

## AN ABSTRACT OF THE THESIS OF

Michael S. Ellis for the degree of Master of Science in Wildlife Science presented on April 1, 2016.

Title: Twenty-year changes in riparian bird communities of east-central Oregon.

Abstract approved: \_\_\_\_\_

Major Professor

Over the last 50 years, riparian zones in the semi-arid West have gained recognition as disproportionately important habitats for both breeding bird communities and agricultural operations. Despite growing interest in exploring avian-habitat relationships in these systems to better inform land management, few studies have attempted to describe temporal changes in the region's riparian bird communities. To provide a frame of reference for these changes, we compared indices of avian abundance and diversity from three streamside vegetation associations in east-central Oregon during the 2014 breeding season with baseline data collected by TA Sanders and WD Edge in 1993 and 1994 (Sanders and Edge 1998). Our objectives were to identify patterns of change in the avian community with a focus on riparian shrub-dependent species, to re-examine previously reported relationships between avian abundance and vegetation volume, and to identify possible causes of declines in abundance and diversity with the goal of providing land managers in the semi-arid region information to be used in the guidance and adaptation of management practices. We combined field protocols used by Sanders and Edge to survey birds along point count transects and to measure riparian vegetation with modern analytical techniques. We found few major differences in overall diversity between study

periods, but documented precipitous declines in detections for two of three riparian shrub-dependent focal species –Yellow Warbler (*Setophaga petechia*) and Willow Flycatcher (*Empidonax trailii*). Song Sparrow (*Melospiza melodia*) detections declined by a smaller margin. Changes in number of detections for these species did not reflect trends in mesic shrub volume, which had been identified as a likely driver of obligate species abundance by Sanders and Edge but remained relatively unchanged between study periods. Declines of all three focal species reflected regional Breeding Bird Survey trends, corroborating our finding that their declines may not be a result of changes in local site conditions. Compositional similarity between avian communities was significantly higher in 2014 than in 1993-1994, exhibiting a shift toward greater homogenization between structurally distinct riparian habitats. Our results suggest managing working lands for riparian shrub cover or volume, an important metric of grazing intensity and riparian system health, may not be enough to conserve diversity and abundance of riparian-shrub dependent birds, and more work should be done to identify and alleviate factors contributing to these species' declines.

**KEYWORDS:** Bird, community, habitat, Hill numbers, historical, Oregon, riparian, time, Intermountain West, willow

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Twenty-year changes in riparian bird communities of east-central Oregon

by  
Michael S. Ellis

A THESIS

submitted to

Oregon State University

in partial fulfillment of  
the requirements for the  
degree of

Master of Science

Presented April 1, 2016  
Commencement June 2016

Master of Science thesis of Michael S. Ellis presented on April 1, 2016

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Michael S. Ellis

## ACKNOWLEDGEMENTS

Funding for this study was provided by the Campbell Family Fisheries & Wildlife Fund and the Eastern Oregon Agriculture & Natural Resource Program (EOANRP). We thank the 96, Gander, Carter and Silvies Valley ranches for their hospitality, access to their properties, and support throughout our study. We also thank our field technician, Samuel Artiaiz, and the supportive staff of both OSU and EOANRP.

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# **Twenty-year changes in riparian bird communities of east-central Oregon**

## **INTRODUCTION**

Riparian zones in the semi-arid Intermountain West cover less than one percent of the region's land area but disproportionately contribute to ecosystem function, natural resource pools, and biodiversity (Knopf et al. 1988, National Research Council 2002, Ward et al. 2002). Vegetation communities in riparian zones are particularly important for breeding birds and may be used by two-thirds to four-fifths of all species in semi-arid landscapes (Carothers et al. 1974, Anderson and Ohmart 1977, Johnson and Haight 1985). A subset of these species, such as the Willow Flycatcher, Yellow Warbler, and Song Sparrow, may be riparian-obligate or -dependent species susceptible to population declines and/or extirpation with loss of woody streamside habitat (Rich 2002, Mac Nally et al. 2008).

Humans in the Intermountain West are similarly dependent upon riparian zones, an estimated 98% of which have been altered from their natural states by damming, agricultural irrigation, livestock grazing, urban development, and recreation (Brinson et al. 1981, Benke 1990, National Research Council 2002). Because of the intersecting and occasionally conflicting needs of landowners and wildlife, government agencies, working lands producers, and other stakeholders have increased research on and protection of the region's riparian zones in recent years, often with bird communities as project foci. Successful conservation, restoration, and management of streamside habitats for avian abundance and diversity necessitate long-term monitoring, and a handful of studies have attempted to track long-term avian responses in riparian zones in the Intermountain West (Norvell et al. 2005, Martin 2007, Earnst et al. 2012, Manuwal 2012). All but one of these four studies (Norvell et al. 2005) examine changes in avian communities in

relation to substantial modifications in riparian habitat brought about by management, climate change, and/or weather events. While examining changes in bird communities and their habitats in tandem provides needed insight into habitat relationships, evaluating temporal changes in bird communities when their habitat has been substantially altered may obscure changes stemming from more enigmatic factors (e.g., regional population trends, climate change, wintering ground loss) which may be misattributed to management action or local habitat changes (Magurran 2011).

In 1993 and 1994, Sanders and Edge (1998) investigated relationships between vegetation structure and avian communities in riparian zones of east-central Oregon. Their work helped elucidate the importance of woody mesic shrubs for diverse and abundant breeding bird communities and established shrub volume as a reliable predictor of both overall and riparian shrub-dependent species abundances. Twenty years later, we repeated their study with the objectives of (1) documenting temporal changes in riparian bird diversity on working rangelands; (2) re-evaluating established relationships between three riparian shrub-dependent species (Yellow Warbler, Willow Flycatcher, Song Sparrow) and mesic shrub volume; (3) examining local changes in avian abundance in the context of regional population trends from the Breeding Bird Survey (BBS); and (4) providing a frame of reference for other long-term riparian bird studies.

We framed our study around an *a priori* null hypothesis that changes in avian abundance and diversity observed in 2014 would reflect changes in riparian shrub volume and would be consistent with volume:species abundance relationships identified by Sanders and Edge – i.e., abundance and diversity of birds would increase with increasing vegetation volume; riparian shrub-dependent species would greatly decrease in abundance where shrub volume was <5,000

m<sup>3</sup>/ha and would not be detected at sites with <1,200 m<sup>3</sup>/ha shrub volume. We tested four sets of alternative hypotheses and predictions: (1) diversity, overall abundance, and abundance of riparian shrub-dependent species would increase or decrease in accordance with changes in mesic shrub volume but would no longer follow the same volume relationships identified by Sanders and Edge. Given widespread declines of aerial insectivores – species often associated with riparian zones – across North America (Nebel et al. 2010, Sauer et al. 2014) and the detrimental effects on riparian habitats and their bird communities predicted as a result of a warming climate (Both et al. 2009, Perry et al. 2012, Catford et al. 2013, Kominoski et al. 2013, Friggens and Finch 2015), we suspected species abundances may be declining even in sites where vegetation remained relatively unchanged. (2) Changes in the avian community would occur independently of vegetation changes, driven by factors beyond the scope of this study, such as climate change or loss of wintering habitat, to such an extent vegetation volume no longer would explain the majority of variation in local riparian bird abundance. (3) Any declines we observed in riparian shrub-dependent species abundances would be steepest in “lower suitability” sites with less mesic shrub cover where we predicted riparian shrub-dependent species would be more vulnerable to environmental change. And (4) changes in focal species abundances would follow regional trends from the BBS independently of local site changes.

## **STUDY AREA**

We conducted our study in Bear and Silvies valleys of southern Grant County and northern Harney County in east-central Oregon. The town of Seneca (N44.134909, W118.972278) is central to the two valleys, which are composed primarily of privately-owned ranchlands surrounded by Malheur National Forest. Ditch- and flood-irrigated riparian grasslands dominate

the valley floors (elevation 1,380-1,475m), giving way to sagebrush steppe and conifer forests dominated by Ponderosa Pine (*Pinus ponderosa*) in upland areas. The Silvies River runs north-south through the centers of both valleys and is fed by a number of streams, including Scotty and Bear creeks from the west and east sides, respectively, of Bear Valley. Portions of both creeks and the Silvies River are lined with woody mesic vegetation consisting primarily of willow species (*Salix* spp.), rose (*Rosa* spp.) and currant (*Ribes* spp.). This system was previously described by Sanders and Edge (1998) and Sanders (1995), who provide additional details and a figure of the study area.

Climate in Bear and Silvies valleys is typified by cold winters and warm, dry summers. Streamflow in the Silvies River drainage has, historically, been driven primarily by melting snowpack throughout the spring and summer months, though occasional thunderstorms may provide some moisture. Sanders and Edge (1998) reported annual precipitation averaging 34.3 cm falling mostly as snow from November through June and ambient temperatures averaging -6.2°C in January and 15.0°C in July. In the 20-years following their study, annual precipitation in Seneca, OR averaged 36.6 cm. Average ambient temperatures during this time were -3.4°C in January and 16.6°C in July (PRISM Climate Group 2015). Droughts were governor-declared in Grant County in 1994, 2002, 2007, and 2014 and in Harney County in 2001, 2002, 2003, 2007, and 2014 (Oregon Water Resources Department 2015).

## **METHODS**

### **Sampling Design**

Sanders and Edge (1998) established four replicate sampling areas within each of three riparian habitats differentiated by vegetation. These habitats were termed *continuous*,

*discontinuous*, and *herbaceous vegetation associations* and were distinguished by visual estimates of riparian shrub zone continuity and density. We were unable to regain access to two of the private properties on which all eight of Sanders and Edge's continuous and herbaceous sampling areas were located, but we were able to resample all four of their discontinuous sampling areas. Under the assumption their original sampling areas were representative of riparian habitats throughout Bear and Silvies valleys, we established new continuous and herbaceous sampling areas in visually similar habitats under the same constraints employed by Sanders and Edge (1998), e.g., landowner permission,  $\geq 100$ -m from upland habitats,  $\geq 900$ -m in length.

Few similar locations met these constraints, so we were unable to randomly select new sampling areas. Instead we sampled all available habitat within Bear and Silvies valleys meeting the aforementioned criteria. Following Sanders and Edge (1998), in every sampling area we established one 700-m transect, each with eight sampling sites at 100-m intervals for a total of four transects and 32 sampling sites per vegetation association. Like Sanders and Edge, we use sampling areas as experimental units and replicates.

