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From declines to recovery: 52 years of changes in autumn migratory songbird abundance at an island stopover site in southern New England

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ABSTRACT

Migratory birds are declining across North America, and many studies have used long-term datasets to estimate trends in abundance for migratory birds. To prioritize at-risk populations most efficiently, it is critical to know the current trend status of migratory species, which involves accurately assessing the most appropriate timescale for monitoring changes in populations. In this study, we analyzed autumn bird banding data from the Block Island Banding Station on Block Island, Rhode Island, USA, over a 52-year period (1970–2021), focused exclusively on hatch-year birds, to examine long-term abundance trends for 22 species of migratory songbirds and compared our results to trends from Manomet Conservation Sciences and the USGS Breeding Bird Survey. We ran 4 models (no change, linear, quadratic, and breakpoint) for each species and used model selection with AIC_c to determine the best model. Eighteen of the 22 species were best represented by breakpoint models, indicating that they experienced a sudden change in slope during the study period. All 18 breakpoint species experienced the largest decline in the first 2 decades of the study (between 1976 and 1986), and 17 of 18 species were stable or recovering after the breakpoint. Of the 4 species without a significant breakpoint, 1 had no change over the study period, 2 were best represented by linear models, and 1 was best represented by a quadratic model. Trend classifications varied across time, methods, and the geographic regions of the subpopulations sampled. These results present evidence that despite initial declines, many species are currently stable or in recovery. We also highlight that modern conservation efforts need to account for abrupt changes in trend direction within long-term time series analyses to most accurately assess current abundance trends for the conservation of vulnerable species.

Keywords: abundance, bird banding, breakpoint, conservation, migration, recovery, songbird

How to Cite

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LAY SUMMARY

- Migratory songbirds have declined across North America, although the extent of decline and resulting conservation status depend on the chosen timescale.
- We assessed long-term changes (1970–2021) in relative abundance of hatch-year migratory songbirds of 22 species at the Block Island Banding Station on Block Island, Rhode Island, USA, and compared these to trends from another long-term coastal banding station and the North American Breeding Bird Survey.
- We evaluated 4 models (no change, linear, quadratic, and breakpoint) for 22 species and found that trends for 82% of species had breakpoints indicating sudden changes in trajectory over the 52 years.
- The pattern for these 18 species was a sharp decline in the 1970s and 1980s followed by stabilization, recovery, or slower decline for the remainder of the study. Only 2 species declined throughout the entire study.
- Conservation prioritization schemes should consider these abrupt changes in long-term trends when determining which species are most vulnerable.

De los declives a la recuperación: 52 años de cambios en la abundancia de aves canoras migratorias de otoño en un sitio de parada insular en el sur de Nueva Inglaterra

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RESUMEN

Las aves migratorias están disminuyendo en toda América del Norte, y muchos estudios han utilizado bases de datos a largo plazo para estimar las tendencias de abundancia de estas aves. Para priorizar de manera eficiente las poblaciones en riesgo, es fundamental conocer el estado actual de las tendencias en las especies migratorias, lo cual implica evaluar con precisión la escala temporal más adecuada para monitorear los cambios poblacionales. En este estudio, analizamos datos de anillado de aves de otoño provenientes de la Estación de Anillado de Block Island, en Block Island, Rhode Island, EE.UU., durante un período de 52 años (1970–2021), centrándonos exclusivamente en aves de hasta un año, para examinar las tendencias de abundancia a largo plazo de 22 especies de aves canoras migratorias y comparamos nuestros resultados con las tendencias obtenidas por el centro de Ciencias de la Conservación de Manomet y el Censo de Aves Reproductoras del Servicio Geológico de Estados Unidos. Ejecutamos 4 modelos (sin cambio, lineal, cuadrático y de punto de quiebre) para cada especie y utilizamos la selección de modelos con el criterio de información de Akaike corregido (AICc por sus siglas en inglés) para determinar el modelo más adecuado. Dieciocho de las 22 especies fueron mejor representadas por modelos de punto de quiebre, lo que indica que experimentaron un cambio repentino en la pendiente durante el período de estudio. Las 18 especies con punto de quiebre experimentaron el mayor descenso en las dos primeras décadas del estudio (entre 1976 y 1986), y 17 de las 18 especies estuvieron estables o en recuperación después del punto de quiebre. De las 4 especies sin un punto de quiebre significativo, 1 no presentó cambios durante el período de estudio, 2 estuvieron mejor representadas por modelos lineales, y 1 estuvo mejor representada por un modelo cuadrático. Las clasificaciones de tendencias variaron según el tiempo, los métodos y las regiones geográficas de las subpoblaciones muestreadas. Estos resultados ofrecen evidencia de que, a pesar de los descensos iniciales, muchas especies actualmente están estables o en recuperación. También destacamos que los esfuerzos actuales de conservación deben considerar los cambios abruptos en la dirección de las tendencias dentro de los análisis de series temporales a largo plazo para evaluar con mayor precisión las tendencias actuales de abundancia para la conservación de especies vulnerables.

Palabras clave: abundancia, anillado de aves, aves canoras, conservación, migración, punto de quiebre, recuperación

INTRODUCTION

Bird populations in North America are declining at an alarming rate, with an estimated loss of 3 billion birds since the 1970s (Rosenberg et al. 2019). This monumental loss has disproportionately affected migratory birds (which account for over 80% of the decline) due in part to the heightened risks and physiological strain associated with migration (Newton 2006, 2024, Rosenberg et al. 2019). With ever-increasing environmental pressures on migratory bird species (Rosenberg et al. 2019, Nemes et al. 2023, Newton 2024), it is critical to assess the trajectories of abundance over time to determine which species are presently at the greatest risk and how to best meet the needs of at-risk populations and species.

Accurately assessing abundance trends first requires defining an appropriate timescale because this choice can drastically impact the conclusions that are drawn from time series data (Wolkovich et al. 2014, Ryo et al. 2019). The choice of timescale is critical across fields, from economics to medicine to climate science, as it shapes whether patterns can be detected and how these patterns are interpreted. For example, short-term trends in climate data or economic markets poorly indicate longer-term trends on which important policy and investment decisions rely (Da and Warachka 2011, De Elía et al. 2014). In ecology, timescale choice is particularly critical when studying population dynamics (Frankham and Brook 2004, Wolkovich et al. 2014, White 2019, Ryo et al. 2019) because natural population cycles and regime shifts are typically only evident with longer time series (Hill and Hagan 1991, Scheffer et al. 2001, Folke et al. 2004, Andersen et al. 2009, Pélissié et al. 2024). Given that time series data are used in species prioritization, conservation planning, and wildlife management decision-making (García-Barón et al. 2021), using the wrong timescale can lead to misinterpretations of the *current* vulnerability status of avian species, particularly when abrupt changes or recoveries are overlooked (Hill and Hagan 1991). Thus, defining the status of populations requires both a long-term perspective and identifying appropriate time periods that best reflect the current state of the population of interest.

Bird banding data from long-term banding stations, many of which have been operating since the 1960s, are one of the most consistent sources of long-term data on migratory birds (Osenkowski et al. 2012, Dunn 2016, McDermott and DeGroote 2017, Kamm et al. 2019). Long-running bird banding stations present an invaluable data resource to estimate population-level trends in abundance, understand the effects of climate change and other anthropogenic threats on a diversity of bird species, and examine how changes in the environment may alter bird condition, behavior, and abundance, all of which must be understood to conserve threatened species effectively (Osenkowski et al. 2012, Dunn 2016, Kamm et al. 2019). Additionally, comparing abundance trends derived from other systematic, long-term studies such as the United States Geological Survey (USGS) North American Breeding Bird Survey (BBS) with long-term bird banding data can paint a clearer regional picture of which species are declining over certain timescales, how consistent trends are across species' ranges, and which species should be prioritized in conservation and management efforts (Lloyd-Evans and Atwood 2004, Osenkowski et al. 2012, Kamm et al. 2019).

