Summer 2020

Effects of non-commercial thinning on Great Horned Owls on Turnbull National Wildlife Refuge

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EFFECTS OF NON-COMMERCIAL THINNING ON GREAT HORNED OWLS ON TURNBULL NATIONAL WILDLIFE REFUGE

A Thesis

Presented To

Eastern Washington University

Cheney, Washington

In Partial Fulfillment of the Requirements

For the Degree

Master of Science

By

Christopher P. Brady

Summer 2020

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MASTER'S THESIS

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Abstract

To sustain forest health, increase species diversity and reduce wildfire events in the ponderosa pine (*Pinus ponderosa*) forests of at Turnbull National Wildlife Refuge (TNWR) managers have implemented a variety of approaches including prescribed burning and non-commercial thinning. The impacts of thinning on owl species diversity and habitat occupancy have not been studied in these ponderosa pine forests. My study had the following objectives 1) compare forest stand metrics between treatment plots, 2) compare owl species richness between treatment plots, 3) examine if occupancy and detection patterns of Great Horned Owls varied with forest condition and season, respectively, and 4) determine how the frequency of calls and frequency of detection of calls for Great Horned Owls varied between treatments and seasons. I established three stations within three sites for each of three treatments: control (no management activity; thinned 5 years and thinned 11 years prior to study. I measured 13 habitat metrics associated with trees and ground cover at each station. I used SongMeter SM2+ digital recorders to collect owl calls in nightly sessions from 30 June 2014 to 10 August 2015. I identified 112,025 territory hoots (calls) between 27 sampling stations over 1,107 sessions using Raven Pro 1.5 spectrographs. Although there were more live trees and snags in the Control plots, the diameter at breast height did not differ between treatments. There were more stumps in the 5 year since thinning treatment. I identified only Great Horned Owl calls, although Northern Pygmy Owls, *Glaucidium gnoma,* were observed in areas adjacent to my treatment sites. Occupancy and detectability analysis conducted using Program Presence revealed no differences in occupancy between the control and thinned plots but detectability was least during fall and early winter. Chi square analysis of the frequency of call detections and of calls also indicated a decrease in the number of detections during fall and early winter as well as an increase in call frequency in late winter and lasting until spring. These results indicate that

Great Horned Owls dominate the ponderosa pine forests of TNWR and that thinning has little impact on their use of the forests or calling behavior. Monitoring of owls in all refuge habitats over multiple years is recommended too understand species richness of owls across the entire refuge.

Acknowledgements

First, I would like to thank Michael I. Rule with Turnbull National Wildlife refuge for the opportunity to study at TNWR. I would like to give special thanks Dr. Margaret A. O'Connell for all your help, insight, and tireless efforts. Thanks to Dr. Krisztian Magori for assistance and guidance with R and with data analysis . Thank you to Dr. Stacy Warren for help with ArcGIS and ArcMAP. Thanks to the three of you for your help with this manuscript. Many volunteers helped with moving callboxes, conducting habitat surveys, and entering data: Kimberly La Mora, Shelby Hunter, Robert Shadix, Shamus Riley, Justin Donahue, Sam Clark, David Edwards, Bronwyn Edwards, Cayce Martin, Gunner Davies, Marissa Medina, Alexandria Anderson, Jenifer McGovern, Brittany Moore, and Jared Pyeatt. Your efforts are greatly appreciated. This study was supported by Eastern Washington University Turnbull Laboratory of Ecological Studies Barry Research Grant (2014), EWU Biology Department Mini Grant (2014) and through the Society for Ecological Restoration Northwest Student Research Grant (2014).

Finally, and most importantly, I would like to thank my wife, Rhianna, and children, Dominic, Alyssa, Devin and Ian, for your help, patience, and support as I completed this project.

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Introduction

Restoration of habitats is an increasingly important approach to counter the negative impacts of habitat loss and degradation on biodiversity and ecosystem function. Indeed, the United Nations (UN) declared 2021-2030 as the "UN Decade on Ecosystem Restoration" in March 2019 (UN Environment 2019). Worldwide nearly two billion hectares are managed for some form of restoration (Stanturf et al. 2014). Although most restoration projects address the similar overarching goals of restoration of species, restoration of whole ecosystems or landscapes, and the restoration of ecosystem services (Ehrenfeld 2000), objectives and management strategies of specific restoration projects must address local conditions. As forests cover more than 30% of the global land surface, forest restoration is a key component to address global restoration goals (FAO 2016).

In the Pacific Northwest, pre-European settlement ponderosa pine, *Pinus ponderosa*, forests historically achieved successional change through low-severity, low-intensity wildfires that occurred every 5-15 years (Agee 1996). Naturally-occurring wildfires opened large patches of canopy, gaps, within older ponderosa pine stands, by killing off unhealthy mature trees, reducing the density of juvenile stands and removing litter loads (Agee 1996; Pasanen et al. 2015; Abella and Springer 2015). Surviving, fire-tolerant trees had reduced competition and resulting assemblages were comprised of trees with high-crowns and variable-spacing (Hessburg et al. 2015). After European settlement of the Pacific Northwest, fire suppression and modified land-uses interrupted the natural succession provided by wildfires (Kennedy and Wimberly 2009; Gaines et al. 2010), resulting in increased fuel loads in the form of detritus, deadfall, snags, and dense juvenile tree stands (Youngblood 2001). These changes altered the natural structure and composition and impacted forest diversity and health (Caldararo 2002).

To sustain forest health and restore ecological function in these forests, prescribed burning, and other fuel reduction approaches, such as non-commercial thinning (hereafter, thinning), have been implemented (Allen et al. 2002, Briggs et al. 2017). Although the implementation of prescribed burning, thinning, or combination of both varies with local conditions (e.g., amount of fuel loading, proximity to urban areas, topography), the goal is to introduce disturbance that simulates the historic thinning regimes and leads to successional changes brought about by wildfire (Certini 2005; Fule et al. 2012). Both approaches initiate early secondary successional changes within the treated areas where gaps in the canopy are opened, thereby increasing understory growth of forbs and grasses, altering soil chemistry and light regimes, and increasing the heterogeneity of the forest by increasing biodiversity of plants (McConnell and Smith 1970; Widenfalk and Weslien 2009; Ruiz-Mirazo and Gonzalez-Rebollar 2013; Muscolo et al. 2014). Modeling suggests that thinning benefits overall forest health as the surviving trees exhibit greater resistance and resilience to fire (Fill et al. 2015). Monitoring of forest condition and wildlife response to restoration approaches is a critical component contributing to our understanding of restoration of ponderosa pine forests (e.g., Briggs et al. 2017).