### **Vegetation Sampling and Riparian Zone Attributes**

Following Sanders and Edge (1998), we sampled vegetation and riparian zone attributes at each sampling site once between 14 June and 5 August 2014. Whereas Sanders and Edge sampled or derived 20 variables to describe riparian zone characteristics and vegetation, we focused our efforts on variables they found or suspected were key predictors of avian abundance and diversity, i.e., mesic shrub volume, height, and percent cover; riparian zone width; riparian shrub zone width; and stream/river width. We verified accuracy of their field techniques for

estimating percent shrub cover at each sampling site by digitizing riparian shrubs on the most recent publicly available aerial imagery (Landsat 2013) and calculating shrub cover in a 40-m radius circle around each sampling site – an area chosen based on Sanders and Edge’s estimation of effective area surveyed for birds (1998). Suspecting that ground cover may be an important explanatory variable correlated to number of bird detections, we followed techniques used by Sanders (1995) to estimate percent cover for grasses and sedges, forbs, rushes, litter, bare earth, and standing water.

### **Bird Sampling**

We followed Sanders and Edge’s protocols for sampling riparian bird communities, however, we only had one field season, 25 May to 21 June 2014, to their two (1993 and 1994). Like Sanders and Edge, we used two trained observers to conduct point counts four times during peak breeding season at each sampling site. To account for differences in detectability among vegetation associations, Sanders and Edge (1998) used program DISTANCE (Laake et al. 1993, Buckland et al. 2005), methods 1 and 5 of Ramsey and Scott (Ramsey and Scott 1979), and cumulative detection curves (Scott et al. 1986). They conservatively estimated all birds detected within 40-m of the observer were equally detectable across all vegetation associations and consequently included only these birds in their analyses. Though we could not adjust for observer differences or differences in detectability between study periods without Sanders and Edge’s raw data (which unfortunately are no longer available), cumulative detection curves for the 2014 study period also estimated detectability to be equal for all birds within 40-m of the observer across all vegetation associations. Therefore, we too truncated our data post hoc to



include only birds detected  $\leq 40$  m from our sampling sites for all comparisons between study periods (i.e., 2014 and 1993-1994) and analyses using detections from the 2014 breeding season.

## **Data analyses**

### *Vegetation and riparian zone characteristics*

Without Sander's and Edge's raw data, we were unable to make statistical comparisons of vegetation and riparian zone characteristics between study periods. The majority of their reported means and standard errors were based on non-normally distributed data with small sample sizes ( $n = 4$  transects per vegetation association). Our own vegetation data from 2014, when averaged over sites within each transect, were non-normally distributed with unequal variances. No standard transformations improved our transect-level data, and we knew little about the shape of the 1993-1994 data, so we concluded too many assumptions would be violated to reliably assess differences between study periods using  $t$ -tests. Sample sizes were too small for  $t$ -test alternatives to detect significant differences between study periods, so we report our means and standard errors for vegetation and riparian zone characteristics alongside those from Sanders and Edge (1998). We believe there is great value in this side-by-side comparison of modern and historical data even without  $p$ -values.

Because our data at the transect level were not conducive to analyses using the techniques employed by Sanders and Edge (1998), to evaluate differences in vegetation and riparian zone characteristics among vegetation associations in 2014, we ran each habitat variable through a 1-way analysis of variance (ANOVA) using linear mixed-effect models in which vegetation and riparian zone characteristics were dependent variables, vegetation association was a fixed effect, and transects were assigned a random effect to account for a potential lack of independence

among sites within a transect (3 vegetation associations x 4 transects x 8 sampling sites;  $n = 96$ ). We excluded the herbaceous vegetation association, where woody mesic vegetation was largely absent, from analyses involving shrub cover, volume, and shrub zone width (2 vegetation associations x 4 transects x 8 sampling sites;  $n = 64$ ). When residuals were not normally distributed or suffered from heteroscedasticity, we applied square root or natural log transformations to meet ANOVA test assumptions. We report test statistics based on transformed values but untransformed means and standard errors for comparison with the 1993-1994 study period. We adjusted p-values for multiple comparisons using False Discovery Rates (FDR; Benjamini and Yekutieli 2001).

### *Birds*

Although they determined the same bird species were equally detectable within 40m of the observer across each vegetation association, Sanders and Edge did not account for detectability in their abundance estimates. Instead, they estimated abundance of each bird species by summing detections from repeated sampling events over sampling sites and transects. When detectability is not accounted for, summed detections are less than ideal indices for abundance (territorial individuals are re-counted, present but undetected individuals and observer effects are not accounted for, etc.). We therefore refer to Sanders and Edge's estimates of abundance simply as "detections" or "number of detections" in this paper. Although true abundance estimates could be calculated from our data, and from Sanders and Edge's raw data were it still available, we continue to use detections summed across sites and sampling events for the sake of comparisons with the historical data.

Without Sanders and Edge's raw data, we were unable to make statistical comparisons between avian detections in 1993-1994 and 2014 using *t*-tests or their alternatives for the same reasons encountered during vegetation analyses (above), so we again rely on their reported means and standard errors. We back-transformed these means to determine total number of detections within each vegetation association for the 1993-1994 study period using the appendices from Sanders' thesis (1995). However, we could not separate detections from the 1993 breeding season from those they recorded in 1994. We therefore refer to their lumped data in our analyses as the "1993-1994 study period."

To compare number of detections temporally between study periods, it was necessary to account for differences in sampling effort (two seasons of avian sampling in 1993-1994 vs. 2014). Rarefaction and extrapolation are two of the methods most commonly used for post hoc statistical adjustments for differences in sampling effort (Chao and Jost 2012, Dornelas et al. 2013). Both methods generate samples of equal size or coverage convenient for analyses, but they do not always reflect equal sampling effort in the field. Instead, to standardize effort in terms of number of point counts and sampling hours for detection comparisons, we divided all reported bird detections from the 1993-1994 study period by two under the assumptions there was no significant annual variation in detection probability and each detection was equally likely to occur in either 1993 or 1994. This approach was supported by Sanders and Edge's finding that, of all species with  $\geq 20$  detections, statistically significant inter-annual variation in detections was observed only for Red-winged Blackbirds (*Agelaius phoeniceus*). Averaging the 1993-1994 dataset preserved differences in numbers of detections between study periods, which may have arisen as a result of natural changes in avian abundance over time, though this cannot be separated from differences in observer detection ability without Sanders and Edge's raw data.

Sanders and Edge (1998) used observed richness of detected species to examine differences in avian diversity among vegetation associations. Species richness is a poor stand-alone index for community diversity, however, because it fails to account for evenness among species frequencies within a community (Jost 2006, Maurer and McGill 2011). To better test for differences in avian diversity among vegetation associations and between study periods, we estimated and compared Hill number diversities ( ${}^qD$ ), i.e. effective numbers of species, for the orders  $q = 0, 1$ , and  $2$  for both the 1993-1994 and 2014 study periods. Hill numbers, which account for both species richness and evenness, mathematically unify a number of commonly used diversity indices using the formula put forth by Jost (2006) based on the work of Hill (1973):

$${}^qD \equiv \left( \sum_{i=1}^S p_i^q \right)^{1/(1-q)}$$

When  $q = 1$ , this formula is undefined, but the limit as  $q$  approaches 1 exists and can be used here (Jost 2006):

$${}^1D = \exp\left(-\sum_{i=1}^S p_i \ln p_i\right)$$

The orders of  $q$  we use are close relatives of species richness ( $q = 0$ ), Shannon entropy ( $q = 1$ ), and Gini-Simpson ( $q = 2$ ) indices and place different weights on rare species ( $q = 0$ ), “typical species” neither common nor rare ( $q = 1$ ), and common species ( $q = 2$ ) (Jost 2006, Chao et al. 2014). We use overlap in 95% confidence intervals derived from 2000 bootstrap replications for each of these indices to identify statistically significant differences in diversity among vegetation associations and study periods following Chao et al. (2014). When comparing diversity among study periods with unequal sampling effort, it is necessary to standardize communities to a standardized sample coverage or sample size (Chao and Jost 2012, Chao et al. 2014). Bird communities from all vegetation associations from both sampling periods had sample coverage

between 98.8-99.9% before standardization. We interpolated or extrapolated each vegetation association to a sample coverage of 99.5%, which we selected because it was the maximum coverage achievable through extrapolation without doubling sample size (Chao et al. 2014). When comparing total community diversity between study periods, we interpolated the 1993-1994 dataset and extrapolated the 2014 dataset to a coverage of 99.8%.

To compare compositional similarity, or overlap, of avian communities between and within study periods, we used three non-parametric similarity indices from the  $C_{qN}$  family – Sørensen ( $C_{0N}$ ), Horn ( $C_{1N}$ ) and Morisita-Horn ( $C_{2N}$ ) (Chao et al. 2008, Jost et al. 2012). All three of these indices are mathematically unified by Hill numbers, and each examines the same aspects of community structure. However, these indices apply different weights to species frequencies following the same pattern with the order  $q$  detailed above and are therefore most useful for interpreting compositional similarity when used together (Jost 2006, Jost et al. 2012). We also use overlap in 95% confidence intervals derived from 2000 bootstrap replications for each  $C_{qN}$  index to identify statistically significant differences in compositional similarity among pairwise comparisons of avian communities from each vegetation association.

We calculated Hill numbers and compositional similarity indices using all bird detections recorded  $\leq 40$  m of the observer during the combined 1993-1994 study period despite the difference in sampling effort because calculation methods for both Hill numbers and  $C_{qN}$  are robust to differences in sample size when samples are large and coverage is relatively high (Chao et al. 2008). We confirmed this with tests on our full and halved datasets, which exhibited differences on the order of 0.01%. We therefore report all diversity-related findings using the full 1993-1994 dataset. We calculated all compositional similarity measures and their 95% CIs using

package *SpadeR* in program R (Chao et al. 2015) and all Hill numbers and their 95% CIs using package *iNEXT* in program R (Hsieh et al. 2015).

As with our vegetation data, we also faced sample-size, shape, and heteroscedasticity issues with our avian detection data—our equivalent to Sanders and Edge’s abundance—when summing within transects. We therefore used ANOVAs and generalized linear mixed effect models (GLMMs) with Poisson distributions to compare differences in total avian detections and differences in focal riparian specialist species (all Poisson-distributed counts) among vegetation associations in 2014. For these models we used counts of birds observed  $\leq 40$  m of the observer summed over four visits at each sampling site as the response variable. Vegetation association was a fixed effect and transects were again assigned a random effect. Since our focal species were not detected in the herbaceous vegetation association in 2014, we excluded herbaceous sampling sites from our analyses for those species (2 vegetation associations x 4 transects x 8 sampling sites;  $n = 64$ ). All vegetation associations were included in the analysis of total detections summed across all observed species ( $n = 96$  sites). We recorded only 11 Willow Flycatcher detections at ten sampling sites in 2014, which was too few to adequately model with a Poisson distribution, so we instead used a binomial presence-absence distribution with a complementary log-log link to run ANOVAs and models for this species. We again used FDR to adjust  $p$ -values for multiple comparisons (Benjamini and Yekutieli 2001).