For this study, we analyzed 52 years (1970–2021) of banding data collected in the autumn for hatch-year (HY) individuals of 22 migratory songbird species at the Block Island Banding Station (BIBS), Rhode Island, USA. Given the high proportion of HY birds captured at BIBS (>97% of captures), we had a unique opportunity to assess HY abundance independently of adult abundance on Block Island. A focus on HY birds specifically may provide greater insights into changes in relative productivity over time that could be diluted by including adult birds in the analysis and can provide complementary insights into long-term changes in bird abundance when compared to datasets that use different survey methods and sample different demographics and subpopulations. Our objectives were (1) to detect trends for each species and quantify whether abundance changed gradually or abruptly over time, (2) to classify each of these trends into categories based on their current status (increasing, declining, or stable) that are relevant for conservation, and (3) to compare these trends to abundance trend estimates from Manomet Conservation Sciences (Kamm et al. 2019), a geographically similar bird banding station in the northeastern USA, and the BBS (Ziolkowski et al. 2023) in an effort to better understand regional trends of migratory songbirds.

METHODS

Study Site

We conducted fieldwork at the BIBS, located on Block Island, Rhode Island, USA. Block Island (25 km^2) is located in the Block Island Sound, 17.5 km south of mainland Rhode Island. Block Island was formerly predominantly forested, while the present landscape is the product of hundreds of years of human habitation and modification, beginning with the original inhabitants, the Manissean Indigenous Tribe (Hammond 1992, Rosenzweig et al. 1992). Block Island is currently dominated by a maritime shrub community comprised of mostly native fruiting shrubs such as *Viburnum dentatum* (northern arrowwood), *Parthenocissus quinquefolia* (Virginia creeper), *Ilex verticillata* (winterberry), *Arionia melanocarpa* (black chokeberry), and *Myrica pensylvanica* (northern bayberry), as well as invasive shrubs including *Celastrus orbiculatus* (oriental bittersweet) and *Rosa multiflora* (multiflora rose) (Reinert et al. 2002, Osenkowski et al. 2012). The maritime shrubland on Block Island covers about one-third of the island, and is exposed to ocean winds and salt spray that, along with the well-drained glacial outwash and till soils, maintains the shrubland habitat (Rosenzweig et al. 1992).

Block Island serves as a key stopover site for migratory songbirds, >97% of all captures being HY birds undertaking their first migration, with the highest numbers passing through during autumn migration (Baird and Nisbet 1960, Reinert et al. 2002). Despite the significant contribution of HY populations to overall species population dynamics (White et al. 2021), little is known about long-term changes in HY abundance and how these trends may differ from the abundance trends of breeding adult birds, such as those from the BBS, which are commonly used to determine the conservation status of migratory songbird species (Rosenberg et al. 2017). Due to the “coastal effect,” which concentrates HY birds at coastal stopover sites (Ralph 1978, 1981, Reinert et al. 2002), coastal bird banding stations on major migratory routes provide a unique opportunity for studying population dynamics and patterns of abundance in HY birds.

BIBS was established by Elise Lapham in 1967 to monitor birds on the island and has been continued by Helen Lapham and Kim Gaffett. These 3 women were the lead banders during the data collection phase of this study and were supported by volunteer labor when available. This constant-effort long-term bird banding station is located 2.5 km from the northern end of the island on privately owned land within the Clay Head Sanctuary (Reinert et al. 2002). BIBS operated each autumn from late-August/early-September to late-October/mid-November. Birds were captured using up to 21 nylon mist nets (30 mm, 4-panel) within a ~0.7 ha study area. Each mist-net was 2.6 m high and either 6- or 12-m long. Each fair-weather day, banding staff typically opened nets 30 min before sunrise and closed them by sunset. The number of nets, the duration and timing of daily effort, and the number of days nets were operated each year varied annually during the study (Supplementary Figure S1). To account for this variation through time, banders recorded daily effort in the form of net-hours, where one 12-m net open for 1 hr was equivalent to 1 net-hour and one 6-m net open for 1 hr was counted as 0.5 net-hour. On average, BIBS recorded 3,381 net-hours (SE = 180) using 12 nets (range: 8–21 nets) in 45 banding days (SE = 1.76) each autumn. The vegetation at BIBS matured over the course of the study period, thus also changing the local habitat present around the net lanes. In 1976, a ~0.6-ha area of shrub habitat surrounding the net lanes was mowed to <1 m height and then was left to undergo natural succession undisturbed other than routine clearing of the net lanes and mowing of paths. By the early 2000s, the vegetation around the net lanes was ~2–3 m high and several stands of mature *Pinus thunbergiana* (Japanese black pine), planted in the early 1970s, were within 200 m of the banding station (Reinert et al. 2002).

Data Processing

We used the BIBS autumn banding records from 1970 to 2021 for this analysis. We limited the “autumn” banding season to September 15 through November 15 to standardize the banding period among seasons. We only included initial captures of HY birds, which accounted for >97% of all captures.

During high-volume days (n = 221 days over 52 years), *Setophaga coronata coronata* (Yellow-rumped “Myrtle” Warbler), *Dumetella carolinensis* (Gray Catbird), *Certhia americana* (Brown Creeper), and *Regulus satrapa* (Golden-crowned Kinglet) were released without banding or aging if the number of captured birds exceeded the number that could be banded safely. Birds that were released unbanded were tallied and recorded for each day of banding. These birds were then incorporated into the dataset on the day that they were caught. Age was not recorded for released birds, so we estimated age ratios for these birds using the average age ratios across the study period for each species. We first calculated the average percentage of HY birds across all 52 years for each of the 4 species (99.2% ± 0.16 (SE) for *S. c. coronata*, 96.3% ± 0.30 for *D. carolinensis*, 99.1% ± 0.37 for *C. americana*, 99.8% ± 0.11 for *R. satrapa*). Then, this average percentage of HY birds for each species was multiplied by the total number of released birds of each species for each autumn.

We included species in our analysis if they had at least 5 captures per year in 80% of study years (≥41/52 years; Osenkowski et al. 2012) to ensure that each species was sampled adequately to detect meaningful trends during the study period and avoid detecting potentially stochastic declines in already small cohorts. This filtering step resulted in 22 of 126 species being included in our final analysis (Supplementary Table S1 provides the full list of species). Given that autumn migration phenology differs across the 22 songbird species, we estimated the migratory window for each species that accounted for 98% of captures of a given species in a given autumn season (i.e., we removed 1% of captures at the start and end of the season; Lloyd-Evans and Atwood 2004, Kamm et al. 2019). We calculated migratory windows on a yearly basis to account for the influence of annual changes in frequency of north-westerly winds that can affect the arrival of birds to Block Island (Baird and Nisbet 1960, Able 1977, Parrish 1997). The total number of individuals of each species caught during their autumn migration window was then divided by total net-hours within each species' autumn migration window, regardless of whether that species was captured on a given day, to determine annual capture rate (captures per 100 net-hours for each autumn). A zero was included in the dataset in years where a given species was not captured.

We log-transformed (natural log) annual capture rates prior to estimating abundance trends over time; we added a constant of 1 before log transformation to account for years when capture rate was zero (Osenkowski et al. 2012). We centered banding year around 0 and scaled all values with an SD of 1 to avoid multicollinearity issues in our quadratic models (Smith and Paton 2011). Annual banding effort (percentage of days closed during a species' migratory window) was not related to annual capture rates over the 52 years (Supplementary Figure S2). Given the goals of this study, we maintain that using capture rate (birds captured per 100 net-hours each autumn) as an estimate of annual abundance provides a reliable, sufficiently unbiased index to standardize abundance estimates, and further, this metric is consistent with how such banding data have been summarized in other studies (Hagan et al. 1992, Lloyd-Evans and Atwood 2004, Osenkowski et al. 2012, Kamm et al. 2019).

