

Initial experimental effects of intensive forest management on avian abundance

Matthew G. Betts*, Dept. of Forest Ecosystems and Society, Oregon State University, Corvallis, Oregon, 97331 (corresponding author: matthew.betts@oregonstate.edu)

Jake Verschuyt*, National Council for Air and Stream Improvement, Inc., P.O. Box 1259, Anacortes, WA 98221, USA

Jack Giovanini, Statistics, Mathematics, and Operations Research, Weyerhaeuser NR, Federal Way, Washington, United States of America

Thomas Stokely, Dept. of Forest Ecosystems and Society, Oregon State University, Corvallis, Oregon, 97331

Andrew J. Kroll, Weyerhaeuser NR, WTC 1A5, PO Box 9777, Federal Way, WA 98063, USA

*The contribution of these authors to the manuscript was equal

1 Abstract

2 Components of biodiversity in intensively managed forest stands may be reduced in
3 comparison to naturally regenerated stands. Use of herbicides to suppress herbaceous
4 and woody plant species that compete with planted seedlings has been implicated in
5 negative impacts. We designed a large-scale experimental study to test the influence of
6 intensive forest management on the abundance of early seral bird species in the Oregon
7 Coast Range, U.S.. Experimental applications consisted of ‘Intensive’ (i.e., heavy use of
8 herbicides), ‘Moderate’ and ‘Light’ treatments, as well as controls with no herbicide
9 application. In relation to the control, abundance of six out of thirteen bird species was
10 significantly reduced in at least one of the three treatments. Leaf-gleaning insectivorous
11 birds were more negatively affected by heavier herbicide treatments in general than bird
12 species with other foraging behavior. Long-term bird population trends, derived from the
13 Breeding Bird survey, were correlated with the effect of intensive treatment; species
14 more negatively associated with intensive treatments at the stand scale, were more likely
15 to be in decline across the Pacific Northwest, U.S.. Our results also indicate that reducing
16 intensity of herbicide applications has positive effects on early seral bird abundance
17 during the first two years of stand growth – particularly those exhibiting negative
18 population trends. To balance biodiversity conservation and timber production, research
19 examining the tradeoffs between reduced application of herbicide and tree growth is
20 required.

21 Keywords: biodiversity, early successional forest, herbicide, imperfect detection,
22 intensive forest management, plantation forestry, songbirds

1 Introduction

2 Structurally and compositionally diverse early seral forest is declining in some
3 locations worldwide (Angelstam, 1998; Najera and Simonetti, 2006; Thomas *et al.*, 2006)
4 and, in many instances, declining below the historic range of variability (Spies and
5 Johnson, 2007). This trend is of conservation concern because early successional stages
6 are generally associated with high species diversity and food web complexity (for review
7 see Swanson *et al.*, 2011). Further, many species seem to be linked to this forest
8 condition for critical parts of their life histories (Hagar, 2007). For example, previous
9 research has linked changes in the availability of early seral habitat with population
10 trends of vertebrate species (Litvaitis, 1993; Hunt, 1998; Betts *et al.*, 2010).

11 Decline in availability of complex early seral forest has been attributed to two
12 primary factors. First, fire suppression and reductions in timber harvest in many
13 developed countries have reduced the amount of early seral forest being created
14 (Kennedy and Spies, 2005; Kauppi *et al.*, 2006; Spies *et al.*, 2007). Second, stands
15 disturbed by both timber harvest and natural disturbance tend to be managed intensively
16 under an industrial model in order to produce wood fiber as rapidly as possible. Intensive
17 forest management (IFM) in the Pacific Northwest of the United States (Oliver and
18 Larson 1996), as in many parts of the globe (Najera and Simonetti, 2006), temporarily
19 inhibits development of herbaceous plants and early seral broadleaf shrubs, and reduces
20 competition with commercially valuable planted conifers (Adams *et al.*, 2005). Such
21 practices increase wood production, but may simplify the forest ecosystem both spatially
22 and temporally. Though species diversity of intensively managed plantations may be
23 similar to less intensively managed stands during some periods of their development

1 (Ellis and Betts, 2011), it has been argued that such plantation forestry truncates the
2 longevity of pre-canopy closure establishment period (Donato *et al.*, 2012).

3 Conservation and management programs require more information about bird
4 response to herbicide treatments in the Pacific Northwest. Available evidence on this
5 topic is generally circumstantial (Lautenschlager and Sullivan, 2004); it is therefore not
6 possible to infer causality from these findings because apparent relationships between
7 IFM and biodiversity could be related to confounding factors. Also, forest practice
8 policies in many areas, including the Pacific Northwestern U.S., prevents natural
9 regeneration following harvest, requiring landowners to replant quickly and provide a
10 ‘free to grow’ condition for planted seedlings. Such policies reduce the possible range of
11 early seral composition represented by sampling, which limits inference to stands within
12 a fairly narrow range of IFM that does not include unmanaged controls (Jones *et al.*,
13 2012). The few manipulative studies that exist tend to examine only one or two
14 treatments in relation to a control (Easton and Martin, 1998) and are poorly replicated
15 (Lautenschlager, 1993). As a result, statistical power may be low, thus weakening
16 inference about any potential management recommendations from a study. Ideally,
17 gradients in management intensity should be reflected in sampling designs, allowing for
18 the detection of potential *degrees* of IFM that might minimize trade-offs between timber
19 production and biodiversity (Iglay *et al.*, 2012).

20 Here, we report results of a two-year manipulative experiment designed to address
21 the question of how a gradient in IFM influences biodiversity in early seral stands of the
22 Oregon Coast Range. In this paper, we capitalized on a well-replicated randomized block
23 design, conducted with samples at the scale of entire forest stands, to test whether IFM

1 influences the abundance of passerine bird species. Birds are considered to be
2 biodiversity indicators (Schulze, 2004; Venier and Pearce, 2004; Gregory *et al.*, 2006)
3 and perform important ecosystem services (Sekercioglu *et al.*, 2004). Many of the bird
4 species examined in the present study have previously shown strong sensitivities to IFM
5 in correlative studies conducted at stand (Morrison and Meslow, 1983; Jones *et al.*, 2011;
6 Ellis *et al.*, 2012) and landscape scales (Betts *et al.*, 2010). These sensitivities are
7 hypothesized to be indirectly caused by declines in broadleaf shrubs. Compared to
8 conifers, the leaves of these hardwood species may support more abundant arthropods,
9 which are important prey food sources for insectivorous birds (Hammond and Miller,
10 1998; Hagar, 2007; Hagar *et al.*, 2012). Likewise, many Neotropical passerines nest in
11 dense broadleaf shrubs. Therefore, we expected leaf-gleaning, insectivorous and shrub
12 nesting birds to respond more negatively to IFM than species more generalized in their
13 foraging and nesting habits.

14 Methods

15 *Study Area*

16 We established 32 study stands, ranging in size from 12-16 ha each, in the Coast
17 Range of Oregon, U.S. Study stands occurred in 8 distinct blocks spanning a 100 km (N-
18 S) portion of the northern Oregon Coast Range (Fig. 1). To reduce within-block variation,
19 all four stands within a block are located within 5 km of each other. Within all block but
20 one (Tillamook), all experimentally treated stands were designed to be >1 km apart to
21 avoid influence from adjacent treatments (see *treatments* below). One study block with
22 treated stands < 1 km apart was selected due to the unavailability of alternatives on
23 Oregon Dept. of Forestry land. We note that in this case, adjacency could have the effect

1 of making our results more conservative (individuals from the control stand could
2 potentially move to the treated stands reducing effect sizes). The climate of the Oregon
3 Coast Range consists of cool, wet winters and mild, dry summers. All sampled stands are
4 in the western hemlock zone (Franklin and Dyrness 1973) and range in elevation from
5 210 - 850 m. Early-seral plantations in this area are dominated by Douglas-fir
6 (*Pseudotsuga menziesii*) saplings, with minor components of grand fir (*Abies grandis*),
7 western hemlock (*Tsuga heterophylla*), and western redcedar (*Thuja plicata*). Dominant
8 shrub/woody species include California hazelnut (*Corylus conuta* sub-spp. *californica*),
9 oceanspray (*Holodiscus discolor*), vine maple (*Acer circinatum*), big-leaf maple (*Acer*
10 *macrophyllum*), cascara (*Rhamnus purshiana*), salmonberry (*Rubus spectabilis*) and red
11 alder (*Alnus rubra*). Smaller understory broadleaf species include *Vaccinium* spp., salal
12 (*Gaultheria shallon*), and Oregon grape (*Mahonia nervosa*) which can dominate stands
13 post-harvest. The herbaceous community is comprised of many native and non-native
14 herbaceous plants with swordfern (*Polystichum munitum*) and brackenfern (*Pteridium*
15 *aquilinum*) often dominating.

