10.1111/2041-210X.14123
20. Landsat 8-9 OLI/TIRS: *Vermote, E., Justice, C., Claverie, M., & Franch, B. (2016). Preliminary analysis of the performance of the Landsat 8/OLI land surface reflectance product. Remote Sensing of Environment, 185, 46-56.*
21. Martha M. Gomez-Sapiens, Eduardo Soto-Montoya and Hinojosa-Huerta O. (2013). Shorebird abundance and species diversity in natural intertidal and non-tidal anthropogenic wetlands of the Colorado River Delta, Mexico. Ecological Engineering. Volume 59, pages 74-83. https://doi.org/10.1016/j.ecoleng.2013.03.004
22. Miura, Y., Shamsudduha, M., Suppasri, A., & Sano, D. (2025). A Global Multi-Sensor Dataset of Surface Water Indices from Landsat-8 and Sentinel-2 Satellite Measurements. Scientific data, 12(1), 1253. https://doi.org/10.1038/s41597-025-05562-z
23. Nieto, S., Flombaum, P., & Garbulsky, M. F. (2015). Can temporal and spatial NDVI predict regional bird-species richness? Global Ecology and Conservation, 3, 729-735. https://doi.org/10.1016/j.gecco.2015.03.005
24. Harrity, E. and Conway, C. (2020). Satellite transmitters reveal previously unknown migratory behavior and wintering locations of Yuma Ridgway's Rails. Journal of Field Ornithology. Volume 91, pages 300-312. https://doi.org/10.1111/jofo.12344
25. Hinojosa-Huerta, O., DeStefano, S. and Shaw, W. (2001). Abundance and distribution of the Yuma Clapper Rail (*Rallus longirostris yumanensis*) in the Colorado River delta, Mexico. Journal of Arid Environments. Volume 49, pages 171–182. https://doi.org/10.1006/jare.2001.0841
26. Hinojosa-Huerta, O., Rivera-Díaz, J.J., Iturribarría-Rojas, H. and Calvo-Fonseca, A. (2008). Population trends of Yuma Clapper Rails in the Colorado River delta, Mexico. Studies in Avian Biology. Volume 37, pages 69-73.
27. Hinojosa-Huerta, O., R. Guzmán-Olachea, J. Butrón-Méndez, J.J. Butrón-Rodríguez, and A. Calvo-Fonseca. (2013). Status of marsh birds in the wetlands of the Colorado River delta, Mexico. Ecological Engineering. Volume 59, pages 7-17. https://doi.org/10.1016/j.ecoleng.2013.04.058
28. Mexicano L., Glenn E.P, Hinojosa-Huerta O., Garcia-Hernandez J., Flessa K and Hinojosa-Corona A. (2013). Long-term sustainability of the hydrology and vegetation of Cienega de Santa Clara, an anthropogenic wetland created by disposal of agricultural drain water in the delta of the Colorado River, Mexico. Ecological Engineering. Volume 59, pages 111-120. https://doi.org/10.1016/j.ecoleng.2012.12.096
29. Pitt, J. e Hinojosa-Huerta, O. (2022). How the Colorado River Delta Might Survive the Compact. https://doi.org/10.2307/j.ctv2z6qfp2.10