Model Fitting

Our primary goal for this analysis was to determine the long-term trends in autumn abundance from 1970 to 2021 for each of our 22 species. We used model selection to simultaneously evaluate 4 different models for each species: "no change," "linear," "quadratic," and "breakpoint" with banding year (centered around 0 and scaled with a SD of 1) as the independent variable and abundance (log-transformed capture rate) as the dependent variable (Pélissié et al. 2024). This method allowed us to compare the "breakpoint" model to simpler, more traditional models to ensure that the inclusion of the breakpoint added value to the model. The "no change" model served as our null model and assumed no relationship between banding year and abundance; therefore, we report only an intercept for these models. "No change," "linear," and "quadratic" models were fitted using the base R *lm* function.

We used piecewise regression, referred to here as "breakpoint" models, to identify meaningful changes in abundance trend direction over time in an effort to characterize the current trend status of each species most accurately (Rozek et al. 2017, Habel et al. 2022). Although applying breakpoint models to long-term bird banding data is novel, these models have been widely used in ecological research to detect sudden changes in populations, identify ecological thresholds, analyze biotelemetry data, and assess shifts in phenology (Homan et al. 2004, Rozek et al. 2017, Askeyev et al. 2022, Wolfson et al. 2022, Habel et al. 2022). We fit a piecewise linear regression with a single breakpoint to each time series using the *segmented* function with default parameters from the *segmented* R package (version 2.1.0, Muggeo 2008). This function requires the inclusion of an "initial" estimation of the breakpoint, though this does not mean that the breakpoint will closely reflect this initial breakpoint estimation. Our initial breakpoint estimation was set to the year 1980 (Scaled Banding Year = −1.032) based on a preliminary analysis of our data as well as trend estimates from prior studies that suggested a possible decline in abundance for many species of migratory songbirds in the late 1970s and early 1980s (Robbins et al. 1989, Hagan et al. 1992, Kamm et al. 2019). Importantly, the *segmented* package fits a breakpoint regardless of whether it is appropriate for the data. To prevent the identification of false breakpoints, we used a Davies' test to evaluate whether there was a statistically significant change in slope before and after the identified breakpoint (Muggeo 2008). If the Davies' test was not significant ($P > 0.05$), we concluded that the breakpoint model was not well supported, and it was removed from the model selection process. We decided a priori to reject breakpoint location estimates if they were within 5 years of the beginning or end of the study period (Pélissié et al. 2024), although this was not the case for any of the 22 songbird species we considered.

Model Comparison

To determine the best model for each species, we used Akaike Information Criterion corrected for small sample size (AIC_c) to compare each of the 4 initial models (Burnham and Anderson 2004, Pélissié et al. 2024). AIC_c values were calculated using the *MuMIn* package (version 1.48.4, Bartoń 2023). If the AIC_c indicated a clear "best" model, defined as a difference of >2 AIC_c units between the top models, we identified that model as the top model. If we could not identify a single model using AIC_c, we chose the simplest model in an effort to maximize parsimony and avoid over-fitting (Arnold 2010). We then assessed the AIC_c weights to ensure that the selected top model had the strongest support. We calculated the normalized root mean square error (NRMSE), normalized by the standard deviation, for each of the final best models as an additional measure of model performance (Pélissié et al. 2024). The *segmented* package did not calculate *P*-values for each segment of the breakpoint model, so we calculated these manually by running a linear model on the first segment (including the breakpoint) and another on the second segment after the breakpoint. Notably, this approach aims to provide a broad classification of the most prevalent trend shape and direction, rather than aiming to maximize model fit for each species (Pélissié et al. 2024).

Leave-One-Out Cross-Validation

We ran a leave-one-out cross-validation (LOOCV) to confirm the reliability of our model selection and to identify any year(s) that disproportionately influenced the trends over time for each of our species (Pélissié et al. 2024). To do this, we reran all models and model selection criteria but removed a single year of data during each iteration. This was performed for all 52 time points for each species. We used this to determine the proportion of time that the "best" model from each LOOCV iteration matched the overall "best" model. A LOOCV value of 1 indicated that each iteration of the LOOCV produced the same best-supported model, whereas a LOOCV value of 0 indicated that no iterations matched the best model (Pélissié et al. 2024).

Trend Comparison

Overall trends from BIBS were compared to trends estimated from Manomet Conservation Sciences (Manomet, 1969–2015; Kamm et al. 2019), a long-term constant-effort bird banding station located 115 km northeast of Block Island on the coast of Massachusetts, and the BBS (1966–2022; Ziolkowski et al. 2023). Because we did not know the breeding locations of the migratory birds stopping over on Block Island, we compared our annual trends to the BBS trends from 4 bird conservation regions (BCRs) that represented the potential breeding range for the 22 species we modeled: New England Mid-Atlantic Region (BCR 30), the Atlantic Northern Forests Region (BCR 14), the Boreal Hardwood Transition (BCR 12), and the Boreal Softwood Shield Region (BCR 8) (Osenkowski et al. 2012, NABCI 2016). We only used BBS trends in our study if they had medium-to-high survey confidence (Kamm et al. 2019).

TABLE 1. Model selection results and annual trends in log-transformed capture rates for 22 autumn migratory bird species from the Block Island Banding Station (1970–2021) on Block Island, Rhode Island, including slopes with 95% CI, breakpoint year, and NRMSE values. See Supplementary Table S2 for the full model comparisons for each species.

Species	Slope 1 ± 95% CI[a]	Breakpoint year[b]	Slope 2 ± 95% CI[a]	NRMSE[c]
Breakpoint—1970s				
Vireo solitarius (Blue-headed Vireo)	**−1 ± 1.05**	1978	**0.23 ± 0.1***	0.8
Vireo olivaceus (Red-eyed Vireo)	**−2.36 ± 1.4***	1978	−0.14 ± 0.13*	0.67
Corthylio calendula (Ruby-crowned Kinglet)	**−1.85 ± 1.63**	1979	**0.21 ± 0.16***	0.86
Dumetella carolinensis (Gray Catbird)	**−1.81 ± 1.63**	1978	**−0.16 ± 0.13***	0.73
Junco hyemalis hyemalis (Dark-eyed Junco)	**−1.43 ± 1.23***	1979	−0.02 ± 0.12	0.81
Melospiza melodia (Song Sparrow)	**−2.7 ± 0.69***	1979	**0.2 ± 0.08***	0.5
Melospiza georgiana (Swamp Sparrow)	**−2.26 ± 1.18***	1978	**0.06 ± 0.09**	0.7
Pipilo erythrophthalmus (Eastern Towhee)	**−2.32 ± 1.31***	1976	**0.09 ± 0.06***	0.72
Geothlypis trichas (Common Yellowthroat)	**−1.22 ± 0.84***	1979	0.09 ± 0.1	0.82
Setophaga ruticilla (American Redstart)	**−1.27 ± 1.04***	1978	0.02 ± 0.08	0.81
Breakpoint—1980s				
Sayornis phoebe (Eastern Phoebe)	**−0.69 ± 0.83**	1980	**0.19 ± 0.1***	0.84
Troglodytes aedon (House Wren)	**−0.52 ± 0.34***	1985	**0.09 ± 0.09***	0.82
Troglodytes hiemalis (Winter Wren)	**−0.76 ± 0.41***	1984	**0.13 ± 0.11***	0.79
Catharus ustulatus (Swainson's Thrush)	**−2.04 ± 0.99***	1980	−0.12 ± 0.14*	0.63
Catharus guttatus (Hermit Thrush)	**−1.61 ± 1.08***	1982	**0.28 ± 0.21***	0.83
Turdus migratorius (American Robin)	**−1.32 ± 0.46***	1985	**0.23 ± 0.12***	0.62
Zonotrichia albicollis (White-throated Sparrow)	**−1.24 ± 0.63***	1983	0.06 ± 0.14	0.74
Setophaga striata (Blackpoll Warbler)	**−0.97 ± 0.4***	1986	**0.22 ± 0.14***	0.71
No change				
Setophaga caerulescens (Black-throated Blue Warbler)	0 ± 0	–	–	0.99
Linear				
Regulus satrapa (Golden-crowned Kinglet)	**0.23 ± 0.19***	–	–	0.93
Setophaga coronata coronata (Myrtle Warbler)	**−0.86 ± 0.23***	–	–	0.68
Quadratic				
Certhia americana (Brown Creeper)	**1.25 ± 0.84***	–	**0.99 ± 0.84***	0.87

[a]Starred slopes represent statistical significance ($P < 0.05$). Bolded slopes represent species whose trends were increasing or declining based on 95% CIs. Slope 1 represents the only slope for linear models, the linear term coefficient for quadratic models, and the "before" breakpoint slope for breakpoint models. Slope 2 represents the "after" breakpoint slope for breakpoint models and the quadratic term coefficient for quadratic models.

