



## Research Paper

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# Integrating habitat models for threatened species with landownership information to inform coastal resiliency and conservation planning

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**Summary**

Sea-level rise threatens both human communities and vulnerable species within coastal areas. Joint spatial planning can allow conservation and social resiliency goals to work in synergy. We present a case study integrating distribution information of a threatened saltmarsh bird, the eastern black rail (*Laterallus jamaicensis jamaicensis*), with social information to facilitate such joint planning. We constructed a distribution model for the species within an urbanizing coastal region (New Jersey, USA) and integrated this with publicly available parcel and protected area data to summarize ownership patterns. We estimated that c. 0.3–2.8% (c. 260–2200 ha) of available saltmarsh is occupied by eastern black rail, most of which is publicly owned (79%). Privately owned saltmarsh was spread across nearly 5000 individual parcels, 10% of which contained areas with the highest likelihood of rail presence according to our model (top quartile of predicted occupancy probabilities). Compared with all privately owned saltmarsh, parcels with probable rail habitat were larger (median: 5 versus 2 ha), contained more marsh (87% versus 59%) and were less economically valuable (US\$11 200 versus US\$36 100). Our approach of integrating species distributions with landownership data helps clarify trade-offs and synergies in species conservation and coastal resiliency planning.

**Introduction**

Sea-level rise compounds extinction risk for coastal species while posing an existential threat to human communities (Hallegatte et al. 2013, Veron et al. 2019). Actions to protect human life and property in response to sea-level rise, however, can further exacerbate threats to biodiversity, creating a vexing trade-off between climate resiliency and species conservation (Chown & Duffy 2017). Joint spatial planning can help reconcile resiliency and conservation efforts, but this requires the integration of spatially explicit data from disparate domains, namely species biology and occurrence (Maslo et al. 2016), the geography of physical threats (Horton et al. 2018) and the social landscape in which management decisions will occur (Field et al. 2017b). While the landscape of physical threats has been well characterized in the form of sea-level and inundation projections in many locations (e.g., Horton et al. 2018), much less progress has been made integrating the geography of species occurrence and social factors associated with landownership (Field et al. 2017b, Klingbeil et al. 2021). With the continued urbanization of coastlines worldwide, the need to integrate species conservation efforts with complex and heterogeneous landownership patterns has become acute (McGranahan et al. 2007, Hallegatte et al. 2013). We integrate spatial data on coastal saltmarsh ownership and the occupancy of these marshes by a bird of conservation concern (eastern black rail, *Laterallus jamaicensis jamaicensis*) to illustrate a profitable avenue to address coastal conservation planning.

Tidal saltmarshes illustrate the interrelated threat of sea-level rise to human and ecological systems. Rising seas and land subsidence, coupled with the ‘coastal squeeze’ of shoreline development, threaten the long-term persistence of saltmarshes worldwide (Craft et al. 2009). These habitats support threatened vertebrate taxa (Greenberg et al. 2006) and provide protection for human settlements (Narayan et al. 2017). In densely populated eastern coastal regions of the USA, for example, multiple endemic tidal marsh birds are threatened with imminent extinction due to habitat loss and nest flooding associated with sea-level rise (Greenberg et al. 2006, Field et al. 2017a, Klingbeil et al. 2021). At the same time, these coastal marshes provide protective services against tidal flooding, especially during storm surges, which substantially reduce negative impacts to private and public property (Craft et al. 2009, Narayan et al. 2017). As interest grows in restoring saltmarsh habitat for threatened bird species (USFWS 2018, Klingbeil et al. 2021) and protecting human communities (Narayan et al. 2017), there will be increasing

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attention paid to ensuring that these activities do not conflict, but instead provide a ‘win–win’ outcome for people and biodiversity. While the geography of species occurrence (Veloz et al. 2013, Stevens & Conway 2020) and social factors (Lathrop et al. 2014, Field et al. 2017b) in saltmarshes have been investigated independently with respect to the risks posed by sea-level rise, integrating all of these components remains rare despite the potential benefits (Arkema et al. 2017). There is thus a critical need to create spatially explicit frameworks that allow simultaneous consideration of the biological, social and physical risk factors associated with sea-level rise in saltmarshes.

Probabilistic maps of species occurrence represent the most common vehicle for making biodiversity information available to spatial conservation planners (Guisan et al. 2013, Burkhalter et al. 2016, Sofaer et al. 2019). These maps result from statistical models, known as species distribution models (SDMs), and they can be especially useful when species occurrence knowledge is incomplete due to rarity, elusiveness or inaccessibility (Veloz et al. 2013, Stevens & Conway 2020). SDMs are most likely to find planning utility if they are ‘fit for purpose’ and if they transparently represent uncertainty (García-Díaz et al. 2019, Sofaer et al. 2019). ‘Fit for purpose’ in this context suggests that models spatially depict metrics that are easily interpretable for project siting and evaluation of risk reduction and are created and calibrated at spatial scales matching those at which planning decisions will be made (tens or hundreds of hectares; Guisan et al. 2013, Sofaer et al. 2019). SDMs created using site occupancy models (Mackenzie et al. 2017) fitted within a Bayesian framework perform particularly well in this regard (Kéry & Royle 2015, Stevens & Conway 2019), especially in complex coastal environments (Seavey et al. 2011, Maslo et al. 2016, Roach et al. 2017) and for elusive and difficult-to-survey species (Veloz et al. 2013).