30. R Core Team. 2025. R: A language and environmental for statistical computing. R Foundation for Statistical Computing, Vienna, Australia. Version 4.5.2. URL https://www.R-project.org/
31. Secretaría de Medio Ambiente y Recursos Naturales (SEMARNAT). (2025). *PROY-NOM-059-SEMARNAT-2025: Protección ambiental Especies nativas de México de flora y fauna silvestres: Categorías de riesgo y especificaciones para su inclusión, exclusión o cambio*. Diario Oficial de la Federación, 14 de abril de 2025
32. Smith P. M. (1975). Habitat requirements and observations on the Clapper Rail (*Rallus longirostris yumanensis*). M.Sc. thesis, Arizona State University. Tempe.
33. SMN (2025). Estaciones Meteorológicas Automáticas (EMAS). https://smn.conagua.gob.mx/es/observando-el-tiempo/estaciones-meteorologicas-automaticas-ema-s. Accessed January 2025.
34. Todd, R.L. 1986. A saltwater marsh hen in Arizona: a history of the Yuma Clapper Rail (*Rallus longirostris yumanensis*). Arizona Game and Fish Department, Federal Aid Project W-95-R, Completion Report, Phoenix, AZ.
35. Tucker, C. J. 1979. Red and photographic infrared linear combinations for monitoring vegetation. Remote sensing of Environment, 8(2), 127-150. https://doi.org/10.1016/0034-4257(79)90013-0
36. Yuma Clapper Rail Recovery Team. (2000). Yuma Clapper Rail survey protocol. U.S Fish and Wildlife Service, Phoenix, AZ.
37. Zengel S.A., Meretsky V.J., Glenn E., Felger R.S., and Ortiz D. (1995). Cienega de Santa Clara, a remnant wetland in the Rio Colorado Delta (Mexico): vegetation distribution and the effects of water flow reduction. Ecological Engineering. https://doi.org/10.1016/0925-8574(94)00050-F

Figures

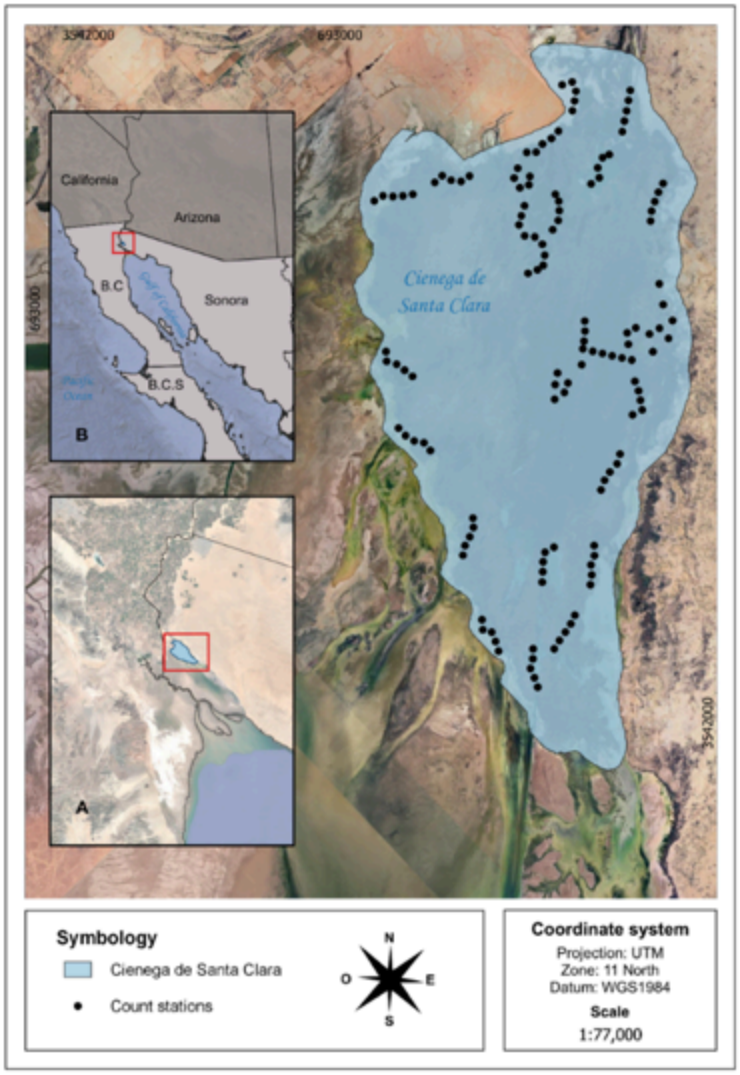


Figure 1

Map of northwest Mexico (inset A) showing the Colorado River delta with the location of Cienega de Santa Clara (inset B), and the Cienega de Santa Clara, Sonora, Mexico (main map). Dots indicate marsh birds monitoring transects along the Cienega.