[b]95% CIs for the breakpoint year are shown in Figures 1 and 2.

[c]NRMSE is the normalized root mean square error, normalized by the standard deviation, for each model.

To be able to compare the resulting species trends over time to the results of previous studies, we classified the qualitative trends for the best-supported model for each species in our study as either increasing, declining, or stable by comparing the 95% confidence intervals (CI) around the model estimate for the first year to the model estimate in the last year of each time frame or section of time. For our study, if the abundance model estimate for the most recent year of the time frame or time-frame segment (either 2021 or breakpoint) exceeded the upper 95% CI for the first year (1970 or breakpoint), we classified it as increasing; if it was below the lower 95% CI, we classified it as declining; and if the abundance estimate was within the CI for the first year of the trend, we classified it as stable. For breakpoint models, we classified trends before and after the breakpoint (i.e., we compared the breakpoint year estimate to 1970 95% CI and the 2021 estimate to the breakpoint 95% CI to yield 2 trend classifications). Kamm et al. (2019) also used this method to classify trend trajectories for their analysis of long-term banding data, while the BBL relied instead on *P*-value significance and trend direction to determine the overall trend classification (Ziolkowski et al. 2023).

To provide context for interpreting the results of these models, we present capture rate estimates for the first year, last year, and breakpoint for each breakpoint model, as well as the percent decline and percent recovery. We back-transformed model predictions for log capture rate ($e^{(\text{Log Capture Rate})} - 1$) to capture rate (captures per 100 net-hours) for the first year of the study (1970), last year of the study (2021), and breakpoint year for each species. We calculated total percent change over the study period for linear and quadratic models ((Capture Rate$_{2021}$ – Capture Rate$_{1970}$)/Capture Rate$_{1970}$). For breakpoint models, we calculated "percent decline" as the percent change from 1970 to the breakpoint ((Capture Rate$_{bp}$ – Capture Rate$_{1970}$)/Capture Rate$_{1970}$) and we calculated "percent recovery" as the percent change from the breakpoint to 2021 relative to 1970 ((Capture Rate$_{2021}$ – Capture Rate$_{bp}$)/Capture Rate$_{1970}$). We performed all analyses using R statistical software (version 4.4.0, R Core Team 2024).

RESULTS

Model Selection Results and Abundance Trends

BIBS was operated for 156,126 total net-hours from 1970 to 2021. We analyzed autumn bird banding captures for a total of 66,288 HY birds of 22 selected species. Our model selection framework identified breakpoint models as the best fit for the

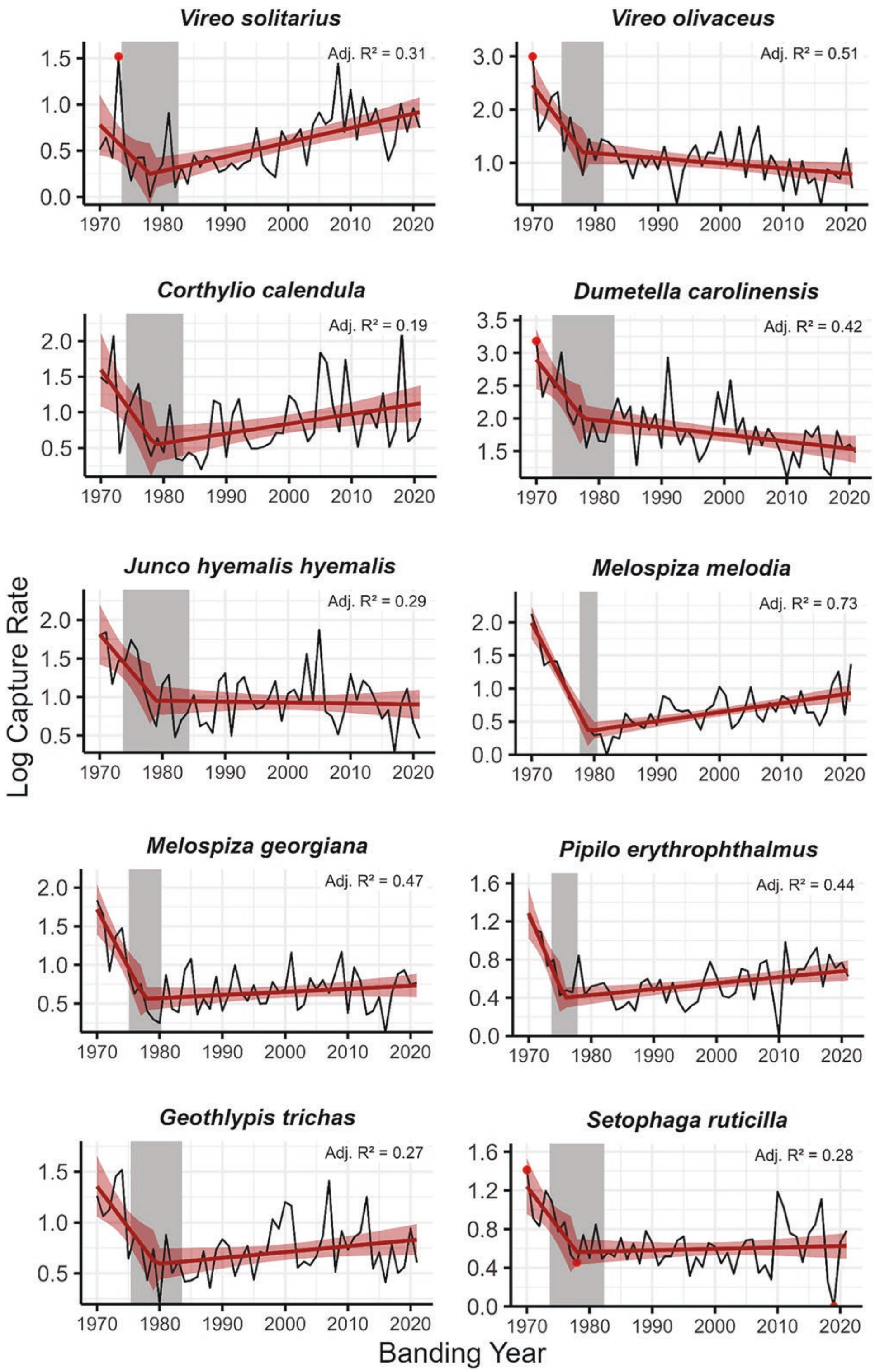

FIGURE 1. Relationship between log-transformed capture rate (birds per 100 net-hours) and banding year (1970–2021) for 10 autumn migratory songbird species with breakpoints in the late 1970s at the Block Island Banding Station, Rhode Island. Model selection was based on AIC_c and AIC_c weight. Shaded regions represent 95% CI and vertical rectangles represent the 95% CI for the estimated breakpoint year. Red points for 4 species mark influential years in leave-one-out cross-validation.