Ameliorating, halting or reversing the threats posed by sea-level rise to saltmarshes can require major engineering feats, including the creation of jetties and ‘hardening’ of shorelines (van Slobbe et al. 2013). Less engineered techniques to increase saltmarsh resiliency include sediment deposition, the creation of ‘living shorelines’ and assisting marsh migration into nearby uplands (van Slobbe et al. 2013, Saleh & Weinstein 2016). How such projects are viewed by local communities, their potential costs and their logistical feasibility will depend on the social landscape that is overlaid onto pre-existing physical and ecological geographies (Arkema et al. 2017, Field et al. 2017b, Wigand et al. 2017). While social information can be obtained via questionnaire-style surveys (Field et al. 2017b), ample publicly available social information also exists in the form of landownership (e.g., public or private) and land parcel characteristics data (e.g., size, market value; Newburn et al. 2005). Often derived from tax records, these data can be readily mapped and integrated with ecological and physical factors (Burkhalter et al. 2016). Such an approach has proven useful in identifying barriers and opportunities for conservation planning in terrestrial systems (Newburn et al. 2005, Burkhalter et al. 2016, Robinson et al. 2019). Applying it to coastal resiliency and conservation planning is a natural extension.

Here we present a framework for integrating species conservation and landownership within a spatially explicit framework to inform saltmarsh resiliency planning using a highly pertinent case study of coastal saltmarshes in New Jersey, USA. We created a SDM for eastern black rails within New Jersey, which is the subspecies’ northernmost coastal population stronghold. The resultant habitat suitability map was calibrated at a spatial extent (c. 774 km<sup>2</sup>) and resolution (30 × 30 m) that is the spatial scale most

relevant to local planning decisions. We then integrated this map with landownership information to estimate the proportion of the rail’s saltmarsh habitat that fell within public and private ownership categories. Finally, we characterized privately owned rail habitat based on parcel area and both absolute and per-area net market value, all factors that probably influence the feasibility of engaging landowners in resiliency and conservation efforts (Newburn et al. 2005, Burkhalter et al. 2016, Field et al. 2017b). Our work informs formal regional conservation planning efforts and could serve as a model for other efforts to spatially integrate species conservation and social geography within climate resiliency planning.

## Methods

### *Study area and species*

Our study region included the New Jersey (USA) coast as defined by the National Oceanic and Atmospheric Administration’s 1.2-m (4-foot) mapped flood zone (<https://coast.noaa.gov/slrdata/>). We also added a 200-m buffer around this region to ensure the inclusion of survey locations that occurred on higher-elevation features embedded within saltmarsh habitat (e.g., berms). New Jersey experienced significant saltmarsh loss to land reclamation prior to the 1970s, and its current marsh extent is threatened by rapid sea-level rise exceeding 20 cm since 1970 (Craft et al. 2009, Spanger-Siegfried et al. 2014). Coastal areas are densely settled over much of the state (e.g., land cover fringing Barnegat Bay, the largest water body in the state, is 70% urban), while coastal communities, such as Atlantic City and Cape May, experience among the highest rates of coastal flooding regionally; a rate expected to increase ninefold by 2045 (Lathrop & Bogner 2001, Spanger-Siegfried et al. 2014). Saltmarshes specifically were estimated to have saved over US\$400 million in economic losses to New Jersey’s coastal communities during Superstorm Sandy, which caused billions of dollars in damage in 2010 (Narayan et al. 2017).

Saltmarshes within this region also support one of the four key breeding populations of the eastern black rail, a secretive, nocturnal marsh bird primarily distributed along the Atlantic and Gulf coasts of North America (Watts 2016). The subspecies was recently listed as threatened under the United States Endangered Species Act (USFWS 2018). Eastern black rails were once abundant in New Jersey’s marshes, but after several decades of regional declines in abundance (–9%/year) they are now considered rare (Watts 2016). This decline is thought to be primarily a result of nest losses due to rising sea levels and storm surges, but development and habitat succession have also played a role (Watts 2016, USFWS 2018).