We re-examined correlations between Song Sparrow, Willow Flycatcher, Yellow Warbler, total observed species detections and riparian shrub volume using the same techniques and  $R^2$  coefficients employed by Sanders and Edge (1998). Because of the previously mentioned issues with our detection data at the transect level, we were skeptical of this technique’s accuracy, so we repeated these tests with Poisson-distributed GLMMs (binomial for Willow Flycatcher)

where numbers of detections at the site level were response variables, shrub volume was the explanatory variable, and transects were assigned a random effect ( $n = 64$  for focal species analyses;  $n = 96$  for total detections). Willow Flycatcher models were adjusted for zero-inflation. We calculated correlation coefficients ( $R^2_{GLMM}$ ) for these models using methods outlined by Nakagawa and Schielzeth (2013). We found shrub cover estimated from digitized aerial imagery explained more variation in bird detections (5-10%) than shrub volume and report this relationship below.

By running our data at the site level rather than across transects as Sanders and Edge did, we increased our analytical sample size and were able to add multiple covariates to our models examining relationships between bird detections and habitat characteristics. We established *a priori* GLMMs based on Sanders and Edge's findings and avian ecology for each riparian shrub-dependent focal species, total bird detections among sites with shrub cover, and total detections of birds in herbaceous sites without shrub cover. We followed a stepwise removal method to identify models with the lowest  $AIC_c$  values. We considered all models within two  $AIC_c$  units of the minimum  $AIC_c$  value to have substantial support and report these models in Table 2 (Burnham and Anderson 2004). Covariates we considered in addition to shrub cover in our global models included volume for each 1m height interval, riparian zone width (distance to adjacent upland habitats), shrub zone width, stream width, grass and sedge cover, forb cover, herbaceous cover height, and litter cover. We checked for collinearity and multi-collinearity of covariates with exploratory plots, Pearson correlation coefficients, and variance inflation factors (VIF); strongly correlated covariates ( $R \geq 0.70$ ,  $VIF \geq 4$ ) were not included in the same model (O'Brien 2007). We examined fit of our models and identified covariates in need of log/square

root transformations and/or polynomial terms using diagnostic plots of detections, residuals, and fitted values.

BBS routes in eastern Oregon are infrequently sampled and overlap with few riparian zones, so we utilize trend data aggregated to the Oregon, Great Basin, and Northern Rockies levels. Our study system is found on the edge of the latter two bioregions. All data were collected from the BBS website and span from 1993 to 2013 (Sauer et al. 2014).

## **RESULTS**

### *Vegetation and riparian zone characteristics*

Differences in woody riparian vegetation between discontinuous and continuous mesic shrub associations in 2014 closely followed patterns observed by Sanders and Edge in their 1993-1994 study period (Table 1). Shrub cover was greater in the continuous mesic shrub association both overall and in each 1-m height interval from 0 to 4 m but not statistically so for the 3- to 4-m layer. Total shrub volume at each site and shrub cover as estimated by aerial imagery were also higher in the continuous mesic shrub association. The herbaceous vegetation association had little woody cover and nothing above 1 m in height.

After accounting for relatedness of sites within transects, we found no statistically significant differences in physical features of riparian zones between herbaceous and discontinuous mesic shrub associations and herbaceous and continuous associations in 2014 (Table 1). Stream width, stream depth, and riparian zone width did not differ between the continuous mesic shrub and discontinuous shrub associations. Though average shrub zone width for the continuous mesic shrub association was over 60% greater than discontinuous mesic shrub association, variation among transects was so high that this difference was not significant (Table 1).



We measured a number of ground cover variables but report only those relevant to our avian detection analyses. Average forb and grass cover and sedge cover were substantially higher during 1993-1994 while average litter cover was higher in 2014. We found differences among vegetation associations in 2014 only for forb cover, which was significantly lower in continuous than herbaceous sites (Table 1).

### *Birds*

In 1993 and 1994 combined, Sanders and Edge (1998) recorded 7,707 total detections of birds. They included 4,016 detections (52%) in their analyses after removing fly-by species and detections beyond 40 m. We recorded 6,972 total detections in 2014. Thirty-five percent of these detections were within 40 m of the observer. After excluding Bank Swallows (*Riparia riparia*), whose presence was primarily a result of one transect's proximity to a colonial nesting site, and single fly-by detections of a Prairie Falcon (*Falco mexicanus*) and a Double-crested Cormorant (*Phalacrocorax auritus*), we were left with 2,330 detections for our comparisons. This was 322 detections more than the average number of birds detected per year (2,008) by Sanders and Edge. We provide side-by-side comparisons of detections for all species by vegetation association and study period in Appendix 1.

The increase in overall detections between study periods was driven by increases in detections in herbaceous and discontinuous mesic shrub associations relative to average annual detections from the 1993-1994 study period (Figure 1). The number of detections we recorded in the continuous mesic shrub association in 2014 was lower than the annual average from 1993-1994 but still nearly 40% higher than those recorded in the herbaceous vegetation association, which had the next most detections in 2014 (Figure 1). Despite this difference, we found no

statistically significant differences in number of detections among 2014 vegetation associations using the analytical techniques employed by Sanders and Edge (one-way ANOVA;  $df = 2, 9$ ;  $F = 1.75$ ;  $p = 0.281$ ). After accounting for multiple comparisons, our GLMM suggested this difference between detections in our continuous and discontinuous mesic shrub associations was suggestive of a trend ( $z = 2.02$ ,  $p = 0.165$ ), but that there was no difference between continuous and herbaceous ( $z = 1.70$ ,  $p = 0.236$ ) or herbaceous and discontinuous mesic shrub associations ( $z = 0.32$ ,  $p = 0.750$ ).

Sanders and Edge recorded detections of 64 bird species at all distances, of which 56 were included in their truncated dataset. In 2014, we recorded 75 species at all distances and 54 species within 40 m of the observer. We found no significant differences in overall richness ( $q = 0$ ), “typical species” ( $q = 1$ ; Chao et al. 2010, Chiu et al. 2014), or abundant species ( $q = 2$ ) when summing total detections and species among vegetation associations for each study period (Figure 2). This suggests a high degree of similarity in terms of both richness and evenness and is supported by similar rank-detection distributions for both study periods (Figure 3). Differences in species richness ( $q = 0$ ) among vegetation associations in 2014 followed the same pattern reported for 1993 by Sanders and Edge (1998) (Figure 2). However, orders  $q = 1$  and 2 provided evidence the avian community in continuous habitats was more diverse than in the discontinuous and herbaceous vegetation associations in 2014. Our discontinuous mesic shrub association did not differ from the herbaceous in diversity of typical bird species ( $q=1$ ) or abundant species ( $q=2$ ; Figure 2).

We found no evidence of a difference in avian species richness ( $q = 0$ ) among discontinuous or continuous mesic shrub associations of the 2014 and 1993-1994 study periods (Figure 2). However, all four of these vegetation associations had higher richness than the 2014 herbaceous

association, and there was some evidence the latter was more species rich than the herbaceous association sampled in 1993-1994. Considering Hill numbers of order  $q = 1$  and 2 provided more insight into differences in diversity between study periods. In both cases, the 2014 continuous mesic shrub association fell out as the single most diverse among all associations from either study period, while the 1993-1994 herbaceous vegetation association was the least diverse. We found evidence that typical and common bird species in the discontinuous mesic shrub association sampled in 1993-1994 were more diverse than the 1993-1994 continuous mesic shrub association. The 1993-1994 discontinuous mesic shrub association's bird community was more diverse than the 2014 discontinuous community as well (Figure 2).

Species turnover was fairly pronounced both overall and within vegetation associations. Seventeen species recorded during the 1993-1994 study period were not documented in 2014 while 15 previously unreported species were detected – a replacement of ~30% of all species over 20 years. Many of the lost and newly detected species were observed only once or twice in either study period (Appendix 1), though some species such as Northern Pintail (*Anus acuta*, 14 detections in 1993-1994), Vaux's Swift (*Chaetura vauxi*, 26 detections in 1993-1994), Forster's Tern (*Sterna forsteri*, 29 detections in 2014), and Marsh Wren (*Cistothorus palustris*, 23 detections in 2014) were fairly common in one study period but entirely absent from the other, even when detections from all distances were considered.

Compositional similarity estimated with  $q = 0$  reflected this turnover (Figure 4). When subsetting for typical and common species, however, compositional similarity was much higher (85-90%). Compositional similarity between study periods was lowest for continuous mesic shrub associations, which were significantly less similar to each other than herbaceous, discontinuous, and overall bird communities were. Also of note is the high degree of

compositional similarity among vegetation associations in 2014. Though we found no conclusive evidence of differences in similarity indices for species richness ( $q = 0$ ) among 2014 vegetation associations relative to those sampled in 1993-1994, pairwise comparisons from 2014 using orders  $q = 1$  and  $2$  were significantly higher than they were in 1993-1994 for all comparisons. (Figure 4).

We found no statistically significant difference in the number of Willow Flycatcher and Yellow Warbler detections between discontinuous and continuous mesic shrub associations in 2014 ( $z = 1.45, p = 0.241$  and  $z = 1.00, p = 0.352$ , respectively), but Song Sparrow detections were significantly higher in continuous habitat than in discontinuous ( $z = 3.96, p < 0.001$ ). For all three species, we observed declines in detections relative to the average from the 1993-1994 study period (Figure 5). These declines occurred mainly in the continuous mesic shrub association and were especially marked for the Willow Flycatcher (- 93%) and Yellow Warbler (- 80%). As expected, we did not detect any of these riparian shrub-dependent species in the herbaceous vegetation association.

Using data from the 2014 study period, we re-tested Sanders and Edge's finding that detections of riparian specialist species and total avian detections were positively correlated with mesic shrub volume. Initial tests of our data revealed shrub cover, as estimated from digitized aerial imagery, was a better predictive metric with less error than vegetation volume measured using the techniques employed by Sanders and Edge, so we report our findings here using aerial cover in lieu of volume. Our GLMMs supported Sanders and Edge's conclusions for detections of all birds ( $z = 2.92, p = 0.015$ ), Song Sparrows ( $z = 6.4, p < 0.001$ ), and Yellow Warblers ( $z = 2.61, p = 0.025$ ) but not Willow Flycatchers ( $z = 1.36, p = 0.175$ ).

We found, however, striking differences in the amount of variation described by volume and shrub cover in our models relative to Sanders and Edge's. After accounting for relatedness among points within transects, shrub cover as the sole predictor variable only explained a substantial amount of variation in number of Song Sparrow detections. Marginal  $R^2_{GLMM}$  values associated with our shrub cover fixed effect for Willow Flycatcher, Yellow Warbler, and total bird detections were substantially lower (Table 2). Transforming count data for obligates and total bird detections to fit a normal distribution for analysis with linear mixed models and volume following Sanders and Edge (1998) increased  $R^2$  values for each species, but we remain skeptical of the predictive accuracy of this statistical technique because of the sample size, variance, and residual issues. We report these values in Table 2 alongside Sanders and Edge's and those derived from our GLMMs to illustrate differences between the two statistical techniques and the disparate inferences they may lead to.