relationship between banding year (scaled) and abundance (log capture rate) for 18 of 22 species (Table 1; see Supplementary Table S2 for full AIC_c results). All breakpoints occurred between 1976 and 1986, with 10 species experiencing breakpoints in the 1970s (Figure 1) and the remaining 8 species in the early 1980s (Figure 2). The long-term trends for all 18 species with breakpoints exhibited a consistent pattern: a steep decline in abundance within the first 2 decades of the study period, followed by stabilization, recovery, or a more gradual decline (Figures 1 and 2). After the estimated breakpoint, 17

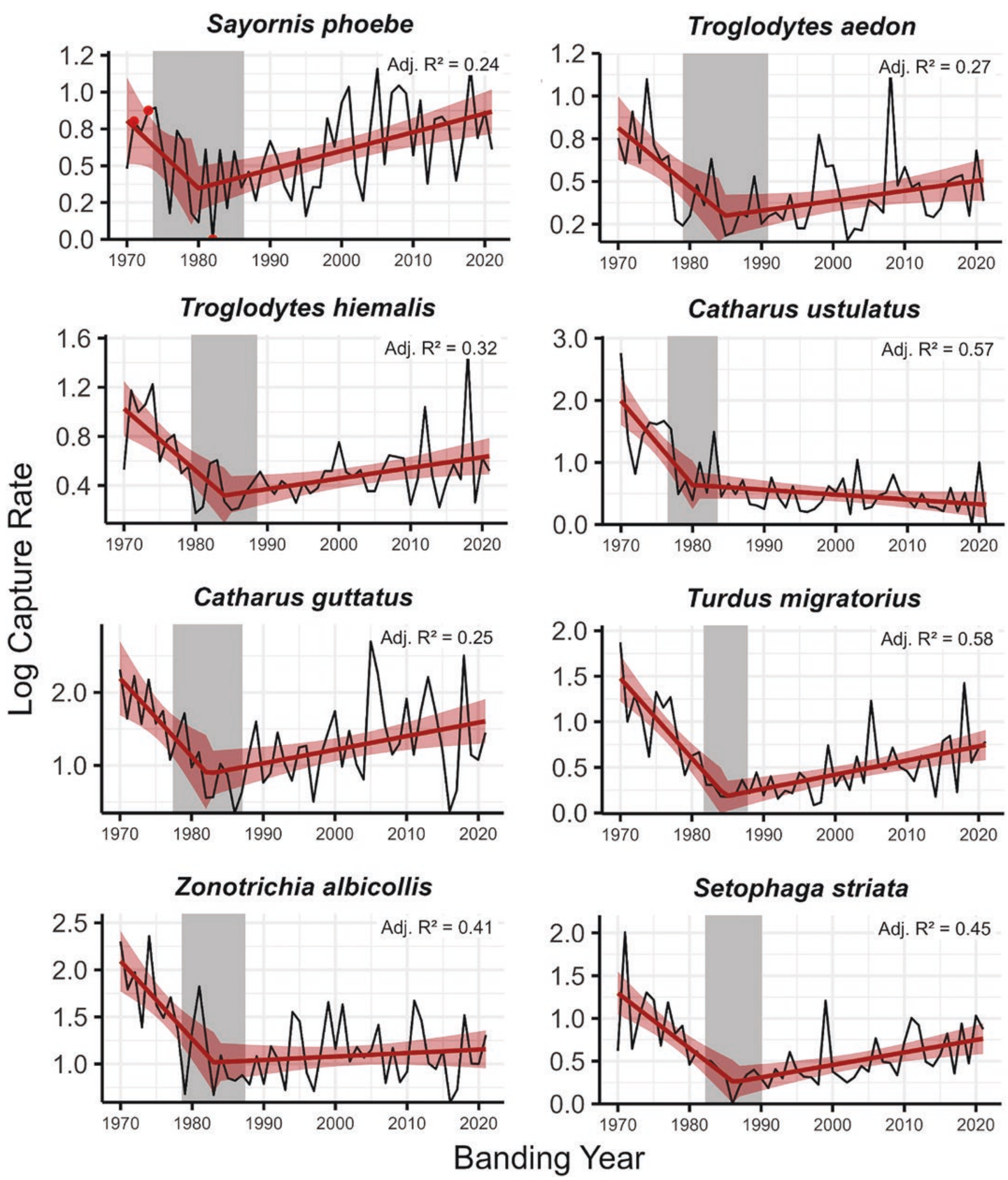

FIGURE 2. Relationship between log-transformed capture rate (birds per 100 net-hours) and banding year (1970–2021) for 8 autumn migratory songbird species with breakpoints in the early 1980s at the Block Island Banding Station, Rhode Island. Model selection was based on AIC_c and AIC_c weight. Shaded regions represent 95% CI and vertical rectangles represent the 95% CI for the estimated breakpoint year. Red points for 1 species mark influential years in leave-one-out cross-validation.

out of 18 species were stable (33%) or significantly increased (61%), and *D. carolinensis* continued to decline (Table 1). The LOOCV for each species confirmed that these trend estimates were not influenced by unusual years. The overall "best" model matched each LOOCV model iteration 100% of the time for 15 of the 22 species and the match rate for the other 7 remaining species ranged from 92% to 98%.

Four species did not exhibit breakpoints and were best represented by a simpler model (Table 1, Figure 3). *Setophaga caerulescens* (Black-throated Blue Warbler) was the only species that had no significant change over the 52-year period and was best represented by the null model, despite substantial fluctuations in yearly captures. *Regulus satrapa* increased linearly by ~168%, though there was considerable interannual variation, especially in the first half of the study period. Notably, *S. c. coronata*, which was the most common species captured during autumn on Block Island, had a consistent and dramatic linear decline over the course of the study period. By 2021, *S. c. coronata* declined by 96% at BIBS, a loss of ~65 birds per 100 net-hours over 52 years. *Certhia americana* was best described by a quadratic model, showing relative stability until captures began gradually increasing in the 1990s (Table 1). Overall, *C. americana* increased by 122% over 52 years.

For the 18 species with trends that were best represented by breakpoint models, the percentage of decline in untransformed capture rate (i.e., birds per 100 net-hours) between 1970 and the breakpoint for each species ranged from 63% to 94% (Table 2). In the period after the breakpoint, 14 of the 18 species recovered relative to their 1970 abundance, though the degree of recovery varied widely (5–103% recovery) and only *Vireo solitarius* (Blue-headed Vireo) and *Sayornis phoebe*

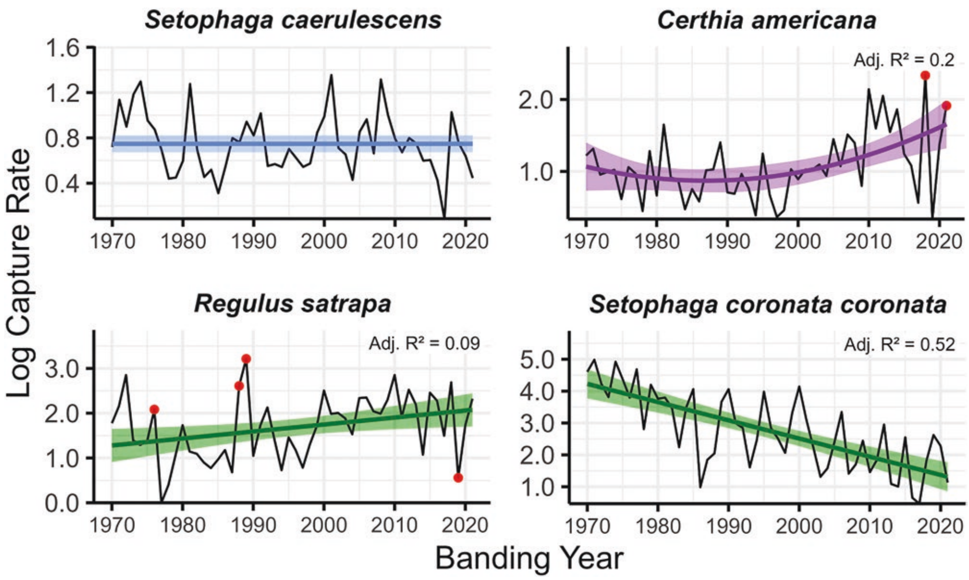

FIGURE 3. Relationship between log-transformed capture rate (birds per 100 net-hours) and banding year (1970–2021) for 4 autumn migratory songbird species best represented by non-breakpoint models at the Block Island Banding Station, Rhode Island. Colored lines indicate the best model: no change (blue), linear (green), and quadratic (purple). Model selection was based on AIC_c and AIC_c weight. Shaded regions represent 95% CI. Red points for 2 species mark influential years in leave-one-out cross-validation.

