Analysis of combined data sets yields trend estimates for vulnerable spruce-fir birds in northern United States

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ABSTRACT

Continental-scale monitoring programs with standardized survey protocols play an important role in conservation science by identifying species in decline and prioritizing conservation action. However, rare, inaccessible, or spatially fragmented communities may be underrepresented in continental-scale surveys. Data on these communities often come from decentralized, local monitoring efforts that differ in their goals and survey protocols. We combine 16 point count datasets, controlling for differences in protocol and detection probabilities to estimate regional trends for 14 spruce-fir forest bird species across Northeastern and Midwestern United States, a vulnerable community threatened by numerous anthropogenic stressors and widely considered a priority for conservation. Our analyses indicated that four species considered as ecological indicators for this community, Bicknell's Thrush (*Catharus bicknelli*), Magnolia Warbler (*Setophaga magnolia*), Blackpoll Warbler (*Setophaga striata*) and Yellow-bellied Flycatcher (*Empidonax flaviventris*), each exhibited significant declines. Olive-sided Flycatcher (*Contopus cooperi*), a species of concern in parts of its range, and two additional species for which no previous concern existed, the Evening Grosbeak (*Coccothruastes vespertinus*) and the Gray Jay (*Perisoreus canadensis*), each also showed significant overall declines. Five out of nine species with sufficient data for analyses from Northeastern and Midwestern surveys showed significant differences in trends between these regions. Spruce-fir obligate species were more likely to decline significantly than species that use spruce-fir in addition to other habitat types. These results demonstrate the value of combining disparate data sources for analyzing regional patterns of population trends to confirm and extend conservation concern for some species and identify others for which additional attention may be needed.

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1. Introduction

Monitoring of plant and animal populations and their environments is a fundamental component of conservation science (Nichols and Williams, 2006; Lovett et al., 2007). Long-term monitoring data can be used to identify species in decline, track the spread of invasive species, assess the effectiveness of management practices, and understand species' responses to environmental disturbances (Niemi and McDonald, 2004; Marsh and Trenham, 2008; Lindenmayer and Likens, 2009). For birds, the most extensively monitored animal taxon on the planet, continental-scale monitoring programs such as the North American Breeding Bird Survey (BBS; Sauer et al., 2014) have been invaluable in assessing population trends and assigning conservation priorities (Robbins et al., 1989; Sauer and Droege, 1992; Rich et al., 2004). However, rare, inaccessible, or spatially fragmented habitats may be underrepresented in road-side continental-scale surveys (Hanowski and Niemi, 1995). Data on bird assemblages that breed in these habitats therefore come from decentralized, local, and sometimes ad hoc monitoring efforts that differ in their goals and protocols (Marsh and Trenham, 2008). In such cases, when data from larger geographic scales is absent, local data collected using a diversity of methodologies and at shorter time scales can be combined to estimate long-term trends in abundance (Houlahan et al., 2000; Loh et al., 2005; Van Strien et al., 2013; Pagel et al., 2014). Such

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analyses allow local conservation and management considerations to be placed in broader geographic contexts (Houlahan et al., 2000). Here, we combine local and regional point count survey data, controlling for inter-survey differences in protocol and detection probabilities (Sólymos et al., 2013a), to estimate population trends for a group of spruce-fir forest birds.

Vulnerable and threatened spruce-fir forest birds of the upper Midwestern and Northeastern regions of the United States are an example of an assemblage that is poorly covered by continental-scale monitoring programs. High-elevation spruce-fir forests occur on the tops and sides of mountains on steep, difficult terrain, and accessibility is largely limited to hiking or ski trails. Low-elevation spruce-fir forests are dense and boggy with few roads to interior patches. The BBS often misses species that breed largely in these inaccessible forests. For example, the Bicknell's Thrush (*Catharus bicknelli*), a globally vulnerable species (IBTCG, 2010) and an indicator of montane spruce-fir habitat (US Forest Service, 2006), has not been detected on a BBS route in the United States since 1996 (Sauer et al., 2014). As a result, little information exists regarding long-term population trends of spruce-fir birds at broad geographic scales (Niven et al., 2004; King et al., 2008). Spruce-fir forest birds in the United States are affected by anthropogenic development (Glennon and Porter, 2005; Zlonis and Niemi, 2014), commercial timber harvests (Titterington et al., 1979), defoliation from episodic insect pest outbreaks (Venier and Holmes, 2010), atmospheric deposition of environmental toxins (Rimmer et al., 2005), and may be especially vulnerable to modern climate change (Atwood et al., 1996; Rodenhouse et al., 2008; Ralston and Kirchman, 2013). Spruce-fir forest ecotones may already be shifting upwards in elevation (Beckage et al., 2008). Birds at their southern periphery are shifting their distribution northward (Zuckerberg et al., 2009), occupying unsuitable habitats (DeLuca, 2013), and suffering losses in reproductive success as a result of modern warming (Waite and Strickland, 2006). Climate change may also be causing an increase in occupancy of an important nest predator, the red squirrel (*Tamiasciurus hudsonicus*), in montane spruce-fir forests (Rimmer et al., 2001; DeLuca, 2013). It is therefore important to establish population baselines for these climate vulnerable species, especially at their southern periphery.