Our model evaluation process allowed us to identify multiple additional covariates explaining variation in detection numbers. For Willow Flycatchers, Yellow Warblers, and total detections, our new models more than doubled marginal  $R^2$  values beyond those from our models with shrub cover area as the lone explanatory variable (Table 2). For Song Sparrows, no other covariate significantly improved upon the shrub cover area model. Not surprisingly, riparian zone and shrub zone widths were important predictor variables positively contributing to detection numbers. Percent forb and litter cover seemed to be negatively correlated with avian detections while percent grass and sedge cover was positively correlated with detections in the herbaceous vegetation association. Heights of these cover types were ruled out as covariates during the model selection process as was shrub cover by 1-m height intervals.

Oregon trend estimates of declines for our three focal species from the BBS closely reflected percent yearly declines we observed between the 1993-1994 and 2014 study periods (Table 3; Sauer et al. (2014)). Our estimates for both Song Sparrow and Willow Flycatcher declines were within the confidence intervals generated by the BBS for the state of Oregon. Yellow Warblers declined slightly more in our study area than was predicted by the BBS for the state of Oregon. Estimates generated for the Great Basin and Northern Rockies bioregions were quite different from our findings and even predicted fairly large annual increases for some species.

## **DISCUSSION**

We framed our study around a null hypothesis that changes in our study area's riparian bird community between the 1993-1994 and 2014 study periods would reflect changes in riparian shrub volume, but we found no connection between the substantial changes we observed for riparian shrub-dependent bird detections and riparian shrub cover, which changed little. All three of our riparian shrub-dependent focal species declined while riparian shrub cover remained relatively unchanged. The only marked decline in woody vegetation cover between the 1993-1994 and 2014 study periods occurred at the 3-4-m height interval, but shrub cover by height was not an explanatory variable identified during the model selection process for any of our focal species. Indeed, by at least two metrics (shrub zone and riparian zone width), the discontinuous and continuous mesic shrub associations we sampled in 2014 could be considered more suitable for riparian shrub-dependent species than those associations sampled during the 1993-1994 study period, but this did not translate into increased detections.

Our results provide evidence for both our first and second alternative hypotheses: shrub cover is still one of the leading predictor variables for riparian shrub-dependent and overall bird detections, but neither shrub cover nor any other covariate we measured satisfactorily explains

the Song Sparrow, Yellow Warbler, and Willow Flycatcher declines we observed. Several hypotheses have been put forth by other authors to explain declines of neotropical migrant insectivores like our focal species. Destruction of wintering and migratory ground habitat has been suggested as a cause of declines for many species, including the Willow flycatcher, but their wintering needs are apparently highly plastic, so this may be unlikely (Paxton et al. 2011). Parasitism by Brown-headed Cowbirds (*Molothrus ater*) is a well-documented cause of nest failure for all three species (Burgham and Picman 1989, Whitfield et al. 1997, Smith et al. 2002), but we recorded four fewer Cowbird detections in 2014 than the yearly 1993-1994 average, and Cowbirds have been a part of Eastern Oregon rangeland ecosystems since at least the early 1900s (Schweitzer et al. 1996). Without nest success data we cannot further evaluate this potential mechanism.

Decreased breeding ground fidelity has been noted in Willow Flycatchers as a response to reproductive failure following periods of severe drought and extreme temperatures (Sedgwick and Grubb Jr 2004, Paxton et al. 2007). We hypothesize that the prevalence of droughts and increasing temperatures in our study system may be having a similar, cumulative effect over time, leading to decreased nest success and site fidelity and resulting in the low number of detections we recorded in 2014. Pesticide use has been identified as a possible driver behind the decline of aerial insectivores in North America (Nebel et al. 2010), but historical changes in pesticide use in our study area and our species' wintering grounds were beyond the scope of this study. Furthermore, while Yellow Warblers and Willow Flycatchers declined in our study area, neither Cliff Swallows (*Petrochelidon pyrrhonota*) nor Northern Rough-winged Swallows (*Stelgidopteryx serripennis*) – both aerial insectivores with overlapping breeding and wintering grounds – exhibited similar declines.

Contrary to our third alternative hypothesis, riparian-shrub-dependent species declined most in “high quality” continuous habitat, not in the “lower quality” discontinuous vegetation association. Because we resampled the same discontinuous transects originally sampled by Sanders and Edge but established new continuous transects for the 2014 study period, this could mean there was a difference we were unable to detect between the original and newly established continuous sites. This difference may have been important for riparian shrub-obligate birds even though shrub cover was similar and riparian and shrub zone widths increased. If this is the case, identifying this difference between Sanders and Edge’s continuous sites and our own could reveal the cause of the declines in riparian shrub-dependent birds we observed. However, vegetation and site characteristics followed the same patterns in our replicated (discontinuous) sites as those observed in our non-replicated (continuous) sites. It is perhaps more plausible that the discontinuous vegetation association is not truly “lower quality” or less-preferred by riparian shrub-dependent birds, but that it may instead simply have a lower carrying capacity as a result of minimum territory needs of our focal species and greater interspersions between foraging and nesting substrates, i.e., riparian shrubs.

The close similarity between trends we observed for our three focal species and trend estimates derived from Oregon BBS results provide some evidence supporting our alternative hypothesis that changes we observed in number of detections for Song Sparrows, Willow Flycatchers, and Yellow Warblers may be driven, at least in part, by overarching regional shifts and not local site conditions. If true, then the declines we observed for our focal species may not actually be solely reflective of changes in our study area’s habitat suitability or carrying capacity. However, for each of our focal species, BBS results for Oregon State include multiple subspecies



living in a wide range of conditions on either side of the Cascades Mountains, so its predictive value is somewhat questionable.

The increase in overall detections we observed was driven by increases in herbaceous and discontinuous mesic shrub associations relative to those from the 1993-1994 study period, which together outweighed the decline we observed in the continuous mesic shrub association. The species with the greatest increases in detections suggest our newly established herbaceous transects were wetter, and perhaps more productive, habitat than those sampled by Sanders and Edge. Of the 11 species we identified with  $\geq 100\%$  increases in detections over the 1993-1994 average, one was a grassland obligate (Bobolink, *Dolichonyx oryzivorus*), two were associated with human development (European Starling, *Sturnus vulgaris*; Cliff Swallow), and eight were wetland-dependent species (e.g., Yellow-headed Blackbird, *Xanthocephalus xanthocephalus*; Wilson's Snipe, *Gallinago delicata*; Black Tern, *Chlidonias niger*) (Appendix 1).

The conclusion that we sampled wetter vegetation associations in 2014 and that this was responsible for the increased bird detections we observed is further supported by differences in habitat characteristics between the 1993-1994 and 2014 study periods. Covariates positively correlated with detections in the herbaceous vegetation association (riparian zone width, grass and sedge cover) were substantially higher in 2014 than in the 1993-1994 study period, while forb cover—negatively correlated with detections—was lower. We observed similar patterns in our newly established continuous mesic shrub association as well as in Sanders and Edge's original discontinuous transects. Increased detections here were driven by many of the same species they observed, increases in wetland-dependent species, and large increases in our two most common species, the Red-winged Blackbird and Savannah Sparrow (*Passerculus sandwichensis*). This suggests a system-wide improvement in habitat suitability for many

grassland and wetland bird species has occurred over the past 20 years, and our increased detections likely were not simply a result of sampling new transects or observer differences.

Diversity in terms of species richness ( $q = 0$ ) did not change substantially between study periods, but we observed increases in diversity orders  $q = 1$  and  $2$  in the continuous vegetation community from 1993-1994 to 2014, and these increases can be attributed to increased evenness. This increased evenness was a result of declines in our riparian shrub-dependent focal species, which were some of the most frequently detected birds in 1993-1994. Increased evenness is generally perceived as a hallmark of biodiverse communities and may therefore be viewed as a positive goal for management activities, but we caution against this interpretation when greater evenness may in fact be a result of declines in specialist or habitat-dependent species like Song Sparrows, Yellow Warblers, and Willow Flycatchers in our study system.

Our finding of increased compositional similarity among 2014 vegetation associations relative to 1993-1994 is also cause for concern. We have shown a significant decrease in heterogeneity of bird communities in structurally distinct habitats over the past 20 years even though structure has itself remained relatively stable and not converged to a more homogenous endpoint. Several authors have predicted increased homogeneity among plant and animal communities across landscapes—and thus lower beta and gamma diversity—could be a result of modern land management practices and climate change, especially in rangeland and riparian systems (Fuhlendorf and Engle 2001, Catford et al. 2013, Kominoski et al. 2013). In most of these scenarios, increased homogenization of wildlife communities may be expected to occur as a result of or in tandem with homogenizing habitats, but our work suggests, for bird communities at least, homogenization may begin even before physical habitats have been substantially altered.

Our results have unveiled a precipitous decline in riparian shrub-dependent birds that was not linked to changes in mesic shrub structure/cover, but our scope of inference is limited to two valleys in the Intermountain West whose riparian communities were surveyed just twice in 20 years, and inferences are further hampered by our ability to resample only one of Sanders and Edge's original vegetation associations. Although we identified several new covariates useful in predicting detections for our focal species, our lack of success identifying local factors causing their declines, coupled with negative BBS trends, could be indicative of a larger-scale issue. Regional and state-wide breeding bird surveys coupled with strong citizen science data in the Intermountain West, e.g. Oregon 2020 (2015), could help determine the current state of riparian shrub-dependent bird populations throughout the region and identify other areas of conservation concern. Without historical data, however, it will be difficult to determine whether the declines we observed have occurred elsewhere. Our results suggest new methods of analyzing BBS trend data in conjunction with habitat change estimated from aerial imagery may not identify causes for riparian-dependent species declines, but combining spatially-explicit modeling of population trends with historical climatic data could provide some insight (Jiménez-Valverde et al. 2011, Bled et al. 2013, Mattsson et al. 2013).

We recommend biologists and land managers in the region draw upon the wealth of knowledge gathered on riparian bird declines in the American Southwest, particularly for the Southwestern Willow Flycatcher (*E. t. extimus*), to inform long-term and expanded monitoring in the region. Concerns raised about small population and metapopulation sizes of *E. t. extimus* may now be applicable to the Willow Flycatcher subspecies in our study area, *E. t. adastus* (Finch and Stoleson 2000). Studies focused on demographics, reproductive success, and population

viability could deliver the key to ensuring the long-term persistence of riparian shrub-dependent birds in our study system and the wider region (Finch and Stoleson 2000).