16

17 *Treatments*

18 We used a randomized complete block design and randomly applied one of four
19 treatments to each of the four stands in each of the eight blocks ($n=32$). All 32 stands
20 were clearcut in fall 2009 and were planted in spring 2010 with Douglas-fir (*Pseudotsuga*
21 *menziesii*), the major commercial species in the region. Our objective was to test
22 combined effects of the suite of herbicides and surfactants used in typical operations
23 rather than to examine the effect of a particular chemical. Therefore, we applied a full

1 suite of chemicals to sites with the aim of creating a gradient in management intensity
2 across four treatments (Table 1, Fig. 2). Importantly, within a treatment, the same
3 amount and type of chemicals were applied across all blocks. The ‘site preparation’
4 treatment occurred before stands were planted and consisted of 0.10 kg ha⁻¹ Escort
5 (DuPont, Willmington, Delaware; active ingredient (ai) 60 percent *metsulfuron methyl*),
6 7.01 L ha⁻¹ Accord (Dow AgroSciences LLC, Indianapolis, Indiana; a.i. 41.5 percent
7 *glyphosate*), 1.75 L ha⁻¹ Chopper (BASF Corporation, Florham Park, NJ; a.i. 27.6 percent
8 *imazapyr*), 0.21 kg ha⁻¹ Oust (DuPont, Willmington, Delaware; a.i. 75 percent
9 *sulfometuron methyl*), and 1.75 L ha⁻¹ MSO (methylated seed oil, as surfactant) applied
10 aerially via helicopter. First year (2011) spring herbaceous release spray consisted of 2.98
11 kg ha⁻¹ Velpar (DuPont, Willmington, Delaware; a.i. 75 percent *hexazinone*), and 2.24 kg
12 ha⁻¹ 2,4-D (Dow AgroSciences LLC, Indianapolis, Indiana; a.i. 97.5 percent 2,4-
13 *dichlorophenoxy acetic acid*) applied aerially via helicopter or with ground-based
14 backpack sprayers. Second year (2012) spring herbaceous release spray consisted of 0.14
15 kg ha⁻¹ Oust XP (DuPont, Willmington, Delaware; a.i. 75 percent *sulfometuron methyl*),
16 0.42 kg ha⁻¹ Transline (Dow AgroSciences LLC, Indianapolis, Indiana; a.i. 40.9 percent
17 *clopyralid*), and 1.49 kg ha⁻¹ Velpar (DuPont, Willmington, Delaware; a.i. 75 percent
18 *hexazinone*) applied aerially via helicopter or with ground-based backpack sprayers. Only
19 two treatments had been applied prior to summer 2011 (Table 1). Thus, Moderate and
20 Intensive stands had not yet been differentiated during the first year of data collection
21 (Table 1).

22

23 *Sampling*

1 We used a stratified random approach to select three point count plots in each
2 stand and to maximize the distance between survey locations and stand edge while
3 sampling representative portions of the treatment area. In analysis, we used the average
4 of the three counts within a stand as our response variable (see *Analysis* below). We
5 sampled birds at each of the 96 point count locations in 2011 and 2012. Each point was
6 sampled four times during the breeding season (May 28 – July 3rd). The survey order and
7 observer were varied throughout the season to avoid associated biases. Point count survey
8 guidelines followed Ralph *et al.* (1995) except that we used a 10-minute time interval for
9 sampling. Censuses began at sunrise and were completed by 10 am. Every bird seen or
10 heard was recorded with an associated behavior. First and closest detection distances
11 from the census point were estimated with detections beyond a 50m radius distance band
12 considered “out” of the survey plot.

13 Ocular estimates of broadleaf shrub and herbaceous cover were taken for all non-
14 coniferous plants by species within 3-3m radius subplots each centered 20 m from avian
15 census locations. The bearing to the initial subplot was selected at random; remaining
16 plots were located to maintain 120 degrees separation from other plots. Cover estimates
17 for each species, taken within the 3m radius plots were then summed to achieve total
18 (sometimes overlapping) shrub cover estimates for the three 3m-radius subplots (Ellis et
19 al. 2012). This method was chosen to help quantify the three-dimensional nature of the
20 woody vegetation. As a result of its use, summed point level cover estimates across
21 species were allowed to exceed 100 percent.

22
23 *Analysis*

1 We assume that the replicated counts n_{ij} are obtained from R spatially dispersed
2 plots (i) on each of J sampling visits (j), where the (unobserved) plot population, N_i , is
3 closed during the period of sampling. The N -mixture model (Royle, 2004) describes
4 counts arising from a hierarchical model with two components: a state process where the
5 true plot abundances N_i are assumed to be random variables with distribution $f(N_i, \lambda)$
6 and an observation process where counts n_{ij} follow a binomial distribution, conditional on
7 the unobserved N_i and detection probability p . A general description of the model with
8 Poisson counts is:

$$9 \quad N_i \sim \text{Poisson}(\lambda_i)$$

$$10 \quad n_{ij} | N_i \sim \text{Binomial}(N_i, p_{ij})$$

11 In the N -mixture model, the plot-specific abundances N_i are considered nuisance
12 parameters and are numerically integrated from the likelihood function to obtain joint
13 estimates of λ and p . One advantage of this framework is that the parameters λ and p
14 can be allowed to vary as a function of covariates, typically via a link function for the
15 mean parameter (e.g., log-link for abundance, logit-link for detection probability).
16 Important assumptions for these models include 1) in-plot population closure during the
17 period of sampling, 2) independence of counts across plots, 3) the assumed distribution of
18 plot-level abundance across the area of interest (e.g., Poisson), and 4) the structural form
19 of parameterizations for mean abundance and detection probability. We did not assess the
20 assumption of within-season closure at the individual point-count scale due to the
21 sparsity of data for most species in our study (Rota *et al.*, 2009) If the closure assumption
22 is violated, this would result in an upward bias in our abundance estimates across all
23 treatments.

1 For all analyses, we fit N -mixture models using the ‘pcount’ function in the
2 package ‘unmarked’ in the software program R (R Development Core Team, 2010). We
3 obtained approximate asymptotic variances of parameter estimates from the inverse
4 Hessian evaluated at the maximum likelihood estimates (Royle, 2004). For all analyses
5 reported here, we used a Poisson distribution for the plot-level abundances, N_i . Several
6 approaches are available to obtain estimates of abundance from the fitted model, as
7 described in Royle (2004). We estimate expected abundance from $\hat{\lambda}$ directly, using the
8 pooled count from the three 50 m radius plots.

9 We considered the model based on the experimental design, in which abundance
10 varied by year and treatment. The effect of elevation was not a focus of our study, but
11 prior studies have found associations between avian abundance and elevation (Terborgh,
12 1977) and we wanted to control for this source of variation; we therefore included it as a
13 predictor of abundance in all models. We modeled detection probability as a function of
14 percent cover of broadleaf plants. This was a more parsimonious way to include an effect
15 of the treatments (which primarily reduce broadleaf cover) on detection than a categorical
16 treatment effect. In addition, we included linear and quadratic terms for Julian date
17 (January 1 = 1, December 31=365) because avian detection rates are known to vary
18 seasonally (Kéry *et al.*, 2005). We centered and scaled the continuous covariates. Our
19 specific model was:

20

21 $N_i \sim \text{Poisson}(\lambda_i)$

22 $n_{ij} : \text{Binomial}(N_i, p_{ij})$

23

1 $\text{logit}(p_{ij}) = \alpha_0 + \alpha_1 \cdot \text{Percent Broadleaf Cover} + \alpha_2 \cdot \text{Julian Date} + \alpha_3 \cdot \text{Julian Date}^2$