TABLE 2. Breakpoint model-predicted capture rates (birds per 100 net-hours) for 18 migratory songbird species from the Block Island Banding Station (1970–2021) at 3 timepoints (1970, the breakpoint year identified in Table 1, and 2021) and the corresponding percent decline between 1970 and the breakpoint and the percent recovery between the breakpoint and 2021.

Species	1970 Capture rate[a]	Breakpoint Capture rate[a]	2021 Capture rate[a]	Percent decline [b]	Percent recovery[c]	Net change[d]
Breakpoint—1970s						
Vireo solitarius	1.18	0.28	1.50	**–76.07**	**103.30**	27.22
Vireo olivaceus	10.72	2.33	1.21	**–78.31**	–10.42	–88.73
Corthylio calendula	3.95	0.73	2.09	**–81.53**	**34.41**	–47.11
Dumetella carolinensis	17.19	6.36	3.61	**–62.97**	**–16.02**	–78.99
Junco hyemalis hyemalis	5.13	1.60	1.47	**–68.90**	–2.47	–71.37
Melospiza melodia	6.34	0.43	1.53	**–93.21**	**17.32**	–75.90
Melospiza georgiana	4.58	0.75	1.08	**–83.54**	**7.24**	–76.30
Pipilo erythrophthalmus	2.62	0.49	0.98	**–81.20**	**18.69**	–62.52
Geothlypis trichas	2.88	0.81	1.29	**–72.05**	16.99	–55.06
Setophaga ruticilla	2.46	0.76	0.87	**–69.28**	4.68	–64.60
Breakpoint—1980s						
Sayornis phoebe	1.24	0.41	1.38	**–66.53**	**77.94**	11.41
Troglodytes aedon	1.26	0.35	0.66	**–72.17**	**25.07**	–47.10
Troglodytes hiemalis	1.79	0.38	0.89	**–79.01**	**29.01**	–50.00
Catharus ustulatus	6.35	0.89	0.38	**–85.95**	–8.14	–94.08
Catharus guttatus	7.96	1.42	3.98	**–82.12**	**32.15**	–49.97
Turdus migratorius	3.38	0.20	1.11	**–94.04**	**26.80**	–67.24
Zonotrichia albicollis	7.11	1.76	2.18	**–75.18**	5.81	–69.37
Setophaga striata	2.66	0.29	1.14	**–89.14**	**32.20**	–56.94

[a]Capture rates were calculated from back-transformed breakpoint model-predicted log capture rates.
[b]Percent Decline = ((Capture $Rate_{bp}$ – Capture $Rate_{1970}$)/Capture $Rate_{1970}$) × 100. Bolded slopes represent species whose trends were increasing or declining based on 95% CIs. Trends that are not bolded are considered stable.
[c]Percent Recovery = ((Capture $Rate_{2021}$ – Capture $Rate_{bp}$)/Capture $Rate_{1970}$) × 100. Bolded slopes represent species whose trends were increasing or declining based on 95% CIs. Trends that are not bolded are considered stable.
[d]Net Change = Percent Decline + Percent Recovery.

(Eastern Phoebe) returned to their 1970 abundance (Table 2). Four of the 18 species continued to decline after the breakpoint (2–16%), though at a markedly slower rate than their initial decline before the breakpoint (*D. carolinensis*, *Vireo olivaceus* [Red-eyed Vireo], *Junco hyemalis hyemalis* [Dark-eyed Junco], and *Catharus ustulatus* [Swainson's Thrush];

TABLE 3. Comparison of trend trajectory categorization for the Block Island Banding Station (BIBS), Manomet Conservation Sciences, and 4 BBS bird conservation regions.[a]

Species	Overall trend[b]	Initial trend[b]	After-breakpoint trend[b]	Manomet[c]	NEMA	ANF	BSS	BHT
Breakpoint—1970s								
Vireo solitarius	Stable	Decline	Increase	Increase	Stable	Increase	Stable	Increase
Vireo olivacus	Decline	Decline	Stable	Decline	–	Decline	Stable	Stable
Corthylio calendula	Stable	Decline	Increase	Decline	Stable	Increase	Stable	Increase
Dumetella carolinensis	Decline	Decline	Decline	Stable	Stable	Decline	Stable	Decline
Junco hyemalis hyemalis	Decline	Decline	Stable	Decline	–	Decline	Stable	Decline
Melospiza melodia	Decline	Decline	Increase	Decline	Decline	Decline	Decline	Decline
Melospiza georgiana	Decline	Decline	Increase	Stable	–	Decline	Decline	Decline
Pipilo erythrophthalmus	Decline	Decline	Increase	Decline	Decline	Decline	–	Decline
Geothlypis trichas	Decline	Decline	Stable	Decline	Decline	Decline	Decline	Decline
Setophaga ruticilla	Decline	Decline	Stable	Decline	Stable	Decline	Stable	Stable
Breakpoint—1980s								
Sayornis phoebe	Stable	Decline	Increase	Increase	Stable	Decline	Stable	Stable
Troglodytes aedon	Decline	Decline	Increase	–	Stable	Increase	Stable	Stable
Troglodytes hiemalis	Decline	Decline	Increase	Decline	Decline	Decline	Decline	Decline
Catharus ustulatus	Decline	Decline	Stable	Stable	Stable	Stable	Stable	Increase
Catharus guttatus	Decline	Decline	Increase	Stable	Decline	Stable	Stable	Stable
Turdus migratorius	Decline	Decline	Increase	Decline	Decline	Decline	Stable	Decline
Zonotrichia albicollis	Decline	Decline	Stable	Decline	Stable	Stable	Stable	Stable
Setophaga striata	Decline	Decline	Increase	Decline	–	Decline	Decline	Decline
No change								
Setophaga caerulescens	Stable	–	Stable	Stable	Increase	Increase	Increase	Stable
Linear								
Regulus satrapa	Increase	Increase	–	Decline	–	Stable	Stable	Increase
Setophaga coronata coronata	Decline	Decline	–	Decline	Stable	Stable	Stable	Stable
Quadratic								
Certhia americana	Increase	Increase	–	Decline	Stable	Increase	Increase	Increase

[a]North American Breeding Bird Survey (BBS) trends, classified using trend direction and statistical significance, from 4 separate bird conservation areas where our species may breed: New England/Mid-Atlantic (NEMA), Atlantic Northern Forest (ANF), Boreal Softwood Shield (BSS), and Boreal Hardwood Transition (BHT).

[b]Overall trend represents autumn BIBS trends from 1970 to 2021, where we compared the model estimate for the last year of the study period to the 95% CI of the first year (Kamm et al. 2019). The initial trend compared the model estimate for the breakpoint year to the 95% CI of the first year of the study, whereas the after-breakpoint trend compared the last year of the study to the 95% CI of the breakpoint year.

[c]Manomet trends represent autumn trends from Manomet Conservation Sciences in Plymouth, Massachusetts, from 1969 to 2015 and compare the model estimate for the last year of the study period to the 95% CI of the first year (Kamm et al. 2019).

