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Bird Returns on Public Investment in the Conservation Reserve Program

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Article

Keywords:

Posted Date: June 3rd, 2026

DOI: https://doi.org/10.21203/rs.3.rs-9648037/v1

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Additional Declarations: Yes there is potential Competing Interest. Dr. Ivan Rudik received funding from the Cornell Atkinson Center for Sustainability, Cornell University.

Bird Returns on Public Investment in the Conservation Reserve Program

Abstract: Participatory science offers unprecedented opportunities to evaluate biodiversity outcomes of policy instruments at low cost. Leveraging data from eBird, we evaluated outcomes of the U.S. Farm Bill's Conservation Reserve Program (CRP) from 2008–2019. Results provide the first causal evidence that CRP enrollment increases abundance and richness of grassland birds across 13 Midwestern states. Counterfactual analyses indicate that ~100M grassland birds may have been lost amid enrollment declines since 2007. However, alternative targeting strategies can dramatically and cost-efficiently improve conservation outcomes. Specifically, cost- and benefit-cost targeting could more than double bird returns per dollar spent, especially when allocations are flexible across states. Overall, results demonstrate that strategic, data-driven targeting of CRP allocations can improve both biodiversity benefits and economic efficiency of conservation investments.

The ongoing debate about the federal government's role in science suggests that the current era of downsizing is likely to continue to diminish agencies' capacity to collect the data needed to guide and evaluate federal programs. Impacts are expected to be particularly severe for programs related to environmental protection and conservation, both of which have historically suffered from insufficient data on benefits (Keiser et al. 2019). This lack will further challenge the implementation of economically justified policies, especially amid ideological debates and opposition. In this manner, cutting budget allocations for science and monitoring can accelerate the removal of valuable programs, simply because their value cannot be evaluated.

One program that is frequently in political crosshairs is the U.S. Farm Bill, which provides the nation's largest source of federal funding for conservation. Its flagship environmental program – the Conservation Reserve Program (CRP) – distributes ~$1.8 billion dollars annually for private land conservation (USDA-ERS 2024). To meet its goals of improving water quality, preventing soil erosion, and reducing habitat loss, the CRP compensates landowners for voluntarily retiring environmentally sensitive farmland from intensive crop production and implementing conservation practices, such as increasing vegetative cover (USDA-FSA 2025). The available evidence suggests that the CRP has been beneficial; the most recent national assessment estimated $1.32 billion USD of annual benefits to soil conservation and wildlife from 1985-1997 (Hansen 2007). However, shrinking allocations have resulted in a decline of CRP enrollments from a peak of 15M ha in 2007 to ~8M ha over the last two decades (Fig. 1; Hellerstein 2017). Reductions have been most pronounced in the Northern Great Plains (Stubbs 2014; Otto et al. 2018), a continental hotspot for waterfowl and grassland birds. As long-term funding priorities for the Farm Bill continue to be hotly debated (Monke and Stubbs, 2026), rigorous demonstrations of impact are urgently needed.

Fortunately, new data sources – namely, those from participatory science projects – and greatly expanded analytic capabilities are simultaneously creating new opportunities to improve evidence-based decision-making and conduct impact evaluations at a time of tight budgets. This opportunity is revolutionary for estimating biodiversity benefits, which historically required intensive field-based study. Now the growth of eBird, the world's largest biodiversity database powered by participatory science (Sullivan et al. 2014), makes it possible to use grassland birds as a lens to assess how the CRP supports biodiversity within agricultural landscapes. North

America's grassland birds are in the conservation spotlight, given their steep declines (Johnston et al. 2025, Leroy et al. 2026) and loss of roughly two-thirds of their populations since 1970 (Rosenberg et al. 2019). Because most grassland birds (>80%) occur on private lands (NABCI 2013), programs that incentivize landowners to increase vegetative cover and grassland habitat, as does CRP (Vandever et al 2021), are essential to improve habitat conditions for them (Best et al 1997; Ryan et al. 1998; Herkert 2009; Pavlacky Jr. et al. 2021; Packlacky Jr. et al. 2022). In the case of CRP, enrolled land changes from an intensive cropping system to more 'natural' habitat, like native perennial grasses, that supports grassland birds.

In this paper, we show how eBird data can be combined with USDA enrollment information to, for the first time, provide causal evidence that the CRP positively impacts grassland birds across landscapes in 13 midwestern US states (prairie states hereafter; Fig. 1A). Although causal inference can be challenging with datasets like eBird (Johnston et al. 2021; Schrodt et al. 2025; Baylis et al. 2026), statistical and computational advances now fuel new approaches to improve returns on taxpayer investment – a timely demonstration given the growing interest in regenerative agriculture to safeguard ecosystem services as public goods (Koman et al., 2021).

Rather than evaluating specific practices or identifying causal pathways, we used a program evaluation approach to determine the effect of the CRP program on bird abundance. We also constructed four counterfactual scenarios of alternative CRP allocations. For the first scenario, we predicted how many grassland birds could have been conserved had the CRP enrollment remained at 2007 levels. We also created three counterfactuals representing different allocation strategies for the 2007 enrollment levels: a) enrolling the lowest cost land in the region that achieves 2007 enrollment levels (i.e., cost-targeting acreage constraint), b) enrolling the lowest cost land that achieves the 2007 expenditure levels (i.e., cost-targeting budget constraint), and c) enrolling the land with the highest benefit-to-cost ratio of CRP's effect on bird conservation (i.e., benefit-cost-targeting budget constraint).

Approach

Our analyses drew upon the eBird participatory science project and USDA county-level CRP enrollment records. Our bird data derive from 3 million eBird checklists submitted from 2008-2019 across 13 prairie states (Fig. 1A), a region accounting for >75% of all CRP lands (USDA-ERS 2024). We use two key measures of bird communities: abundance (i.e., number of

individuals) and species richness (i.e., number of distinct species). Because raw, semi-structured eBird data (i.e., individual checklists) are prone to sampling biases associated with the observation process (Johnston et al. 2021), we followed Liang et al. (2020) to estimate relative bird abundance and richness while adjusting for birding effort and detectability. We estimated annual abundance and richness for 40 grassland bird species within each county during four stages of their annual cycle – breeding, non-breeding (stationary), pre-breeding migratory, and post-breeding migratory. We then combined these estimates with USDA's CRP enrollment records for each county to assemble a longitudinal dataset describing the year-by-year changes in bird outcomes and CRP enrollment.

Estimating the causal effect of CRP on biodiversity is empirically challenging because CRP enrollment is not randomly assigned across counties. Instead, it reflects voluntary participation by landowners and, by program design, often systematically favors contracts with relatively greater expected environmental value. The Environmental Benefit Index (EBI) is the main metric used to evaluate and rank CRP contracts, and it prioritizes offers based on their expected benefits to soil and water resources, wildlife habitats, and other environmental outcomes. Therefore, counties with inherently greater potential for preserving bird biodiversity (e.g., more grassland habitats) are more likely to receive higher enrollment levels (Hellerstein 2017; USDA-FSA 2025). Without a credible quasi-experimental approach, a naïve comparison of observed bird abundance and richness across counties with different CRP enrollment levels would yield biased estimates that do not reflect the true program effects (Ferraro and Pattanayak 2006; Miteva et al. 2012; Adams et al. 2019).