2

3 $\log(\lambda_i) = \beta_0 + \beta_1 \cdot \text{Year (2012)} + \beta_2 \cdot \text{Light} + \beta_3 \cdot \text{Moderate} + \beta_4 \cdot \text{Intensive} + \beta_5 \cdot \text{Light} \cdot$
4 $\text{Year (2012)} + \beta_6 \cdot \text{Moderate} \cdot \text{Year (2012)} + \beta_7 \cdot \text{Intensive} \cdot \text{Year (2012)} + \beta_8 \cdot \text{Elevation}$

5

6 For each species by year combination, we present estimates (average and 95%
7 confidence interval) of treatment effect contrasts (Kroll *et al.*, 2012). In our
8 parameterization, the treatment contrast compares abundance of each of the three
9 herbicide treatments to the Control. We back-transformed values so that contrasts can be
10 interpreted as either the average percent increase or decrease in abundance due to a
11 specific treatment as compared to the Control. A treatment contrast of 1 indicates that
12 abundance is equal across treatments. In addition, we included contrasts with Moderate as
13 the base-line, given that Moderate is the closest approximation of operational practice in
14 the Oregon Coast Range (Appendix B). We present estimated abundances for species by
15 treatment combination for 2011 and 2012. Following Nichols *et al.* (2009), we interpret
16 λ as the average number of individuals whose home ranges overlap the 3 point stations
17 within a harvest stand, rather than the average total number of individuals who occur in
18 the harvest stand.

19 Using the models above, we estimated the effect size (Control–Moderate) for each
20 species from 2012 data. We summarized these results with box plots of effect sizes
21 grouped by species' life history traits. Regional-scale population trends for the Pacific
22 Northwest were derived from the Breeding Bird Survey (BBS; Sauer *et al.* 2011). The
23 BBS relies on observations made by volunteers along 40 km roadside samples each

1 representing approximately 1° square of latitude and longitude. BBS Trends estimated
2 were for the Pacific Northwest Rainforest (Oregon, California and Washington only)
3 1983-2011 using hierarchical models ([http://www.mbr-](http://www.mbr-pwrc.usgs.gov/bbs/trend/tf11.html)
4 [pwrc.usgs.gov/bbs/trend/tf11.html](http://www.mbr-pwrc.usgs.gov/bbs/trend/tf11.html)).

5 6 Results

7 Cover of non-woody vegetation, including grasses, herbs and ferns, decreased
8 with treatment intensity but increased overall from 2011 to 2012 (Table 2). As expected,
9 Control and Light treatments had higher broadleaf plant cover and species richness than
10 Moderate or Intensive treatments in 2011 and 2012 (Table 2), indicating that herbicide
11 treatments had a strong and consistent effect two years of post-harvest. Broadleaf cover
12 increased in all treatments between 2011 and 2012. Conifer density was similar across
13 treatments and sampling years (Table 2).

14 We detected 63 bird species during the study period with 3044 total detections
15 recorded in 768 10-minute sampling periods. Nine-hundred sixty-eight detections of 51
16 species were recorded in 2011 and 2076 detections from 49 species in 2012. House wren
17 (*Troglodytes aedon*), dark-eyed junco (*Junco hyemalis*) and white-crowned sparrow
18 (*Zonotrichia leucophrys*) made up 50 percent of total detections from both years. We
19 estimated abundance for individual species with greater than 1% (30) of the total
20 detections in 2011 and 2012 combined (Table 3).

21 Thirteen species were abundant enough to be analyzed individually and made up
22 75 percent of total detections (Table 3). Four of those species had 15 or fewer detections
23 in 2011 (orange-crowned warbler (*Vermivora celata*), spotted towhee (*Pipilo maculatus*),

1 Swainson's thrush (*Catharus ustulatus*) and Townsend's solitaire (*Myadestes*
2 *townsendi*), and so we analyzed 2012 detections only for those species. Six of the 13
3 most abundant species are considered to be early seral broadleaf forest associates (rufous
4 hummingbird, orange-crowned warbler, song sparrow (*Melospiza melodia*), Swainson's
5 thrush, white-crowned sparrow, Wilson's warbler (*Wilsonia pusilla*) (Ellis et al. 2012),
6 whereas the remaining 7 species are more generalized in their distributions (Table 3).

7 Mid-season species-specific detection probabilities ranged from 0.04 to 0.60 for
8 an individual point count visit. Twelve of 13 species had detection probabilities below
9 0.35. Species-specific abundance estimates increased for most species between 2011 and
10 2012 as species colonized the stands post-disturbance (Appendix A). Confidence
11 intervals of abundance estimates were broad for many species, especially those with low
12 numbers of detections.

13 Treatment contrasts with the Control were statistically significant for at least one
14 treatment×year interaction for four species that have previously been identified as being
15 strongly associated with early seral broadleaf forest (rufous hummingbird, Swainson's
16 thrush, white-crowned sparrow, and Wilson's warbler). Wilson's warbler and rufous
17 hummingbird were the most sensitive species to the Intensive treatment, with abundance
18 estimates ~— for both years – 5-20% of those in Control (Fig. 3a). Two species that are
19 more generalized in their distributions (American goldfinch [*Spinus tristis*] and house
20 wren) (Figs. 3 and 4; Appendix C) also showed sensitivity to intensive forest
21 management treatments but the magnitude and direction of these effects differed;
22 contrasts with the Control for American goldfinch were more variable, indicating lower
23 abundance estimates for Light and Moderate treatments but not for the Intensive

1 treatment (Fig. 3b). House wren treatment contrasts indicated lower abundance estimates
2 for Moderate and Intensive treatments but not the Light treatment (Fig. 3b).

3 We did not detect significant differences between treatments and the Control for 6
4 species (dark-eyed junco, orange-crowned warbler, song sparrow, spotted towhee,
5 Townsend's solitaire, and western bluebird (*Sialia mexicana*)) (Figures 3a and 3b). Due
6 to the relatively low number of detections for three of these species, spotted towhee, song
7 sparrow and orange-crowned warbler were imprecise, so we are not able to reject the
8 possibility of a biologically meaningful effect of herbicide treatments. However, the
9 remaining four species generally showed equal or greater abundances in treated stands.
10 Violet-green swallow (*Tachycineta thalassina*) showed a significant positive response to
11 the moderate treatment in the first year of the study.

12 When modeled as a group, both the 6 early seral associated species (Fig. 4
13 Appendix C). Treatment contrasts to the Control indicate lower abundance estimates in
14 Light, Moderate and Intensive treatments for this group. We reduced life history traits to
15 coarse categories due to the limited number of species in our study. Shrub-nesting birds
16 and foliage gleaning insectivores responded more strongly to the moderate treatment in
17 general than ground nesting species, cavity nesters, and aerial insectivores (Fig. 5a and
18 5b).

19 Finally, the species with the largest negative effect sizes in our study (primarily
20 early-seral associates, especially shrub nesting, leaf gleaning insectivores) are also ones
21 with the greater estimated long-term population declines (Sauer *et al.*, 2011; Fig. 6).

22

23

24

1 Discussion

2 Abundance of six of thirteen bird species that are common in early-seral forests of the
3 Pacific Northwest was significantly reduced in at least one of the herbicide treatments in
4 relation to the Control. The Moderate and Intensive treatments reduced the cover of
5 broadleaf shrubs, which generally contain greater abundances of lepidopteron larvae than
6 coniferous and graminoid vegetation (Hammond and Miller 1988). If greater amounts of
7 hardwood cover result in more prey, the result could be increased foraging efficiency and
8 reduced territory size, potentially explaining the greater bird abundance we observed in
9 the Control and Light treatments (the food value theory of territoriality; Stenger, 1958).
10 Also, increased foliage volume in Control stands may accommodate a greater number
11 and diversity of nesting sites than hardwood-impoverished stands (Morrison and Meslow,
12 1983). This interpretation is supported by the results that shrub nesters and insectivorous
13 birds tended to be more negatively influenced by our heavier treatments. One of the few
14 previous manipulative studies to examine herbicide effects on bird abundance Easton and
15 Martin (1998) found increased abundance of conifer shrub nesters in treated stands. Our
16 results do not support this finding, but this is likely due to differences in the ages of
17 treated stands (Easton and Martin 1998: 11-22 years, current study: 1-2 years). Conifers
18 in the current study are not sufficiently large in most cases to support a nest.