### *Species distribution model*

We modelled eastern black rail occurrence in a Bayesian occupancy modelling framework based on six spatial covariates, three of which were used within a range-wide distribution model for the species (Stevens & Conway 2020) and three additional covariates that were used in a customized, coastal version of the same model constructed by researchers on behalf of the Atlantic Coast Joint Venture (B Stevens & C Conway, personal communication 2020). The variables all represented proportional land cover and were previously ‘scale-optimized’ to represent the most predictive spatial scale of a 100-, 224- or 500-m radius (see Stevens & Conway 2019). These variables included the proportional coverage of high marsh, terrestrial marsh border and impoundments within 500 m from the Saltmarsh Habitat and Avian Research Program (SHARP) marsh zonation classification (Correll et al. 2019);

low-intensity development within 100 m from the Gap Analysis Project (GAP) land-cover dataset (USGS GAP 2016); and scrub-shrub wetland and artificially flooded land from the National Wetlands Inventory (USFWS 2019). There was no artificially flooded land near the survey point locations in our dataset, so we excluded this covariate from our final model. We included a categorical variable 'year' (2008–2019) to account for annual variation in eastern black rail site occupancy. All spatial covariates were represented in raster format with 30 m × 30 m cell resolution created following Stevens and Conway (2019).

We obtained bird survey data from two sources: a structured survey (2015–2019) and the eBird citizen science database (2008–2019). Structured surveys were 10–15-min counts at 373 locations coordinated by state biologists, while eBird data were counts of varying duration performed at 977 locations by citizen scientists. We integrated both sources into a single dataset for analysis (see Supplementary Appendix S1, available online). Due to relatively few eastern black rail detections during the surveys, we only included what we considered to be the two most important variables in the detection probability sub-model: ordinal date (Bobay et al. 2018) and duration of survey. Thus, the structure for the detection probability and occupancy sub-models were as follows:

$$\text{Logit}(p) \sim \text{intercept} + \alpha_1 * \text{ordinaldate} + \alpha_2 * \text{duration} \quad (1)$$

$$\begin{aligned} \text{Logit}(\psi) \sim & \text{intercept} + \text{sum}(\beta_i * \text{year}_i) + \beta_1 * \text{shrub cover} \\ & + \beta_2 * \text{high marsh} + \beta_3 * \text{terrestrial border} \\ & + \beta_4 * \text{impoundment} \end{aligned} \quad (2)$$

where  $p$  is detection probability, the  $\alpha$  values are coefficients for the two detection probability covariates,  $\psi$  is occupancy probability, the  $\beta$  values are slopes for the occupancy probability covariates and  $\text{year}_i$  represents binary ('dummy') variables indicating each year ( $i$ ) included in the survey.

We used JAGS within the R programming environment (Kellner 2019, R Core Team 2020). Non-informative prior distributions were assumed for all parameters (Kéry & Royle 2015, p. 607). Point estimates and credible intervals for the mapped prediction surface were calculated as the median, 2.5th and 97.5th quantiles of posterior distributions based on 3000 posterior draws each (3 chains, each with 10 000 burn-in and 10 000 subsequent iterations, thinned by 10). Model convergence was assessed using the Gelman–Rubin statistic ( $R\text{-hat} < 1.1$ ) as well as the examination of trace plots.

We used our model to predict eastern black rail occupancy over all emergent tidal marsh habitat in the state as defined by the National Wetlands Inventory (774 km<sup>2</sup>). Thus, the resulting map consisted of 860 298 grid cell values (30 × 30 m), each containing a measure ranging from 0 to 1 that can be interpreted as estimates of the true probability of eastern black rail occurrence within hearing distance of each location. All model predictions were made for 2015, the year with the greatest amount of data (Table 1). Following Kéry and Royle (2015), we summed cell values and then multiplied by the cell area (0.09 ha) to obtain estimates of 'area of occurrence'. However, as the area surveyed per point is larger than the cell size of our map, we acknowledge that this is actually an estimate of the area from which the rails can be heard by an observer. Thus, we term this metric the 'coarse-scale' area of occurrence ( $A_c$ ), in contrast to the 'fine-scale' area ( $A_f$ ) that describes land actually inhabited by the rails. We estimated  $A_f$

**Table 1.** Number of survey locations (survey points or eBird stationary checklist locations) visited and eastern black rail (*Laterallus jamaicensis jamaicensis*) detections by year in New Jersey, USA, saltmarshes.