We addressed these concerns by using a shift-share instrumental variables (IV) research design. The IV approach requires a source of exogenous variation (i.e., the instrument) that affects CRP enrollment but is otherwise assumed to be unrelated to the ecological factors that influence bird abundance and richness. We constructed the instrument using temporal changes in the national CRP enrollment cap authorized by successive Farm Bills as well as past program participation in each county, which we can credibly treat as exogenous to local contemporaneous biodiversity outcomes. Specifically, we multiplied the exogenous enrollment cap by the historical share of a county area enrolled in CRP, which affects county-level CRP enrollment differently depending on historical participation. The main assumption is that a parcel that was historically in CRP, but

no longer is, will not influence current levels of bird abundance because once a parcel returns to cropland, the habitat benefits disappear immediately. We discuss the IV approach in detail in the Methods Section. Under our model assumptions, the predicted current enrollment driven by the exogenous instrument is independent of unobserved confounding factors such as other conservation programs and land use policies, which allows us to estimate the causal effect of CRP on relative bird abundance and richness.

Results

We find positive, statistically significant effects of CRP on grassland bird abundance and richness during the breeding season. A one standard deviation (SD) increase in CRP share, defined as the share of county land enrolled in the program, leads to a 0.482 SD ($p = 0.03$) increase in grassland bird abundance during the breeding season (Fig. 2A). Equivalently, this means that a 0.15% increase in bird abundance will result from a 1% increase in CRP share (e.g., CRP share in a county increases from 1% to 1.01%). Likewise, we find that grassland bird species richness increases by 0.442 SD ($P = 0.04$) in response to a 1 SD increase in CRP share during the breeding season (Fig. 2B). There were no significant effects of CRP on grassland bird abundance or richness in the non-breeding stationary period (abundance: 0.407 SD, $P = 0.48$; richness: 0.261 SD, $P = 0.56$) nor in the pre-breeding (abundance: -0.238 SD, $P = 0.29$; richness: -0.19 SD, $P = 0.41$) or post-breeding migration seasons (abundance: 0.199 SD, $P = 0.42$; richness: 0.194 SD, $P = 0.4$) (Fig. 2A, 2B).

To better understand the scale of these effects, note that a county can only enroll a maximum of 25% of its cropland in CRP, and cropland cover in counties in our sample averaged 33% (from MODIS data). Taken together, this means that, in most cases, only 8% of all land in a county can be enrolled in CRP. Based on this scale of enrollment, grassland bird abundance would be 2.2 times greater, and richness 6.6 times greater, in a county that enrolls all its eligible land into CRP during the breeding season in prairie states, versus a county that does not enroll any land. These effect sizes are comparable with those observed in previous localized field experiments evaluating the impacts of CRP on birds (Best et al., 1997; Ryan et al., 1998), supporting the interpretation that large-scale conservation programs can generate meaningful biodiversity gains at the county level.

We similarly constructed relative abundance and richness estimates for two other groups of birds – wetland-associated (e.g., waterfowl, herons) and game birds (i.e., hunted species) – to examine if co-benefits extend to other groups that regularly use CRP land to meet some, though not all, of their habitat needs. For wetland birds, neither abundance (0.133 SD, $P = 0.28$) nor richness (0.129 SD, $P = 0.33$) during the breeding season was significantly related to CRP (Fig. 3A, 3B), but this may be due primarily to relatively little data on wetlands in our CRP sample rather than their lack of efficacy.

For breeding game birds, the pattern was weaker still for both abundance (0.094 SD, $P = 0.53$) and richness (0.011 SD, $P = 0.94$; Fig. 4A). However, we detected strong negative effects of CRP share on game bird abundance (-0.227 SD, $P < 0.1$) and richness (-0.349 SD, $P < 0.01$) in pre-breeding migration season (Fig. 4A, 4B). Because many game birds use crop residue as a high-energy food source during migration – especially in years with late thaw or low natural seed availability, this pattern may reflect reductions in row crops in counties with high CRP enrollment.

Counterfactuals to Inform Policy

Using simple counterfactual scenarios, we provide insight into the important role that the CRP has historically played in the conservation of grassland species and the program's potential to provide future critical habitat to protect grassland birds. We began by using results from the IV analyses to simulate the impacts on abundance if the CRP cap and associated enrollment had not declined beginning in 2007. We used the estimated breeding-season coefficient (Fig. 2A) to predict grassland bird populations that could have been conserved if CRP enrollment had been maintained at its historical maximum. To do so, we substituted the 2007 CRP enrollment level for the 2024 level in each county whenever the latter was lower and defined the counterfactual as the additional acreage required that would close the gap between 2024 enrolled acreage and the historical 2007 allocation (Fig. 5a).

The geographic distribution of restored acreage confirms that reported reductions in enrollment have not been spatially balanced across the 13 states (Fig. 5a). To replace this acreage in CRP, an additional 4.9 million hectares would need to be enrolled at a total cost of 752 million dollars. Our estimates suggest that this would have added 102 million birds (95% CI [6, 197]). An

equivalent interpretation is that by disinvesting in 4.9 million CRP hectares, the US has about 100 million fewer grassland birds than it would have had with the continued investment. This counterfactual approach also can be applied to even smaller groups of or individual species. For example, if we focus only on Bobolink, a grassland species of concern, our counterfactual suggests that there would be about 1.7 million (23%) more Bobolinks in 2024 than the observed number if CRP acreages had not declined relative to 2007.

In three other counterfactuals, we demonstrated the effects of alternative targeting constraints on the costs, benefits, and land allocation of CRP acreage. These counterfactuals were not intended to be a prescriptive, but rather to highlight the potential efficiency benefits of targeting. The cost-targeting acreage constraint (Fig. 5b) replicated the same aggregate increase of 4.9 million CRP hectares, but allocated those hectares by prioritizing counties with the lowest CRP rental rates, subject to the cap that no more than 25% of a county's cropland can be enrolled. This cost-minimizing scenario demonstrated that for substantially less cost (only $242 million relative to $752 million), nearly the same number of additional birds could be saved (92 million, 95% CI [6, 178]).

For the cost-targeting budget constraint (Fig. 5c), a similar exercise was undertaken but here we target the same total budget of 752 million dollars, while again prioritizing counties with the lowest CRP rental rates; that is, we allowed enrollments to expand until total rental payments reach $752 million. With the same expenditures, this approach supported the enrollment of 10.8 million hectares resulting in the conservation of 192 million birds (95% CI [12, 373]), more than twice the acreage and nearly double the birds compared to Scenario 1.

Finally, benefit-cost-targeting budget constraint (Fig. 5d) similarly allocated hectares until the budget of $752 million was met, but prioritized counties with the highest birds-per-dollar spent. We rank-ordered all counties by the effectiveness of a hectare of CRP land for bird abundance and divided by the cost of a hectare of land. Through this dual targeting strategy, 9.7 million additional hectares could be enrolled at the same cost ($752 million) resulting in the conservation of 219 million birds (95% CI [14, 425]), more than doubling the bird conservation impact relative to first scenario. Together, these scenarios highlight the substantial conservation gains that could be achieved not only by sustaining CRP enrollment but also by strategically targeting enrollment to maximize cost-effectiveness.