19 The association between species' life history traits and magnitude of species'
20 response to the Moderate and Intensive treatments supports the hypothesis that IFM
21 effects are mediated through availability of food and nest sites. The fact that all species
22 examined did not show consistent declines in relation to herbicide treatments suggests
23 that the reduced abundances we observed for some species were not a direct function of

1 herbicide toxicity. This finding is supported by experimental toxicological studies on the
2 primary herbicides used in our study (Tatum 2004, McComb et al. 2008). Regardless, our
3 life-history results are correlative, so more research is required to assess whether there are
4 cumulative direct effects of herbicide on bird demography.

5 Our results come with the important caveat that we estimated bird abundance
6 only; it is well known that abundance is not necessarily an indicator of reproductive
7 success or survival (Van Horne, 1983). Several previous manipulative forest management
8 studies have found males returning to remain in altered habitats during the first year post-
9 disturbance, while females dispersed to new, potentially higher-quality habitat
10 (Woodcock *et al.*, 1997). Both Mackinnon and Freedman (1993) as well as Easton and
11 Martin (1998) showed temporal lags in response to herbicide treatments; effects did not
12 emerge for at least one year. We expect that because our study was initiated immediately
13 post-harvest our abundance estimates reflect colonization by a new avian community (the
14 previous mature stand is unlikely to have supported most of the early seral species
15 reported in this study). Thus, the potential for such temporal lags is mostly eliminated.
16 Further, recent results indicate that density may be correlated with per capita productivity
17 in managed stands and follows an ideal free distribution (Hache *et al.*, 2013). In other
18 words, depressed densities in treated stands may reduce per hectare productivity (i.e.,
19 fewer birds producing young), but not necessarily lower individual-level reproduction.
20 Though we have just initiated demographic studies in our experiment, results from a
21 retrospective study in the same region suggest that per capita productivity does not vary
22 across a gradient in management intensity (Ellis *et al.*, 2012; Rivers *et al.*, 2012).
23 Nevertheless, critical future work on our study plots will test the relationship between

1 density and productivity in intensively managed stands.

2 Donato *et al.* (2012) argued that plantation forestry severely truncates the
3 longevity of pre-canopy closure establishment period. Negative responses by some
4 species to our most intense treatments, in the very early stages of succession, indicates
5 that truncation can occur at the ‘front end’ of stand regeneration as well. How long does
6 this truncation continue, particularly in Light and Moderate stands, which approximate
7 operational standards in the Oregon Coast Range? For instance, in eastern Canada,
8 Mackinnon and Freedman (1993) found that bird abundance for all species was similar in
9 stands that had been sprayed with herbicide to that of the unsprayed control after four
10 years. However, in a retrospective study conducted in the Oregon Coast Range, Ellis et
11 al. (2012) found that counts of orange-crowned warbler, Wilson’s warbler and
12 Swainson’s thrush were reduced in stands with lower levels of broadleaf cover, even 5-9
13 years after initial herbicide treatment.

14 The longevity and degree of this early stage truncation is likely to depend upon
15 intensity of the initial treatment as well as factors such as previous site management,
16 legacy species, seed beds and local seed sources. Our results indicate that the Light
17 treatment, which differs from the control by only one herbicide application (Table 1), has
18 apparent benefits for a number of species (house wren, rufous hummingbird, Wilson’s
19 warbler, orange-crowned warbler) during the first two years of stand growth. This result
20 has been proposed as an untested hypothesis in previous studies (Morrison and Meslow,
21 1983; Santillo *et al.*, 1989), but has only now received empirical support.

22 The dichotomy, for some species, between the Control and Light versus the
23 Moderate and Intensive treatments is consistent with our previous work suggesting

1 thresholds in bird abundance at ~10% hardwood cover (Ellis and Betts, 2011). Both the
2 Moderate and Intensive treatments fell below this threshold in the initial two years of our
3 study, whereas both the Control and Light treatments contained >30% hardwood cover,
4 even just one year after clearcutting and initial treatments.

5 Our finding that species with the largest negative effect sizes also are the ones
6 with estimated long-term population declines may provide insight into demography of
7 these species at the regional scale. This relationship does not necessarily implicate IFM in
8 these declines, but provides insight into the habitat requirements for these species. Other
9 stressors, particularly forest succession and reduced harvesting on Federal lands, may be
10 contributing factors (Betts *et al.*, 2010). During the first two years of stand growth,
11 reducing herbicide treatment intensity may disproportionately benefit those species
12 declining at the greatest rates. For several species, even our Moderate treatment had
13 higher abundance than the Intensive treatment (though not significantly so). The
14 Moderate treatment, more closely approximates the industrial standard on large private
15 landholdings in the PNW, U.S.

16 What remains to be quantified are the trade-offs between reductions in intensity of
17 herbicide application and growth of merchantable trees (Wagner *et al.*, 2004). Such trade-
18 offs should be considered not only within stands, but at landscape and regional scales.
19 Some components of biodiversity are expected to be reduced in those portions of the
20 forest landscape that are managed primarily for timber production rather than biodiversity
21 conservation (Noble and Dirzo, 1997). Importantly, such tradeoffs come with the
22 important benefit of producing more timber on a reduced area (Maguire *et al.*, 2009), thus
23 reducing pressure on less intensively managed and protected areas. Together, these

1 practices follow the key tenets of the TRIAD approach to conserving biodiversity in
2 managed landscapes (Seymour and Hunter, 1992; Hartmann *et al.*, 2010). However, our
3 current study, as well as previous correlative results (Betts *et al.* 2010, Ellis and Betts
4 2011, Ellis *et al.* 2012), suggest that such trade-offs between biodiversity and timber
5 production may not be as large as previously anticipated. Through the first two years of
6 stand growth, the ‘Light’ treatment was indistinguishable from the control for a number
7 of declining early seral associates. To balance biodiversity conservation and timber
8 production, research examining the tradeoffs between reduced application rates of
9 herbicide and tree growth is urgently required.

10

11 Acknowledgments

12 We thank our many field assistants for diligent work. We thank anonymous reviewers for
13 constructive comments on this manuscript. Access to study sites as well as cooperation in
14 implementing the experimental treatments was granted by Weyerhaeuser, Hancock
15 Timber Resource Group, the Oregon Department of Forestry and Plum Creek Timber.
16 Research funding was provided by grants from the United States Department of
17 Agriculture, Agriculture Food and Research Initiative grant (AFRI-2009-04457), the
18 National Council for Air and Stream Improvement, the Nobel Fund and the Oregon State
19 University College of Forestry Fish and Wildlife Habitat in Managed Forests Research
20 Program to M.G. Betts.

Literature Cited

- Adams, W.T., Hobbs, S., Johnson, N., 2005. Intensively managed forest plantations in the Pacific Northwest: Introduction. *Journal of Forestry* 103, 59-60.
- Angelstam, P.K., 1998. Maintaining and restoring biodiversity in European boreal forests by developing natural disturbance regimes. *Journal of Vegetation Science* 9, 593-602.
- Betts, M.G., Hagar, J.C., Rivers, J.W., Alexander, J.D., McGarigal, K., McComb, B.C., 2010. Thresholds in forest bird occurrence as a function of the amount of early-seral broadleaf forest at landscape scales. *Ecological Applications* 20, 2116-2130.
- Donato, D.C., Campbell, J.L., Franklin, J.F., 2012. Multiple successional pathways and precocity in forest development: can some forests be born complex? *Journal of Vegetation Science* 23, 576-584.
- Easton, W.E., Martin, K., 1998. The effect of vegetation management on breeding bird communities in British Columbia. *Ecological Applications* 8, 1092-1103.
- Ellis, T.M., Betts, M.G., 2011. Bird abundance and diversity across a hardwood gradient within early seral plantation forest. *Forest Ecology and Management* 261, 1372-1381.
- Ellis, T.M., Kroll, A.J., Betts, M.G., 2012. Early seral hardwood vegetation increases adult and fledgling bird abundance in Douglas-fir plantations of the Oregon Coast Range, USA. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere* 42, 918-933.