Year	Structured survey data		eBird data	
	Detection	Non-detection	Detection	Non-detection
2008	–	–	1	37
2009	–	–	3	37
2010	–	–	1	63
2011	–	–	1	62
2012	–	–	0	82
2013	–	–	0	117
2014	–	–	1	121
2015	11	226	3	169
2016	19	160	0	197
2017	–	–	2	191
2018	1	39	1	207
2019	0	44	0	279

by assuming a series of maximum distances from which rails can typically be heard ( $d$ ; 200–500 m) and eastern black rail home range sizes ( $r$ ; 3–4 ha; see Table S1). We then produced a range of plausible values for  $A_f$  using the following formula to rescale estimates:

$$A_c \times r / (\pi d^2) \quad (3)$$

### Integrating landownership data

We downloaded spatial data for federal, state, county and local government-owned and non-profit-owned protected areas and the MOD-IV spatial tax parcel database from the Marine Protected Area database (<https://marineprotectedareas.noaa.gov/>) and from state websites (State of New Jersey 2018, NJDEP 2021), respectively. We created a spatial layer of non-protected parcels by subtracting protected area layers from the tax parcel database. To identify parcels that were unprotected and privately owned, we manually scanned and performed text queries to the owner field of tax records to identify and extract all parcels owned by federal, state, county or local governments or non-profits. These parcels were assigned to the appropriate public or non-profit category, ensuring that the final private ownership dataset consisted exclusively of parcels owned by private individuals or corporations.

We used the landownership spatial layer that we created to characterize modelled eastern black rail habitat in relation to ownership categories, as well as to summarize the salient characteristics of privately owned parcels. We summed the cell values in our predicted habitat suitability map to estimate the total amount of eastern black rail habitat occurring in each ownership class (i.e., the various 'area of occurrence' metrics described above). To characterize private parcels, we first filtered out 5793 parcels (134 ha) that did not contain at least 0.1 ha of tidal marshland with the rationale that these were mainly small developed 'edge' parcels that would be unlikely targets for resiliency or restoration actions. For each remaining parcel, we quantified the total area, the proportion of area that is saltmarsh, the tax-assessed property value and the assessed value per hectare. These characteristics were chosen as potential factors likely to influence future coastal resiliency, restoration or conservation prioritization efforts. We then repeated this assessment for each quartile of mapped occupancy probability grid cells, representing areas where our model predicts rails to be least (bottom quartile) to most (top quartile) likely to occur.



## Results

Maps resulting from our occupancy model revealed hotspots of predicted rail occurrence as well as uncertainty in those predictions (Fig. 1). Model coefficients predicted greater occupancy probability with increasing high marsh cover and lower developed cover (Fig. S4). Coefficients for other variables had greater uncertainty (e.g., 80% credible intervals overlapped zero) but predicted greater occupancy with lower shrub cover and slightly higher occupancy with increasing terrestrial border and impoundment cover (Figs S2 & S4). Detection probability decreased over the course of the season and increased with increasing survey duration (Fig. S4). A second occupancy model run without eBird data produced similar estimates but with wider credible intervals (Figs S2 & S3). A third version of the model that treated year as a continuous variable also yielded similar results and revealed a negative trend: a  $-9.8\%$  annual decrease in the odds ratio of occupancy (95% credible interval (CI) =  $-20.5\%$  to  $+0.8\%$ ; Table S2).

Coarse-scale area of occurrence ( $A_c$ ), or the area from which eastern black rails could be heard, was estimated at 6915 ha (95% CI: 3746–12 144) or c. 9% of New Jersey's 77 427 ha of emergent tidal marsh. Estimates of fine-scale area of occurrence ( $A_f$ ), or the area actually occupied by rails, had a mean of 893 ha among the eight scenarios evaluated (range: 264–2201 ha) or 1.2% of all salt-marsh (range: 0.3–2.8%; Table S1). Of occupied eastern black rail habitat, our model estimated that 24% (213 ha in terms of mean fine-scale area of occurrence) occurred on land owned by federal agencies, 45% (399 ha) by the state, 1% (10 ha) by counties, 5% (46 ha) by cities or municipalities, 4% (39 ha) by non-profit organizations and 21% (184 ha) by private owners (see Table S1 for area estimates under different assumptions). When we divided the mapped grid cells (i.e., Fig. 1a) into categories (quartiles) based on predicted suitability, we found that rates of public ownership were markedly higher in marshes with higher predicted occupancy probabilities (Fig. 2). The rate of private ownership was three times higher in areas with the lowest predicted occupancy (i.e., the bottom quartile of cells) compared with the highest predicted occupancy (top quartile; 39% versus 12%), while federal and state ownership showed the opposite pattern (federal: 11% versus 33%; state: 37% versus 47%).

Overlaying parcel data revealed that 4871 privately owned tracts contained emergent tidal marsh, 3566 of which had valuation information available. These parcels averaged 9.2 ha in size (median 2.5 ha; range 0.1–1480 ha) and contained an average of 57% emergent tidal marsh (median 59%; range 0.3–100%; Fig. 3). Excluding 36 parcels with no tax value in the database, we found that saltmarsh parcels had an average net value of US\$630 277 (median US\$36 100; range US\$100–150 630 500) and an average net value per hectare of US\$248 076 (median US\$9830; range US\$12–\$15 623 758). We found a positive relationship between net value and area and a negative relationship between net value and percentage marshland (Fig. 3).