Because these example targeting strategies would significantly alter the geographic location of CRP acreage relative to historical patterns, we further extended the last three targeting constraints in Figure 5 to include state-level constraints on CRP acreage or budget allocations, again using the first scenario (Fig. 5a) as the baseline that sets the total additional acreage and budget required for each state. Instead of allocating additional CRP hectares to the lowest-cost counties across all prairie states, we constrained the prioritization within each state by allocating additional hectares to counties with the lowest rental rates within that state until the acreage or budget cap was reached (Fig. A1) Overall, the results revealed similar qualitative patterns in cost-effectiveness across allocation strategies but notable quantitative differences when state-level constraints were applied (Table 1). Specifically, under the state-constraint scenarios, achieving the same total acreage ($4.9 million hectares) required higher total payment ($414 million versus $242 million under national constraints). Likewise, given the same total budget ($752 million), state-level constraints yielded fewer CRP hectares enrolled (7.7 million and 6.4 million hectares in cost-targeting and benefit-cost-targeting budget constraints, respectively) and fewer birds conserved (128 million and 145 million, respectively) than their national counterparts.

These findings highlight that imposing state-level allocation constraints, though a more administratively realistic approach to distributing CRP funds, reduced overall program efficiency in both cost and bird conservation. When each state had to meet its own acreage or budget constraint, low-cost or high-return opportunities in one state could not offset higher costs or lower returns elsewhere. Consequently, total conservation gains were smaller under state-level constraints even with the same aggregate budget. In contrast, maintaining flexibility in cross-state allocation allowed resources to flow toward counties with the greatest birds-per-dollar conservation return, thereby substantially enhancing both the ecological and economic efficiency of CRP investments.

Discussion

Our analysis provides program-level causal evidence that the Conservation Reserve Program significantly contributes to grassland bird conservation across the U.S. prairie states, while we find no evidence of co-benefits flowing to wetland and game bird species. The positive effects of CRP enrollment on breeding grassland birds likely reflects a combination of short-term

behavioral responses and longer-term demographic processes. In the short term, for example, increases in local CRP acreage could improve nesting cover, reduce pollution exposure from agricultural intensification, and enhance food availability, thereby attracting breeding pairs to restored habitats (e.g., Perlut et al. 2006). Over longer periods, sustained CRP participation may promote population recovery through increased reproductive success and survival (McCoy et al 1999, Fletcher et al. 2006, Sullins et al. 2018), particularly in regions where grassland loss has been extensive.

Although we cannot directly attribute these responses to specific CRP practices, spatial evidence indicates that most CRP parcels lost since 2007 had provided grassland habitat (Fig. 1) used by breeding birds, losses that would be expected to most acutely impact grassland birds and, hence, disproportionately contribute to population declines. In that sense, our results reinforce the recent findings that grassland birds declined most steeply in regions where they were once most abundant (Johnston et al. 2025).

While our work quantified causal effects of CRP on biodiversity outcomes, we recognize that the CRP was implemented to achieve multiple environmental goals, including soil erosion, air quality, and water quality. A holistic analysis of the return on investment would need to assess the full range of environmental benefits provided by CRP. Likewise, our counterfactual analysis only estimated the cost-effectiveness of CRP in conserving bird biodiversity alone and did not monetize the nonmarket benefits of ecosystem services that grassland birds provide (e.g., pollination, pest control, recreation). Estimating the full suite of biodiversity benefits of CRP alongside the intended environmental outcomes is critical for a comprehensive cost-benefit analysis of the program. Our study provides an essential step toward achieving this goal.

As countries face multiple environmental challenges under limited budgets, the design of conservation programs that yield benefits across multiple environmental and ecological fronts becomes increasingly critical. Our work both illustrates how participatory science data can build capacity to evaluate program impacts when facing resource constraints and demonstrates that the CRP, as one specific program, generates measurable biodiversity gains. Encouragingly, our results show that the ecological returns and economic efficiency of program investments can be further improved by strategically targeting enrollment while maintaining spatial flexibility.

Methods:

Data

Our primary bird database is derived from the eBird Basic Dataset (ver. May 2021) maintained by the Cornell Lab of Ornithology (Sullivan et al. 2014). The database is contributed voluntarily by registered users. We restricted our analysis to submissions of 'complete checklists', which include counts of all species detected, along with information about the observation process, such as the location and date of observation, the amount of time spent and distance traveled during the observation, and the number of observers on the same trip.

The CRP database comes from the USDA Farm Service Agency and was further processed by our collaborating partner at the Economic Research Service to aggregate contract-level information to county-level. In this study, we used the total accumulated acreage enrolled in CRP for a county as the share of the county total acreage by the end of each fiscal year as our treatment variable.

We also collected data on other environmental covariates as control variables, including weather, elevation, and land cover outcomes. The weather data, obtained from the PRISM Climate Group, provided daily estimates of precipitation and temperature at the 4km spatial resolution. We used mean temperature and total precipitation in each county-year and calculated mean elevation at a 1km resolution using data from Amatulli et al. (2018). Our land cover data came from Moderate Resolution Imaging Spectroradiometer (MODIS), specifically the annual 500-m resolution products from MODIS MCD12Q1 v061 by Friedl and Sulla-Menashe (2022). We used the International Geosphere-Biosphere Programme (IGBP) legend that classifies land cover into 17 types (e.g., grasslands and croplands). We also aggregated the land cover information to the county level.

For the counterfactual analyses, we used the 2022 relative abundance estimates from the eBird Status and Trends Product (Fink et al. 2023) and the 2024 national bird population estimates from Avian Conservation Assessment and Population Estimates Databases (Partners in Flight 2024) for 33 bird species (Table A1).

Methods: Estimation of Bird Abundance and Richness

We estimated relative bird biodiversity using only complete checklists in the eBird data—submissions where observers reported all species detected. Because bird detectability is affected by the observation process (e.g., time of day, effort), we adjusted for checklist-level variation in observer effort by employing a log-linear Poisson regression framework adapted from Liang et al. (2020).[1]

We modeled both abundance and richness in each checklist as a Poisson process influenced by effort variables, detectability, and true abundance/richness. The model was log-linearized and estimated with ordinary least squares:

$$\log\left(\text{biodiversity index}_{\text{kchdmy}}\right) = \beta X_{\text{kchdmy}} + \Gamma_{\text{cy}} + \xi_{\text{h}} + \lambda_{\text{m}} + \varepsilon_{\text{kchdmy}} \qquad (1)$$

where $\text{biodiversity index}_{\text{kchdmy}}$ was either the counts of individuals or unique species observed by checklist k in county c, at hour of day h, on day d, in month m and year y. The vector X_{kchdmy} was a set of effort variables, including the number of observers, the duration minutes, the distance traveled, and the elevation of the observation location. ξ_{h} was an hour-of-day fixed effect that controls for all variables common across days within an hour of day, and λ_{m} was a month fixed effect that controls for all unobserved variables within the month. The county-by-year fixed effects Γ_{cy} captured bird abundance or richness at the county-year level after conditioning on the effort variables, hour-of-day fixed effect, and month fixed effect. We estimated Equation (1) to recover the estimated fixed effect $\hat{\Gamma}_{\text{cy}}$ as our respective measures of bird abundance and richness in each county-year. Notably, our bird outcome, $\hat{\Gamma}_{\text{cy}}$, does not indicate the actual abundance or richness but, rather, the relative abundance or richness from the sample that are not subject to sampling biases.