Because of heightened conservation concern for this assemblage, several organizations have established monitoring programs that specifically target spruce-fir birds, some of which have now been implemented for over two decades. For example, the Vermont Center for Ecostudies, the White Mountain National Forest (WMNF), and the Wildlife Conservation Society each coordinate long-term survey programs in montane forests or low-elevation boreal spruce bogs (US Forest Service, 2006; Scarl, 2011; Glennon, 2014). In addition, several National Forests, Parks, and Wildlife Refuges throughout the Northeast and upper Midwest have endeavored to monitor spruce-fir forest bird species on local or regional scales (Howe and Roberts, 2005; King et al., 2008; Johnson, 2012; Zlonis et al., 2013; Faccio and Mitchell, 2014). Our goal was to combine and collectively analyze these datasets for the first time in order to estimate broad scale trends in abundance.

2. Methods

2.1. Study area

We describe spruce-fir forests of the eastern United States as forested landscapes in which spruce (red spruce [*Picea rubens*], white spruce [*P. glauca*] and/or black spruce [*P. mariana*]) and balsam fir (*Abies balsamea*) are dominant or codominant. This is a catch-all definition and includes a variety of habitat types (Eyre, 1980; Pastor and Mladenoff, 1992; Sperduto and Nichols, 2011; Edinger et al., 2014) covering over 5 million ha in the upper Midwest (Minnesota, Wisconsin, Michigan; hereafter 'Midwest'), and Northeast (New York, Vermont, New Hampshire, Maine; hereafter 'East'; US Forest Service, 2010; Fig. 1). These Midwestern and Eastern regions correspond closely to physiographic strata used in previous analyses of regional avian trends (Sauer and Droege, 1992; Sauer et al., 2014) and used by Partners in Flight as conservation units (Rich et al., 2004). The Midwestern region of the present study corresponds to the "Boreal Hardwood Transition" physiographic area, and the Eastern region consists primarily of the "Adirondack Mountains" and "Spruce-Hardwood Forests" areas (following Partners in Flight terminology). At forested wetland sites, black spruce dominates with tamarack (*Larix laricina*) and little or no fir. Lowland sites with drier soils are composed of red spruce, balsam fir and occasionally white spruce, or white pine (*Pinus strobus*). In the Midwest, upland spruce-fir forests include varying amounts of quaking aspen (*Populus tremuloides*) and paper birch (*Betula papyrifera*). In the mountainous east, spruce-fir dominates at mid to high elevations and can contain mountain paper birch (*Betula cordifolia*) and mountain ash (*Sorbus americana*). At higher elevations, spruce and broadleaf species decrease in abundance and mountain forests can be nearly pure stands of balsam fir. Due to their ecological distinctiveness and vulnerability, spruce-fir forests have been recognized as a key component of regional biodiversity across Northeastern and Midwestern United States.

2.2. Species selection

We constructed a list of avian spruce-fir forest obligates and associates by consulting authoritative sources that provide matrices of 'preferred' or 'utilized' habitat types for birds in the Midwest (Robbins, 1991) and East (DeGraaf and Yamasaki, 2001). We defined spruce-fir forest 'obligates' as species that prefer and utilize only spruce-fir forest types. We defined spruce-fir 'associates' as species that prefer spruce-fir, but also utilize other forest types. These inclusion criteria excluded a number of species that can be common in spruce-fir forests but do not 'prefer' them and are also broadly distributed in other forest types. Further, we considered only passerines for analysis, as detection of non-passerines during point count surveys can be low. We characterized 18 passerines as either spruce-fir obligates ($n = 8$), or associates ($n = 10$; Table 1). Our list is largely coincident with target species of boreal bird surveys (King et al. 2008; Scarl, 2011; Glennon, 2014), and includes several species considered ecological indicators for high-elevation spruce-fir forest (Bicknell's Thrush, Magnolia Warbler [*Setophaga magnolia*] and Yellow-bellied Flycatcher [*Empidonax flaviventris*]; US Forest Service, 2006). Four species on our list, Bay-breasted Warbler (*Setopahaga castanea*), Rusty Blackbird (*Euphagus carolinus*), White-winged Crossbill (*Loxia leucoptera*), and Pine Siskin (*Carduelis pinus*) were excluded entirely from analyses because of insufficient data, leaving 6 obligates and 8 associates in our analysis.