Given the inherent variability of site conditions, restoration protocols, and that a main goal of restoration of ponderosa pine forest is to promote heterogeneity by simulating successional change, it is not surprising that wildlife response to restoration of these forests is variable. Depending upon their diet, food habits, nesting requirements (Bateman and O'Connell 2006), the diversity and abundance of avian species (George and Zack 2008; Hurteau et al. 2010) have been observed to increase, decrease, or remain unchanged in response to prescribed burning and thinning of ponderosa pine forests (e.g., Easton and Martin 1998; Bull and Wales, 2001; Bayne and Nielsen 2011). Most research has focused on the response of passerine and woodpecker species. In contrast, in the Pacific Northwest, how these restoration activities affect the distribution and abundance of owls has primarily focused on the Northern Spotted Owl (*Strix occidentalis*; e.g., Kennedy and Wimberly 2009; Clark et al. 2013; Henson et al. 2013). The goal of my research was to monitor vocalizations to examine how thinning of ponderosa pine forests in eastern Washington affects the diversity and abundance of owl species with a focus on the occupancy and detection patterns and vocalization behavior of the Great Horned Owl (*Bubo virginianus*).

The potential impacts of thinning on different owl species might vary depending upon each species' nesting requirements, predation habits, and competitive interactions with other owl species (Olsen et al. 2006; Manning et al. 2012). Owls in western pine forests are secondary nesters that do not build their own nests but are dependent on primary nesters for nesting sites. Nests are either open nests constructed by other raptors (*Buteos*), crows (*Corvidae*) or tree cavities excavated primarily by woodpeckers (Piciformes). Impacts of thinning on either primary nester abundance and distribution or on forest structure (e.g., the size and distribution of snags; Zarnowitz and Manuwal 1985; Miller 2010) can alter breeding opportunities for many owl species. Predation rates of owls are determined, in part, by prey availability and behavior. Thinning has different effects on the diversity and abundance of both avian (Olsen et al. 2006; Manning et al. 2012) and small-mammal prey (e.g., Converse et al. 2006; Gitzen et al.2007; Sullivan et al. 2009; Kalies and Covington 2012). Furthermore, smaller owls are prey to larger owl species and thinning can increase predation on smaller owls by the removal of cover that once concealed tree cavity nests (Jones et al. 2002). Owl species typically reduce interspecific competition through differential hunting strategies (Janes and Barss 1985; Holt and Leroux 1996; Trejo and Guthmann 2003), but changing habitat conditions in forests can also affect interspecific competition by changing conditions that favor one species (Wiens et al. 2011).

Turnbull National Wildlife Refuge (TNWR) in eastern Washington provides an excellent opportunity to examine how early successional changes to habitat conditions, created by thinning, impact owl species. Multiple owl species have been reported on TNWR. Van Horn and Kelly (1975) and TNWR biologists reported that owl species historically found at TNWR include the Barn Owl (*Tyto alba*), Great-horned Owl (*Bubo virginianus*), Northern Pygmy Owl (*Glaucidium gnoma*), Northern Saw-whet Owl (*Aegolius acadicus*), Western Screech Owl (*Otus kennicottii*), Long-eared owl (*Asio otus*), and Short-eared Owl (*Asio flammeusless*). Refuge surveys conducted annually (except for 2004 and 2005) from 1983-2006 visually confirmed 438 individuals of six of these species: Great-horned Owl (88% of sightings), Northern Pygmy Owls (5%), Short-eared Owls (4%), Long-eared Owls (2%), Barn Owls (<1%), and Northern Saw-whet Owls (<1%) (Michael I. Rule, TNWR Biologist, personal communication). TNWR initiated restoration of refuge ponderosa pine forests in 1992 using a combination of prescribed burns and non-commercial thinning. There are multiple forest stands on the landscape that have been thinned at different times as well as stands that have had no management activity.

The first objective of my study was to 1) compare forest stand metrics between the control and two sets of treatment plots, the sets were either thinned 5 or 11 years prior to the study. The second objective was to 2) compare owl species richness between control and thinned forest stands. Habitat conditions created by restoration activities in ponderosa pine forests might alter occupancy patterns of individual species (e.g., George and Zack 2008). Therefore, my third objective was to 3) examine if occupancy and detection patterns of the common species, the Great-horned Owl, varied with forest condition and season, respectively. I used passive song recording to monitor owl presence which allowed me to address a final objective, to 4) determine how the frequency of calls and frequency of detection of calls for Great-horned Owls varied between treatments and seasons.

Methods:

Study Area

My study was conducted on Turnbull National Wildlife Refuge (TNWR) which is located 7.24 km (4.5 miles) from Cheney, Washington. The refuge encompasses approximately 7,284 ha. (~18,000 acres) of channeled scablands, ponderosa pine forests, quaking aspen (*Populus tremuloides*) stands, basalt outcroppings and wetland areas (Van Horn and Kelly 1975). Forested areas are dominated by ponderosa pine that have been managed by prescribed burning and thinning, providing areas of different time-since-thinning to use as my treatment sites.

Treatment Sites and Stations

Three treatments were based on the time since thinning: control (no timber management), 5 years since thinning, and 11 years since thinning (Figure 1). No assumptions about the presence of owl species were made. I established 3 sites per treatment with 3 sampling stations per site. Each sampling site contained a single transect containing three sampling stations, in which each station held a single autonomous digital recorder. This yielded a total of 27 replicates (3 stations per site x 3 sites per treatment x 3 treatments; Figure 1). Sampling stations were placed ≥ 200m apart within each site transect to reduce recording overlap of calls by two or more recorders within the same sites (Schieck 1997; Buxton and Jones 2012; Buxton et al. 2013). To prevent duplication of calls between adjacent sites, all sites were spaced a minimum of 1.5 km from adjacent sites and verified using distance measurements in ArcMap.