Of all privately owned parcels that contained saltmarsh, 10% (487) contained marsh classified in the top quartile of mapped grid cells, where our model predicted eastern black rails were most likely to occur. Of these, 382 had detailed valuation information available. Private parcels containing top-quartile saltmarsh were larger (median: 5 versus 2 ha), contained more marsh (87% versus 59%) and were less costly (net value: US\$11 200 versus US\$36 100; per ha net value: US\$1974 versus US\$9830) than the pool of all privately held saltmarsh-containing parcels. The total value for the parcels containing top-quartile grid cells was over

US\$48 million, or 2% of the total value of all marsh-containing private parcels with known value (over US\$2 billion).

## Discussion

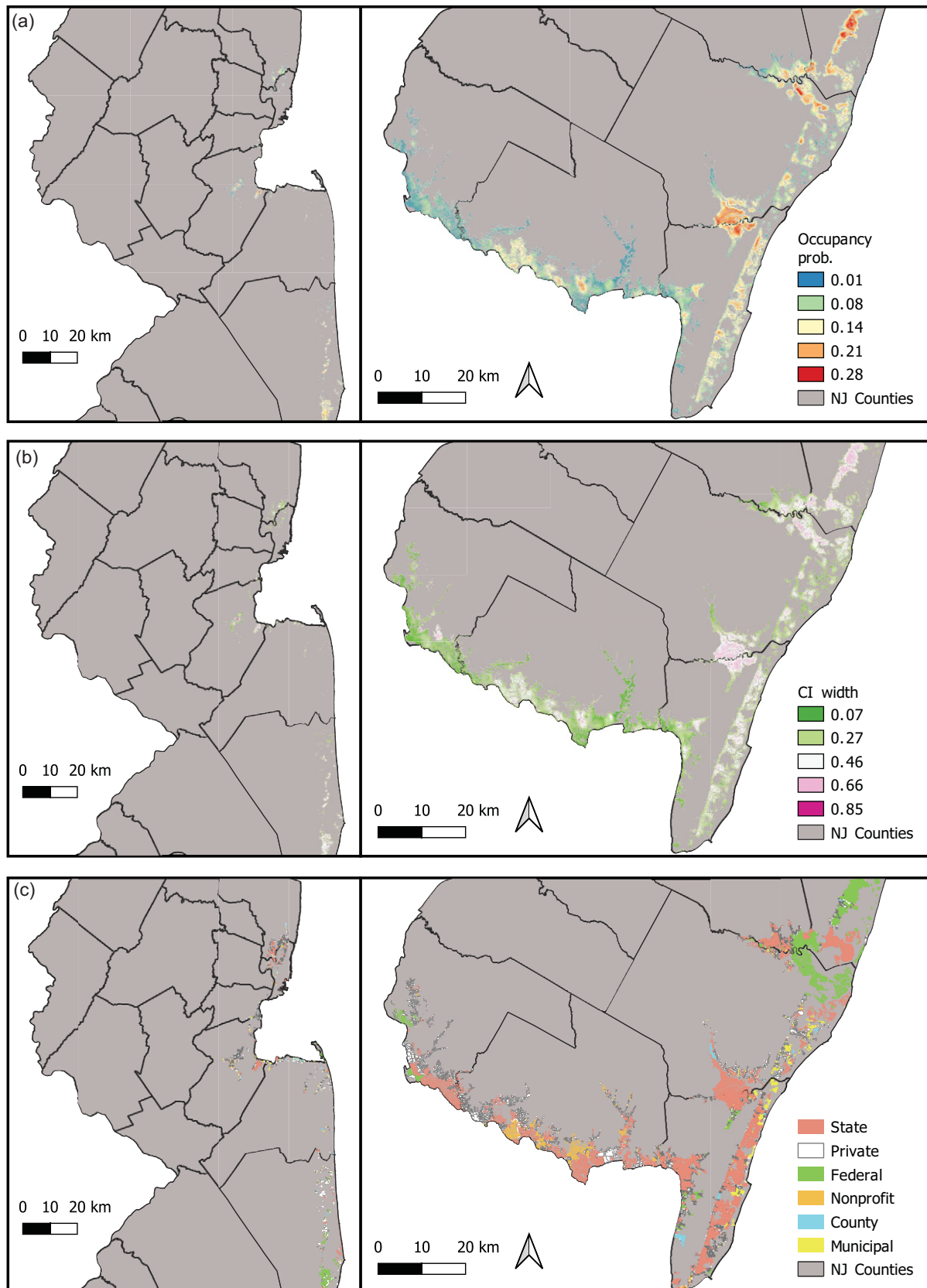
While there has been progress integrating ecosystem restoration into resiliency programmes that safeguard human communities (e.g., 'nature-based' solutions; van Slobbe et al. 2013, Arkema et al. 2017), mechanisms that explicitly integrate the needs and spatial locations of threatened species into this process are rare. Our case study synthesizes social and ecological information for the purpose of joint resilience planning for threatened species and human communities in a densely settled coastal landscape under active threat of sea-level rise. We estimated the amount of rail habitat under private and public ownership (21% and 79%, respectively) and produced quantitative summaries of privately owned marsh parcels. Notably, we showed that private parcels containing habitat identified as most likely to contain eastern black rails were on average larger and less expensive than other parcels. While our case study involved a single species and region, the framework has relevance for clarifying and integrating coastal conservation and resiliency planning efforts worldwide.