2.3. Point count data

Point count data were obtained from 16 monitoring programs (hereafter 'programs') throughout the spruce-fir forest zone of the Midwestern and eastern United States (Fig. 1; Online Appendix Table A1). Point counts took place within the period from 1989 to 2013 and varied across programs in temporal (mean: 13 years; range: 2–24 years) and spatial coverage (median: 3269 km²; range: 159–426,059 km²; Online Appendix Table A1). All surveys included in our analyses are standard single-observer
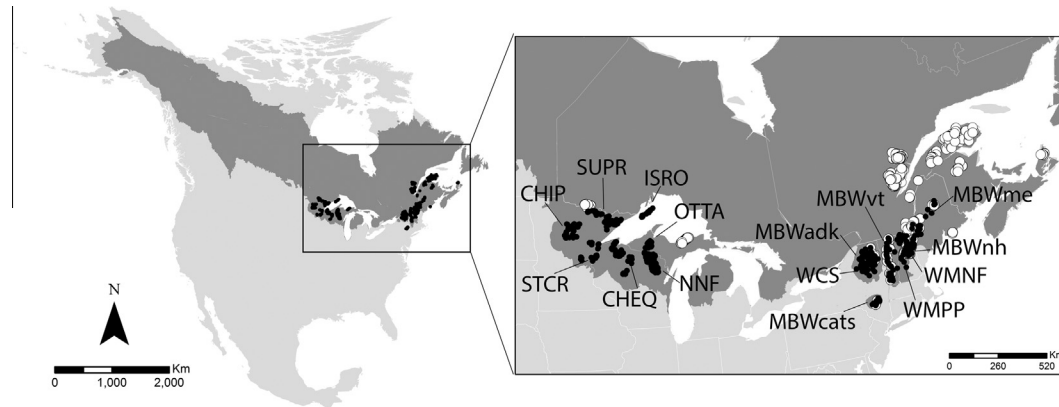
Fig. 1. Location of point count surveys. Dark gray shading is the North American distribution of spruce-fir forests. White circles in inset represent survey locations used to fit detection models, but not used in trend analyses because of insufficient data. Black circles in inset are survey locations used in trend analyses: CHIP = Chippewa National Forest, MN; SUPR = Superior National Forest, MN; STCR = St. Croix State Forest, MN; CHEQ = Chequamegon National Forest, WI; ISRO = Isle Royale, MI; NNF = Nicolet National Forest, WI; OTTA = Ottawa National Forest, MI; WCS = Wildlife Conservation Society, Adirondack Low Elevation Boreal Bird Surveys; MBW = Vermont Center for Ecostudies, Mountain Birdwatch in Adirondacks (MBWadk), Catskills (MBWcats), Green Mountains (MBWvt), White Mountains (MBWnh), and Maine (MBWme); WMPP = White Mountain National Forest PermaPlot Surveys; WMNF = White Mountain National Forest High Elevation Bird Surveys.

Table 1
14 species categorized as spruce-fir forest obligates or associates, and estimated overall and regional trends. Trend estimates less than 1.0 indicated a population decrease, estimates greater than 1.0 indicated an increase, and trends equal to 1.0 indicated a stable population.

Species	Spruce-fir habitat use	Migratory strategy	Overall trend[a]	Midwestern trend[a]	Eastern trend[a]
Olive-sided Flycatcher (*Contopus cooperi*)	Associate	Migratory	0.960 (0.947–0.973)*	0.961 (0.948–0.975)*	0.951 (0.901–1.008)
Yellow-bellied Flycatcher (*Empidonax flaviventris*)	Associate	Migratory	0.976 (0.971–0.981)*	0.980 (0.973–0.987)*[b]	0.971 (0.964–0.978)*[b]
Gray Jay (*Perisoreus canadensis*)	Obligate	Non-migratory	0.981 (0.962–0.999)*	0.921 (0.900–0.941)*[c]	1.037 (1.012–1.065)*[c]
Boreal chickadee (*Poecile hudsonicus*)	Obligate	Non-migratory	1.006 (0.997–1.015)	–	1.006 (0.997–1.015)
Red-breasted Nuthatch (*Sitta canadensis*)	Associate	Non-migratory	1.016 (1.011–1.021)*	1.025 (1.019–1.032)*[c]	0.987 (0.978–0.998)*[bc]
Golden-crowned Kinglet (*Regulus satrapa*)	Associate	Migratory	1.014 (1.009–1.021)*[b]	1.025 (1.017–1.033)*[c]	1.000 (0.992–1.008) [c]
Ruby-crowned Kinglet (*Regulus calendula*)	Associate	Migratory	1.054 (1.046–1.062)*	1.052 (1.037–1.067)*	1.054 (1.046–1.065)*
Bicknell's Thrush (*Catharus bicknelli*)	Obligate	Migratory	0.977 (0.968–0.986)*	–	0.977 (0.968–0.986)*
Swainson's Thrush (*Catharus ustulatus*)	Associate	Migratory	1.035 (1.031–1.039)*	0.988 (0.980–0.996)*[bc]	1.043 (1.038–1.048)*[c]
Magnolia Warbler (*Setophaga magnolia*)	Obligate	Migratory	0.991 (0.987–0.996)*[b]	0.995 (0.989–1.001)	0.995 (0.989–1.003)
Cape May Warbler (*Setophaga tigrina*)	Obligate	Migratory	1.013 (0.998–1.030)	1.013 (0.998–1.030)	–
Palm Warbler (*Setophaga palmarum*)	Associate	Migratory	1.079 (1.046–1.118)*	–	1.079 (1.046–1.118)*[b]
Blackpoll Warbler (*Setophaga striata*)	Obligate	Migratory	0.991 (0.988–0.994)*[b]	–	0.991 (0.988–0.994)*
Evening Grosbeak (*Coccothraustes vespertinus*)	Associate	Non-migratory	0.937 (0.923–0.957)*	0.946 (0.930–0.970)*[c]	0.856 (0.836–0.873)*[c]