Station Mapping

For my base map, I used GIS data provided by TNWR biologists, which included boundaries, roads, and polygons representing the forest management units with attribute tables providing the fire prescriptions used within each unit and/or subunit. From this I was able to filter out all sites that did not fall into the three treatment levels described above. Once the 9 sites were chosen, I walked each site marking the origin point of each station (a tree/snag a minimum of 50 meters from all site borders). I used a Garmin etrex 20 x GPS to determine the GPS coordinates of that tree/snag and save the coordinates as a waypoint to be incorporated into maps of my study. Maps were created in ArcMap using the base maps provided by the TNWR staff and the station location waypoints. Square buffers were created to show the outline of the areas surveyed for the habitat metrics used in my study. Tables containing the calls data and habitat metrics were joined to appropriate layers to allow for use in the ArcGIS online cloud-based GIS mapping software (Figure 2).

Vegetation Surveys

To compare the habitat condition between treatment sites, data were collected at each sampling station for the following habitat metrics: frequency and size of standing live trees and snags, stumps, downed wood, shrubs, groundcover frequency and percent cover (Table 1). At each sampling station, I established a survey plot 100 m in width x 100 m in length centered at the tree where the digital recorder for that treatment site station was located. The perimeter was marked with rebar stakes and along the cardinal directions to create quadrants (50 x 50 m) within the sampling station plot. Nine sampling visual plots 10 x 10 m (100 m^2) were established within each station plot.

Standing Live Trees and Snags

Within each quadrant, the total number of live trees was recorded by species and the diameter at breast height (DBH) was measured and assigned to the following size classes: 1: 4-10 cm; 2: 11-25 cm; 3: 26-50 cm; 4: 51-75 cm; 5: 76-100 cm; 6: > 100 cm. Live trees surveyed were also assigned one of three tree-top classifications as LT: live tree, DT: dead tree top, and

BT: broken tree top. Trees were defined as having a minimum DBH of at least 1 cm and a single stem. If it had multiple stems it was considered a shrub.

The total number of standing dead trees (i.e., snags) was recorded by species, DBH and decay classes. The three classes were 1: recently dead, little decay, retention of bark, branches and top, 2: evidence of decay, loss of some bark and branches and possibly part of the top, and 3: extensive decay, missing bark and most branches, and broken top. Snags < 1.3 m high were considered stumps. Fallen dead tops were considered ‘logs’ and were not measured using the below method. The presence of nests was also noted as either nest (those built within the branches or other external features of the tree) or nest cavity (those hollowed out or formed within the interior of the tree).Total counts per sampling station for Live Tree, Dead Top, Broken Top and Snags along with the mean for Live DBH were recorded.

Shrubs

A total count of shrubs was recorded. Only shrubs that rooted within the 100 m^2 visual plots were measured. The species of each shrub was recorded, being careful not to double count shrubs near the center of the plot. To determine the size of each shrub the height of the shrub was assigned to 1 of 4 categories (1: below knee; 2: knee to waist; 3: waist to shoulder; 4: above shoulder). Sub-shrubs are essentially forb-like plants but with woody stems were not counted as shrubs but instead are measured with groundcover. [Subshrubs include *Linnaea borealwas* (twinflower), *Chimaphila umbellatum* (prince’s pine), *Antennaria spp*. (pussytoes), *Eriogonum spp*. (buckwheat)].

Downed Wood and Stumps

Within the quadrants at each sample station, the total number of fallen logs (downed wood) and stumps were recorded. Downed wood size class information was recorded as 1: >5 m long + ≤15 cm diameter; 2: >5 m long + 16-24 cm dia.; 3: >5 + ≥25 cm dia.; and 4: <5 m long and

>25 cm dia. Decay classes were recorded as follows: DC1: freshly fallen with bark intact, wood solid and no decomposition; DC2: bark beginning to slough off or almost completely gone, decomposition started – sapwood softened but log generally solid; DC3: decomposition to extent that wood soft and breaks into chunks easily; and DC4: wood decomposed to soil like texture. DBH of stumps was measured stumps and were assigned to the above decay classes, and sapling stumps (trees < 4 cm in diameter) were included in the stump count and Stump DBH. Total counts per sampling station for the combined Downed Wood and Stumps for all wood size classes along with Stump DBH were recorded.

Groundcover

I used the nested plot method to estimate the frequency and percent coverage of groundcover by herbaceous plants, rock, marsh, and open ground. To estimate frequency, species of vegetation and substrate were assigned to 1 of 3 categories based on which section of the plot they were first identified (1: first 1% of the plot, 2: 2-10% of plot, or 3: 11%-100% segment). To estimate cover, vegetation was visually estimated into broad categories to determine % composition within the plot (e.g., % Grass, %Forb, % Exposed Rock, % Marsh, and % Open Ground). The mean percent of coverage were recorded per sampling station for the combined Downed Wood for all wood size classes.

Digital Recording:

Owl presence was assessed by passively monitoring owl calls at each station within each treatment level site. Previous diurnal surveys of raptor species at TNWR used visual identification of raptors in various areas of the refuge (Van Horn and Kelly 1975). Given that most owl species are nocturnal, it is more effective to use different methods to detect and identify them (Odom et al. 2013). Owls species may be sampled acoustically using two main methods: active sampling using call broadcasting to elicit response calls or passive monitoring using audio recording equipment to record unsolicited calls (Whitehead 2009). I did not use call broadcasting for several reasons. Broadcast calls can have a negative impact on call response and behavior of some owl species. For example, Spotted Owls have been shown not respond to conspecific calls in areas occupied by or where calls of Barred Owls (*Strix varia*) and Great Horned Owl have been broadcast (Crozier et al. 2005, Crozier et al. 2006). Additionally, factors such as the current moon phase, weather conditions (e.g., wind, rain) and time of year impact the response of owls and the ability to detect these responses (Morrell et al. 1991). Odom et al. (2013) found that the use of passive electronic recorders to monitor Great Horned Owls, especially duetting males and females in the breeding season, was easily done, and has implications for increasing the ability to monitor Great Horned Owls while reducing the time and resources needed for call broadcasting. Therefore, I used passive electronic recorders to monitor the owls. Autonomous digital recording was performed using Song Meter SM2+ (Wildlife Acoustics, Inc.) at each station. Song Meter SM2+ recorded for 10-minute intervals with a 5-minute break between recording periods. Recording began at 1800 hrs and ended at 0600 hrs PST from 30 June 2014 to 2 February 2015, and then from 2200 hrs till 1000 hrs PST from 3 February 2015 to 7 August 2015. Callboxes were moved every Monday, Wednesday, and Friday during daytime hours. Callboxes were placed within each site once every three weeks for two to three nights (sessions). These three-week periods comprised one sampling Round, with a total of 18 Rounds being completed for this study. Times were altered to collect calls from diurnal owl species. Recordings generated 480 minutes of recording time for every session of recording per digital recorder per sampling station. Three breaks in recording occurred between Round 21 and 22, from 24 November until 2 December 2014; within Round 15 from 8-19 May 2015 (due to a moose damage to 6 SMX-II weatherproof acoustic microphones); and within Round 17 8-15 July 2015. The digital recorders provided approximately 14-21 recording sessions (2-3 days in length) per battery change (Buxton and Jones 2012), and battery life typically lasted 18-21 sessions of recording before replacement. Calls were recorded as .wav files to 16GB SDHC cards by each recorder and were collected three times a week during daytime hours to reduce any stress to owls within the treatment sites.

Owl Species Identification

All recorded sounds were displayed as spectrograms and waveforms using Raven Pro versions 1.4 and 1.5 bioacoustics software (Cornell Lab of Ornithology). Recordings were then processed using the manual selection tools provided to manually select for any owl calls that were identified. Early on it was determined that Great Horned Owls was the only species being identified. Adult Great Horned Owls use vocalizations of three types: hoots, chitters and squawks (Kinstler 2009). As defined by Kinstler (2009), hoots include calls that are constructed from the owl making a "hoo" sound through expansion of the gular sac. The "territorial hoot" is different between sexes with males giving 4-5 lower frequency notes and females issuing 6-9 higher frequency notes (Kinstler 2009). Baumgartner (1938) states that males are regularly heard at night throughout the year as a means of defending their territory and to attract and communicate with a mate, while females are typically only heard during the time leading up to and during nesting. Mated pairs will "duet" regularly throughout courtship and nesting allowing for sampling of calls for both sexes (Houston 1999, Kinstler 2009, Odom et al. 2013). Given that Great Horned Owls are best identified using their territorial hoots (Kinstler 2009), I used the territorial hoots (Figure 3) and saved calls as manual selection table text documents.

Data Analysis

Unless otherwise noted, all statistical tests were performed in the statistical program R (version 4.0.1) with $P < 0.05$ significance levels. To examine owl species richness, manual selection tables were aggregated in R to obtain the total number of owl calls identified (vocalizations detected) per session based on the treatment, site, and station.

The habitat metrics for Live Tree, Dead Top, Broken Top, Downed Wood, Snag, Shrub and Stump were not normally distributed were compared between the three treatments using Poisson regressions. Poisson regressions were used because these data. The habitat metrics for Live DBH, Stump DBH, Herbaceous, Rock, Marsh and Open Ground were normally distributed and therefore compared between the three treatments using one-way ANOVA with Tukey HSD tests for pairwise comparisons.

Single-season occupancy models in Presence 13.5 software (Hines 2016, MacKenzie et al., 2002) were used to evaluate probability of site occupancy and detection models for Great Horned Owls. Detection data by round, treatment, and season (0 for absence and 1 for present) were entered into an Excel spreadsheet and saved in .csv format. Presence was recorded if an owl call was detected anytime between 2200 to 0300 hours the proceeding morning and any sessions not surveyed due to technical issues or interruptions to the sampling schedule were missing values. A station was considered occupied if one Great Horned Owl vocalization was recorded during a given round. Covariates for site occupancy were the three treatment categories (Control, 5 Years and 11 Years Since Thinning). Given that frequency of Great Horned Owl vocalizations has been observed to vary depending upon time of year relative to life history (Baumgartner, 1938; Baumgartner, 1939), I used three detection covariates based on seasons. Season 1, Dispersal, was from June–August and corresponded to the time when young owls and mature adults expand their territories beyond the nest site (Baumgartner 1938, Baumgartner

1939). Season 2, Reestablishment, was from September–January and corresponded to the return of males to perches near potential nest sites (Ridgeway 1874 as quoted in Baumgartner 1939). Season 3, Nesting, was from February through June and corresponded to males and females remaining very close to the nest site, females going no further than 0.5 km from the nest (Bennet and Bloom 2005). Four a priori models incorporating different covariates were generated to assess the probability of site occupancy (ψ) and detection (*p*) for Great Horned Owls in relation to the three treatments and seasons.

I tabulated the total number of vocalizations, the frequency of vocalizations per session, the frequency of sessions in which Great Horned Owls were recorded and the frequency of calls per session detected. The frequency of detections was compared between treatments and seasons and the frequency of vocalizations per session detected was compared between treatments and seasons and using Chi square analysis (Table 2).

Results

Vegetation Metrics

The three treatments significantly differed with respect to 8 of the 13 habitat metrics (Table 1). Shrubs, live tree DBH and most ground cover metrics (% herbaceous plants, % marsh, % open ground) did not differ between treatments. The percent rock cover was greater on the Control sites. Tree metrics (both standing and down, live and dead) varied between treatments. The number of live trees and downed wood was greater on Control sites. The number of dead top trees was less, and the number of stumps was greater on the sites thinned 5 years ago. The number of broken top trees and snags were less and the DBH of stumps was greater on sites thinned 11 years ago.

Species Richness

During the study, there was a total of 1,107 sample sessions, recording a total of 112,025 Great Horned Owl vocalizations (Table 2). Great Horned Owls were recorded at every sampling station and during all seasons (Table 2) and were regularly observed within the treatment sites and the surrounding areas throughout the study. No other owl species were recorded during the study. However, the Northern Pygmy Owl was observed and heard regularly in the spring and summer months during the daylight hours in aspen stands adjacent to one treatment site.