Bayesian occupancy models produce a measure of habitat usage that is readily interpretable by stakeholders: the expected probability of occurrence. This approach contrasts with other methods that do not explicitly model imperfect detection and therefore can only represent relative quantities of habitat suitability (Guisan et al. 2013, Kéry & Royle 2015). In the case of the eastern black rail, we predicted habitat occupancy patterns for 'the most secretive of the secretive marsh birds' (Watts 2016, p. 1) across 774 km<sup>2</sup> of difficult-to-survey saltmarsh at a spatial resolution fine enough for most planning applications (30 m × 30 m). Fitting the model locally adds confidence that modelled habitat associations are spatially consistent and do not differ among populations in different ecoregions (e.g., Roach et al. 2017). Furthermore, estimates of occurrence probability produced by our occupancy model allowed us to directly estimate the area of occurrence of eastern black rails (along with measures of uncertainty) in various landownership categories. This readily interpretable metric (area) is a common currency for many coastal resilience efforts and conservation actions, such as marsh restoration and land preservation.

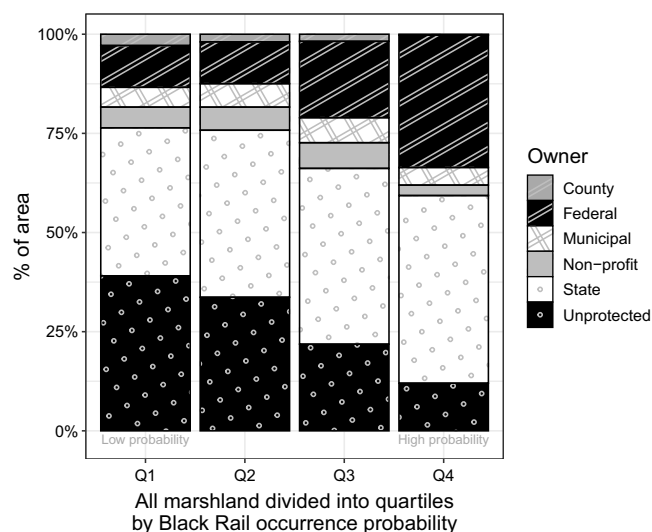
Integrating maps of vulnerable saltmarsh bird species habitat into existing spatial resilience planning workflows (e.g., Veloz et al. 2013, Wigand et al. 2017) is only the first step towards actually implementing the projects that could jointly benefit these species as well as human communities. The cooperation of private landowners and the costs of implementation are also key considerations that will impact the likelihood of success (Field et al. 2017b, Wigand et al. 2017). Important insights into landowner willingness to participate may be gained by mapping the cultural landscape on which resilience and restoration actions are proposed and, more generally, in viewing such actions through the lens of social-ecological systems (Fischer et al. 2021). This framework considers human motivations and actions along with physical drivers, restoration efforts and threatened species ecology in a unified causal network (Allen et al. 2021, Fischer et al. 2021).

While our approach of summarizing ownership and parcel characteristics represents a relatively simple form of social information, it may have real implications for implementing conservation and resiliency efforts. Our finding that parcels containing likely rail habitat were larger means that consensus among fewer landowners may be required in discussions regarding private-lands





**Fig. 1.** Maps of (a) modelled eastern black rail (*Laterallus jamaicensis jamaicensis*) occurrence probability, (b) uncertainty in those estimates and (c) landownership patterns within emergent tidal saltmarshes in New Jersey (NJ), USA. In each map, the left panel shows the northern portion of New Jersey, while the right panel shows the southern portion. CI = credible interval (95%).



**Fig. 2.** Ownership status of New Jersey, USA's 774 km<sup>2</sup> of emergent tidal saltmarsh, divided into quartiles from lowest to highest predicted probability of occurrence for the eastern black rail (*Laterallus jamaicensis jamaicensis*).