- Franklin, J.F., Spies, T.A., Perry, D., Harmon, M.E., McKee, A., (Eds.), pp. 373–379
(Eds.), 1986. Modifying Douglas-fir management regimes for nontimber objectives. USDA Forest Service, Seattle, WA.
- Gregory, R.D., Strien, A., Vorisek, P., 2006. Using birds as indicators of environmental change in Europe. *Journal of Ornithology* 147, 16-16.
- Hache, S., Villard, M.A., Bayne, E., 2013. Experimental evidence for an ideal free distribution in a breeding population of a territorial songbird. *Ecology*. *In Press*.
- Hagar, J.C., 2007. Wildlife species associated with non-coniferous vegetation in Pacific Northwest conifer forests: A review. *Forest Ecology and Management* 246, 108-122.
- Hagar, J.C., Li, J., Sobota, J., Jenkins, S., 2012. Arthropod prey for riparian associated birds in headwater forests of the Oregon Coast Range. *Forest Ecology and Management* 285, 213-226.
- Hammond, P.C., Miller, J.C., 1998. Comparison of the biodiversity of Lepidoptera within three forested ecosystems. *Annals of the Entomological Society of America* 91, 323-328.
- Hartmann, H., Daoust, G., Bigue, B., Messier, C., 2010. Negative or positive effects of plantation and intensive forestry on biodiversity: A matter of scale and perspective. *Forestry Chronicle* 86, 354-364.
- Hunt, P.D., 1998. Evidence from a landscape population model of the importance of early successional habitat to the American Redstart. *Conservation Biology* 12, 1377-1389.

- Iglay, R.B., Demarais, S., Wigley, T.B., Miller, D.A., 2012. Bird community dynamics and vegetation relationships among stand establishment practices in intensively managed pine stands. *Forest Ecology and Management* 283, 1-9.
- Jones, J.E., Kroll, A.J., Giovanini, J., Duke, S.D., Betts, M.G., 2011. Estimating thresholds in occupancy when species detection is imperfect. *Ecology* 92, 2299-2309.
- Jones, J.E., Kroll, A.J., Giovanini, J., Duke, S.D., Ellis, T.M., Betts, M.G., 2012. Avian Species Richness in Relation to Intensive Forest Management Practices in Early Seral Tree Plantations. *Plos One* 7: e43290.
- Kauppi, P.E., Ausubel, J.H., Fang, J.Y., Mather, A.S., Sedjo, R.A., Waggoner, P.E., 2006. Returning forests analyzed with the forest identity. *Proceedings of the National Academy of Sciences of the United States of America* 103, 17574-17579.
- Kennedy, R.S.H., Spies, T.A., 2005. Dynamics of hardwood patches in a conifer matrix: 54 years of change in a forested landscape in Coastal Oregon, USA. *Biological Conservation* 122, 363-374.
- Kéry, M., Royle, J., Schmid, H., 2005. Modeling avian abundance from replicated counts using binomial mixture models. *Ecological Applications* 15, 1450-1461.
- Kroll, A., Giovanini, J., Jones, J., Arnett, E., Altman, B., 2012. Effects of salvage logging of beetle-killed forests on avian species and foraging guild abundance. *Journal of Wildlife Management* 76, 1188-1196.

- Lautenschlager, R.A., 1993. Response of Wildlife to Forest Herbicide Applications in Northern Coniferous Ecosystems. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere* 23, 2286-2299.
- Lautenschlager, R.A., Sullivan, T.P., 2004. Improving research into effects of forest herbicide use on biota in northern ecosystems. *Wildlife Society Bulletin* 32, 1061-1070.
- Litvaitis, J.A., 1993. Response of early successional vertebrates to historic changes in land-use. *Conservation Biology* 7, 866-873.
- MacKinnon, D. S., and B. Freedman. 1993. Effects of silvicultural use of the herbicide glyphosate on breeding birds of regenerating clearcuts in Nova Scotia, Canada. *Journal of Applied Ecology* 30, 395-406.
- Maguire, D.A., Mainwaring, D.B., Rose, R., Garber, S.M., Dinger, E.J., 2009. Response of coastal Douglas-fir and competing vegetation to repeated and delayed weed control treatments during early plantation development. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere* 39, 1208-1219.
- McComb, B.C., Curtis, L., Chambers, C.L., Newton, M., Bentson, K., 2008. Acute toxic hazard evaluations of glyphosate herbicide on terrestrial vertebrates of the Oregon coast range. *Environmental Science and Pollution Research* 15, 266-272.
- Morrison, M.L., Meslow, E.C., 1983. Bird community structure on early-growth clearcuts in western Oregon. *American Midland Naturalist* 110, 129-137.
- Najera, A., Simonetti, J.A., 2006. Enhancing Avifauna in Commercial Plantations. *Conservation Biology* 24, 319-324.

- Nichols, J.D., Thomas, L., Conn, P.B., 2009. Inferences about landbird abundance from count data: recent advances and future directions. In: Thomson, D.L., Cooch, E.G., Conroy, M.J. (Eds.), *Modeling demographic processes in marked populations*. Springer, New York, New York, USA, pp. 201-235.
- Noble, I.R., Dirzo, R., 1997. Forests as human-dominated ecosystems. *Science* 277, 522-525.
- Oliver C.D., Larson B.C., 1996 *Forest stand dynamics*: John Wiley and Sons, New York, New York, USA.
- R Development Core Team, 2010. *R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria.
- Ralph, C.J., Sauer, J., Droedge, S., 1995. *Monitoring bird populations by point counts*. Pacific Southwest Research Station, Albany, CA.
- Rivers, J.W., Liebl, A.L., Owen, J.C., Martin, L.B., Betts, M.G., 2012. Baseline corticosterone is positively related to juvenile survival in a migrant passerine bird. *Functional Ecology* 26, 1127-1134.
- Rota, C.T., Fletcher, R.J., Dorazio, R.M., Betts, M.G., 2009. Occupancy estimation and the closure assumption. *Journal of Applied Ecology* 46, 1173-1181.
- Royle, J.A., 2004. N-mixture models for estimating population size from spatially replicated counts. *Biometrics* 60, 108-115.
- Santillo, D.J., Brown, P.W., Leslie, D.M., 1989. Response of songbirds to glyphosate-induced habitat changes on clearcuts. *Journal of Wildlife Management* 53, 64-71.

- Sauer, J.R., Hines, J.E., Fallon, J., 2011. The North American Breeding Bird Survey, Results and Analysis 1966–2010. Version 2011.2., USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA.
- Schulze, C.H., et al, 2004. Biodiversity indicator groups of tropical land-use systems: comparing plants, birds and insects. *Ecological Applications* 14, 1321–1333.
- Sekercioglu, C.H., Daily, G.C., Ehrlich, P.R., 2004. Ecosystem consequences of bird declines. *Proceedings of the National Academy of Sciences of the United States of America* 101, 18042-18047.
- Seymour, R.S., Hunter, M.L., 1992. New forestry in eastern spruce–fir forests: principles and applications to Maine. Maine Agricultural Experiment Station, Orono.
- Spies, T.A., Johnson, K.N., 2007. Projecting forest policy and management effects across ownerships in Coastal Oregon. *Ecological Applications* 17, 3-4.
- Spies, T.A., Johnson, K.N., Burnett, K.M., Ohmann, J.L., McComb, B.C., Reeves, G.H., Bettinger, P., Kline, J.D., Garber-Yonts, B., 2007. Cumulative ecological and socioeconomic effects of forest policies in Coastal Oregon. *Ecological Applications* 17, 5-17.
- Stenger, J., 1958. Food habits and available food of oven-birds in relation to territory size. *Auk* 75, 335-346.
- Swanson, M.E., Franklin, J.F., Beschta, R.L., Crisafulli, C.M., DellaSala, D.A., Hutto, R.L., Lindenmayer, D.B., Swanson, F.J., 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. *Frontiers in Ecology and the Environment* 9, 117-125.

- Tatum, V.L., 2004. Toxicity, transport, and fate of forest herbicides. *Wildlife Society Bulletin* 32, 1042-1048.
- Terborgh, J., 1977. Bird species-diversity on an Andean elevational gradient. *Ecology* 58, 1007-1019.
- Thomas, J.W., Franklin, J.F., Gordon, J., Johnson, K.N., 2006. The northwest forest plan: Origins, components, implementation experience, and suggestions for change. *Conservation Biology* 20, 277-287.
- Van Horne, B., 1983. Density as a misleading indicator of habitat quality. *Journal of Wildlife Management* 47, 893-901.
- Venier, L.A., Pearce, J.L., 2004. Birds as indicators of sustainable forest management. *Forestry Chronicle* 80, 61-66.
- Wagner, R.G., Newton, M., Cole, E.C., Miller, J.H., Shiver, B.D., 2004. The role of herbicides for enhancing forest productivity and conserving land for biodiversity in North America. *Wildlife Society Bulletin* 32, 1028-1041.
- Woodcock, J., Lautenschlager, R.A., Bell, F.W., Ryder, J.P., 1997. Indirect effects of conifer release alternatives on songbird populations in northwestern Ontario. *Forestry Chronicle* 73, 107-112.