Occupancy Patterns

Of the four models representing different combinations of occupancy and detection covariates and one model with constant probability, two models had a combined model weight of 100% (Table 3). The top model incorporated the three seasons and the second model added the three habitat covariates. The naïve occupancy rate was 1.0. Consequently, the site estimates for the three occupancy estimates were all equal (27.445). In contrast, detection estimates (± SE) varied: 2.833 ± 0.325 for Season 1; 1.012 ± 0.178 for Season 2; 4.167 ± 0.713 for Season 3.

Detection and Vocalization Between Seasons and Treatments.

Great Horned Owls were recorded during all three seasons and treatment levels. There was no difference in the frequency of detections or the frequency of calls per session detected between the three treatments (Table 2). Great Horned Owls called more frequently during Nesting than Reestablishment or Dispersal seasons (Table 2). The frequency of detection of calls was lower during the Reestablishment season (Table 2)

Discussion

TNWR uses a combination of management approaches, including prescribed burns and non-commercial thinning to restore ponderosa pine forests on the refuge and to reduce fuel loads for wildfires. Both approaches introduce successional changes and create positive gains to ecosystem function to these refuge forests (Winford et al. 2015). My study examined how habitat conditions varied with thinning and if thinning affects owls in these ponderosa pine forests. Although habitat metrics, especially those associated with trees, varied between treatments, there is little evidence that these conditions affected the owls.

Similar to observations by Fule et al. (2012), thinning appeared to mimic successional change by reducing stand density. Reduced stand density was evident in the reduced number of live trees and downed wood in the thinned as compared to control sites. In addition, the number of live trees with dead tops, snags increased with the progression of time since thinning. In contrast, the similarity of DBH of live trees across treatments reflects the selection for larger trees to remain after thinning. The number of stumps were highest in the 5 Year treatment attributed to the thinning that occurred most recently, mimicking the die off of saplings and juvenile stands in the event of a wildfire (Certini 2005; Fule et al. 2012). The stumps within the 11 Years treatment were not as numerous most likely due to the decay of the smaller stumps over the longer period since thinning. The reduction in juvenile stands creates greater, open areas (Agee 1996) that are associated with healthier pine forests (Hessburg et al. 2015). Stump DBH though significant between treatments, is likely due to the greater number of sapling stumps that were counted in the 5 Years treatment. It should be noted that over a greater period of time-since-thinned such as in the 11 years treatment, most of these small sapling stumps would have decayed completely. Despite habitat differences between treatments, there were no differences in the species richness of owls or Great Horned Owl occupancy patterns and call behavior between the treatments.

The regular presence of Great Horned Owls, observed in my study as well as in previous studies of Van Horn and Kelly (1975) and earlier TNWR surveys (Michael I. Rule, TNWR Biologist, personal communication) might be the main factor impacting owl diversity. Baumgartner (1939) noted that both Barred Owls (*S. varia*) and Barn Owls (*Tyto alba)* were observed only in areas where Great Horned Owls were not heard or seen, and that *B. virginianus* is known to predate smaller owls. Long-eared Owls (*Asio otus)* also avoid Great Horned Owls because Great Horned Owls are very territorial, usurping nesting sites and, in addition, preying on Long-eared Owls (Knight and Erickson 1977; Errington et al. 1940).

These attributes most likely account for the dominance of Great Horned Owls on TNWR. Although the raptor surveys conducted 1983-2006 on TNWR recorded the presence of six owl species, only Great Horned Owls were recorded during each of the 23 survey years on all 12 survey routes (Michael I. Rule, TNWR Biologist, personal communication). Northern Pygmy Owls were observed during 13 of the 23 survey years on 4 survey routes. The Northern Pygmy Owl is a diurnal species of owl that preys upon small birds and mammals (Hayward and Garton 1988). Although I never recorded Northern Pygmy Owls, I observed them in aspen groves immediately north of South Lower Turnbull along the basalt rock outcropping and drop-off. The owls were perched on bare aspen branches just a few meters above the ground. These owls were immediately adjacent to Great Horned Owl territories, but the two species do not compete for food and predation on Northern Pygmy Owls by Great Horned Owls is reduced due to the Northern Pygmy Owl's diurnal activity and cavity-nesting habits (Hayward and Garton 1988). As Northern Pygmy Owls were regularly observed or heard within the aspen stands on basalt outcroppings found near South Lower Turnbull, future studies to identify the locations of similar aspen stands on the refuge and their use by Northern Pygmy Owls could be useful in helping to clarify habitat restoration goals and efforts of these unique areas.

Short-eared Owls were observed during 10 of the 23 survey years but all but one observation was on a single transect, in the open habitat of the Stubblefield portion of the refuge. Barn owls were observed during only two survey years, also in more open habitat sites. Given that my study focused on ponderosa pine habitat, Barn and Short-eared Owls might not have been recorded due to their preference for more open areas (Van Horn and Kelly 1975; Van Horn and Kelly 1975). Although I sampled more open areas at two sites (North Camas Canyon and East Aspen Meadows), I did not record Barn or Short-eared Owls. These areas might not be large enough to attract these two species, or, because as noted above Barn Owls avoid Great Horned Owl territories.

Long-eared Owls were observed during five survey years on two survey routes and a single Northern Saw-whet Owl was recorded in 2001. I did not record either species during my study. Both species might avoid Great Horned Owl territories to reduce competition and predation.

While the competitive and predator behavior of Great Horned Owls definitely impact other owl species, the low species richness I observed might be also an artifact of my focus on the ponderosa pine habitat and the limitation of my study to one year. The sporadic observations of most owl species during previous surveys on TNWR suggest that some species are more specialized in habitat use than the Great Horned Owls and, perhaps that some are more transient in their use of TNWR. The use of passive electronic recorders was an efficient approach to monitor owls in my study sites, but to fully understand the species richness and diversity of owls on the entire refuge, monitoring would need to be conducted in multiple habitat types over multiple years.