[a] Asterisk (*) indicates trends significantly different from 1.000 (90%CI do not overlap with 1.000).
[b] Trends that become non-significant when considering propagated error as opposed to bootstrap estimated 90% confidence intervals.
[c] Species for which Midwestern and Eastern trends were significantly different.

point count surveys, but differ in their duration (5–10 min), number and length of time intervals (1–10 intervals ranging from 1 to 5 min), survey radius (100 m – unlimited radius), and number of distance intervals (1–5 intervals; Online Appendix Table A1). Surveys were generally conducted in the early breeding season (mean survey date: June 15) and early morning (mean survey time: 1.77 h after local sunrise). All fly-over and other visual observations were removed from analyses. Individual programs were located entirely in the United States and covered local or regional geographic areas (159–8367 km²) except Vermont Center for Ecostudies' Mountain Birdwatch (MBW) which covers high elevation spruce-fir forests in New York and New England and extends into eastern Canada (426,059 km²).

2.4. Estimating trends

Point count data varied across programs in count duration, survey radius, and number of distance sampling categories, variation which could bias estimates of detection probability and population trends (Etterson et al., 2009; Matsuoka et al., 2012; Sólymos et al., 2013a). To account for the heterogeneity in our datasets, we employed the 'QPAD' method (Sólymos et al., 2013a), which uses removal (Farnsworth et al., 2002) and distance sampling (Buckland et al., 2001) methods to control for the effects of survey protocol in the estimation of detection probabilities. Probability of detection is the product of availability (p, the probability that a present individual sings during the survey) and perceptibility (q, the probability that a singing individual is detected by an observer; Marsh and Sinclair, 1989). The expected count of a given species during a point count survey can therefore be written as: $E(C) = Npq$, where N is the true species abundance. QPAD provides conditional maximum likelihood estimates of p and q while allowing survey radius, duration, and the number of distance and time intervals to vary across programs. It also accounts for covariates that might influence detection including time since sunrise, Julian day, and percent tree cover (Sólymos et al., 2013a).

We used the package 'detect' (Sólymos et al., 2013b) in program R version 3.1.2 (R Core Team, 2013) to implement the QPAD approach and fit survey data to nine removal and two distance models (Online Appendix Table A2). Removal models included Julian day, time since local sunrise, and their quadratic terms as covariates of availability. Distance models used percent tree cover as a covariate of perceptibility (Sólymos et al., 2013a). Percent tree cover for each point was obtained from 2010 MODIS Vegetation

Continuous Fields of 250 m resolution (DiMiceli et al., 2011). Fitted removal and distance models with the lowest BIC were selected for each species (Online Appendix Table A2), then applied to the entire data set to obtain estimates of p, q, and the effective survey area (A) for every surveyed point.

To better match spatial scale of other programs, we divided MBW data into five programs for trends analysis: Adirondacks, NY; Catskills, NY; Green Mountains, VT; White Mountains, NH; and Maine. MBW survey protocol changed in 2011, and while our approach accounts for inter-program differences in protocol, we did not allow intra-program variation. Therefore, MBW surveys after 2010, which includes all surveys in Canada, were removed from trend analyses. Survey points at which a species was never found and programs with less than eight years of data were excluded from trend analyses. This resulted in a total of 15 programs for which we analyzed trends for each species. Based on results from a power analysis (Online Appendix), we excluded from trend analyses program data for each species with a sample size (detections and non-detections) of less than 100 or less than 50 detections.

Trends in abundance were calculated for each species and program using a nonlinear regression, implemented in R, with equation $N = ab^y c$, where a is the intercept (abundance at first year of program), b is the slope (trend), y is the survey year, and c is an offset for the detection probability (multiple of p, q, and A) (King et al., 2006, 2008; Sólymos et al., 2013a). Trend estimates less than 1.0 indicated a population decrease, estimates greater than 1.0 indicated an increase, and trends equal to 1.0 indicated a stable population. To estimate confidence intervals around trends we first performed 1000 bootstrap resamples of the survey data for each program. We then conducted a nonlinear regression following the methods above on each bootstrap, and used the 5th and 95th percentiles of the slopes from these regressions, respectively, as the lower and upper bounds of 90% confidence intervals. We considered a trend as significant if 90% CI did not overlap with 1.000. We use 90% CI because of the potential conservation consequences of failing to detect declining trends (Bart et al., 2004; King et al., 2008). We examined plots of residuals over time for each species-by-program combination and found no indication of serial autocorrelation. Because the coverage of points varied across years in some programs, we used linear regression to look for directional changes in latitude, elevation, and tree cover of points surveyed that might bias trend estimates. We tested for the effects of survey methodology on trends by resampling program data to include counts recorded in a subset of the time and distance intervals and comparing trends estimated from whole datasets to those estimated from subsets of the same programs. For example, when possible we calculated and compared trends from entire ten minute surveys and trends from the first 5 min of those same surveys. Similarly, we calculated and compared trends from unlimited radius surveys, to trends estimated from the same surveys when only birds detected within 100 m were included.