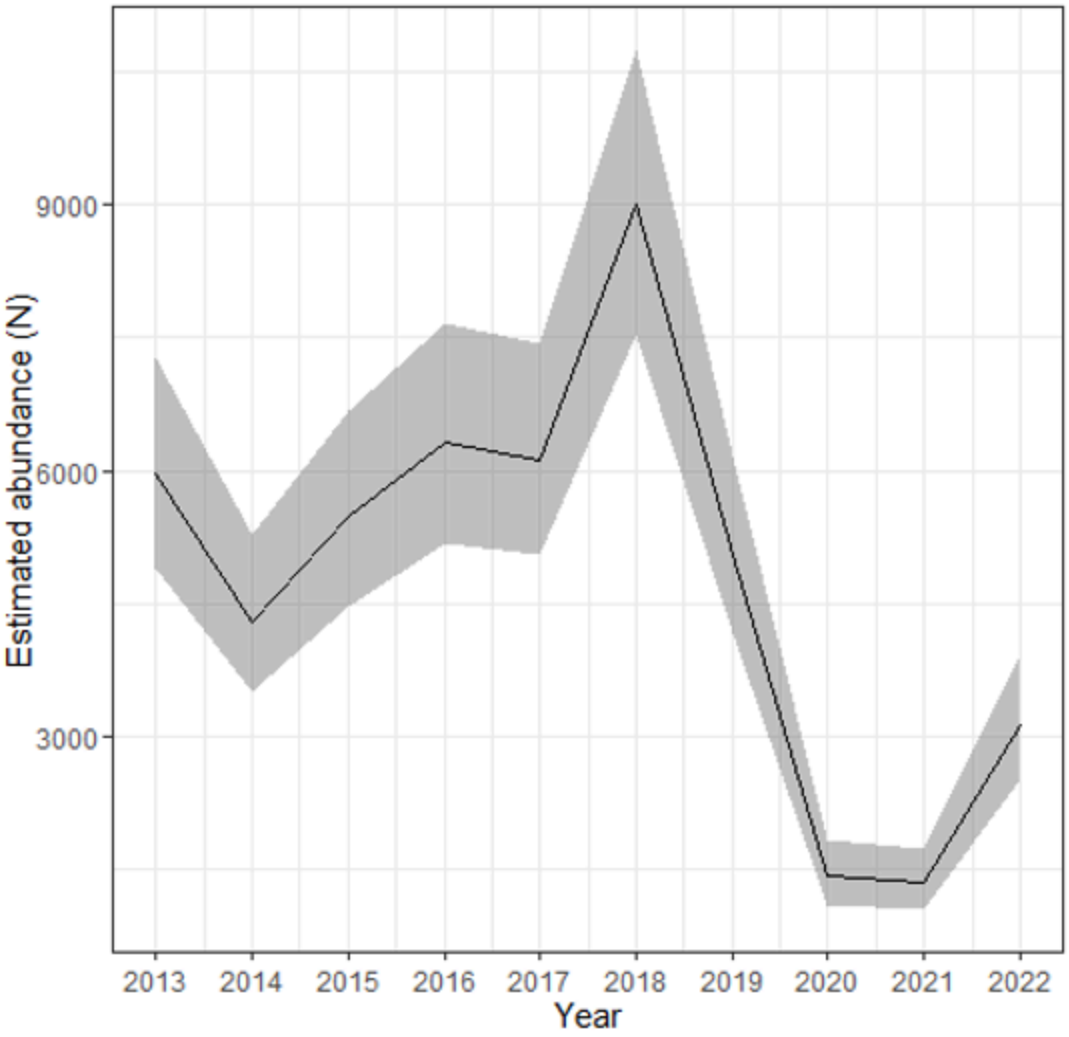


Figure 2

Estimates of total abundance of the Yuma Ridgway's rail in the Cienega de Santa Clara from 2013 to 2022. The solid line represents the predicted mean abundance and the shaded ribbon indicates the 95% confidence interval.