For TNWR refuge managers, my study suggests that restoration efforts using non-commercial thinning have no effect on Great Horned Owls. Great Horned Owls showed no variation in occupancy in relation to any specific treatment level, the time-since-thinning occurred, and there were no differences in the frequency of detections between treatments or the frequency of calls per session between treatments. Some owl species, such as the Spotted Owl, will alter and decrease their use or occupancy of an area after successional change has been introduced (Clark et al. 2013) . In old growth forests of southwestern Virginia, Great Horned Owls were observed to respond to call broadcasts in old stands more frequently than young stands (McGarigal and Fraser 1984). McGarigal and Frasier (1984) suggested that the larger trees in older stands were necessary to Great Horned Owls for hunting, perching, and nesting (1984). Thinning regularly retains larger, healthy trees in a stand while smaller trees are removed to mimic the successional changes historically introduced by wildfire (Agee 1996; Pasanen et al. 2015; Abella and Springer 2015). In my study, there was no difference in the live tree DBH between the control and the thinned sites, suggesting that the owls had access to large trees in each of the three treatment levels. This most likely explains the similarity in occupancy across treatments. In contrast, the Great Horned owls did exhibit seasonal variation in calling behavior.

Time of year affected detectability of the Great Horned Owls. Both detection estimates and the frequency of detections per session were lower during the Reestablishment Season than either the Dispersal or Nesting Seasons. During the Nesting Season, however, the frequency of calls per session was greater than in the other seasons. Similar patterns of seasonality in Great Horned Owl calling has previously been reported. Baumgartner (1939) describes males perching within territories in fall and early winter near possible nesting sites prior to the arrival of females. He describes male owls calling independently, with little to no duetting, and randomly as a means of protecting their territory and possibly attempting to attract females. As Great Horned Owls do not build their own nests, they must search an area for abandoned nests or forcibly take nests from other avian species and will only reuse nests for 2 years (Gardner 1929; Hagar 1957; Knight and Erickson 1978; Smith et al. 1999). To reduce competition during the Reestablishment Season, males might reduce the rate at which they call until a suitable nest is identified, and courtship begins. During courtship and nesting (my Nesting Season), both male and female Great Horned Owls increase the frequency of territorial hoots, sometimes calling and/or duetting for hours each night (Baumgartner 1938, Odom et al. 2013).

In conclusion, species richness of owls in the ponderosa pine forests is dominated by a single species, the Great Horned Owl most likely due to the competitive and predator behavior of this species. To understand the species richness of the entire refuge, monitoring must be conducted in different habitats over many years. My study suggests that thinning has little impact on Great Horned Owl habitat occupancy and calling behavior. However, calling behavior did vary seasonally and reflect the breeding and nesting phenology of the Great Horned Owls.

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Tables

Table 1: Comparison of means (± SE) for the 13 study habitat metrics (defined in text) measured at 27 stations from 3 treatments on Turnbull National Wildlife Refuge, Cheney, WA. Metrics that were not normally distributed were compared using Poisson regression with the residual deviance, df, and Chi-square probability reported. Metrics that were normally distributed were compared using one-way ANOVA with F-value, df and p reported. Means with different letters are significantly different ($p \leq 0.05$ at CI = 95%).

Treatments:	Live Tree	Dead Top	Broken Top	Snag	Downed Wood	Shrub	Stump
Control	322.7±42.6 [a]	15.2±3.6 [a]	17.7±3.3 [b]	28.6±9.8 [b]	104.6±21.6 [a]	7.9±1.9	9.3±3.7 [a]
5 Years	134.8±14.0 [b]	2.1±1.0 [b]	15.1±3.7 [b]	25.7±11.6 [b]	69±20.6 [b]	5.4±1.1	157.3±67.5 [b]
11 Years	133.4±27.9 [b]	11±6.5 [c]	12.1±4.4 [a]	18.1±5.8 [a]	70.7±9.8 [b]	7.7±2.4	32.8±13.3 [c]
Residual Deviance	871.62	272.63	207.0	564.78	856.84	114.12	2,279.4
Chi square	Df 24 P < 0.001	Df 24 P<0.001	Df 24 P=0.009082	Df 24 P<0.001	Df 24 P<0.001	Df 24 P=0.8626	Df 24 P<0.001

Table 1 – continued.

Treatments:	Live DBH (cm)	Stump DBH (cm)	% Herb.	% Rock	%Marsh	% Open Ground
Control	21.3±4.1	5.1±1.7 [b]	32±2.9	8.2±2.3 [a]	5.1±2.4	24±14
5 Years	29.4±2.6	5.0±0.6 [b]	37.2±2.4	16.6±3.8 [b]	2.1±1.3	23.9±2.7
11 Years	28.4±2.9	13.6±2.3 [a]	32.8±4.1	18.9±2.6 [b]	3.2±2.6	26.1±3.4
ANOVA	F=1.836 Df2	F=8.501 Df 2	F=0.751 Df 2	F=3.589 Df 2	F=0.46 Df2	F=0.159 Df2
p-value	P>0.05	p<0.001	P>0.05	P=0.04	P>0.05	P>0.05

Table 2: Number of sample sessions, number of vocalizations, number of sessions calls were detected per number of sessions, and the number of vocalizations per session detected for each of the three treatments and seasons. Chi square statistics provide comparisons of detection frequencies and the frequency of calls per session detected by treatment and season.

	Sample sessions	Total vocalizations	Sessions detected /session sampled (%)	Vocalizations /sessions detected
Treatment				
Control	364	33,800	269/364 (73.9%)	33,800/269 (125.7)
5 Years	365	41,964	260/365 (71.2%)	41,964/260 (161.4)
11Years	378	36,261	275/378 (72.8%)	36,261/275 (131.9)
χ^2			$\chi^2 = 0.0006429$, df = 2, $p > 0.05$	$\chi^2 = 5.2$, df = 2, $p > 0.05$
Season				
Dispersal	371	25,389	294/371 (79.3%)	25,389/294 (86.4)
Reestablishment	378	7,167	190/378 (50.3%)	7,167/190 (37.7)
Nesting	358	79,469	320/358 (89.4%)	79,469/320 (248.3)
χ^2			$\chi^2 = 153.84$, df = 2, $p < 0.0002$	$\chi^2 = 76.137$, df = 2, $p < 0.0002$

Table 3: Occupancy **(**ψ) and detection (p) models of Great Horned Owls based on vocalization sampling on Turnbull National Wildlife Refuge, Cheney, WA 2014-2015. Model descriptions with covariates, number of parameters (K), Aikaike's Information Criteria adjusted for small sample sizes (AIC_c), difference in AIC_c from most parsimonious model (ΔAIC_c), model weights (W_i) and -2log likelihood values are presented.