We repeated the above steps for grassland, wetland, and game bird species, each by breeding, nonbreeding, pre-breeding migration, and post-breeding migration seasons. For example, when we summed across species for a checklist submitted in May, any species whose breeding season

[1] We also take several steps to filter our bird database prior to estimating Equation (1) to impose more consistent structure on the data and to reduce the variation of birding effort that results in the ability to detect birds. In particular, we restrict the sample to checklists with no more than 10 observers, trip duration of less than 5 hours, and traveling distance during observations of less than 5 km.

is outside of May was not included. By breaking up the indices by species annual cycles, the season-specific measurement was not biased by within-year migration patterns. In the end, we estimated county-year abundance and richness measures for each respectively species-season combination.[2]

Methods: The Effects of CRP on Bird Abundance and Richness (IVs)

We took the estimates of $\hat{\Gamma}_{cy}$ for bird abundance and richness from Equation (1), and employed an instrumental variable (IV) approach to estimate the effects of CRP on bird abundance and richness. The IV approach isolated exogenous variation in CRP enrollment that is unrelated to unobserved ecological or policy factors affecting bird outcomes. A valid instrumental variable needs to satisfy two assumptions: relevance (i.e., the instrument is a strong predictor for current CRP enrollment) and exclusion restriction (i.e., the instrument should only affect concurrent bird biodiversity through CRP enrollment today). Under these two assumptions, the variation of concurrent CRP enrollment in a county that was driven by the instrument should estimate consistent causal effects of the program on the bird outcomes.

We instrumented for current CRP enrollment using a Bartik-style shift-share IV to estimate the program impact on relative abundance and richness. A shift-share IV exploits heterogeneity in each unit's initial exposure to the endogenous variable that translates the aggregate shocks into differential predicted changes across space (Adão et al. 2019; Goldsmith-Pinkham et al. 2020; Borusyak et al. 2025). Our shift-share instrument took advantage of the variation of the national CRP enrollment cap over time and the plausibly exogenous past enrollment levels across counties. Specifically, the instrument was formed by the interaction of (i) county-level exposure to the enrollment 10 years prior, and (ii) the national CRP enrollment acreage cap change over the subsequent decade:

$$\text{IV}_{cy} \equiv \text{CRP share}_{cy-10} \times \left(1 + \text{CRP cap change rate}_{y}\right) \qquad (2)$$

Here, CRP share_{cy-10} was the share of a county area enrolled in CRP 10 years prior to year y, and $\text{CRP cap change rate}_{y}$ was the change rate of national CRP enrollment cap dictated by Farm

[2] We have 40 grassland bird species, 178 wetland bird species, and 71 game bird species. The respective season dates are based on the 2020 status and trend products. Information is provided by our collaborators at the Cornell Lab of Ornithology.

Bills since year $y - 10$. For instance, the CRP enrollment cap in 2002 was set at 15.9 million hectares, and the enrollment cap in 2012 was 13 million hectares. The change rate would then be $\frac{13-15.9}{15.9} = -0.182$. In words, our instrument was the prediction of concurrent CRP enrollment for each county using past enrollment (the share) that is subject to a common nationwide policy shock (the shift)[3].

The IV approach involved two steps. The first stage of the IV approach regressed the endogenous CRP enrollment variable on our instrument:

$$\text{std}(\text{CRP Share}_{cy}) = \beta_{IV}\text{std}(\text{IV}_{cy}) + \theta_1\text{std}(\text{controls}_{cy}) + \gamma_c + \lambda_y + \varepsilon_{cy}^{1^{st}\text{ stage}} \quad (3)$$

$\text{std}(\text{CRP Share}_{cy})$ was the share of county c's land enrolled in CRP in year y standardized to mean 0 and SD 1 (i.e., a Z score). β_{IV} reflected the predicted impact of national CRP cap changes, weighted by the historical enrollment in each county. The second stage of IV then used the predicted values of CRP share from the first stage to estimate its effect on bird biodiversity:

$$\text{std}(\hat{\Gamma}_{cy}) = \beta_{CRP}^{IV}\text{std}(\widehat{\text{CRP Share}}_{cy}) + \theta_2\text{std}(\text{controls}_{cy}) + \gamma_c + \lambda_y + \varepsilon_{cy}^{2^{nd}\text{ stage}} \quad (4)$$

where $\widehat{\text{CRP Share}}_{cy}$ was the fitted value from the first stage. The left-hand-side variable $\text{std}(\hat{\Gamma}_{cy})$ was the estimated bird abundance or richness at the county-year level, similarly standardized to mean 0 and SD 1 to facilitate coefficient interpretation. Here, β_{CRP}^{IV} captured the causal impact of CRP share on bird biodiversity and represented the SD changes in bird abundance or richness in response to a 1 SD increase in CRP share. In both Equation (3) and (4), the vector of controls_{cy} was a set of exogenous control variables that accounted for other unobserved ecological determinants of bird abundance and richness. We include county-level shares of cropland and grassland to account for land cover composition. We also controlled for contemporaneous weather conditions that included average annual temperature and total precipitation in a county-year. γ_c and λ_y were county and year fixed effects that controlled for unobserved spatial

[3] This shift-share style instrument is in the spirit of Bartik (1991). The traditional shift-share instrument is the average of shifts or shocks at the subregion (e.g., industry or sector) level weighted by their respective shares within the region. Our instrument is a variation of this form where only one industry or sector (i.e., CRP enrollment) takes a positive share value in a county. Such "single-shift, single-share" shift–share instrument is a special case that is statistically equivalent to the traditional multiple-share Bartik IV (Goldsmith-Pinkham et al. 2020; Borusyak et al. 2025).

heterogeneity (e.g., soil quality) and temporal shocks common to all counties (e.g., drought). In an alternative specification, we further included state-by-year fixed effects to account for time-varying unobservables, such as state- or NGO-led conservation efforts in grasslands that were correlated with the implementation of CRP (Appendix Table A2 through A4). The standard errors were clustered at the county level to account for serial correlation in unobserved shocks within counties over time.

The causal identification of $\beta_{\mathrm{CRP}}^{\mathrm{IV}}$ hinged on the shift-share instrument ($\mathrm{IV}_{\mathrm{cy}}$) from Equation (2) meeting both the relevance and exclusion restriction assumptions. As the changes in county-level CRP enrollment, especially since 2007, mostly followed the similar declining trend of national enrollment (Fig. 1B), $\mathrm{IV}_{\mathrm{cy}}$ should strongly correlate with the enrollment today ($\mathrm{CRP\ Share}_{\mathrm{cy}}$). Table A2 through A4 presented the direct test of the relevance condition through the first-stage F-statistics. The large Kleibergen-Pcp F-Statistics from the first-stage regressions suggests our instrument from Equation (2) was strong and satisfied the relevance condition.