To estimate the regional (Midwest and East) and overall (all programs) trends for each species we used an approach similar to route regression (Geissler and Sauer, 1990; King et al., 2006), and found the weighted mean (b) of program-level trends using the equation $b = \Sigma(w_i b_i)$. Relative program weights (w_i) were proportional to abundance at the midyear of the program (a_i), length in years of the program (y_i), and inversely by the variance associated with the trend estimate (v_i). So, $w_i = c_i/\Sigma c_i$, where $c_i = a_i y_i/v_i$ (King et al., 2006). 90% CI were estimated for regional and overall trends using bootstrap resampling as described above. We concluded a significant difference in the trends between Midwestern and Eastern regions for a species if the 95% confidence intervals around the difference between these trends did not overlap with 0.00. To ensure that our bootstrap estimated confidence intervals did not unfairly underestimate uncertainty, we additionally propagated uncertainty from program-level trends using a Gaussian error propagation approach (Lo, 2005). Propagated error for regional and overall trends was calculated by dividing the square root of the sum of the squared errors for program-level trends by the number of program level trends (Lo, 2005). This effectively finds the mean error for program level trends and applies it to the regional or overall trend estimates. We used Chi-squared tests implemented in R to determine whether population trends differed between migratory and non-migratory species, or between spruce-fir 'obligates' and 'associates'. We hypothesized that spruce-fir obligates would be more likely to show declines than associates which might be buffered against population declines by the ability to utilize additional habitat types (Kotiaho et al., 2005; Newbold et al., 2013).

3. Results

Trend estimates varied considerably across surveys within each species. For species with data from multiple surveys the mean range in trend was 0.13 (Online Appendix Table A3). Seven of our focal species (50%) demonstrated overall significant declines as determined by weighted means of all program-specific trends and bootstrap estimated 90% confidence intervals (Olive-sided Flycatcher [*Contopus cooperi*], Yellow-bellied Flycatcher, Gray Jay [*Perisoreus canadensis*], Bicknell's Thrush, Magnolia Warbler, Blackpoll Warbler [*Setophaga striata*], and Evening Grosbeak [*Coccothraustes vespertinus*]), five significantly increased (Red-breasted Nuthatch [*Sitta canadensis*], Golden-crowned Kinglet [*Regulus satrapa*], Ruby-crowned Kinglet [*Regulus calendula*], Swainson's Thrush [*Catharus ustulatus*], and Palm Warbler [*Setophaga palmarum*]), and two exhibited no overall change (Boreal Chickadee [*Poecile hudsonicus*] and Cape May Warbler [*Setophaga tigrina*]; Table 1, Fig. 2). Chi-squared tests indicated significant differences in trends of obligate and associate spruce-fir species ($\chi^2 = 7.00$, df = 2, P = 0.030). A larger proportion of obligates showed overall declines (66.7%) compared with associates (37.5%), and no obligate species showed a significant overall increase, while 62.5% of associate species significantly increased. There was no significant difference in the proportion of migratory and non-migratory species showing overall declines ($\chi^2 = 0.63$, df = 2, P = 0.730). Regional trends were significantly different between the Midwestern and Eastern regions for one obligate, and four associate species (Table 1).

Propagated errors for regional and overall trends were largely consistent with bootstrap estimated confidence intervals. For three species, using propagated error resulted in a categorical change in overall trend, relative to interpretations from 90% confidence intervals. Golden-crowned Kinglet, Magnolia Warbler, and Blackpoll Warbler each had propagated uncertainty for overall trends overlapping with 1.000, while they were interpreted as having significant trends using bootstrap estimated confidence intervals (Table 1). If these three species were considered as having stable trends, chi-squared tests still indicated significant differences in trends of obligate and associate spruce-fir species ($\chi^2 = 9.1$, df = 2, P = 0.011), but obligates no longer showed a larger proportion of declining species (33.3%), compared to associated (37.5%). No other conclusions differ based on the use of either 90% confidence intervals or propagated error.

We found little evidence that detected declines were the result yearly variation in suitability of surveyed sites or biased by survey protocol. We did detect significant changes in either elevation, latitude, or percent tree cover for 7 of the survey programs (Online

Appendix). However, the magnitude of changes were minimal, and in most cases the changes were into presumably more suitable conditions (higher elevation, higher latitude, or greater tree cover) meaning any reports of declines in these programs may be conservative estimates of declines. In only two cases were changes in suitability over the duration of the survey program to the effect that reported trends may be negatively biased (significant decreases in surveyed elevation in White Mountain National Forest Perma Plots and latitude in Superior National Forest), but based on the magnitude of these changes we feel any bias in these programs will be negligible. By comparing trends estimated from whole datasets and subsets based on duration and radius, we find no consistent evidence that trend is biased by survey protocol. We detected a significant difference in trend and a categorical change in trend interpretation when using protocol subset data in only 2 of the 76 (2.6%) possible comparisons, and these results were not consistent across species or programs.