Model	K	AIC_c	ΔAIC_c	W_i	--2log likelihood values
p Seasons	4	293.43	0.00	0.8808	285.43
ψTreatments; p Seasons	6	297.43	4.00	0.1192	285.43
Constant ψp	2	344.04	50.61	0.0000	340.04
ψTreatments	4	348.04	50.61	0.0000	340.04

Figures

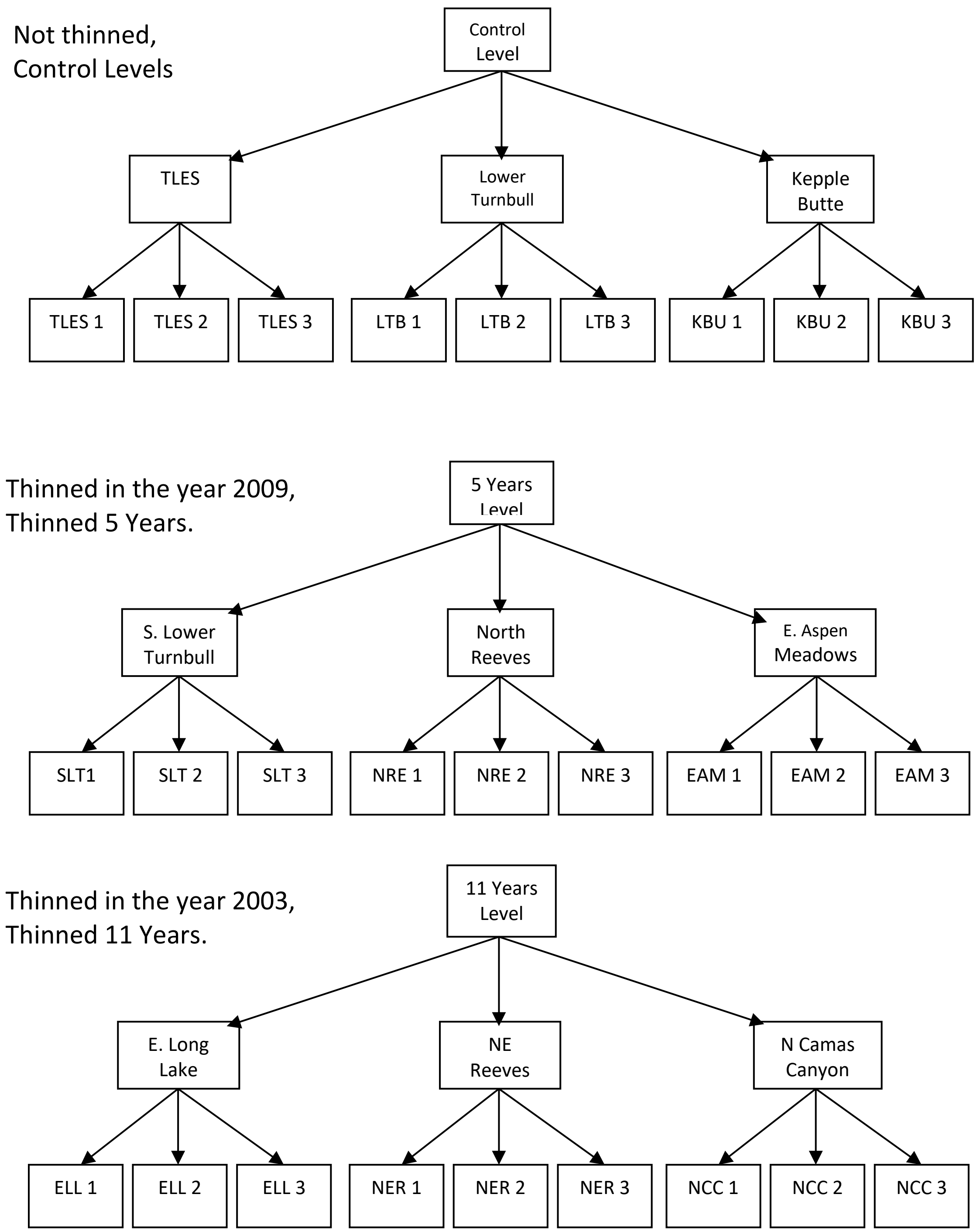

Figure 1: A schematic of the study design used to examine owl response to timber thinning of Turnbull National Wildlife Refuge. The three treatments were Control, 5 Years and 11 Years since thinning. There were 3 sites per treatment and each site had 3 stations for a total of 27 replicates on Turnbull National Wildlife Refuge, Cheney, WA 2014-2015.