The exclusion restriction assumption stated that $E\left[\mathrm{IV}_{cy} \times \varepsilon_{\mathrm{cy}}^{2^{\mathrm{nd}}\ \mathrm{stage}} | \mathrm{controls}_{\mathrm{cy}}, \gamma_{\mathrm{c}}, \lambda_{\mathrm{y}}\right] = 0$. That is, conditional on the set of weather and land cover controls as well as county and year fixed effects, our shift-share IV was assumed to be uncorrelated with any unobserved factors left in $\varepsilon_{\mathrm{cy}}^{2^{\mathrm{nd}}\ \mathrm{stage}}$. The national CRP acreage cap was set at the federal level, which has shifted over time largely in response to commodity market conditions and budgetary reasons (Hendricks and Er 2018), and therefore is unlikely to be affected by local biodiversity outcomes. Importantly, the identification from shift-share designs required the exogeneous component to come from exposure shares in the presence of only few aggregate shocks (Goldsmith-Pinkham et al. 2020; Borusyak et al. 2025), which is true in our case as our instrument exploits a single national policy shifter affecting all counties. In other words, the exclusion restriction had to be satisfied by the exogeneity of the share component (Goldsmith-Pinkham et al. 2020; Borusyak et al. 2025). Additionally, for such share-based instrument to yield a consistent causal estimate, our exposure share needed to be tailored to the endogenous treatment variable, rather than a generic share that captured other ecological drivers similarly contributing to bird biodiversity, such as the share of all agricultural land or vegetative covers in the past (Borusyak et al. 2025).

Leveraging the share of county area enrolled in CRP in the past to construct the share component ensured that our shift-share instrument affected bird outcomes only through the current CRP enrollment levels. However, lagging the shares does not by itself guarantee exogeneity (Borusyak et al. 2025). While the exclusion restriction was fundamentally untestable, we leveraged institutional facts and observed program patterns, as well as a robustness test to lend support to the assumption that our share component was plausibly exogenous. First, a CRP contract generally lasts around 10 years. Therefore, enrollment in the prior decade presumably should not have had a direct impact on current biodiversity outcomes given the short life span of passerines. One hypothetical scenario where lagged CRP enrollment might directly influence current biodiversity outcomes is delayed implementation of practices, especially as delays approach 10 years after the initial enrollment. Fortunately, most CRP contracts require participants to install approved practices within a year or risk non-compliance penalties, contract termination, and/or repayment of current and all prior years' benefits (Stubbs 2018; Commodity Credit Corporation 2025). A USDA national survey confirms that planted covers are typically established early in contracts (Allen and Vandever 2003). Second, our study period experienced a steady decline of enrollment as expiring contracts were mostly not renewed under lower national acreage cap (Bigelow et al. 2020). For example, more than 6.9 million CRP hectares under contract expired between fiscal years 2007 and 2014 and were not re-enrolled (Stubbs 2014). In fact, expired CRP land tended to return to crop production (Bigelow et al. 2020), suggesting practices from past enrollment is unlikely to have a persistent effect on current biodiversity outcomes.

Finally, the share exogeneity assumption for a share-based shift-share instrument fails when there are differential pre-trends across units with different exposure shares (Borusyak et al. 2025). That is, conditional on the controls and fixed effects, counties with higher versus lower past CRP exposure would have followed similar trends in bird outcomes in the absence of the episodes of national enrollment cap changes after 2007. Differential pre-trends can appear when either the exposure shares are persistent over time, or there were earlier shocks of the same nature as those used to construct the instrument (Jaeger et al. 2018). We tested for pre-trends by regressing the baseline bird outcomes on our exposure share component (i.e., lagged CRP share) in the period prior to the major enrollment cap changes, while including the same sets of control variables and fixed effects as in Equation (4). We did not observe statistically significant pre-

trends for bird outcomes leading up to the first reduction of the enrollment cap set by the Farm Bill (Appendix Fig. A2).

Together, these institutional details and the balance test provides supporting evidence that the identification and the exclusion restriction assumptions holds in our setting, in which case the estimate β_{CRP}^{IV} from our shift-share IV isolates the local average treatment effect (LATE) for counties whose CRP enrollment responds to the national acreage cap change in proportion to their historical exposure.

Importantly, in addition to the econometric assumptions on our shift-share instrument, any remaining omitted determinants of bird individual and species counts from Equation (1) are assumed not to be systematically correlated with county-level CRP share as predicted by the instrument. It is possible that birding preferences among eBird users change with the number of CRP parcels in the region, thereby affecting the number of individuals or species being observed and reported. However, as we adjusted for the sampling effort and the detectability of birds when estimating the bird outcomes, our findings should not be influenced by changes in birder behavior.

Under these assumptions and conditioning on the weather and land cover controls, we argued that the coefficient β_{CRP}^{IV} was a consistent estimate of the program effect on the county-year abundance/richness that was free of other unobserved confounders. Appendix Table A5 shows that our results are robust to a different model specification to estimating relative abundance/richness an untransformed Poisson regression:

$$\#\text{birds observed}_{kchdmy} = \exp\left(\beta X_{kchdmy} + \Gamma_{cy} + \xi_{h} + \lambda_{m} + \varepsilon_{kchdmy}\right) \qquad (5)$$

The results confirm that dropping zero-count observations in our preferred log-linear model does not bias results.

Methods: Bird Population Conserved under Counterfactual CRP Enrollment Levels

In this analysis, we used the populations of 33 grassland bird species that could be conserved under four counterfactual scenarios, in which CRP lands were reallocated to counties whose 2024 enrollment levels were below their 2007 levels and were subject to alternative acreage or

budget constraints as well as differing allocation priorities (Fig. 5). For each scenario, the counterfactual population was computed based on the following steps.

We began by computing the current (2024) county-level grassland bird population by integrating the eBird Status data (Fink et al. 2023) with the national grassland bird population estimates from Partners in Flight (2024). The eBird Status models provided species-level relative abundance at a 3 km × 3 km resolution, which we aggregated to the county and national scales. For each species, we calculated a county's share of the national relative abundance and multiplied this share by the national population estimate to derive county-species-level population estimates. Considering that our objective was the effect of CRP on the entire grassland bird group rather than single species per se, we summed across species to produce the total grassland bird population for each county. This approach mirrored our processing of checklist-level eBird data in the previous analysis and facilitated the interpretation of the results in the counterfactual analysis. The procedure to compute county-level grassland bird population is described by:

$$\text{Population}_c = \sum_i \text{rel. abundance share}_{i,c} \times \text{Population}_i \qquad (6)$$

where Population_c was the total grassland bird population in county c, $\text{rel. abundance share}_{i,c}$ was the relative abundance share for species i in county c, and Population_i was the national population estimate for species i.