## **MANAGEMENT IMPLICATIONS**

Though continuous mesic shrub associations in our study system still support the greatest diversity and number of birds, including mesic shrub-dependent species, the declines we have documented provide reason to believe managing riparian land to preserve shrub cover/volume may no longer be enough to ensure the persistence of diverse bird communities and healthy populations of riparian shrub-dependent breeding birds in some places. While it is likely management practices increasing total woody-mesic shrub cover and volume, such as seasonal, short-duration and low-intensity grazing, will continue to prove greatly beneficial for riparian shrub-dependent bird species, land managers, conservationists, and riparian restoration practitioners in the semi-arid Intermountain West should take caution against promoting mesic shrub conservation and regeneration as a panacea for riparian bird declines. Our modeling identified shrub zone width, riparian zone width, stream width, and low forb cover as other habitat characteristics important for riparian shrub-dependent birds, but even managing habitats for these features may not be enough to ensure healthy populations of Willow Flycatchers and Yellow Warblers.

Given the still enigmatic nature of the declines we observed, however, protection and conservation of large, continuous tracts of woody riparian vegetation may be becoming even more critical for slowing, if not halting, the disappearance of riparian shrub-dependent birds in the region. Long-term monitoring, not only of populations and their habitats, but also resource pools and abiotic conditions can help elucidate causal factors behind trends in avian populations

and should be given greater emphasis by land and project managers working to conserve riparian communities. Our study has also provided evidence that managing working lands for livestock production can be compatible with the conservation of many wetland and grassland species. Over the past 20 years, detections of these members of the riparian bird community have significantly increased, especially in places with increased grass and sedge cover, expanded riparian zone widths, and decreased forb cover.

As threats from climate change and anthropogenic modifications of avian wintering, migratory, and breeding ground habitats continue to build, historical datasets like the one this study is based on will continue to increase in value. However, there is only so much that can be done without a study's original data. We therefore wish to echo the calls of many others to preserve data for future researchers and to add to datasets already on hand with continued monitoring and replicated studies (e.g., Igl and Johnson (2005), Tingley and Beissinger (2009)).

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**Table 1.** Shrub and physical feature attributes of riparian habitats in Bear and Silvies valleys, east-central Oregon, measured in 1993 and 2014. Area of riparian shrubs was estimated in a circle with 40 m radius centered around each point; all other structure variables were measured in a circle adjacent to each point with 10 m radius following Sanders and Edge (1998). Test statistics are only for analyses among 2014 vegetation associations; underlined means are significantly different at  $\alpha = 0.05$ . See text for more details.

Habitat attribute	Herbaceous shrub				Discontinuous Shrub				Continuous Shrub				2014 Test statistics		
	1993		2014		1993		2014		1993		2014				
	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	<i>F</i>	df	<i>p</i>
Shrub structure															
Cover (%)	0.9	0.3	<0.1	<0.1	10.1	1.9	<u>13.9</u>	3.2	30.0	4.9	<u>36.8</u>	4.0	22.36	1, 56	0.011
0-1 m	0.9	0.3	<0.1	<0.1	4.3	1.1	<u>10.0</u>	2.1	19.9	3.6	<u>32.5</u>	4.1	31.27	1, 56	0.008
1-2 m	0.0	0.0	0.0	0.0	6.6	1.4	<u>10.9</u>	3.0	26.8	4.5	<u>31.9</u>	4.0	19.89	1, 56	0.012
2-3 m	0.0	0.0	0.0	0.0	7.7	2.1	<u>8.3</u>	2.2	24.9	4.2	<u>20.5</u>	2.9	13.67	1, 56	0.025
3-4 m	0.0	0.0	0.0	0.0	3.9	1.5	4.9	1.9	13.7	2.4	6.0	2.3	1.33	1, 56	0.337
4-5 m	0.0	0.0	0.0	0.0	1.3	0.9	2.2	1.1	2.0	0.3	1.7	0.9	<0.01	1, 56	0.991
5-6 m	0.0	0.0	0.0	0.0	0.3	0.2	0.3	0.2	0.0	0.0	<0.1	<0.1			
Volume (m <sup>3</sup> )	---	---	<0.1	<0.1	---	---	<u>113.1</u>	27.1	---	---	<u>288.0</u>	31.6	22.57	1, 56	0.011
Area – aerial (m <sup>2</sup> )	---	---	0.0	0.0	---	---	<u>227.1</u>	64.2	---	---	<u>790.3</u>	72.1	32.97	1, 56	0.008
Ground cover															
Grass + Sedge (%)	44.5	---	60.0	3.6	40.9	---	60.7	3.9	38.5	---	48.5	4.4	2.86	2, 84	0.149
Forb (%)	25.6	2.7	11.3	4.3	32.6	2.2	6.7	1.0	22.5	3.1	3.4	0.6	4.44	2, 84	0.075
Litter (%)	7.0	1.6	15.6	4.4	5.8	1.6	17.2	4.4	10.4	1.9	9.0	2.3	1.11	2, 84	0.395
Physical features															
Stream width (m)	6.3	0.4	7.1	0.6	6.7	2.4	5.4	0.7	5.1	0.4	14.1	1.2	3.91	2, 84	0.090
Stream depth (cm)	---	---	57.9	5.6	---	---	48.0	6.6	---	---	101.3	10.0	4.83	2, 84	0.071
Riparian width (m)	483.4	43.8	939.3	51.2	494.6	22.6	692.1	41.6	583.1	96.9	1227.8	29.2	5.60	2, 84	0.056
Shrub zone (m)	0.0	0.0	0.0	0.0	43.2	7.5	67.8	9.7	68.2	4.9	109.7	8.5	2.09	1, 56	0.248

**Table 2.** Model selection results and correlation coefficients for riparian shrub-dependent species and total detections from both 1993-1994 and 2014 study periods in Seneca, Oregon. All models include an intercept term; models with  $AIC_c$  values include a transect random effect to account for dependence among sites. Volume models were created following Sanders and Edge's statistical technique and were not used for model selection purposes.

Model Structure <sup>a</sup>	$K$	$AIC_c$	$\Delta AIC_c$	$R^2$	Marginal $R^2_{GLMM}$ <sup>b</sup>	Conditional $R^2_{GLMM}$ <sup>b</sup>
<i>Song Sparrow</i>						
1993-1994 volume <sup>c</sup>	2	-	-	0.858	-	-
2014 volume <sup>c</sup>	2	-	-	0.873	-	-
Shrub cover	3	235.7	0.0	-	0.420	0.420
Shrub cover - forb	4	237.5	1.8	-	0.431	0.431
Intercept + transect	2	261.9	26.2	-	-	0.289
<i>Willow Flycatcher</i>						
1993-1994 volume <sup>c</sup>	2	-	-	0.817	-	-
2014 volume <sup>c</sup>	2	-	-	0.500	-	-
Shrub cover	3	60.0	2.9	-	0.105	0.106
Riparian zone + shrub zone	4	57.1	0.0	-	0.394	0.394
Riparian zone	3	57.3	0.2	-	0.257	0.257
Intercept + transect	2	59.7	2.6	-	-	<0.001
<i>Yellow Warbler</i>						
1993-1994 volume <sup>c</sup>	2	-	-	0.721	-	-
2014 volume <sup>c</sup>	2	-	-	0.251	-	-
Shrub cover	3	192.6	12.4	-	0.130	0.376
Shrub cover + shrub zone	4	180.2	0.0	-	0.345	0.478
Shrub cover + shrub zone + stream	5	182.0	1.8	-	0.346	0.490
Intercept + transect	2	198.3	18.1	-	-	0.296
<i>Total<sub>all</sub></i> <sup>d</sup>						
1993-1994 volume <sup>c</sup>	2	-	-	0.830	-	-
2014 volume <sup>c</sup>	2	-	-	0.219	-	-
<i>Total<sub>shrub</sub></i> <sup>e</sup>						
Shrub cover	3	452.9	25.7	-	0.196	0.441
Shrub + shrub zone - forb - litter	6	427.2	0.0	-	0.399	0.624
Shrub cover - forb - litter	5	428.5	1.3	-	0.403	0.598
Intercept + transect	2	469.5	42.3	-	-	0.372
<i>Total<sub>herb</sub></i> <sup>f</sup>						
Riparian zone + g&s	4	228.5	0.0	-	0.506	0.551
Riparian zone + g&s - forb	5	229.1	0.6	-	0.546	0.602
Intercept + transect	2	250.4	21.9	-	-	0.313

<sup>a</sup> Covariates: 1993-1994 volume, average shrub volume per transect in 1993-1994; 2014 volume, average shrub volume per transect in 2014; shrub cover, site shrub cover area; riparian zone, site riparian zone width; shrub zone, site shrub zone width; stream, site stream width; forb, site percent forb cover; litter, site percent litter cover; g&s, site percent grass and sedge cover.

<sup>b</sup> Marginal and Conditional  $R^2_{GLMM}$  are correlation coefficients explaining fixed and both fixed and random factors respectively.

<sup>c</sup> Volume models are based on simple linear regression between shrub volume and avian detections summed to transect level. All other models use generalized linear mixed effects with bird detections summed by sites.

<sup>d</sup> Models for *Total<sub>all</sub>* include all species and all three vegetation associations regardless of shrub occurrence.

<sup>e</sup> Models for total bird detections among discount. and cont. mesic shrub associations where shrubs are present.

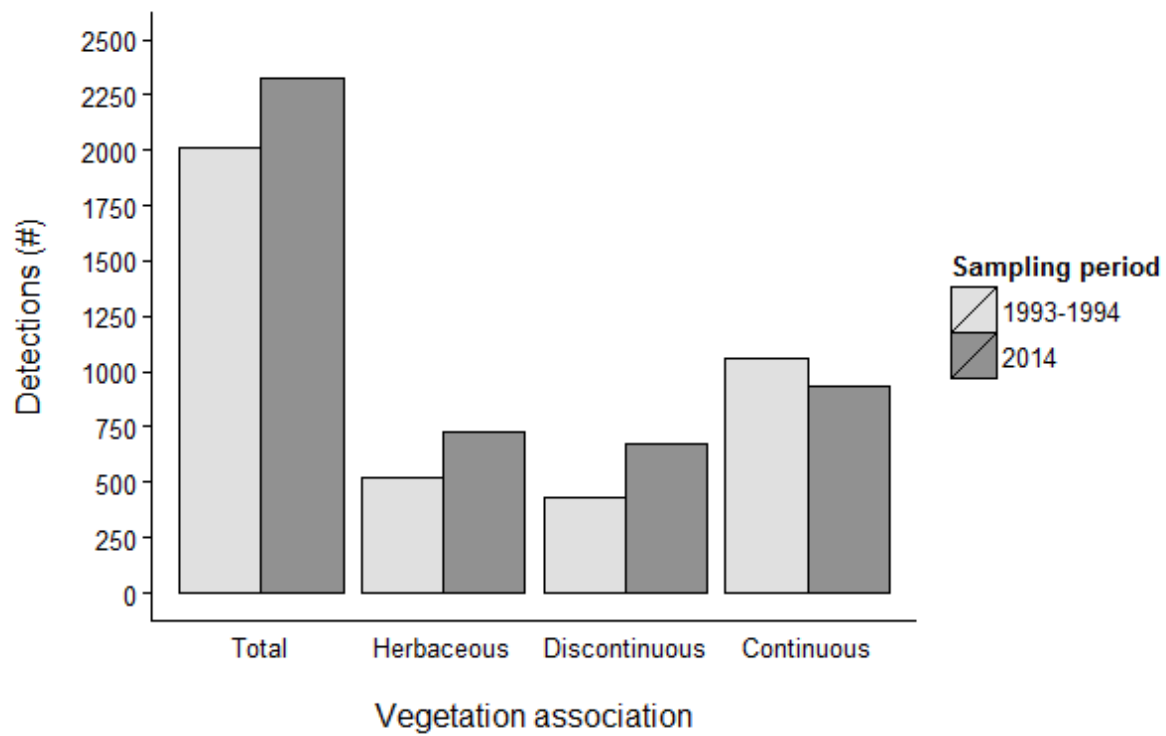
<sup>f</sup> Models for total bird detections within the herbaceous vegetation where shrubs are absent.