Tables and Figures

Table 1. Timing of the application of four treatments (Control, Light, Moderate, Intensive) from 2010-2012 in the Oregon Coast Range, U.S

Treatment	Year post- harvest	Control	Light	Moderate	Intensive
Site-preparation (broadleaf vegetation spray)	0			X	X
Planted: Approx. 1100 trees/ha.	1	X	X	X	X
Herbaceous spray	1		X	X	X
Herbaceous spray	2				X

Table 2. Mean (\pm SD) for vegetation and stand-location attributes in relation to herbicide treatments applied in the Oregon Coast Range, U.S

Treatment	Percent Broadleaf		Broadleaf Species		Non-woody		Conifer Density		Elevation (m)	Slope (%)
	Cover (%)		Richness		Vegetation Cover		(stems/ha)			
	2011	2012	2011	2012	2011	2012	2011	2012		
Control	27.9	51.2	7.3	6.3	37.6	55.4	1022	1077	496	17.4
	(13.4)	(17.1)	(1.1)	(1.9)	(16.6)	(15.4)	(377)	(739)	(182)	(6.4)
Light	32.2	57.2	6.7	5.5	14.6	34.5	786	790	484	20.7
	(11.0)	(17.5)	(2.6)	(1.5)	(9.4)	(21.4)	(311)	(389)	(180)	(6.7)
Moderate	4.5	7.6	2.3	2.4	3.5	27.4	928	763	485	16.3
	(3.3)	(7.5)	(0.9)	(1.2)	(3.1)	(18.8)	(263)	(228)	(158)	(7.6)
Intensive	3.0	11.0	2.0	1.9	3.0	10.3	900	881	528	16.2
	(3.6)	(8.9)	(1.5)	(1.1)	(2.1)	(6.7)	(153)	(244)	(151)	(9.6)

Table 3. Individual detections by treatment and year for species making up greater than 1 percent of total detections, Oregon Coast Range, U.S., 2011-2012. Species considered previously as being strongly associated with early seral broadleaf forest are designated with a *.

Species	Control		Light		Moderate		Intensive		Total
	2011	2012	2011	2012	2011	2012	2011	2012	
American goldfinch (<i>Spinus tristis</i>)	13	37	4	17	1	13	4	28	117
Dark-eyed junco (<i>Junco hyemalis</i>)	44	88	60	84	58	80	57	86	557
House wren (<i>Troglodytes aedon</i>)	71	126	49	100	28	104	36	82	596
Orange-crowned warbler * (<i>Vermivora celata</i>)	0	13	3	12	1	2	2	1	34
Rufous hummingbird * (<i>Selasphorus rufus</i>)	10	14	4	20	2	9	1	2	62
Song sparrow * (<i>Melospiza melodia</i>)	15	26	13	11	3	8	2	4	82
Spotted towhee (<i>Pipilo maculatus</i>)	2	16	3	17	9	10	0	4	61
Swainson's thrush * (<i>Catharus ustulatus</i>)	0	21	1	8	0	4	0	3	37
Townsend's solitaire (<i>Myadestes townsendi</i>)	4	5	2	6	4	5	5	3	34
Violet-green swallow (<i>Tachycineta thalassina</i>)	5	22	4	22	16	40	6	39	154
Western bluebird (<i>Sialia mexicana</i>)	3	12	4	6	8	19	10	26	88
White-crowned sparrow * (<i>Zonotrichia leucophrys</i>)	50	90	16	76	21	51	16	57	377
Wilson's warbler * (<i>Wilsonia pusilla</i>)	11	23	7	18	4	3	2	3	71

Figure Captions

Fig. 1. Geographic location of eight study blocks containing four treatments in Oregon Coast Range, U.S.

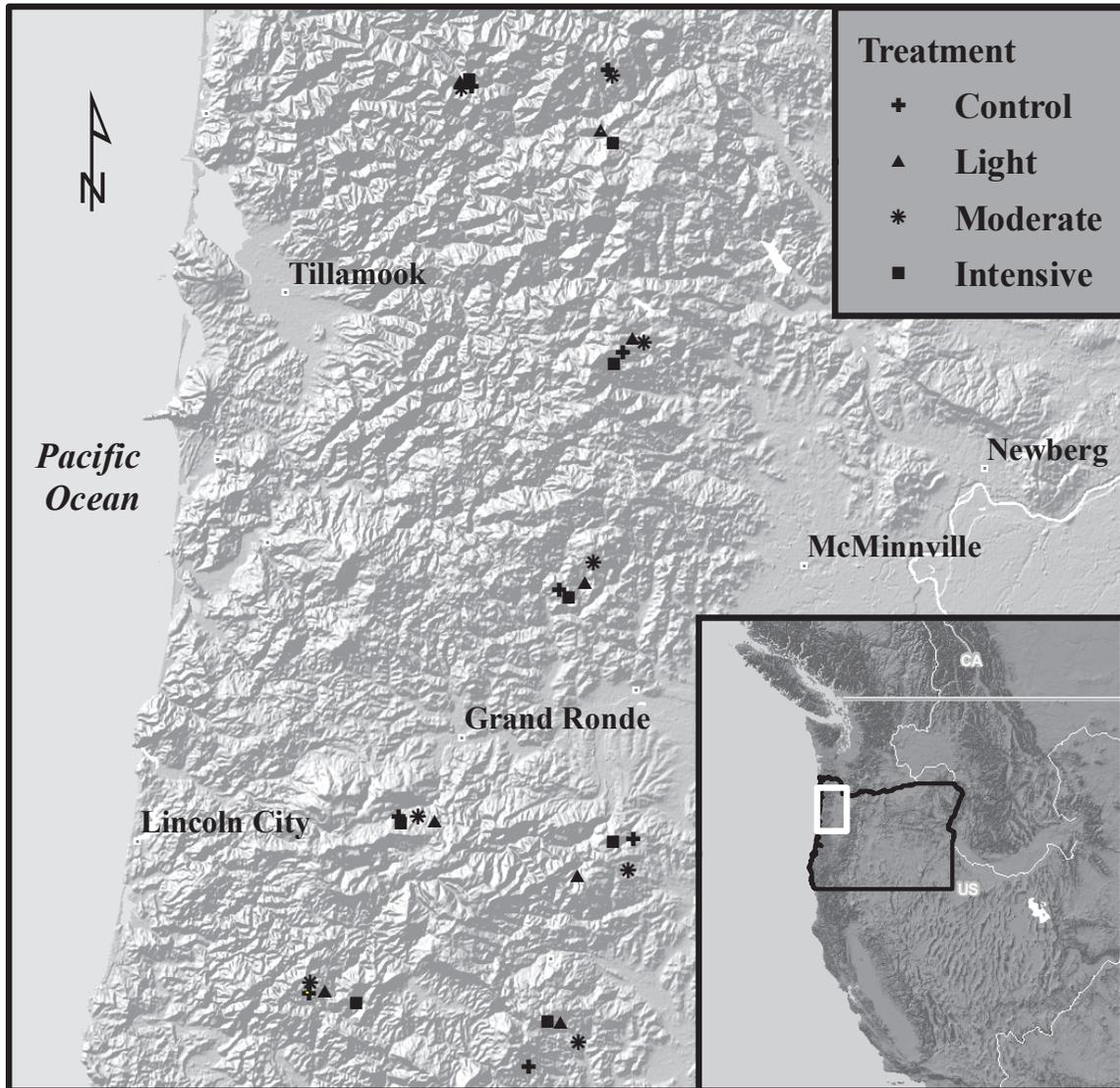


Fig. 2 Representative examples of four intensive forest management treatments applied in this study in the second year of the study ranging from Control (A), Light (B), Moderate (C) and Intensive (D) herbicide application, Oregon Coast Range, U.S.