Next, we calculated the percentage difference in grassland bird population for a county c given the counterfactual allocation of CRP land as

$$\Delta\%(\text{Population}_c) = \beta_{CRP}^{IV(\%)} \times \left(\text{CRP Share}_c^{\text{counterfactual}} - \text{CRP Share}_c\right) \qquad (7)$$

where $\beta_{CRP}^{IV(\%)}$ was the percentage change in grassland bird abundance per 1 percentage point change in share of land area enrolled in CRP, an elasticity version of the original β_{CRP}^{IV} estimate on an SD bird – SD CRP scale that we estimated in Equation (4). $(\text{CRP Share}_c^{\text{counterfactual}} - \text{CRP Share}_c)$ was the percentage point difference between current CRP share in county c and that under the counterfactual CRP enrollment level. Combined with the county-level population estimates, we then converted the percentage population difference to the population gain in each county under the counterfactual CRP enrollment level, given by

$$\text{Population conserved from CRP}_c^{\text{counterfactual}} = \text{Population}_c \times \Delta\%(\text{Population}_c). \quad (8)$$

Finally, we applied the above equation to compute the populations of grassland birds conserved under four alternative CRP allocation scenarios. In scenario 1 (Fig. 5a), we identified counties where current CRP enrollment is below the 2007 level and define counterfactual scenario as the additional acreage required for each county to return to that benchmark.[4] From this, we calculated the total additional acreage, associated rental payments, and the population of grassland birds conserved. Rental rate information was unavailable for counties currently without enrolled CRP lands, in which case we used the most recent rate from the last year the county enrolled CRP lands. All rental rates were converted to 2024 dollars using the monthly Producer Price Index. Scenario 2 (Fig. 5b) targeted the same aggregate increase in CRP acreage as scenario 1 but allocated additional lands by prioritizing counties with the lowest CRP rental rates. Each county may enroll up to 25% of its total eligible cropland into CRP, equivalent to about 8% of total land area for the average county in our sample. We began with the lowest-rate county, expanding enrollment until its acreage ceiling was reached, and proceeded sequentially to the next lowest-rate county until the total additional acreage matches that of scenario 1. Scenario 3 (Fig. 5c) followed the same algorithm but continued enrollment until the aggregate rental payment equaled that in scenario 1, rather than total acreage. Lastly, scenario 4 (Fig. 5d) also exhausted the same budget as scenario 1, except it allocated acreage starting with the county offering the highest birds-per-dollar spent. The birds-per-dollar spent for each county was derived from dividing the grassland bird population per hectare conserved under counterfactual CRP acreage allocation in Equation (8) by its per hectare rental rate.

[4] In 2024, 799 counties are below their 2007 enrollment levels. Counties whose 2024 enrolled levels are above the 2007 benchmark are excluded from our counterfactual analyses.

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Tables and Figures

Table 1: Counterfactual Comparison between National and State Constraints

Scenarios	Total Payments ($million)	Total Additional Hectares (million)	Total Birds Conserved (million)
Region-wide Constraints			
Restore	752	4.9	102 [6, 197]
Cost-targeting acreage constraint	242	4.9	92 [6, 178]
Cost-targeting budget constraint	752	10.8	192 [12, 373]
Benefit-cost-targeting budget constraint	752	9.7	219 [14, 425]
State Constraints			
Restore	752	4.9	102 [6, 197]
Cost-targeting acreage constraint	414	4.9	88 [6, 172]
Cost-targeting budget constraint	752	7.7	128 [8, 248]
Benefit-cost-targeting budget constraint	752	6.4	145 [9, 282]

Note: The entries in this table correspond to scenarios under region-wide constraints (Figure 5) and those under state-level constraints (Appendix Figure A1). Region-wide constraint scenarios allocate additional CRP hectares across all counties in our sample based on lowest rental rates or highest birds-per-dollar values, while state-level scenarios apply the same allocation rules within each state's own acreage or budget limits. Total payments, additional hectares, and birds conserved are aggregated across the 13 prairie states for direct comparison. Numbers in the brackets denote 95% confidence intervals for the estimated number of birds conserved.

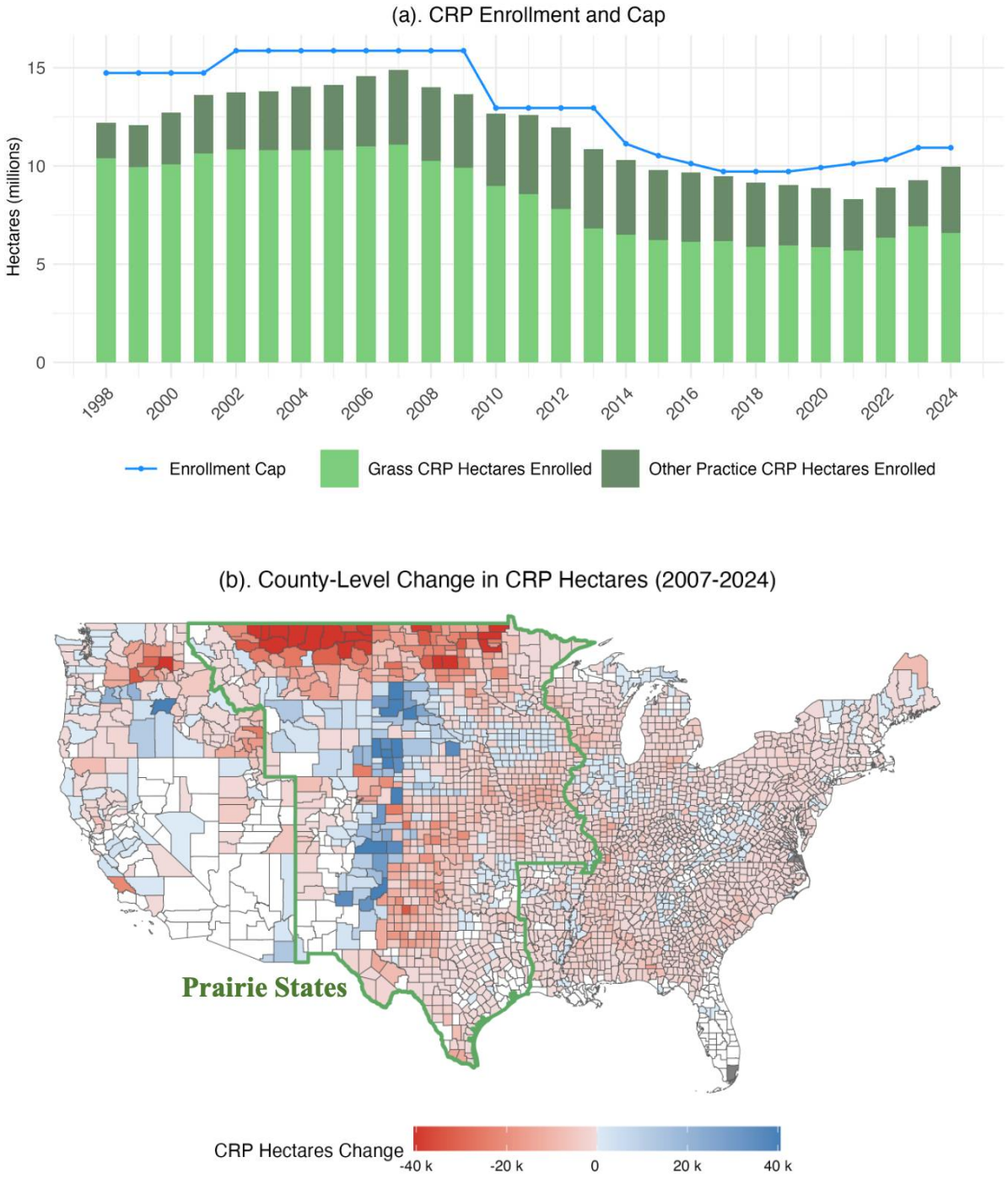


Figure 1(A). Enrollment in the CRP from 1998 to 2024, showing total hectares enrolled (light green plus dark green bars), hectares managed with practices that favor grassland habitat (light green bars), and the program enrollment cap (blue line with dots). (B). County-level change in CRP hectares between 2007 and 2024 across the contiguous United States. Red shading indicates counties with CRP losses, and blue shading indicates gains. The 13 prairie states (Colorado, Iowa, Kansas, Minnesota, Missouri, North Dakota, Montana, Nebraska, New Mexico, Oklahoma, South Dakota, Texas, Wyoming) are outlined in green.