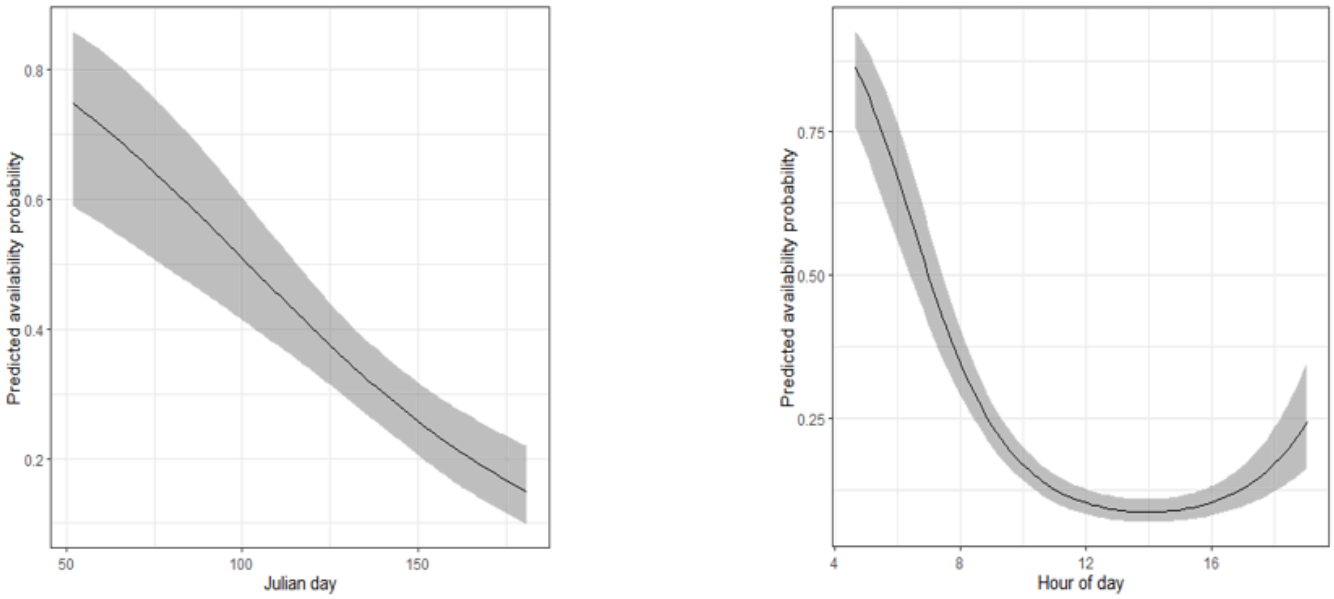


Figure 3

Availability of yRIRA detection according to Julian date (3a) and hour of day (3b).