**Table 3.** Average yearly Breeding Bird Survey population trends<sup>a</sup> for Song Sparrow<sup>b</sup>, Willow Flycatcher<sup>b</sup>, and Yellow Warbler<sup>b</sup> from 1993-2013 and estimated percent change per year in Seneca, Oregon from 1993-1994 to 2014.

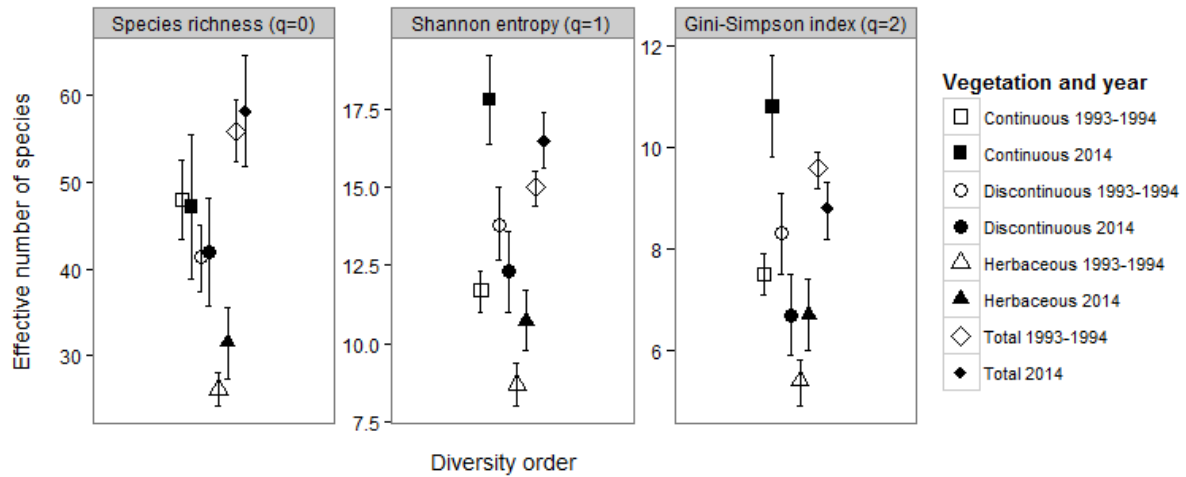
<b>Species (% change/yr in Seneca, OR)</b>	<b>BBS Estimate</b>	<b>95% CI</b>	<b>Region</b>
Song Sparrow	-1.14	-1.72, -0.47	Oregon
(-0.55)	1.45	0.65, 2.26	Great Basin
	0.03	-0.55, 0.61	N. Rockies
Willow Flycatcher	-4.60	-5.96, -3.11	Oregon
(-4.55)	-1.27	-2.10, -0.33	Great Basin
	-0.02	-0.96, 1.02	N. Rockies
Yellow Warbler	-2.43	-3.33, -1.42	Oregon
(-3.58)	0.75	-0.06, 1.57	Great Basin
	-1.00	-1.52, -0.48	N. Rockies

<sup>a</sup> BBS trend data from Sauer et al. (2014).

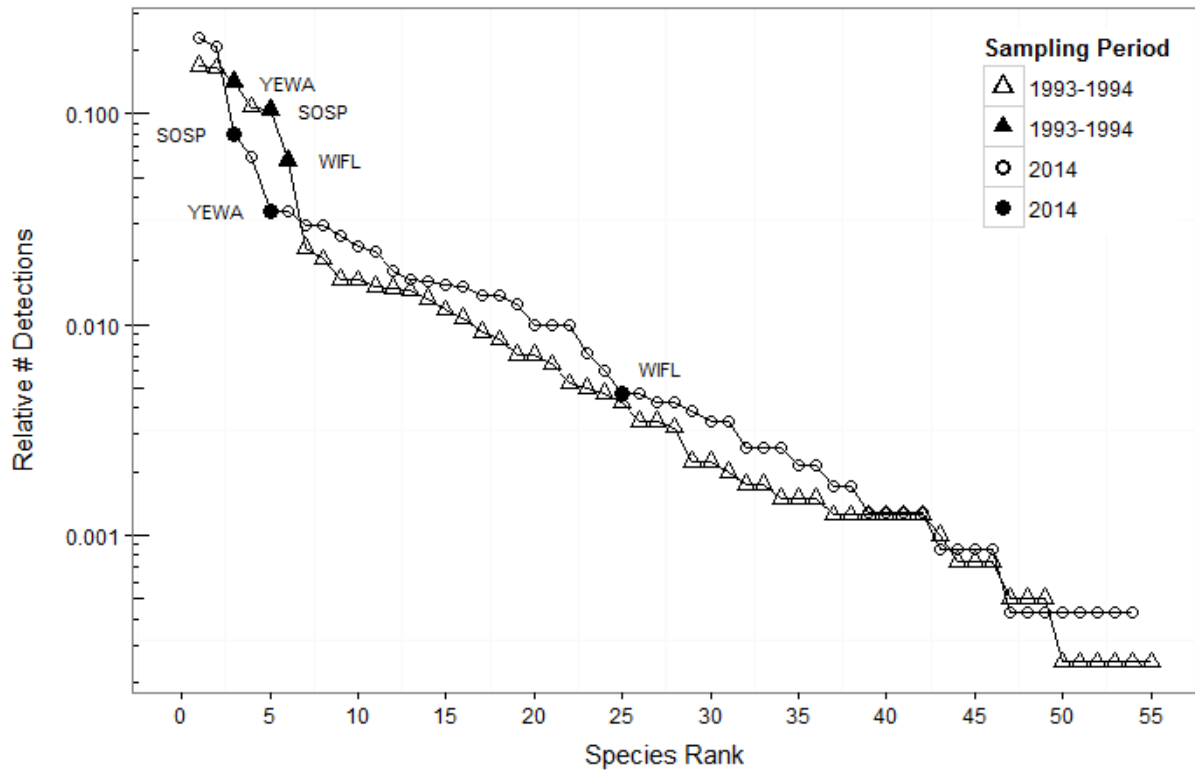
<sup>b</sup> Riparian-shrub dependent focal species identified by Sanders and Edge (1998).



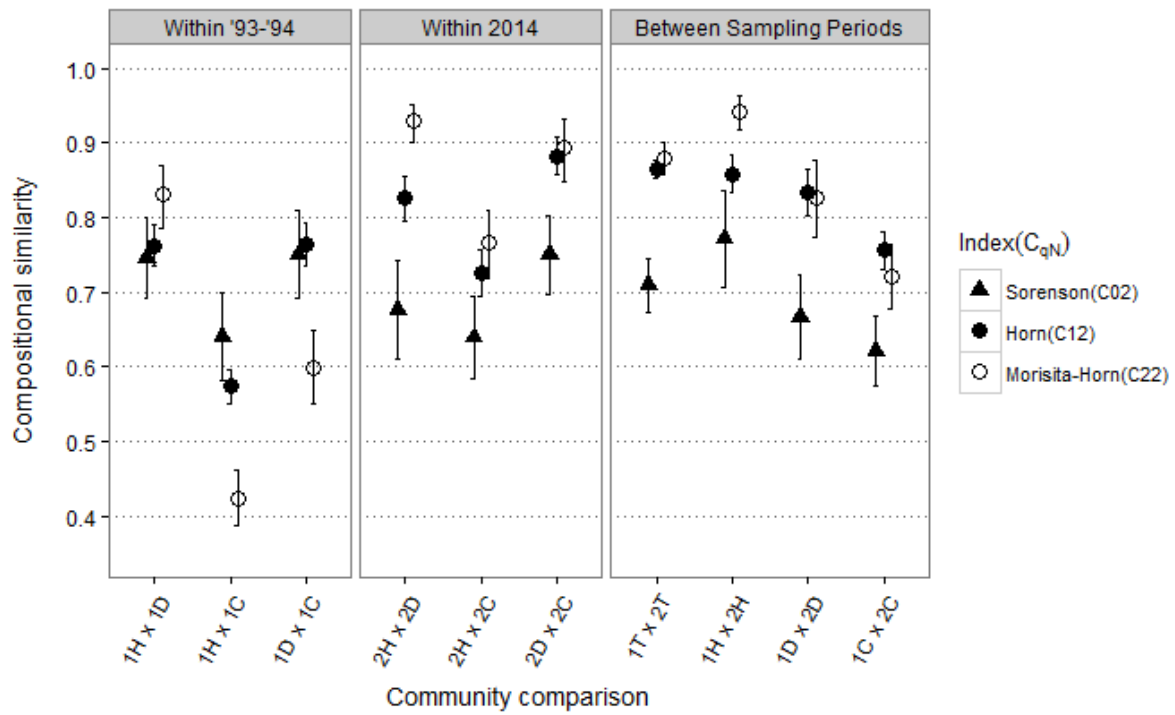
**Figure 1.** Detections of birds observed overall and within each riparian vegetation association sampled during 1993-1994 and 2014 in Bear and Silvies valleys, Seneca, Oregon. Values we present from the 1993-1994 study period were halved for comparison with the 2014 study period.