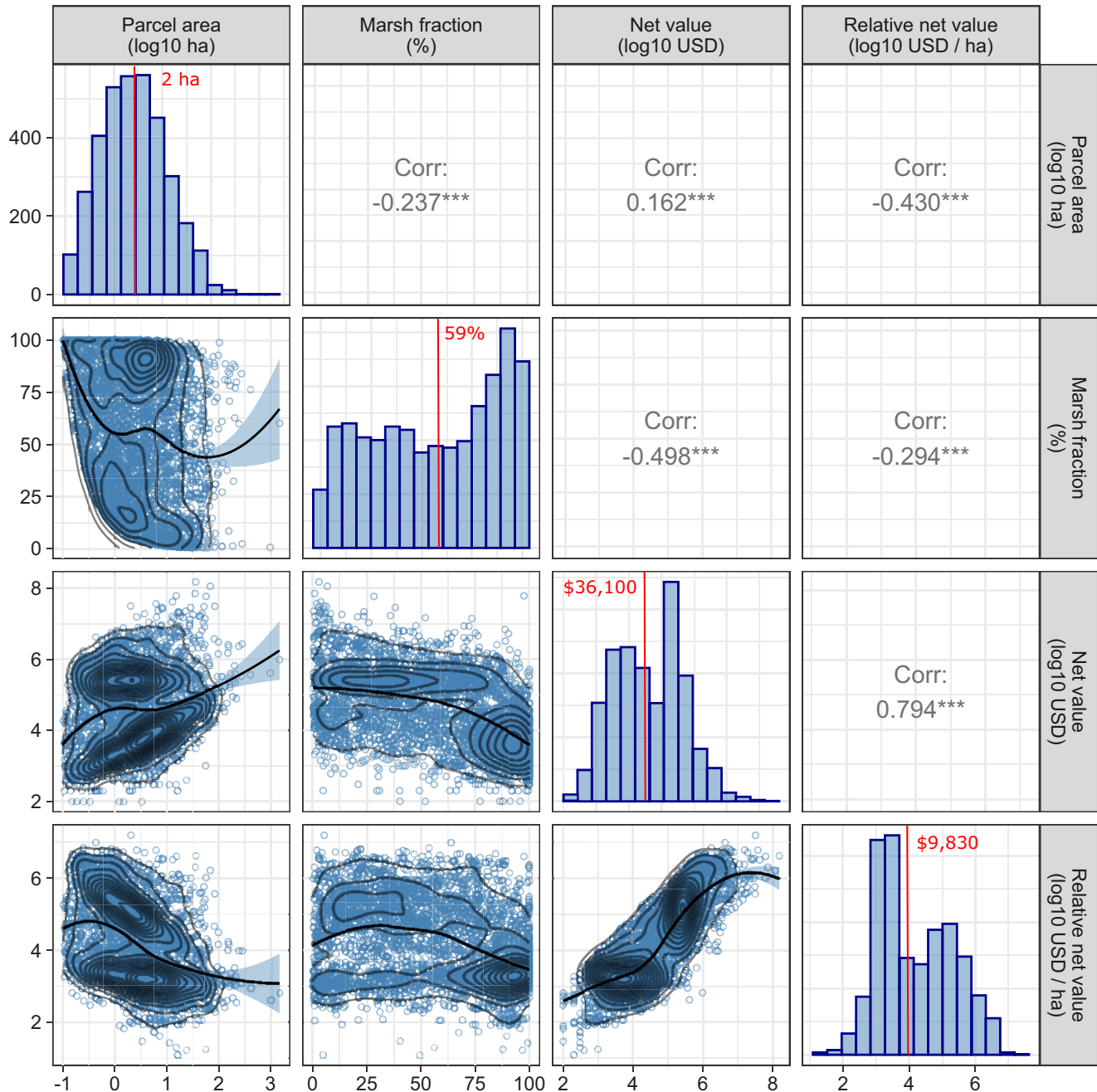
incentive programmes or similar actions to jointly address coastal resiliency and conservation. Furthermore, other factors that we measured, such as land value or the fraction of a parcel that is saltmarsh, could conceivably be correlated with landowner willingness to cooperate in such programmes. Indeed, attitudes and beliefs regarding saltmarsh restoration (as revealed in mail-in surveys) have been shown to exhibit geographical patterns (Field et al. 2017b). However, connecting the dots between the geography of landownership patterns (including parcel characteristics), stakeholder attitudes and beliefs and effective coastal resiliency and conservation strategy is far from straightforward. There are limits to how much researchers or planners can or should infer from such data without a deeper understanding of the social and cultural factors influencing landowner–land relationships, an understanding best obtained through direct stakeholder engagement (Field et al. 2017b, Wigand et al. 2017). In the end, successful resiliency and conservation efforts should recognize that private saltmarsh is held by a diverse range of individuals, trusts and small to large businesses and corporations, many of which have complex connections to the land and water resources of the area (Wigand et al. 2017).