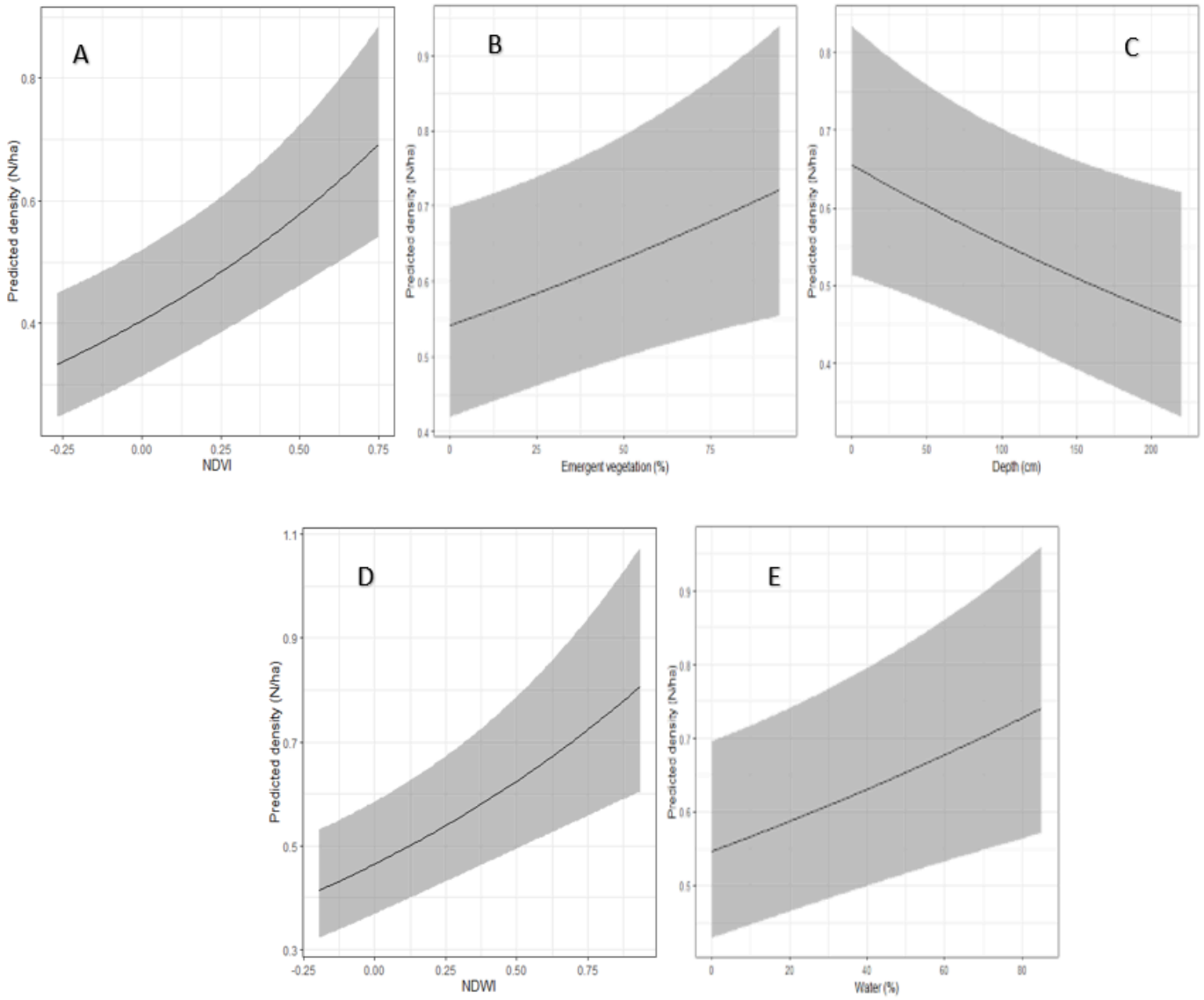


Figure 4

Effect of environmental covariates on yRIRA density in the Cienega de Santa Clara: NDVI (A), percent cover of emergent vegetation (B), water depth (C), NDWI (D), and percent water cover (E)