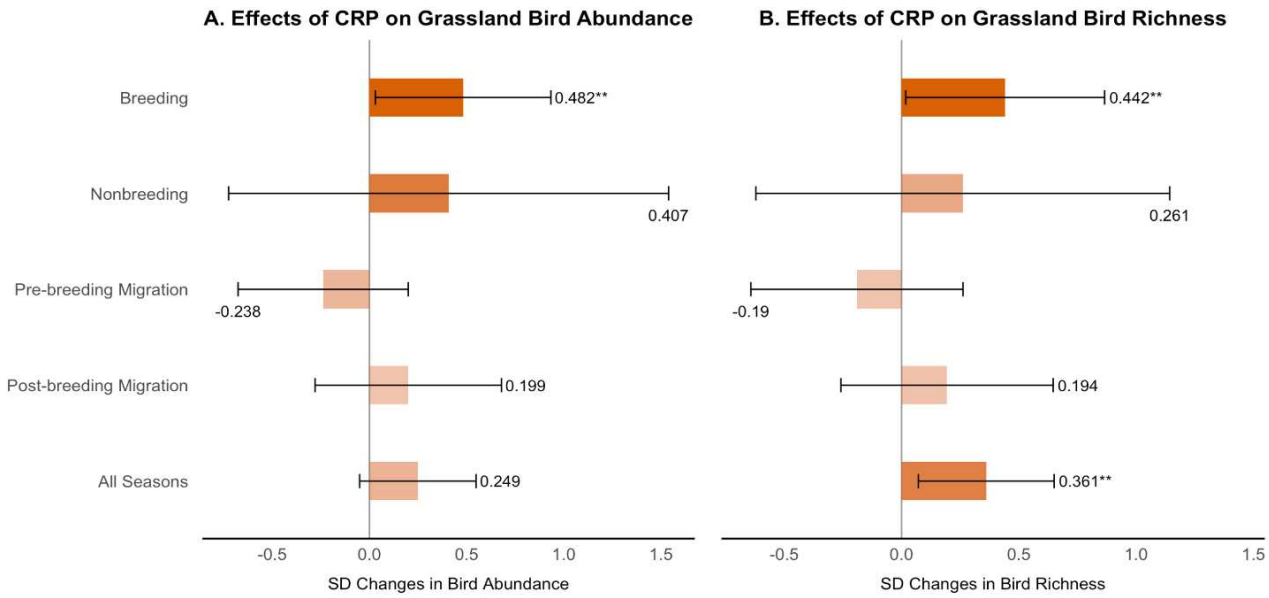


*Figure 2. (A). Effects of the CRP on grassland bird abundance in SD units. (B). Effects of CRP on grassland bird richness in SD units, as calculated using an IV approach. Birds' annual cycle is classified into four season groups (breeding, nonbreeding, pre-breeding migration, and post-breeding migration seasons) and a whole year group. The black bars indicate 95% confidence intervals. SEs are clustered at the county level. Appendix Table A2 reports the IV first stage F statistics in estimating the effect of CRP on grassland bird abundance and richness. ***P < 0.01, **P < 0.05, *P < 0.10.*

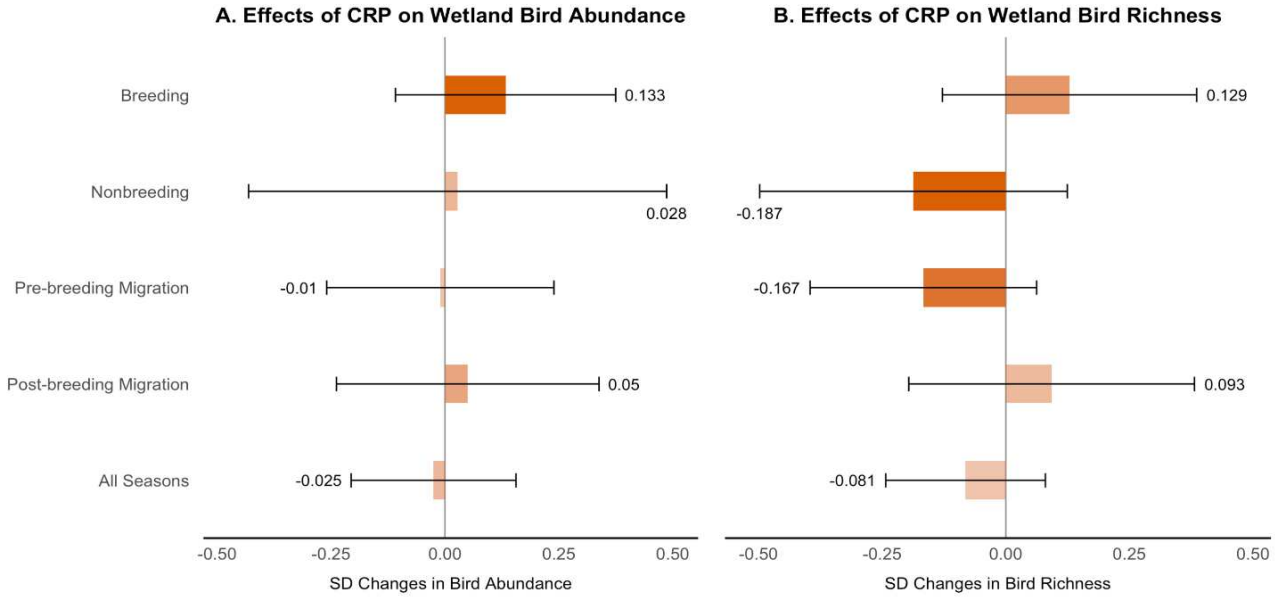


*Figure 3. (A). Effects of the CRP on wetland bird abundance in SD units. (B). Effects of CRP on wetland bird richness in SD units, as calculated using an IV approach. Birds' annual cycle is classified into four season groups (breeding, nonbreeding, pre-breeding migration, and post-breeding migration seasons) and a whole year group. The black bars indicate 95% confidence intervals. SEs are clustered at the county level. Appendix Table A3 reports the IV first stage F statistics in estimating the effect of CRP on wetland bird abundance and richness. ***P < 0.01, **P < 0.05, *P < 0.10.*