4. Discussion

This study demonstrates how analysis of multiple datasets can be coordinated to identify population trends across multiple geographic regions and scales. Because ecological data from multiple geographic regions and spatial scales are valuable in effectively managing threatened species and communities (Poiani et al., 2000; Johnson et al., 2004; Wallace et al., 2010), this approach can be used to increase the utility of local monitoring efforts and their contribution to conservation planning. This may be especially useful for rare species or those that inhabit inaccessible or geographically sparse habitats. These species tend to be underrepresented in continent-scale monitoring programs but may be extensively monitored by ad hoc surveys at a local or regional scales (Hanowski and Niemi, 1995).

In the case of avian spruce-fir species, the combination of local and regional data to examine population trends has allowed us to
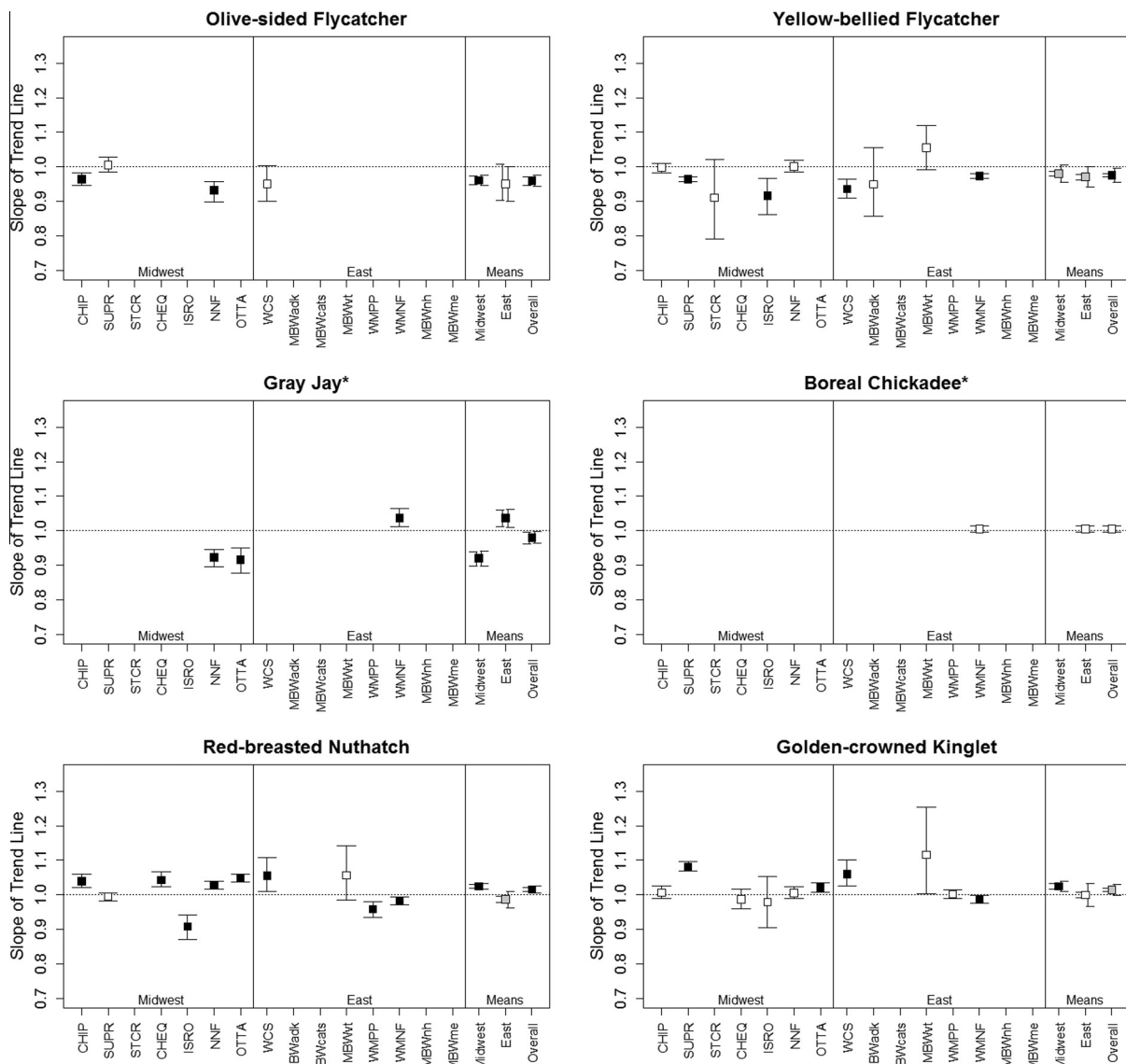