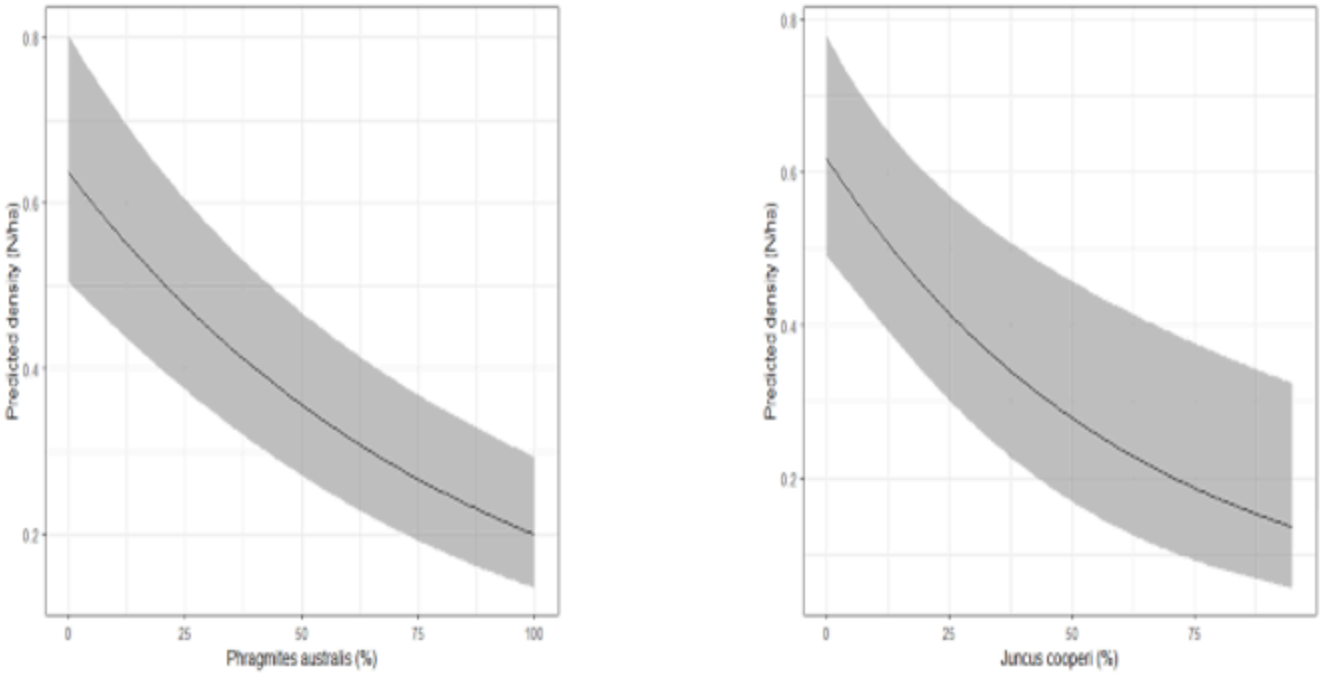


Figure 5

Effect of the species *Phragmites australis* (left) and *Juncus cooperi* (right) on the density of yRIRA in the Cienega de Santa Clara

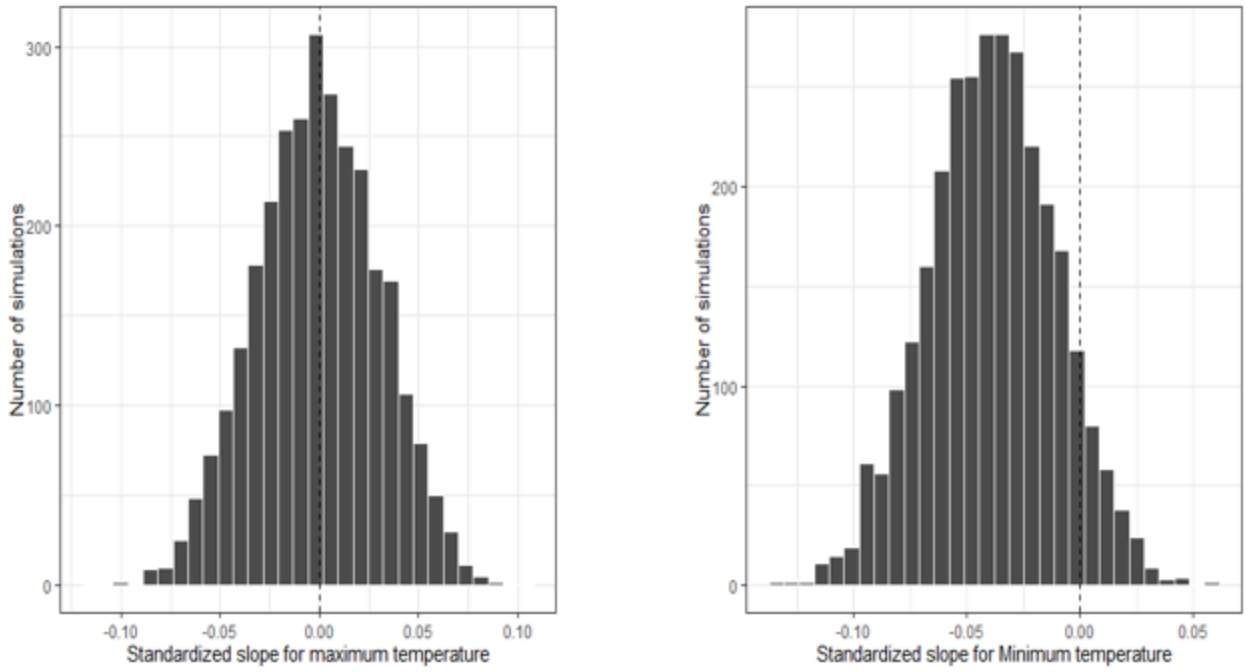


Figure 6

Standardized slope of the effect of maximum (left) and minimum temperature (right) on the density of the yRIRA in the Cienega de Santa Clara during 2013-2022.