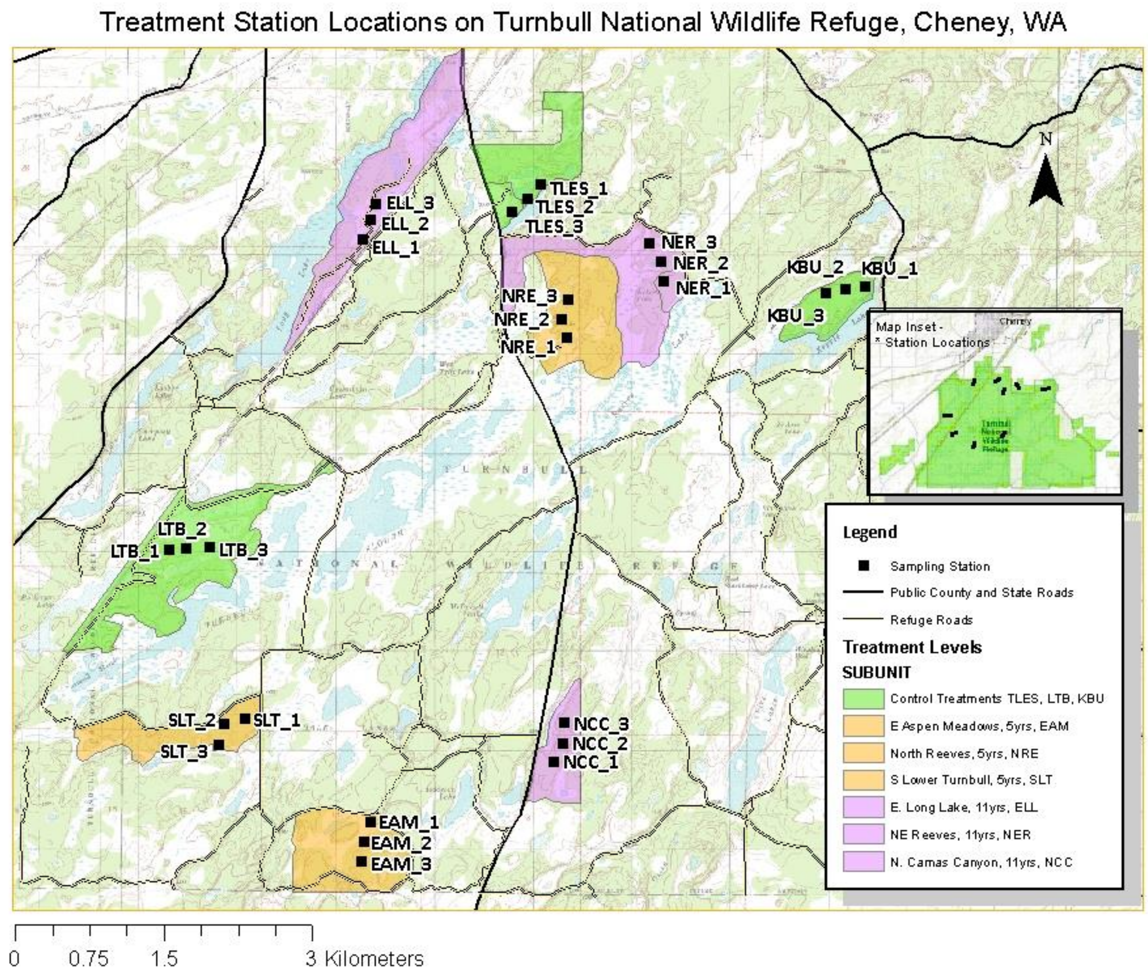

Figure 2: ArcGIS map of the location of the 27 sampling stations within each study site for each of the 3 Treatments to monitor owl vocalizations on Turnbull National Wildlife Refuge, Cheney, WA.

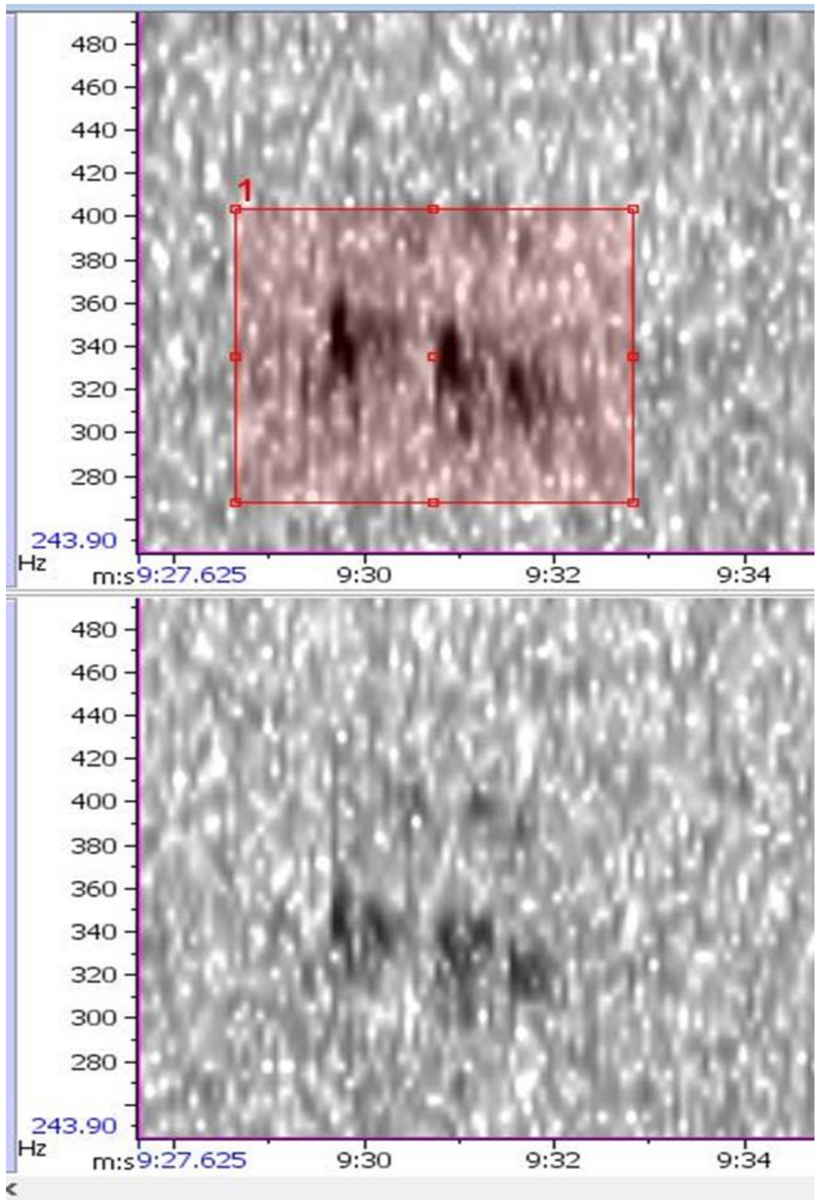

Figure 3: A typical territorial owl hooting call, in a spectrographic image captured using Raven Pro showing frequency (Hz) on the y-axis and the time (m:s) on the x-axis. The call was selected from a saved .wav file from recordings done for this study on Turnbull National Wildlife Refuge, Cheney, WA.

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