The monetary value of parcels is another social factor with clear implications for conservation and resiliency project implementation. At the most basic level, our distinction between public and private lands, coupled with the tax-assessed valuation of properties, can give planners an idea of the total economic value of private lands that are likely to be included in eastern black rail conservation or sea-level rise resiliency efforts (Burkhalter et al. 2016, Klingbeil et al. 2018). Our analysis estimated that the combined value of privately held land most likely to support eastern black rails was at least US\$48 million or only c. 2% of the total valuation of all marsh-containing properties. While this number is likely to be an underestimate (some parcels lacked valuation information), and while tax-assessed value may not perfectly reflect market value, the proportionally low economic value of likely rail habitat relative to that of all marsh-containing properties may be viewed as an encouraging sign from a conservation and resiliency planning perspective. The value of land may correlate with the resources required, not just for outright land purchases, which can be

controversial if implemented without sensitivity to the community, but also for incentive-based tools within privately owned saltmarshes. In either case, achieving coastal resiliency and conservation goals within a patchwork of public and privately owned saltmarsh will still probably prove to be a difficult challenge. Surveys of private landowners along nearby Long Island Sound (USA) have revealed extreme reluctance to participate in habitat restoration incentive programmes that support endangered saltmarsh birds and offer property protection, even when the programmes offered monetary compensation (Field et al. 2017b). This, again, points to the vital importance of supplementing ‘arm-chair’ investigations such as ours with direct landowner engagement to find mutually beneficial arrangements for all parties.

Though we found a relatively high proportion of eastern black rail habitat in public ownership (79%), the complexities of managing habitat in dynamic urbanized coastal environments are formidable. With ongoing sea-level rise, habitat that is suitable today may not be suitable in 30 years’ time, and vice versa (Seavey et al. 2011, Veloz et al. 2013). In addition, coastal systems are uniquely connected in terms of hydrology and sedimentary dynamics. Thus, actions taken to conserve or restore a publicly owned saltmarsh parcel, such as applications of thin-layer sediment, prescribed burning or changes to hydrology, will often impact the dynamics of adjacent privately owned parcels. Therefore, close coordination is required among a range of stakeholders, including private landowners (Wigand et al. 2017). Integrating predictive habitat maps with tax parcel cost information, as we have done here, has the potential to inform planners exactly as to where areas of fragmented private ownership overlap with areas of interest for conservation or resiliency actions, and also to inform them about where their financial investments may go the farthest to ensure species protection and resilient communities.

Our study illustrates an approach for integrating species habitat with social information so that they can be jointly considered in a spatially explicit coastal resiliency planning workflow. Such integration is particularly needed in saltmarsh ecosystems exposed to sea-level rise, where restoration and conservation is increasingly necessary, expensive and heavily dependent on stakeholder input (Veloz et al. 2013, Field et al. 2017b, Wigand et al. 2017, Klingbeil et al. 2021). Decisions to site sea-level rise marsh resiliency projects do not always take into consideration the presence of threatened species, which runs the risk of such efforts conflicting with the needs of these species, at least in the short term (Chown & Duffy 2017, Wigand et al. 2017, Klingbeil et al. 2021). Alternatively, marsh resiliency projects might restore or create marsh where no threatened species are likely to occur even if there are payoffs for local human communities. Similarly, an approach to marsh restoration that ignores the geography of social factors risks implementing projects to aid threatened species in locations that are socially undesirable. Integrated planning is needed to allow conservationists to communicate with engineers, planners and other coastal stakeholders. We posit that species occupancy models can enable real-world spatial planning applications that take the habitat needs of threatened species into account (e.g., within local spatial planning applications such as NJFloodMapper.org; Lathrop et al. 2014). Of course, even the clearest and most accurate quantitative maps of species distribution may still fail to engender buy-in without direct engagement and communication with stakeholders, both during and after the model construction process (Wigand et al. 2017, Sofaer et al. 2019). Ultimately, all coastal resources



**Fig. 3.** A scatterplot matrix showing characteristics for 3566 tax parcels that contain saltmarsh habitat in New Jersey, USA. ‘Parcel area’ is the total area of each parcel in hectares; ‘Marsh fraction’ is the percentage of each parcel that is saltmarsh; ‘Net value’ is the assessed valuation according to the state’s MOD-IV tax parcel database; and ‘Relative net value’ is the assessed valuation per hectare. Panels below the diagonal show scatterplots with locally weighted smoothing lines ( $\pm 95\%$  confidence intervals). Panels above the diagonal show Spearman’s rank correlations ( $***p < 0.001$ ). Panels along the diagonal are histograms with solid vertical lines showing the median. Scatterplots and histograms all share an  $x$ -axis within each column, the description for which is at the top of the figure. The  $y$ -axes are shared by all scatterplots within each row, with the label occurring at the right side of the figure. The upper-left  $y$ -axis shows the scale for all histograms (i.e., the frequency of observations in each bin). Where indicated, axes show  $\log_{10}$  values, such that 0 = 1, 1 = 10, 2 = 100, 3 = 1000, etc.

threatened by sea-level rise would benefit from the fullest range of interacting components being placed on the map and thus made visible to all participants in the planning process.

**Supplementary material.** For supplementary material accompanying this paper, visit <https://doi.org/10.1017/S037689292200039X>.

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