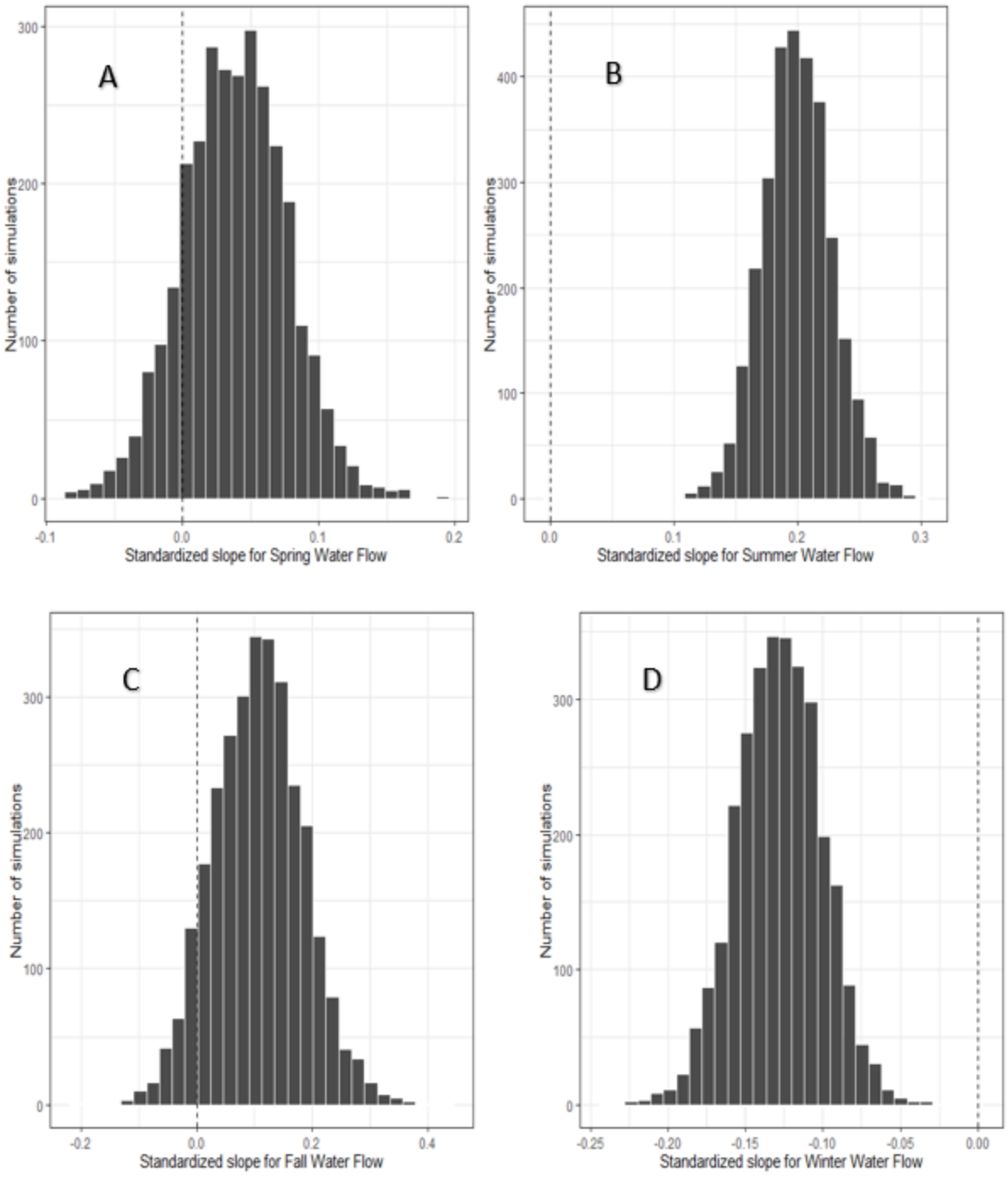


Figure 7

Standardized slope of the effect of water flows during different seasons on the density of the yRIRA in the Cienega de Santa Clara during 2013-2022: spring (A), summer (B), fall (C), winter (D).

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- Supplementarymaterialvfinal.docx