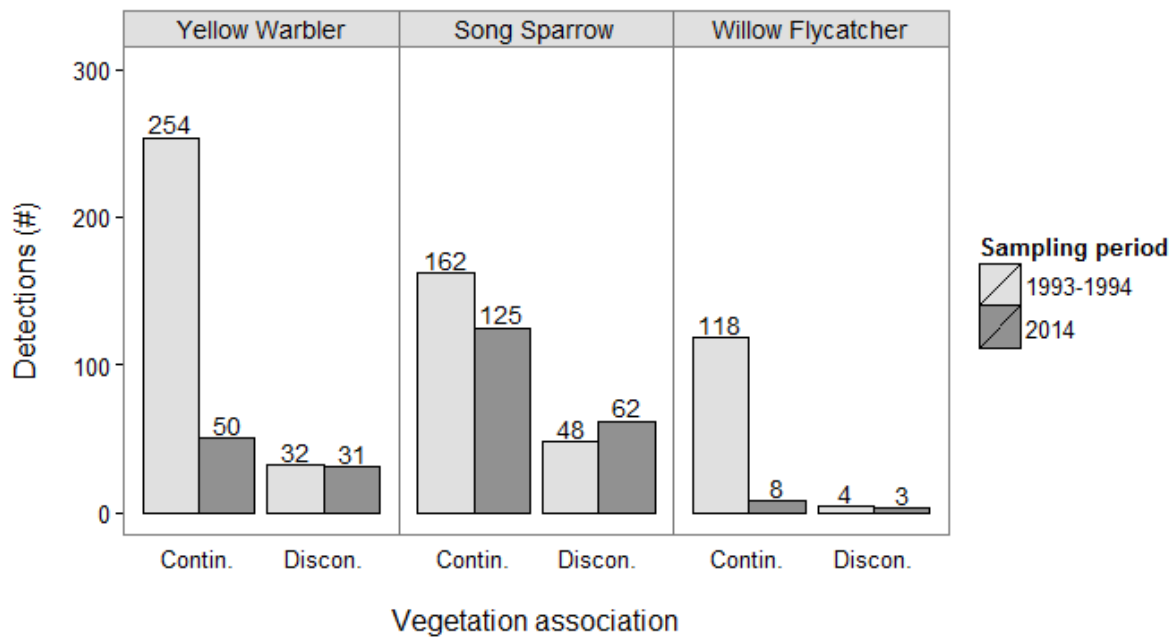
**Figure 2.** Coverage-based diversity estimates of orders  $q = 0, 1$ , &  $2$  with 95% CIs for riparian bird communities sampled in 1993-1994 and 2014 in Bear and Silvies valleys, Seneca, Oregon. Sample data from the 2014 study period were interpolated or extrapolated to sample coverage of 99.5% for vegetation associations and 99.8% for the total bird community for each study period.



**Figure 3.** Rank-detections distributions for riparian birds sampled in 1993-1994 and 2014 in Bear and Silvies valleys, Seneca, Oregon. Ranked relative number of detections,  $\hat{p}_i$ , are reported on the y-axis on a logarithmic scale. Changes in relative abundances of focal species Yellow Warbler (YEWA), Song Sparrow (SOSP), and Willow Flycatcher (WIFL) are highlighted with shaded points.



**Figure 4.** Pairwise comparisons of compositional similarity ( $C_{qN}$  indices with 95% CIs) between avian communities sampled during 1993-1994 and 2014 in Bear and Silvies valleys, Seneca, Oregon. On the x-axis, we use the number 1 to represent a bird community from the 1993-1994 study period and 2 to indicate a community sampled in 2014. We use H to denote bird communities from herbaceous vegetation associations, D for discontinuous mesic shrub associations, C for continuous, and T for total communities from a given study period, i.e. communities from all vegetation communities combined.



**Figure 5.** Number of Yellow Warbler, Song Sparrow, and Willow Flycatcher detections in continuous (Contin.) and discontinuous (Discon.) riparian vegetation associations in Bear and Silvies valleys, Seneca, Oregon during the 1993-1994 and 2014 study periods. Values we present from the 1993-1994 study period were halved for comparison with the 2014 study period.



**Appendix 1.** Bird detections in herbaceous, discontinuous mesic shrub, and continuous mesic shrub vegetation associations in riparian zones of Bear and Silvies valleys, Seneca Oregon, during the 1993-1994 and 2014 study periods. Data from the 1993-1994 study period are adapted from Appendix A of Sanders (1995).

Species (Scientific name)	Detections				Herbaceous				Discontinuous				Continuous			
	'93-'94 <sup>a</sup>		2014		'93-'94		2014		'93-'94		2014		'93-'94		2014	
	$N_t^b$	$N_r^c$	$N_t$	$N_r$	$\bar{x}^d$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
American Bittern ( <i>Botaurus lentiginosus</i> )	6	1	13	0	-	-	-	-	-	-	-	-	0.1	0.1	-	-
American Coot ( <i>Fulica americana</i> )	0	0	3	1	-	-	-	-	-	-	-	-	-	-	0.3	0.3
American Crow ( <i>Corvus brachyrhynchos</i> )	104	19	129	14	-	-	0.3	0.3	2.3	1.2	0.3	0.3	0.4	0.3	3.0	1.3
American Robin ( <i>Turdus migratorius</i> )	140	82	19	6	-	-	-	-	0.3	0.3	1.5	1.0	10.0	2.0	-	-
American Wigeon ( <i>Anas americana</i> )	29	21	24	9	1.9	0.5	0.3	0.3	0.6	0.4	1.0	0.7	0.3	0.2	1.0	1.0
Barn Swallow ( <i>Hirundo rustica</i> )	34	29	21	10	1.4	0.6	0.5	0.3	1.7	0.9	1.3	0.5	0.8	0.3	0.8	0.3
Belted Kingfisher ( <i>Megaceryle alcyon</i> )	12	6	2	0	0.4	0.3	-	-	0.1	0.1	-	-	0.3	0.2	-	-
Black Tern ( <i>Chlidonias niger</i> )	44	29	80	37	3.0	0.6	2.0	1.4	0.7	0.4	0.8	0.5	-	-	6.5	2.2
Black-billed Magpie ( <i>Pica hudsonia</i> )	29	5	45	11	-	-	-	-	-	-	0.3	0.3	0.6	0.3	2.5	1.5
Black-crowned Night-Heron ( <i>Nycticorax nycticorax</i> )	2	1	8	6	-	-	-	-	0.1	0.1	-	-	-	-	1.5	0.9
Black-headed Grosbeak ( <i>Pheucticus melanocephalus</i> )	0	0	4	3	-	-	-	-	-	-	-	-	-	-	0.8	0.5
Blue-winged Teal ( <i>Anas discors</i> )	0	0	8	4	-	-	1.0	1.0	-	-	-	-	-	-	-	-

Species (Scientific name)	Detections				Herbaceous				Discontinuous				Continuous			
	'93- '94 <sup>a</sup>		2014		'93- '94		2014		'93- '94		2014		'93- '94		2014	
	$N_t^b$	$N_r^c$	$N_t$	$N_r$	$\bar{x}^d$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Bobolink ( <i>Dolichonyx oryzivorus</i> )	126	34	289	38	-	-	4.5	3.5	-	-	1.8	0.8	4.3	2.4	3.3	1.8
Brewer's Blackbird ( <i>Euphagus cyanocephalus</i> )	676	432	285	145	20.4	5.6	15.0	7.9	21.7	9.1	8.8	5.5	14.6	6.6	12.5	4.4
Brewer's Sparrow ( <i>Spizella breweri</i> )	37	5	5	0	-	-	-	-	0.6	0.6	-	-	0.1	0.1	-	-
Brown-headed Cowbird ( <i>Molothrus ater</i> )	150	93	94	42	-	-	-	-	7.3	2.8	2.3	0.9	5.3	2.0	8.3	1.4
Canada Goose ( <i>Branta canadensis</i> )	60	5	77	0	0.3	0.3	-	-	0.3	0.3	-	-	0.1	0.1	-	-
Cassin's Finch ( <i>Haemorhous cassinii</i> )	0	0	1	1	-	-	-	-	-	-	0.3	0.3	-	-	-	-
Cedar Waxwing ( <i>Bombycilla cedrorum</i> )	0	0	2	2	-	-	-	-	-	-	0.5	0.5	-	-	-	-
Cinnamon Teal ( <i>Anas cyanoptera</i> )	88	61	54	32	4.8	0.9	1.0	0.7	0.7	0.3	4.0	2.4	2.3	0.6	3.0	1.7
Cliff Swallow ( <i>Petrochelidon pyrrhonota</i> )	64	53	172	80	1.8	0.9	14.5	6.7	2.3	1.5	3.5	1.8	2.9	1.2	2.0	1.7
Common Merganser ( <i>Mergus merganser</i> )	5	5	1	0	-	-	-	-	-	-	-	-	0.6	0.4	-	-
Common Raven ( <i>Corvus corax</i> )	8	0	26	2	-	-	0.3	0.3	-	-	0.3	0.3	-	-	-	-
Common Yellowthroat ( <i>Geothlypis trichas</i> )	0	0	39	5	-	-	-	-	-	-	0.5	0.5	-	-	0.8	0.8
Dark-eyed Junco ( <i>Junco hyemalis</i> )	1	1	0	0	-	-	-	-	-	-	-	-	0.1	0.1	-	-
European Starling ( <i>Sturnus vulgaris</i> )	56	14	309	52	0.1	0.1	11.3	9.0	1.7	1.7	-	-	0.1	0.1	1.8	1.0

Species (Scientific name)	Detections				Herbaceous				Discontinuous				Continuous			
	'93-'94 <sup>a</sup>		2014		'93-'94		2014		'93-'94		2014		'93-'94		2014	
	$N_t^b$	$N_r^c$	$N_t$	$N_r$	$\bar{x}^d$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Forster's Tern ( <i>Sterna forsteri</i> )	0	0	65	29	-	-	0.3	0.3	-	-	4.3	2.8	-	-	2.8	0.9
Gadwall ( <i>Anas strepera</i> )	17	9	88	32	0.3	0.2	2.0	1.7	1.0	0.7	1.0	0.4	-	-	5.0	2.1
Great Blue Heron ( <i>Ardea herodias</i> )	3	2	7	1	-	-	-	-	0.1	0.1	-	-	0.1	0.1	0.3	0.3
Green-winged Teal ( <i>Anas crecca</i> )	22	20	32	17	0.9	0.3	1.0	0.6	0.7	0.4	-	-	1.0	0.5	3.3	1.7
Hooded Merganser ( <i>Lophodytes cucullatus</i> )	17	17	10	3	2.1	1.4	0.5	0.5	-	-	-	-	-	-	0.3	0.3
House Wren ( <i>Troglodytes aedon</i> )	0	0	3	3	-	-	-	-	-	-	0.3	0.3	-	-	0.5	0.3
Killdeer ( <i>Charadrius vociferus</i> )	21	9	43	2	0.4	0.4	0.5	0.3	0.7	0.4	-	-	0.1	0.1	-	-
Long-billed Curlew ( <i>Numenius americanus</i> )	83	13	25	3	0.8	0.4	0.5	0.5	0.9	0.7	0.3	0.3	0.1	0.1	-	-
Long-eared Owl ( <i>Asio otus</i> )	6	6	2	0	-	-	-	-	0.9	0.9	-	-	-	-	-	-
MacGillivray's Warbler ( <i>Geothlypis tolmiei</i> )	0	0	2	1	-	-	-	-	-	-	-	-	-	-	0.3	0.3
Mallard ( <i>Anas platyrhynchos</i> )	143	58	87	35	3.6	1.4	3.0	1.8	0.9	0.5	1.3	1.3	2.9	1.1	4.5	1.7
Marsh Wren ( <i>Cistothorus palustris</i> )	0	0	42	23	-	-	-	-	-	-	-	-	-	-	5.8	1.8
Mourning Dove ( <i>Zenaida macroura</i> )	3	2	5	0	-	-	-	-	-	-	-	-	0.3	0.3	-	-
Nashville Warbler ( <i>Oreothlypis ruficapilla</i> )	0	0	2	1	-	-	-	-	-	-	-	-	-	-	0.3	0.3