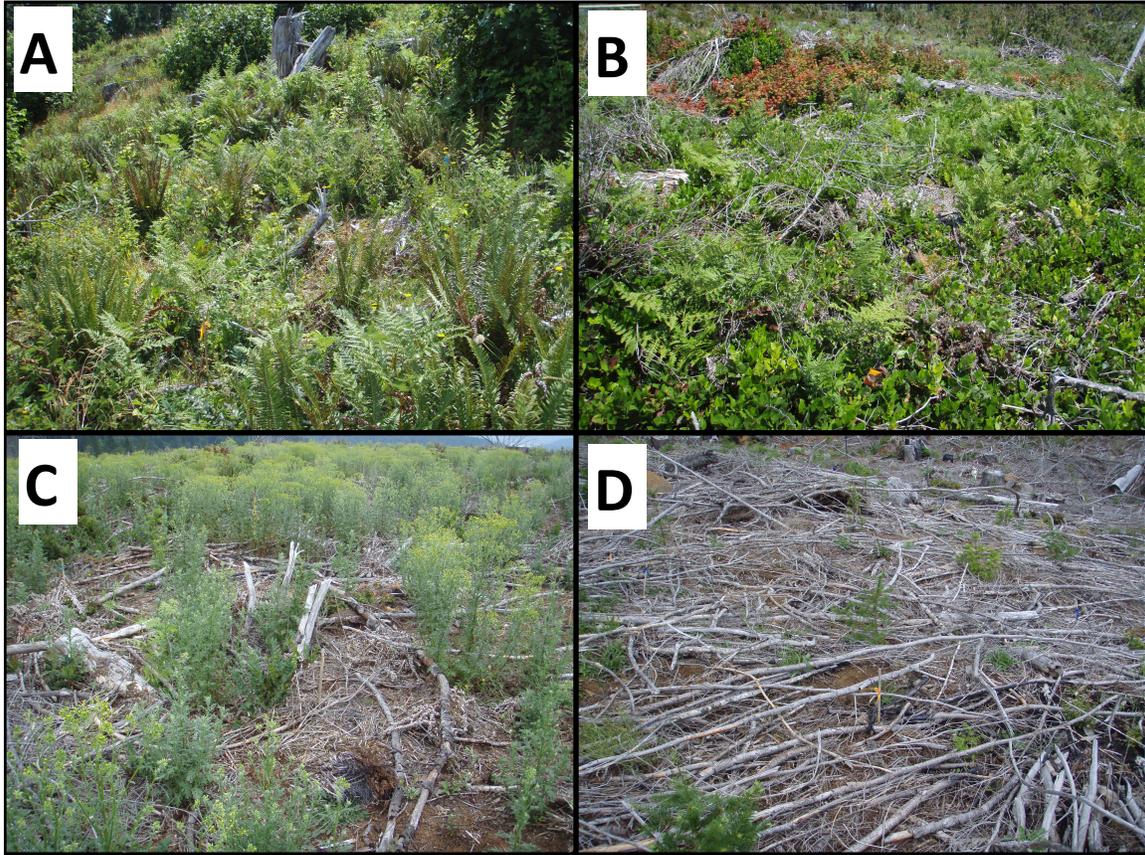
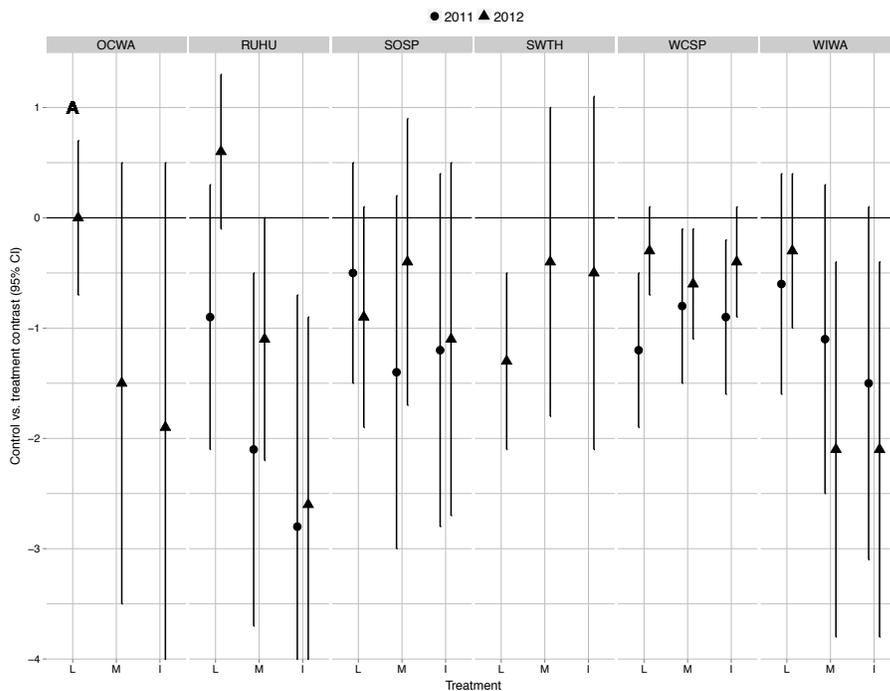


Fig. 3 Back-transformed model estimates (95% confidence interval) for three intensive forest management treatments in relation to untreated controls, Oregon Coast Range, U.S., 2011-2012, for (A) six common species expected, based on previous research, to be associated strongly with early seral hardwood forest (orange-crowned warbler [OCWA], rufous hummingbird [RUHU], song sparrow [SOSP], Swainson’s thrush [SWTH], white-crowned sparrow [WCSP] and Wilson’s warbler [WIWA] and (B) seven of the other most common species detected in our study (American goldfinch [AMGO], dark-eyed (Oregon) junco [DEJU], house wren [HOWR], spotted towhee [SPTO], violet-green swallow [VGSW], western bluebird [WEBL] and Townsend’s solitary [TOSO]). Treatments comprise Light (L), Moderate (M) and Intensive (I).



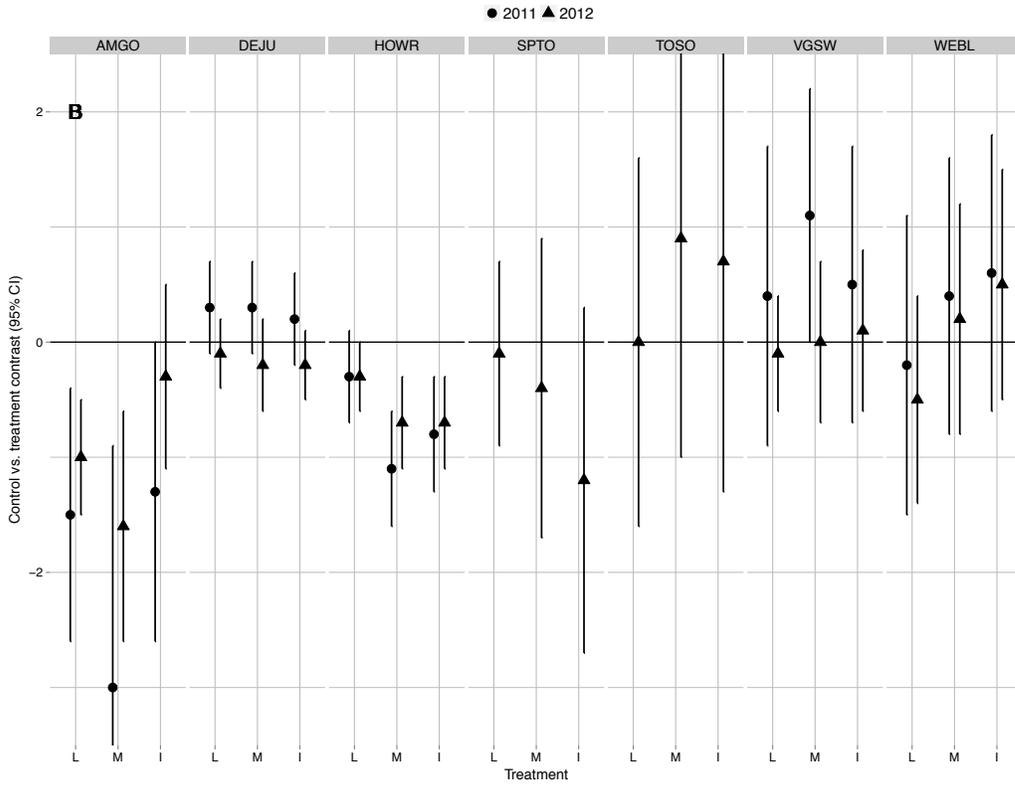


Fig. 4. Back-transformed model estimates (95% confidence interval) for three intensive forest management treatments in relation to untreated controls for species expected to be associated with early serial hardwood forest, Oregon Coast Range, U.S., 2011-2012.