Table 2). Only *D. carolinensis* continued to decline enough to be classified as "declining" by our CI criteria.

Trend Comparisons

Given that for 18 of 22 species, we detected a breakpoint in the first 2 decades of the study (Table 1, Figures 1 and 2), comparing to other studies illuminates the contrast in trend trajectories when considering decadal-scale changes in trends (Table 3). When considering the entire time series, 16 of our 22 species declined, 4 were stable, and 2 increased over the 52-year period, and our trends were largely consistent with those found at Manomet and the BBS. However, when we considered intermediate timescales (determined by our breakpoint analysis), the interpretation of the abundance trends of many of our species changed considerably. At BIBS, only *S. c. coronata* linearly declined during the entire study period, and only *D. carolinensis* consistently declined both before and after the identified breakpoint. The declining trend that we documented for *S. c. coronata* at BIBS also occurred at Manomet, whereas the trend for *D. carolinensis* at Manomet was classified as stable. Trends from Manomet identified 13 other species as declining whereas at BIBS the trends for these species, when accounting for breakpoints, were either stable (5 species) or increasing (8 species) after the breakpoint. Trends from Manomet and BIBS were consistent for only 5 species in total, including increases for 2 species (*V. solitarius* and *Sayornis phoebe*) and stable trends for 2 others (*Catharus ustulatus* and *Setophaga caerulescens*), and declines for 1 (*S. c. coronata*).

Trend comparisons between the 2 banding stations (BIBS and Manomet) and the broader-scale BBS also revealed a few general patterns (Table 3). For example, *S. c. coronata* capture rates have declined considerably at BIBS and Manomet, but their status as breeding birds is considered stable at all 4 BBS locations. When considering the entire time series, 5 species (*Setophaga striata* [Blackpoll Warbler], *Geothlypis trichas* [Common Yellowthroat], *Pipilo erythrophthalmus* [Eastern Towhee], *Melospiza melodia* [Song Sparrow], and *Troglodytes hiemalis* [Winter Wren]) were consistently declining at BIBS, Manomet, and all 4 BBS regions. However, when considering only the period after the breakpoint, trends for these species at BIBS were either increasing (4 of 5 species) or stable (Table 3). The most common scenario (for 15 of the 22 BIBS species) was that trend trajectories were distinctly different across the 4 regions of BBS. For these 15 species, the most common trend pattern across the BBS regions was consistent with the trend from Manomet for 5 species (*Turdus migratorius* [American Robin], *V. solitarius*, *D. carolinensis*, *Catharus guttatus* [Hermit Thrush], and *J. h. hyemalis*), was consistent with the overall BIBS trend for 6 species (*T. migratorius*, *Sayornis phoebe*, *D. carolinensis*, *Corthylio calendula* [Ruby-crowned Kinglet], *J. h. hiemalis*, and *Certhia americana*), was consistent with the after-breakpoint trend from BIBS for 4 other species (*Setophaga ruticilla* [American Redstart], *V. olivaceus*, *Corthylio calendula*, and *Zonotrichia albicollis* [White-throated Sparrow]), and inconsistent with either banding station for 5 species. For example, *R. satrapa* was classified as increasing at BIBS, declining at Manomet, and stable in some BBS regions, while increasing in others. Similarly, *Certhia americana* was considered increasing at BIBS, declining at Manomet, and showed a mix of stable and increasing trends across the BBS regions.

DISCUSSION

We determined that the abundance trends over the last half century for 18 of 22 species at BIBS had significant breakpoints: they declined steeply during the first 17 years of the study, followed by gradual recovery (61%), stabilization (33%), or a continued decline but at a far slower pace (6%; Table 1, Figures 1 and 2). However, the abundance of only 2 species (*V. solitarius* and *Sayornis phoebe*) recovered to their previous 1970 abundance, indicating that while many species have been gradually recovering since their initial decline, effort is still required to monitor, and potentially facilitate, their recovery (Table 2). These results suggest that given an adequately long-term dataset, breakpoint detection and careful timescale consideration can reveal current, relevant abundance patterns and may be useful for determining conservation priorities for migratory songbirds.

Abundance Trends—Breakpoints Are Common but Rarely Highlighted

The pattern of decline in the 1970s and 1980s that we detected 82% of study species at BIBS was also observed in a recent analysis of long-term autumn bird banding data (which included both HY and adult birds) at Manomet, a coastal banding station located 115 km northeast of BIBS in Massachusetts (Kamm et al. 2019), as well as in an analysis of BBS data, which consisted of breeding adult birds (Robbins et al. 1989). Though not explicitly described in their analysis, 18 of 62 autumn species at Manomet showed a sharp decline in the 1970s and 1980s, followed by either relative stability, slower declines, or gradual recovery. Of these 18 species, 5 species also had breakpoints in our analysis (*Setophaga striata*, *G. trichas*, *P. erythrophthalmus*, *J. h. hiemalis*, and *T. hiemalis*), 3 showed possible breakpoints at Manomet but not at BIBS (*C. americana*, *R. satrapa*, and *S. c. coronata*), and the remaining species did not have apparent breakpoints at BIBS or were not included in our study (Kamm et al. 2019). Two Manomet species (*Seiurus aurocapilla* [Ovenbird] and *Setophaga virens* [Black-throated Green Warbler]) showed signs of a possible breakpoint in the 1990s, though neither species was included in our analysis. An additional 8 species monitored during spring migration also showed possible breakpoints within the same time frame, including *C. americana*, *Contopus virens* (Eastern Wood-Pewee), *P. erythrophthalmus*, *Icterus galbula* (Baltimore Oriole), *M. melodia*, *M. georgiana* (Swamp Sparrow), *J. h. hiemalis*, and *Molothrus ater* (Brown-headed Cowbird). Signs of breakpoints in the 1970s and 1980s were also seen in an analysis of 32 years (1970–2001) of Manomet data, using entirely different statistical methods (Lloyd-Evans and Atwood 2004) compared to Kamm et al. (2019). A 22-year analysis of BBS data from eastern North America that compared changes in abundance between 1966–1978 and 1978–1987 found that most species had a period of significant decline during these 2 decades (Robbins et al. 1989). Notably, many of the songbird species that declined most dramatically in the first decade of the BBS study showed a population increase or drastic slowing of decline in the second decade, which may indicate that these species had a breakpoint in their trends. In sum, the trend dynamics of the late 1970s and early 1980s were exhibited across different population sampling techniques, seasons, statistical methodologies, and geographic locations, and underscore the importance of considering time series length when the conservation status of species are assessed.

The substantial declines in abundance of most songbird species in the late 1970s and early 1980s could be influenced by long-term changes in weather, changing local land use and habitat, or cyclical, decadal patterns of breeding success. On Block Island, climate change, vegetation growth, land use changes, and shifts in plant community composition may have potentially impacted yearly capture rates of migratory birds. For example, if vegetation is consistently above the maximum net height of 2.6 m, it could lead to fewer captures overall. While natural succession has occurred at BIBS over the last 52 years, the presence of similar patterns in other datasets, coupled with the predominantly increasing or stable recent trends for our species, indicate that these factors are unlikely to be primary drivers of local abundance. While changes in regional weather patterns could affect the abundance of birds stopping over on Block Island, the diversity of trends seen across the 22 study species and evidence of similar patterns from multiple mainland data sources (Robbins et al. 1989, Hill and Hagan 1991, Lloyd-Evans and Atwood 2004, Kamm et al. 2019) suggest that the trends we detected are not predominantly driven by long-term changes in weather patterns. Higher capture rates at eastern North American banding stations in the 1970s were speculated to be linked to *Choristoneura fumiferana* (Eastern spruce budworm) outbreaks in parts of Canada and the USA, which may have boosted breeding productivity through increased resource availability (Hill and Hagan 1991, Hagan et al. 1992, Lloyd-Evans and Atwood 2004). Additionally, Hill and Hagan (1991) analyzed spring bird counts in eastern Massachusetts from 1937 to 1989 and found that abundance for many songbird species cycled every ~10 years (e.g., *Setophaga tigrina* [Cape May Warbler], *Leiothlypis peregrina* [Tennessee Warbler], *S. castanea* [Bay-breasted Warbler], *S. americana* [Northern Parula], *S. caerulescens*, *Cardellina pusilla* [Wilson's Warbler], and *Mniotilta varia* [Black-and-white Warbler]). Thus, it is possible that the high capture rate at BIBS in the early 1970s reflected a peak in the natural population cycle for many species (possibly associated with increased food availability on the breeding grounds), followed by a cyclical decline, although assessing this requires even longer-term, pre-1970s, banding records which are not available. Regardless, the lack of full recoveries in the 4 decades since for many of the species in this study suggests that changing environmental conditions may have disrupted natural population cycles, making it more challenging for species to recover to 1970s levels.