Species (Scientific name)	Detections				Herbaceous				Discontinuous				Continuous			
	'93-'94 <sup>a</sup>		2014		'93-'94		2014		'93-'94		2014		'93-'94		2014	
	$N_t^b$	$N_r^c$	$N_t$	$N_r$	$\bar{x}^d$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Northern Flicker ( <i>Colaptes auratus</i> )	5	3	1	0	-	-	-	-	-	-	-	-	0.4	0.2	-	-
Northern Goshawk ( <i>Accipiter gentilis</i> )	1	1	0	0	-	-	-	-	-	-	-	-	0.1	0.1	-	-
Northern Pintail ( <i>Anas acuta</i> )	38	14	0	0	0.8	0.3	-	-	0.6	0.4	-	-	0.5	0.3	-	-
Northern Rough-winged Swallow ( <i>Stelgidopteryx serripennis</i> )	45	43	32	23	0.3	0.2	2.5	1.0	1.7	1.1	2.5	1.5	3.6	3.1	0.8	0.8
Northern Shoveler ( <i>Anas clypeata</i> )	15	8	36	4	0.5	0.4	-	-	0.3	0.3	0.8	0.5	0.3	0.3	0.3	0.3
Osprey ( <i>Pandion haliaetus</i> )	1	0	1	1	-	-	0.3	0.3	-	-	-	-	-	-	-	-
Pied-billed Grebe ( <i>Podilymbus podiceps</i> )	0	0	16	5	-	-	-	-	-	-	0.5	0.3	-	-	0.8	0.8
Red-winged Blackbird ( <i>Agelaius phoeniceus</i> )	1532	676	1320	489	28.8	3.0	38.5	11.7	10.1	3.1	39.8	2.4	46.9	7.3	44.0	19.0
Ring-necked Duck ( <i>Aythya collaris</i> )	7	5	22	8	0.5	0.5	-	-	-	-	0.8	0.5	0.1	0.1	1.3	1.3
Sandhill Crane ( <i>Grus canadensis</i> )	78	4	84	1	-	-	0.3	0.3	-	-	-	-	0.5	0.4	-	-
Savannah Sparrow ( <i>Passerculus sandwichensis</i> )	1133	660	1103	530	42.3	4.1	51.0	13.4	29.3	2.1	46.0	8.5	14.6	2.9	35.5	5.7
Sharp-shinned Hawk ( <i>Accipiter striatus</i> )	0	0	1	1	-	-	-	-	-	-	0.3	0.3	-	-	-	-
Song Sparrow ( <i>Melospiza melodia</i> )	525	420	391	187	-	-	-	-	13.7	4.4	15.5	1.6	40.5	3.2	31.3	4.5
Sora ( <i>Porzana carolina</i> )	26	2	39	0	-	-	-	-	-	-	-	-	0.3	0.3	-	-

Species (Scientific name)	Detections				Herbaceous				Discontinuous				Continuous			
	'93- '94 <sup>a</sup>		2014		'93- '94		2014		'93- '94		2014		'93- '94		2014	
	$N_t^b$	$N_r^c$	$N_t$	$N_r$	$\bar{x}^d$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Spotted Sandpiper ( <i>Actitis macularius</i> )	9	7	92	55	-	-	6.8	3.7	0.6	0.4	5.0	3.5	0.4	0.4	2.0	1.1
Townsend's Solitaire ( <i>Myadestes townsendi</i> )	1	1	0	0	-	-	-	-	0.1	0.1	-	-	-	-	-	-
Tree Swallow ( <i>Tachycineta bicolor</i> )	46	37	14	10	0.5	0.3	2.3	1.3	2.3	1.2	0.3	0.3	2.1	0.5	-	-
Vaux's Swift ( <i>Chaetura vauxi</i> )	30	26	0	0	-	-	-	-	0.3	0.3	-	-	3.0	1.0	-	-
Vesper Sparrow ( <i>Pooecetes gramineus</i> )	14	5	4	0	-	-	-	-	0.7	0.5	-	-	-	-	-	-
Virginia Rail ( <i>Rallus limicola</i> )	5	3	15	6	-	-	-	-	-	-	-	-	0.4	0.3	1.5	0.9
Western Meadowlark ( <i>Sturnella neglecta</i> )	357	60	344	23	3.5	1.5	0.8	0.5	2.1	0.7	2.8	0.8	2.1	0.8	2.3	1.1
Western Tanager ( <i>Piranga ludoviciana</i> )	3	3	0	0	-	-	-	-	-	-	-	-	0.4	0.4	-	-
Willet ( <i>Tringa semipalmata</i> )	175	65	171	36	4.1	1.4	3.8	1.5	4.0	1.0	1.3	0.6	0.5	0.3	4.0	1.6
Willow Flycatcher ( <i>Empidonax traillii</i> )	381	242	19	11	-	-	-	-	1.0	0.6	0.8	0.5	29.4	3.1	2.0	0.4
Wilson's Phalarope ( <i>Phalaropus tricolor</i> )	115	66	142	69	5.0	1.8	11.0	5.8	0.3	0.3	3.5	2.0	3.0	1.6	2.8	1.3
Wilson's Snipe ( <i>Gallinago delicata</i> )	393	47	392	61	1.8	0.6	5.3	1.6	0.4	0.4	5.8	1.8	3.8	0.7	4.3	1.7
Wilson's Warbler ( <i>Cardellina pusilla</i> )	1	1	11	8	-	-	-	-	0.1	0.1	-	-	-	-	2.0	1.2
Wood Duck ( <i>Aix sponsa</i> )	9	7	3	2	-	-	-	-	-	-	0.5	0.5	0.9	0.9	-	-

Species (Scientific name)	Detections				Herbaceous				Discontinuous				Continuous			
	'93- '94 <sup>a</sup>		2014		'93- '94		2014		'93- '94		2014		'93- '94		2014	
	$N_t^b$	$N_r^c$	$N_t$	$N_r$	$\bar{x}^d$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Yellow Warbler ( <i>Setophaga petechia</i> )	732	570	213	81	-	-	-	-	9.0	0.7	7.8	2.8	63.4	4.9	12.5	3.7
Yellow-headed Blackbird ( <i>Xanthocephalus xanthocephalus</i> )	26	6	209	69	0.3	0.3	0.5	0.5	0.1	0.1	0.5	0.5	0.4	0.3	16.3	5.4
Yellow-rumped Warbler ( <i>Setophaga coronata</i> )	2	2	1	0	-	-	-	-	-	-	-	-	0.3	0.2	-	-

#### Beyond 40-m detection radius

American Kestrel ( <i>Falco sparverius</i> )	4	0	0	0	-	-	-	-	-	-	-	-	-	-	-	-
Bald Eagle ( <i>Haliaeetus leucocephalus</i> )	0	0	1	0	-	-	-	-	-	-	-	-	-	-	-	-
Caspian Tern ( <i>Hydroprogne caspia</i> )	0	0	1	0	-	-	-	-	-	-	-	-	-	-	-	-
Common Nighthawk ( <i>Chordeiles minor</i> )	2	0	0	0	-	-	-	-	-	-	-	-	-	-	-	-
Golden Eagle ( <i>Aquila chrysaetos</i> )	1	0	0	0	-	-	-	-	-	-	-	-	-	-	-	-
Great Egret ( <i>Ardea alba</i> )	0	0	1	0	-	-	-	-	-	-	-	-	-	-	-	-
Mountain Chickadee ( <i>Poecile gambeli</i> )	0	0	1	0	-	-	-	-	-	-	-	-	-	-	-	-
Orange-crowned Warbler ( <i>Oreothlypis celata</i> )	0	0	1	0	-	-	-	-	-	-	-	-	-	-	-	-
Red-tailed Hawk ( <i>Buteo jamaicensis</i> )	6	0	0	0	-	-	-	-	-	-	-	-	-	-	-	-

Species (Scientific name)	Detections				Herbaceous				Discontinuous				Continuous			
	'93-'94 <sup>a</sup>		2014		'93-'94		2014		'93-'94		2014		'93-'94		2014	
	$N_t^b$	$N_r^c$	$N_t$	$N_r$	$\bar{x}^d$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Ring-necked Pheasant ( <i>Phasianus colchicus</i> )	0	0	1	0	-	-	-	-	-	-	-	-	-	-	-	-
Swainson's Hawk ( <i>Buteo swainsoni</i> )	0	0	2	0	-	-	-	-	-	-	-	-	-	-	-	-
Turkey Vulture ( <i>Cathartes aura</i> )	1	0	0	0	-	-	-	-	-	-	-	-	-	-	-	-

#### Excluded from analyses<sup>e</sup>

Bank Swallow ( <i>Riparia riparia</i> )	0	0	150	86	-	-	21.0	14.5	-	-	0.5	0.3	-	-	-	-
Double-crested Cormorant ( <i>Phalacrocorax auritus</i> )	2	1	1	1	-	-	-	-	0.1	0.1	-	-	-	-	0.3	0.3
Prairie Falcon ( <i>Falco mexicanus</i> )	0	0	1	1	-	-	-	-	-	-	-	-	-	-	0.3	0.3

<sup>a</sup> Detections from both 1993 and 1994 sampling seasons combined; twice the sampling effort of the 2014 study period.

<sup>b</sup>  $N_t$  = Total number of individual detections at all distances.

<sup>c</sup>  $N_r$  = Number of detections where detection probabilities were assumed to be constant ( $\leq 40$  m from observer).

<sup>d</sup> Mean number of detections recorded during four visits/transect/year ( $n = 8, 7, 8$  and  $4, 4, 4$  for '93-'94 and 2014 vegetation associations respectively).

<sup>e</sup> Species excluded were unassociated with herbaceous and woody riparian vegetation (Double-crested Cormorant and Prairie Falcon flybys, Bank Swallow presence determined by proximity to nesting colonies).