Treatments comprise Light (L), Moderate (M) and Intensive (I).

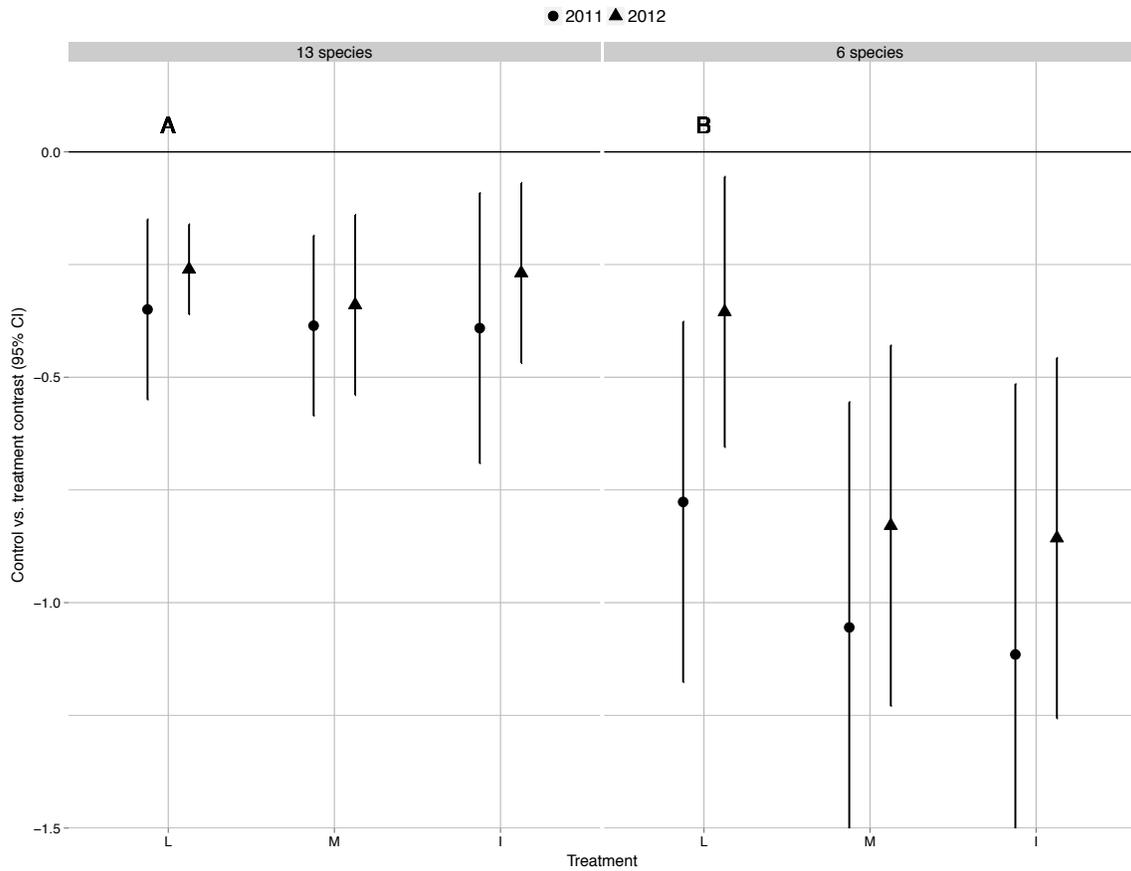


Fig. 5. Boxplots of the association between two life-history traits on modeled effect size (log scale) of the Moderate intensity forest management treatments, Oregon Coast Range, U.S., 2011-2012. (A) Nest type: shrub (S) vs. other types (ground, cavity; O), (B) Foraging mode: leaf gleaner (G) versus other modes (ground foraging, areal insectivore; O).

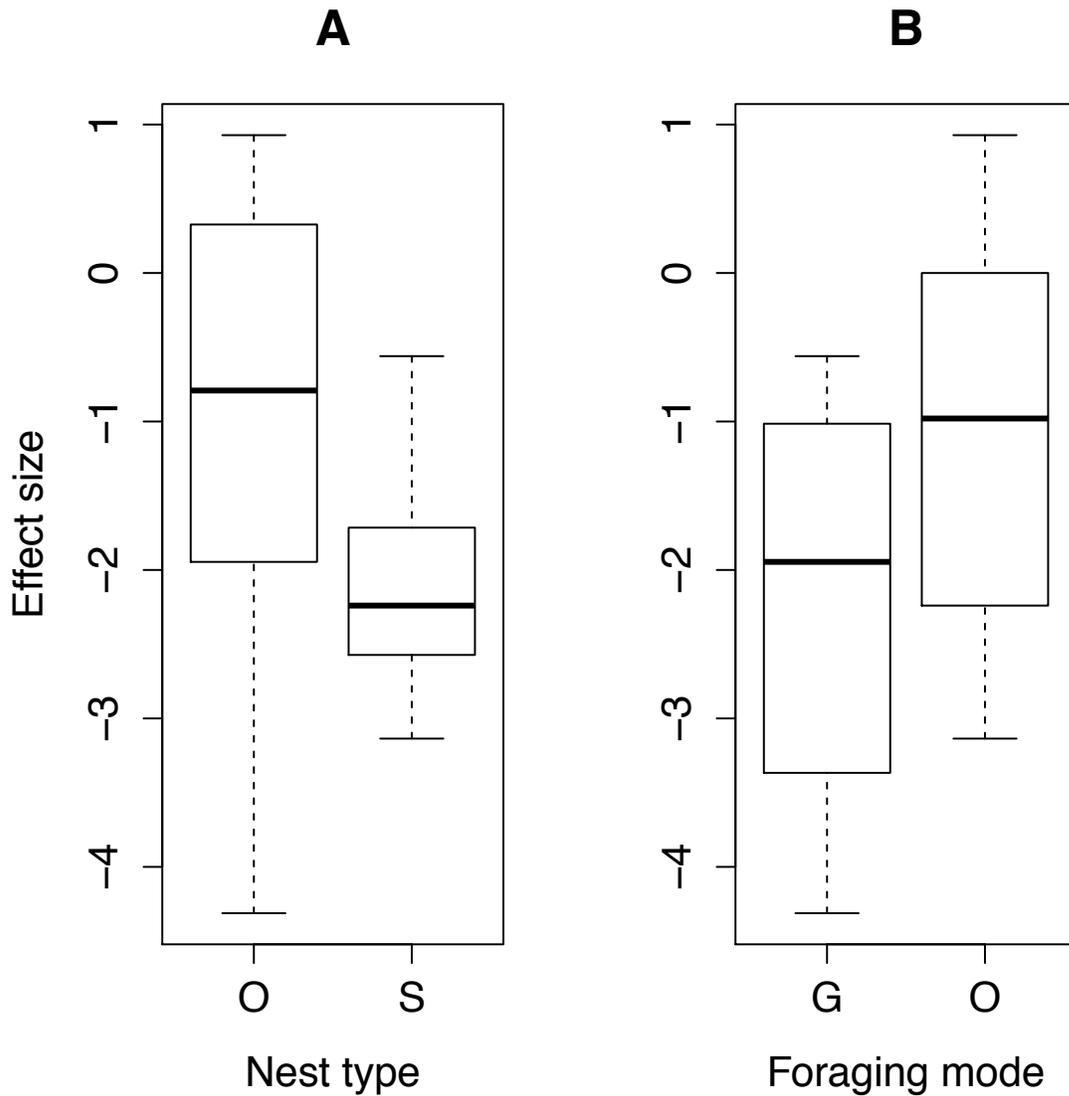


Fig. 6. Correlation between modeled effect size (log scale) of the Moderate intensity treatment (see text for details) and the 30-year population trend for 13 species of forest birds in the Pacific Northwest estimated from the BBS monitoring program. Species most strongly influenced by intensive forest management showed the greatest population declines. Species most and least influenced by intensive treatments are labeled according to species codes in Fig. 3.