Fig. 2. Trend estimates for each species and surveys with a sample size greater than 100 and greater than 50 detections. Error bars around Midwest and East surveys represent bootstrap estimated 90% confidence intervals. Error bars around regional and overall mean trends are divided to display both represent bootstrap estimated 90% confidence intervals (left) and propagated uncertainty from survey-level trends, calculated using Gaussian error propagation (right). Closed black boxes represent significant trends (90% CI do not overlap with 1.0), and open boxes are non-significantly different from 1.0. Gray boxes represent mean trends with 90%CI that do not overlap with 1.0, but that are considered non-significant according to propagated error. Asterisks next to species common names indicate spruce-fir obligate species. Surveys along X axis are ordered roughly west to east.
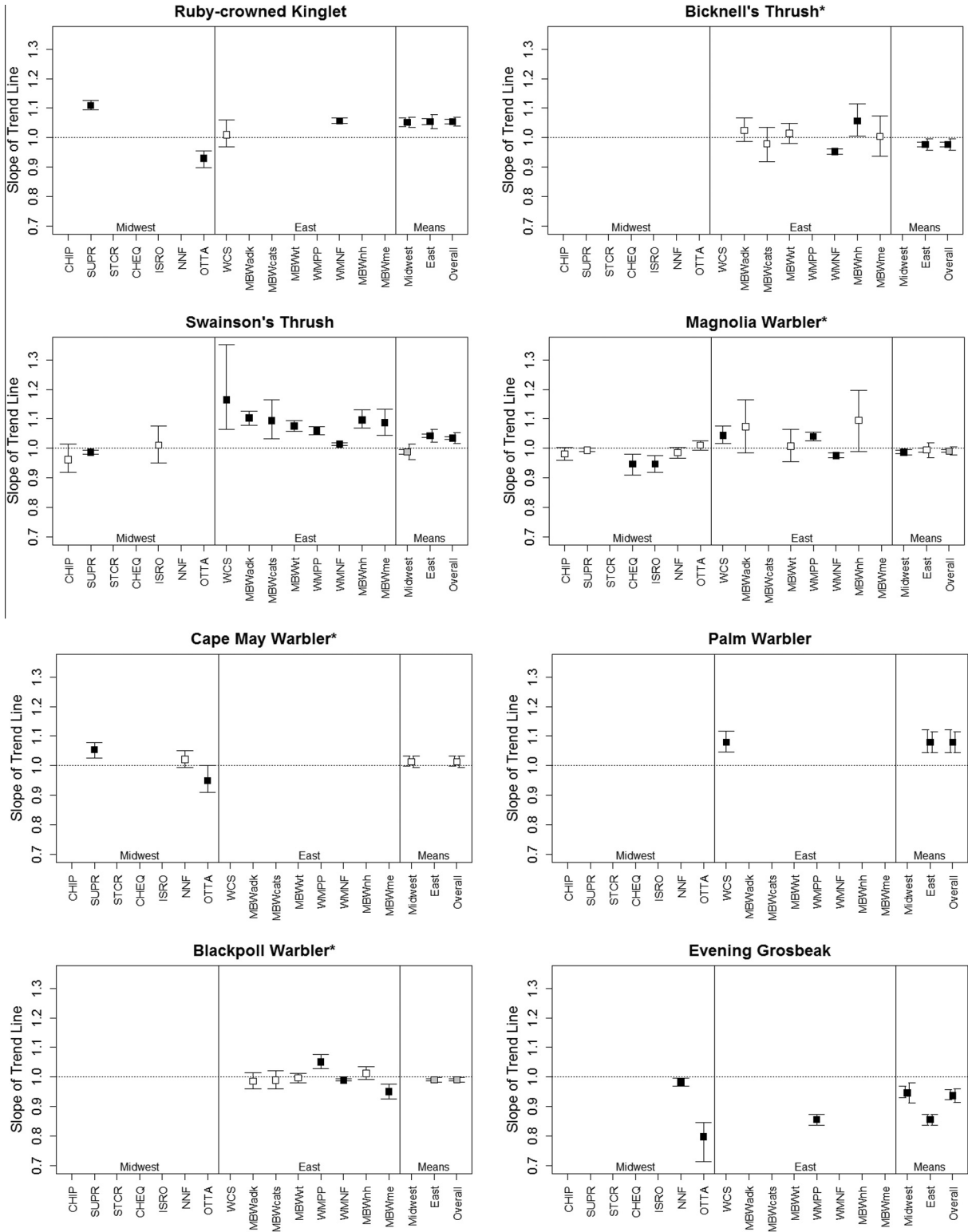
Fig. 2 (*continued*)

refine our understanding of declines in species of conservation concern and to identify declines in species previously thought to be secure. Examples of the former include Bicknell's Thrush and Olive-sided Flycatcher, both listed by the Committee on the Status of Endangered Wildlife in Canada as "Threatened", and by the International Union for the Conservation of Nature (IUCN) as "Vulnerable" and "Near Threatened", respectively. Recent

decreases in occupancy have been reported for Olive-sided Flycatcher in the Adirondacks (Glennon, 2014) and long-term declines have been reported in other parts of the species' distribution (Altman and Sallabanks, 2012; Sauer et al., 2014). A significant overall decline reported here extends concern for this species in our study area. Bicknell's Thrush has received extensive conservation attention (IBTCG, 2010), and our results suggest this should

continue, and that similar attention would be justified in Olive-sided Flycatcher. Gray Jay, Yellow-bellied Flycatcher, Magnolia Warbler, Blackpoll Warbler, and Evening Grosbeak are all categorized by IUCN as species of "Least Concern" and have received minimal conservation attention, though population declines have previously been reported for some of these species (Bonter and Harvey, 2008; King et al., 2008; Sauer et al., 2014; Glennon, 2014). We find evidence for overall declines in all of these species, though propagated errors for Magnolia Warbler and Blackpoll Warbler indicate declines are possibly non-significant. That we find at least some evidence for decline in these species suggests further investigations of their statuses in the eastern and Midwestern United States is warranted. For one species, Evening Grosbeak, dramatic modern declines in abundance and range contractions (Bonter and Harvey, 2008; Sauer et al., 2014) follow an equally large range expansion in the late 19th and early 20th centuries (Gillihan and Byers, 2001). More work is needed to determine whether the current trajectory is part of naturally dynamic population processes in this irruptive species, or whether it is indicative of more widespread environmental degradation.