The modeling approach we adopted not only allows for identification of breakpoints but also identified simpler abundance trends. *Regulus satrapa*, for example, was best described by a linear trend and increased significantly over the study period, though the final model was a poor fit to the data due to the high interannual variation. *Certhia americana* also increased over time, instead showing a quadratic trend with a gradual decline in the first 2 decades followed by a significant increase since the 1990s. *Setophaga caerulescens* was the only species that was stable throughout the entire study period and were best represented by a "no change" model, despite considerable decadal population cycling (Lewis et al. 2023). Most notably, *S. c. coronata*, which are designated as Least Concern globally (BirdLife International 2023) and are considered stable in all 4 BBS BCRs (Table 3), have declined 96% in the last 5 decades on Block Island. This means that in an average BIBS season with 3,000 net-hours, we could expect to capture ~1,950 fewer *S. c. coronata* annually than in 1970. Given that *S. c. coronata* has also declined considerably over the last half century at Manomet (Kamm et al. 2019), more research is needed to understand whether the dramatic decline in autumn abundance in southern New England represents changes in regional migratory behavior or actual population declines.

The Timescale Matters When Estimating Abundance Trends for Conservation Action

Developing relevant conservation plans for songbirds requires knowing the long-term abundance trajectories of species, and as our results have shown, these estimates are often sensitive to the timescale chosen. Kamm et al. (2019) calculated trend classifications for songbirds at Manomet based on the 95% CIs, effectively comparing the capture rate in the last year of the study to the CI of the first year. In contrast, Ziolkowski et al. (2023) calculated trend trajectory for BBS data based on estimated trend direction and significance for the entire 57-year study period. Although these results provide useful context on long-term trends for a wide variety of species, identifying interim changes in slope direction, as we have provided, may be important when considering current species prioritization for conservation. For example, trend estimates, using the entire time series, from Manomet, BIBS, and the BBS found that *M. melodia*, *P. erythrophthalmus*, *S. striata*, and *T. hiemalis* were declining across all locations (Table 3). However, our breakpoint analysis indicated that this decline occurred over the first 2 decades of the study, followed by a period of gradual recovery (Table 1). While these species have not yet fully rebounded to their 1970 abundance (Table 2), their sustained recovery indicates that present environmental conditions may be adequately supporting these populations. Had we interpreted these trends using the entire 52-year time series, as other studies have, we would have concluded that these species had declined—rather than acknowledging that though they may have declined overall, they have been recovering over the last several decades. Therefore, while our overall patterns were similar to those seen in other long-term datasets, accounting for abrupt changes in slope can significantly impact the interpretation and conclusions drawn from long-term abundance data. Nevertheless, the most "relevant" and appropriate timescale will vary depending on the dataset being used and the goals of the analysis. For example, our analysis determined that the most "relevant" timescale for conservation planning on Block Island is the period after the breakpoint for most species, as it reflects the current sustained abundance trajectory. Given that long-term abundance datasets, such as the BBS, are commonly used for species prioritization (Rosenberg et al. 2017), the choice of timescale should be carefully considered so that changes in abundance can be classified in the most ecologically meaningful way to support relevant conservation action (Rigal et al. 2020, Pélissié et al. 2024).

One unique aspect of our study was that all of the identified breakpoints occurred shortly after the start of the study. As a result, across all trends (including before- and after-breakpoint trends), 5 trends that were classified as increasing or declining based on 95% CIs were not statistically significant, while 2 statistically significant trends were classified as stable based on 95% CIs. For some species at BIBS, the "before" breakpoint segments were classified as stable based on *P*-values, despite steep declines, due to the small sample size (n = 7–17 years before the breakpoint). Due to the susceptibility of *P*-values to misinterpretation (especially with low sample sizes), the use of CIs to estimate trend direction may provide a more reliable estimate for breakpoint models because it reduces the risk of type 2 errors that could impact correct model interpretation (Steel et al. 2013).

Different Data Sources Provide Complementary Abundance Insights

Bird abundance trend estimates may differ across studies because of differences in the sampled populations, which can ultimately provide complementary insights into population dynamics. For example, the BBS offers a broad, regional view of changes in breeding bird abundance, which reflects the combined effects of adult survival, the previous year's reproductive output, and recruitment. In contrast, autumn bird banding stations typically sample HY birds in higher proportions than they naturally exist in the general population (Mills 2016). Coastal banding stations, such as Manomet and BIBS, are even more biased towards HY birds compared to inland sites, with estimates that 88% of coastal banding station captures, on average, are HY birds (Ralph 1981, Reinert et al. 2002). This means that autumn banding data may provide a more immediate signal of annual reproductive output and juvenile survival into initial migratory stages. For example, if adult abundance remained stable despite declining autumn HY abundance, then it could indicate that HY annual survival rates and recruitment may be increasing over time, possibly due to density-dependent mortality on the wintering grounds. Despite the close geographic proximity of Manomet to BIBS, our trends seemed to differ considerably for a number of species. While it is difficult to assess exactly why these differences exist without further research, they may reflect differences in sampled subpopulations, local vegetation, changes in migratory routes over time, or regional landcover changes that could affect preferred stopover locations. While trends in bird abundance from a single data source or banding station should be interpreted with care, future collaboration between banding stations to analyze their data using consistent methods could offer valuable insight into the prevalence of breakpoint patterns across North America—and how they may vary by region, demographic, method, or season. Additionally, assessing data sources representing both breeding adults and juveniles may allow for a more nuanced understanding of population trends, helping to disentangle changes in adult survival, juvenile recruitment, and productivity (White et al. 2021).

Conclusions

Accurate long-term abundance trends are crucial for bird conservation, and the chosen timescale and method for estimating abundance can significantly influence trend estimates and the conclusions drawn from them. Given that abrupt changes in bird abundance have been documented for a number of species in North America (Robbins et al. 1989, Lloyd-Evans and Atwood 2004, Kamm et al. 2019), breakpoint models provide a powerful tool for identifying relevant timescales in long-term abundance datasets, particularly when trends are used to guide conservation priorities. Our study underscores the need for careful consideration of the most relevant timescale when assessing population trends, as selecting the most appropriate timescale can lead to more accurate assessments of species' status and inform targeted, effective conservation strategies.

Supplementary material

Supplementary material is available at *Ornithological Applications* online.

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Ethics statement

All research was conducted under appropriate state and federal permits.

Conflict of interest statement

We have no conflicts of interest.

Author contributions

L.E.M. and S.R.M. determined research questions, statistical methods, and study design. S.R.M. acquired funding. S.E.R. entered data, managed the database, and provided valuable background and resources. L.E.M. analyzed the data and wrote the manuscript with help from S.R.M. S.R.M. and S.E.R. reviewed and edited the manuscript.

Data availability

The data used for all analyses is available at Michael et al. (2025).

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