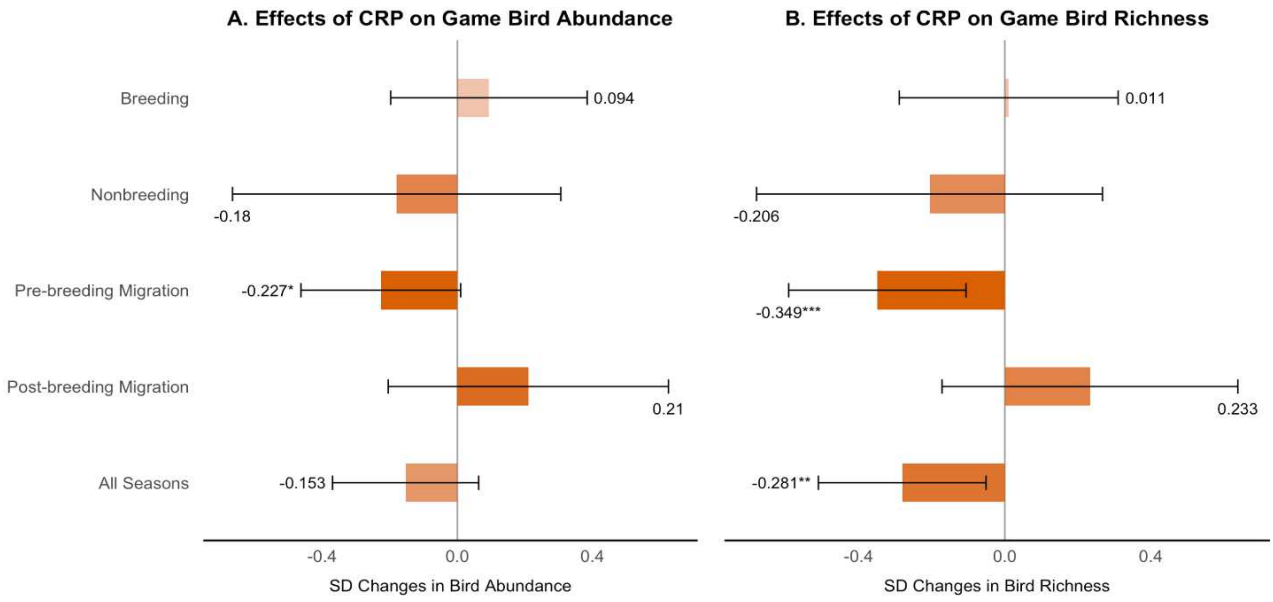


*Figure 4. (A). Effects of the CRP on game bird abundance in SD units. (B). Effects of CRP on game bird richness in SD units, as calculated using an IV approach. Birds' annual cycle is classified into four season groups (breeding, nonbreeding, pre-breeding migration, and post-breeding migration seasons) and a whole year group. The black bars indicate 95% confidence intervals. SEs are clustered at the county level. Appendix Table A4 reports the IV first stage F statistics in estimating the effect of CRP on game bird abundance and richness. ***P < 0.01, **P < 0.05, *P < 0.10.*

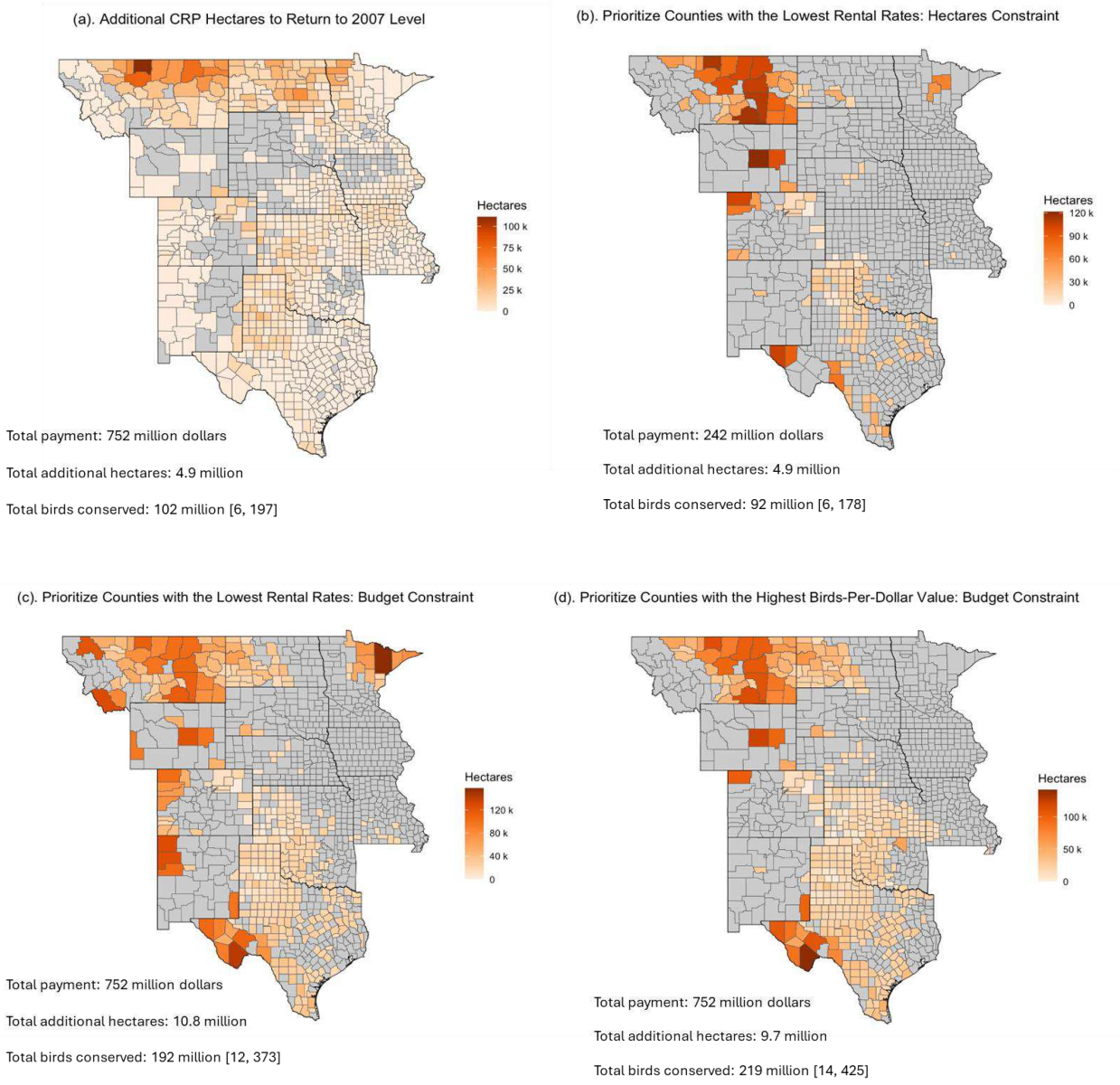


Figure 5 (a). Allocation of hectares needed for counties to return to their 2007 enrollment levels. (b). Hectares needed from counties with the lowest rental rates to recover the same amount of CRP acreage as in (a). (c). Hectares needed from counties with the lowest rental rates to match the same amount of additional CRP expenditure as in (a). (d). Hectares needed from counties with the highest birds-per-dollar value. Numbers in the brackets denote 95% confidence intervals for the estimated number of birds conserved. Note that the sample only includes counties where the current enrollment levels are below their respective 2007 levels.

Supplementary Files

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

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