Examining trends at multiple scales and geographic regions has allowed us to better understand the population status of several species. For example, by combining data from multiple survey programs that access underreported high elevation forests, we were able to detect a significant increase in Swainson's Thrush in the Northeast, where all eight survey programs showed a significant increase in this species. Using data from BBS and Christmas Bird Counts, BirdLife International (2015) reports a range-wide non-significant decline for Swainson's Thrush, similar to what we observe in our Midwest datasets. Swainson's Thrush has significantly increased in high elevations in recent decades (Scarl, 2011), and we believe our regional Eastern trend captures this population growth at higher elevations that may be missed in large scale national datasets. Similarly, by using a weighted mean of program-level trends, we find support for declines in both Yellow-bellied Flycatcher and Blackpoll Warbler. Previous analyses using data from only large scale data sets, or from a single local survey program failed to detect significant declines for these species (King et al., 2008; Scarl, 2011; BirdLife International, 2015). Despite the potential value of combining datasets to estimate regional and overall trends, our results for Bicknell's Thrush demonstrate that weighted means should be interpreted cautiously and only in the context of program-level trends. The significant overall trend for Bicknell's Thrush in this study is driven by WMNF High Elevation Surveys, the only program showing a significant decline. Contrasting with this, MBW surveys reveal no significant changes for Bicknell's Thrush. This seeming contradiction might be partly explained by a difference in program years (King et al., 2006). Because programs differ in the temporal coverage, differences in trends across programs may represent both geographic and temporal variation. It is possible that for Bicknell's Thrush, MBW data (2003–2010) indicate a recent leveling off of populations following declines on a longer time scale as indicated by WMNF data (1993–2013). Examining program level trends in concert allows us to be specific about where and when populations have been declining.

Spruce-fir forests in the United States have been affected in recent history by commercial forestry, anthropogenic development, disturbance from insect pest outbreaks, atmospheric deposition, and climate change (Miller-Weeks and Smoronk, 1993; Glennon and Porter, 2005; Fraver et al., 2007; Rodenhouse et al., 2008). Any of these factors, in addition to those faced during migration and on the wintering grounds, might be contributing to the observed trends. Although our analyses cannot explicitly identify causal mechanisms underlying population trends, we argue the present study does provide a framework in which causal

mechanisms operating at multiple spatial scales can be investigated in future studies. Trends for five species were significantly different between Midwest and East regions, and variation in trends exists across programs within species. Environmental stressors, land use history, management practices, and effects of climate change also vary regionally and locally and might contribute to geographic variation in avian trends. Future studies that correlate population trends with regional and local environmental factors may be able to identify the mechanisms underlying population trends and the best management practices for this avian assemblage. For example, the effects of climate change on spruce-fir communities are, in particular, of great concern to managers and conservationists (Rich et al., 2004; Rodenhouse et al., 2008). Future studies that compare trends from the present study with those in more northerly Canadian populations may be able to determine whether climate change is directly contributing to trends in avian abundance. Similarly, the severity and periodicity of spruce budworm (Choristoneura fumiferana) outbreaks differs considerably between the Midwestern and Eastern regions (Fraver et al., 2007; Robert et al., 2012), providing an opportunity to determine how this disturbance agent might be impacting long-term trends in associated bird species. Further investigations to identify point-level environmental and disturbance factors driving trends and how climate change affects those factors are currently under way.

We present the most data-rich and detailed examination of avian spruce-fir population trends to date for a large portion of the eastern and Midwestern regions of the United States. Results indicate that conservation concern for this group is warranted with 50% of spruce-fir species significantly declining across the study region. This includes significant declines in 66.7% of spruce-fir obligates, which were more likely to decline than associates. We were able to confirm and extend species-specific conservation concern within the spruce-fir forest community and identify species for which additional attention may be needed. However, we emphasize that despite the combination of local data, several spruce-fir species still lack enough data to thoroughly evaluate their status. For example, while we find no significant overall decline for Boreal Chickadee, this trend estimate comes from a single survey program in the Northeast. BirdLife International (2015) reported significant declines in Boreal Chickadee populations, and it is possible that our failure to detect these trends comes from a lack of geographic coverage in our dataset. For four species, data were insufficient to estimate trends anywhere in the study area. This includes Rusty Blackbird which may be one of the most precipitously declining species in North America (Greenberg and Droege, 1999; Greenberg and Matsuoka, 2010). This emphasizes the need for continued and expanded efforts to monitor this threatened assemblage.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.biocon.2015.04.029.

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